9. ANALYSIS OF ECOLOGICAL EFFECTS (ERAGS STEP 6)

In accordance with USEPA guidance, this chapter describes information on observed effects for ecological components of the Onondaga Lake ecosystem. The groups of ecological receptors discussed in Section 9.1 are aquatic macrophytes, phytoplankton, zooplankton, fish, herpetofauna (amphibians and reptiles), and wildlife (birds and mammals). Most of the information on these groups was collected by other parties and is presented in detail in the publications cited herein.

Benthic invertebrates are discussed in detail in Section 9.2, which contains a detailed evaluation of sediment toxicity and benthic macroinvertebrate communities. Most of the data were collected by Honeywell in 1992 and 2000 as part of the remedial investigation (RI). This section also contains a key element of the effects characterization, the development of site-specific sediment effect concentrations (SECs) for Onondaga Lake, using the empirical information collected on sediment chemistry and sediment toxicity during the RI. The development of site-specific SECs is consistent with the recommendations of NYSDEC (1994b, 1997a) to conduct site-specific evaluations based on sediment toxicity tests when chemical concentrations in sediments are found to exceed the state sediment screening values.

The information on benthic macroinvertebrate communities was not used to develop site-specific SECs, because it was found that benthic communities at every station in the lake are impaired to some degree; thus, it is not possible to calculate an SEC because these calculations require that a certain proportion of stations have no effects. Although this lakewide impairment is typical of eutrophic lakes, it can reduce the value of these communities for developing site-specific SECs, particularly if there is ambiguity as to the identity of the factors primarily responsible for any observed community effects. The results of the benthic macroinvertebrate evaluations for Onondaga Lake were, therefore, used primarily to interpret the magnitude and significance of any potential sediment toxicity predicted on the basis of the site-specific SECs.

The last section of this chapter, Section 9.3, presents the effects characterization for aquatic and terrestrial vertebrates. It describes the methodology used and the toxicity reference values (TRVs) selected for fish, avian, and mammalian receptors.

9.1 Onondaga Lake Field Studies/Observations

9.1.1 Aquatic Macrophytes

Aquatic macrophytes form an important part of lake ecosystems. They serve as food for other aquatic organisms and provide habitat for insects, fish, and other aquatic and semi-aquatic organisms. Most aquatic macrophytes are rooted or attached to the sediment, although some free-floating forms exist (Auer et al., 1996a). Little quantitative information existed on macrophyte distributions in Onondaga Lake prior to 1991, when Madsen et al. (1993) conducted the first quantitative survey of macrophyte distributions in the lake, as discussed in Chapter 3, Section 3.2.5.1.

In addition to conducting a field survey of lake macrophytes, Madsen et al. (1993) collected different kinds of sediments from various parts of the littoral zone and used them for greenhouse experiments in which macrophyte growth was evaluated in each sediment type. Two additional field surveys of macrophytes in the lake were conducted by Honeywell in 1992 and 1995 as part of the RI.

During the 1991 field survey conducted by Madsen et al. (1993), five species of submerged macrophytes species were found in Onondaga Lake:

- *Ceratophyllum demersum* (coontail).
- *Heteranthera dubia* (water stargrass).
- *Myriophyllum spicatum* (Eurasian water-milfoil).
- *Potamogeton crispus* (curly-leaf pondweed).
- *P. pectinatus* (Sago pondweed).

An additional five aquatic macrophyte species were found by Madsen et al. (1998) in 1993. These include *Potamogeton diversifolius* (waterthread pondweed), *Elodea canadensis* (Canadian pondweed), *Lemna minor* (duckweed), *Sparganium* sp. (bur-reed), and *Zannichellia palustris* (horned pondweed). All of the newly reported species were relatively rare, with the possible exception of *Elodea canadensis*. The distribution of these species throughout the littoral zone of the lake was found to be relatively limited. The number of macrophyte species found in the lake (i.e., ten) is low, since up to 15 were present in Onondaga Lake before 1940 (Auer et al., 1996a). For a eutrophic lake in New York State, 15 species of vascular plants is typical (Madsen et al., 1993), which is lower than the New York State average of 18 (Madsen et al., 1996).

Madsen et al. (1993, 1996) also conducted laboratory studies to evaluate the role of sediments in limiting plant growth. They planted *P. pectinatus,* the dominant aquatic plant in Onondaga Lake, in lake sediments collected from nine locations that corresponded to areas of high, medium, and low macrophyte abundance, as well as four different sediment types (oncolite, organic, sand, and silt). In addition, the authors also collected a silty reference sediment from a lake in Texas. Madsen et al. (1993, 1996) found that growth on the fertile reference sediment was significantly higher than growth on Onondaga Lake sediments. They predicted that improvement in water clarity or quality alone would not improve plant growth, as sediment degradation is directly related to the input of $CaCl_2$ (calcium chloride) into the lake and resulting calcium carbonate deposition. Sediments from sites with high, medium, and low percentages of plant cover showed significant differences in their abilities to support plant growth, with laboratory results corresponding to plant cover seen in the field.

The work of Madsen et al. corresponds with the macrophyte transplant study performed in the summer of 1992 by Honeywell (PTI, 1993c). The purpose of this study was to determine the extent to which representative macrophyte species can survive and grow in the sediment and water of the littoral zone of Onondaga Lake (PTI, 1993c). Three macrophyte species (*P. pectinatus, P. richardsonii* [redhead grass], and *Vallisneria americana* [wild celery]) were transplanted at two depths (1 and 1.5 meters [m]) at six locations in the littoral zone of Onondaga Lake (Figure 9-1) and at two potential reference lakes (Otisco and Cross Lakes, of which only the former was determined to be an appropriate reference lake and is discussed here). A total of three seeded racks (one for each macrophyte) were deployed at each location at the two depths in Onondaga Lake. Each rack contained three sediment samples from each of the reference lakes and four from Onondaga lake for a total of 10 samples per rack and 30 samples per depth at each location. Twelve samples (three plant species × four reference lake sediment samples) were deployed at two depths at one location in the reference lakes.

The results, as summarized in the table below, showed macrophyte survival to be minimal at Onondaga Lake. The lakewide survival rate was less than 3 percent at 1 m, and improved to slightly under 12 percent at 1.5 m. In contrast, 75 and 58 percent survival rates were seen at depths of 1 and 1.5 m at Otisco Lake.

Macrophyte Transplant Study Results (PTI, 1993c)				
	Mean Percent Survival		**No. of Plants Surviving**	
Location	**1 m**	**1.5 m**	**1 m**	**1.5 m**
Onondaga Lake M1 (between Harbor Brook and Onondaga Creek)	0	0	0/30	0/30
Onondaga Lake M2 (between Onondaga Creek and Ley Creek)	6.7	6.7	2/30	2/30
Onondaga Lake M3 (south of Tributary 5A)	0	27	0/30	8/30
Onondaga Lake M4 (mouth of Ninemile Creek)	6.7	0	2/30	0/30
Onondaga Lake M5 (northwest corner of lake)	3.3	13	1/30	4/30
Onondaga Lake M6 (south of Sawmill Creek)	0	23	0/30	7/30
Otisco Lake	75	58	9/12	7/12

Based on these results, it appears that stressors limit macrophyte growth in Onondaga Lake. According to Auer et al. (1996a), the major stressors potentially limiting macrophytes in Onondaga Lake include the following:

- **Water transparency:** Reduced water transparency in the water column is a major factor in limiting the depth distribution of submerged macrophytes (Canfield et al., 1985; Chambers and Kalff, 1985). Historically, transparency in Onondaga Lake has been very low (<1 m), primarily because of the eutrophic nature of the lake (Effler and Perkins, 1996b). More recently, however, water clarity in the lake has been improving (Effler and Perkins, 1996a), increasing the amount of time in which a plant can complete its life cycle before parent-plant mortality (Auer et al., 1996a).

- **Salinity:** The high levels of salinity that have prevailed in Onondaga Lake in the past may have prohibited the establishment of many common emergent and floating-leaved species, due to increased stress from evapotranspiration (Auer et al., 1996a). Although the salinity of the lake has declined (Effler et al., 1996) and is now within the tolerance range of more species, the dominance of *P. pectinatus* in many habitats may hinder the colonization of returning species.

- **Calcium carbonate precipitation:** The extremely high rate of calcite ($CaCO_3$) precipitation and deposition onto the surfaces of macrophytes in Onondaga Lake in the past may have been sufficient to completely coat plants (Auer et al., 1996a). This mechanism may have been responsible for or contributed to the disappearance of charophytes from Onondaga Lake (Dean and Eggleston, 1984). Calcium carbonate deposition decreased 64 percent from 1985 to 1989, reflecting the relative decrease in external loading (70 percent) after the closure of the soda ash/chlor-alkali facility in 1986 (Effler et al., 1996). However, high concentrations of calcite are still present in and being released to the lake (Effler et al., 1996).

- **Oncolites:** During the 1991 macrophyte survey conducted by Madsen et al. (1993), macrophyte distributions were found to be limited in areas where oncolites were present. The greenhouse experiments conducted by Madsen et al. (1993) indicated that macrophytes grow as well on oncolite sediment as on the other kinds of sediment in the lake. Madsen et al. (1996a) suggested that oncolites may limit macrophytes in the field because their relatively low density makes them susceptible to movement by wave action; therefore, oncolites may provide an unstable substrate for macrophyte colonization.

- **Water level changes:** As described in Chapter 3, the level of Onondaga Lake is regulated as part of the New York State Barge Canal System. Auer et al. (1996a) noted that although annual variations in lake level are usually less than 1 m, even that magnitude of change can have a substantial effect on macrophytes since light restricts macrophyte growth to shallow depths.

Based on the information presented by Auer et al. (1996a), it appears that many potential stressors have become less limiting to macrophytes in Onondaga Lake in recent years. Recently, Madsen et al. (1998) reported on a series of experiments in Onondaga Lake to evaluate techniques for restoration of littoral areas to improve fish habitat. Two temporary habitat areas were created with wave breaks (i.e., hay bales) and fencing for protection of natural colonization by macrophytes, as well as planted macrophytes. The authors found that survival of most planted species was low. Potential causes of low survival include herbivory (primarily by waterfowl) and changes in water levels by 0.3 m after planting was performed. However, the habitats were successful in enhancing colonization by native macrophyte species.

Between July and September, the authors found large amounts of filamentous algae in the habitats, which is generally detrimental to the growth of rooted macrophytes (i.e., through light limitation, competition for nutrients, and mechanical damage). Madsen et al. (1998) found mechanical damage for several emergent species, where floating masses of filamentous algae, driven by wind and wave action, collapsed emergent macrophyte stems and leaves.

9.1.2 Phytoplankton

Phytoplankton communities have been routinely monitored at two stations in Onondaga Lake since 1970 by Onondaga County, following a detailed study of lake conditions in 1969. This monitoring involves collection of water samples, usually biweekly, over the spring to early fall interval from the south deep station (W1) at the surface and at depths of 3, 6, and 12 m (Auer et al., 1996a). This information has been summarized and interpreted by Auer et al. (1996a).

In general, the characteristics of the phytoplankton communities of Onondaga Lake have reflected the eutrophic nature of the lake. Prior to 1972 (when phosphorus loadings were dramatically reduced due to a local ban on phosphates in detergents), cyanobacteria (formerly known as blue-green algae) were common in the lake during the spring-fall growing season. A seasonal succession was described in which the major groups were diatoms in spring, green algae in early summer, and cyanobacteria in the late summer and fall (Auer et al., 1996a). Although cyanobacteria disappeared from the lake after 1972, it returned in the late 1980s, apparently due to more efficient grazing zooplankton, as cyanobacteria may not be a suitable food for zooplankton (Auer et al., 1996a). No obvious changes were observed in nutrient loading during that period. Auer et al. (1996a) noted that as the degree of eutrophy in the lake has declined since the early 1970s, the intensity of phytoplankton blooms has declined. However, the authors also noted that Onondaga Lake remains highly eutrophic and that concentrations of phosphorus in the lake remain sufficient to sustain near-maximum rates of algal growth over the entire summer. The strong seasonal changes in phytoplankton biomass seen in Onondaga Lake represent imbalances between growth and loss processes.

Based on the information summarized by Auer et al. (1996a), the primary stressors that have affected phytoplankton communities in Onondaga Lake are nutrients, which have influenced both the types of species found in the lake and the densities of those species. Although the effect of mercury contamination on the phytoplankton community is unknown, it is evident from the bioaccumulation investigation (PTI, 1993b) that mercury accumulates in phytoplankton and can be passed on to animals feeding on phytoplankton in Onondaga Lake.

9.1.3 Zooplankton

Zooplankton communities in Onondaga Lake have been routinely monitored by Onondaga County in conjunction with the phytoplankton monitoring. This information has been summarized and interpreted by Siegfried et al. (Auer et al., 1996a; Siegfried et al., 1996).

The number of zooplankton species found in Onondaga Lake has increased substantially since the early 1970s. This increase can be attributed to both the increased sampling effort, which has collected a number of rare species, and the closure of the Honeywell soda ash/chlor-alkali facilities in 1986 (Auer et al., 1996a; Siegfried et al., 1996). The soda ash/chlor-alkali process was in operation from 1884 to 1986 and released large quantities of ionic waste (high in calcium, chloride, and sodium ions) into Onondaga Lake, impacting the ecology so that the salinity of the lake and the rate of calcium carbonate precipitation were artificially high (Siegfried et al., 1996).

Salinity affects the osmoregulation capabilities of zooplankton, while calcium carbonate particles can physically interfere with feeding. The chloride/salinity levels of the lake before the closure of the facility were near the upper limit for freshwater organisms. Since the facility has been closed, the salinity and transparency (clear water phases) of the lake have improved. The relative abundance of species within the zooplankton community has also changed considerably since the 1970s. Prior to the mid-1980s, the zooplankton biomass was dominated by *Cyclops vernalis*, a cyclopoid copepod. Since that time, however, dominance has shifted to large-bodied cladocerans (*Daphnia* spp.) and the calanoid copepod *Diaptomus siciloides*. During the period of peak pollution in the 1970s and 1980s native species of Daphnia were replaced by exotic species, such as *Daphnia exilis* and *D. curvirostris*. These exotic species are found in saline environments in the southwestern US and Europe (Hairston et al., 1999, Duffy et al., 2000).

Mercury has also affected the Onondaga Lake zooplankton community. The period of peak mercury concentrations in the sediments based on ^{210}Pb dating, coincides with zero hatching success of *D. exilis* eggs in laboratory monitoring (Hairston et al., 1999). Whether mercury in the water column caused the eggs to become non-viable at the time they were produced, or mercury and/or other contaminants and stressors in the sediments made the eggs non-viable over the burial period, is uncertain.

Despite recent increases in zooplankton diversity, the zooplankton assemblage of the lake remains depauperate compared to other lakes in the region (Auer et al., 1996a). Further reductions in the loadings of ionic waste-associated stressors may result in additional changes to the zooplankton community.

9.1.4 Fish

Most quantitative studies of fish communities in Onondaga Lake have been conducted by researchers at the State University of New York College of Environmental Science and Forestry (SUNY ESF) since 1989, and results of those studies have been summarized by Ringler et al. (Auer et al., 1996a). Only qualitative information exists on fish communities in the lake prior to 1989.

A total of 55 fish species have been collected in Onondaga Lake since 1989 (see Table 3-7). Although this number represents a considerable increase over the numbers found in historical studies, comparison of current species richness with historical values is difficult because considerably more sampling effort was used in the more recent studies, increasing the probability of collecting more species (Auer et al., 1996a; Tango and Ringler 1996). The current level of species diversity in Onondaga Lake is similar to values found in other New York lakes, and growth rates, age distributions, and mortality rates of several species are similar to those observed in other northeastern US lakes (Auer et al., 1996a).

However, in contrast to comparison lakes, many of the species found in Onondaga Lake do not reproduce there and recruitment rates are unknown. Only 16 of 48 species captured in 1991 were found to reproduce in the lake, and reproduction within the lake varied by location. Based on the absence of juveniles in the catches of shoreline seine hauls, species such as the walleye and northern pike are thought not to reproduce in the lake. Areas characterized by the presence of aquatic macrophytes and submerged structures (e.g., near the lake outlet) supported the largest populations of juveniles. Areas with heavy silt loads and that are unprotected from wind are undesirable as spawning areas, as silt loads or wave action may cause eggs to be covered or removed from optimal areas (Auer et al., 1996a).

As discussed in Section 3.2.5.1, Onondaga Lake supports a warmwater fish community that is dominated by the pollution-tolerant gizzard shad (*Dorosoma cepedianum*), freshwater drum (*Aplodinotus grunniens*), carp (*Cyprinus carpio*), and white perch (*Morone americana*) (Auer et al., 1996a). Food-habit studies documented important prey species in the lake, including fish, fish eggs, zooplankton (primarily copepods and cladocerans), and benthic macroinvertebrates (primarily chironomids). However, the absence of macrophytes from many areas of the lake reduces the use of those areas by fish.

The composition of the fish community in the lake varies seasonally, with migration between the Seneca River and the lake being an important contributor to the variability. One of the major changes in the fish community occurs during fall turnover, when concentrations of dissolved oxygen decline throughout the water column. Based on reduced catches conducted during fall turnover and a complimentary increase in Seneca River catches, it is likely that many fish leave the lake to avoid the stress of low dissolved oxygen concentrations (Auer et al., 1996a). Species moving out of the lake include channel catfish (*Ictalurus punctatus*), gizzard shad, white perch, and smallmouth bass (*Micropterus salmoides*) (Auer et al., 1996a).

According to Ringler et al. (Auer et al., 1996a), the major stressors that are potentially limiting to fish in Onondaga Lake include the following:

- **Dissolved oxygen** – The absence of dissolved oxygen concentrations in the hypolimnion of the lake during stratification and the unusually low concentrations throughout the water column during fall turnover represent significant constraints on fish in the lake.

- **Absence of macrophytes in parts of the lake** – Macrophytes play a critical role in supporting the fish community of Onondaga Lake, as demonstrated by the absence of juveniles and adults in areas of low macrophyte density.

Ringler et al. (1995) evaluated the nesting activity of fish in the nearshore zone of Onondaga Lake, and found that the highest densities of nests were located on the northwestern shoreline between the lake outlet and the mouth of Ninemile Creek. They noted that this shoreline is composed primarily of oncolytic sediments and is relatively well protected from the predominantly northwest winds. Ringler et al. (1995) stated that conclusions regarding relationships between nest densities and habitat variables cannot be drawn without more quantitative data on distribution and abundance of habitat types. From a qualitative standpoint, however, they noted that the flat, unvegetated habitat off the wastebeds on the western shoreline of the lake provides minimal cover for spawning fish, and relatively few nests were found in that area. Ringler et al. (1995) also noted that the paucity of nests in the entire southern part of the lake may be related to the sparse distribution of macrophytes in certain areas, turbidity from Onondaga Creek, or ammonia mainly from the Metropolitan Syracuse Sewage Treatment Plant (Metro) effluent.

9.1.5 Amphibians and Reptiles

Between April and October of 1994, Ducey and Newman (1995) conducted a herpetological survey along the perimeter of Onondaga Lake. They made 30 visits to the lake and expended approximately 235 person-hours during the surveys. This study is the first and only comprehensive assessment of herpetofauna near the lake.

Ducey and Newman (1995) documented the presence of seven amphibian species (i.e., five frog and two salamander species) and six reptilian species (i.e., three snake and three turtle species) near Onondaga Lake during their 1994 survey. They found that habitats around the lake differed dramatically in the species supported, with the lake itself and many other areas nearly devoid of herpetofauna. The terrestrial areas around the lake were divided into five regions for evaluation. The results were as follows:

- **Region A:** This region is located along the northwest shoreline of Onondaga Lake, and includes Maple Bay, Ninemile Creek, several wooded areas, a large swamp, many temporary wetlands, two ponds (one connected to the lake and the other isolated from the lake), and several fields with grass and shrubs. According to Ducey and Newman (1995), this region supported, by far, the largest numbers of individuals and the greatest number of species. However, even within this region, herpetofaunal populations varied. No herpetofauna were found at the pond with hydrological connections to the lake.

- **Region B:** This region is located along the northeast shoreline of Onondaga Lake, and includes a park area along Willow Bay, a drainage ditch, a seasonally flooded forest, Sawmill Creek, and a dump area for the Parks Department. According to Ducey and Newman (1995), this region supported few amphibians and reptiles, despite extensive searching by the authors in spring, summer, and fall. The authors were particularly surprised by the fact that no frogs were ever heard calling, even during times when they were very active elsewhere. The authors also reported that no tadpoles or frogs were found within the forest or along Sawmill Creek, despite apparently ample water, vegetation, and insects. They also noted that no snakes were found in the dump area, despite the fact that it contained many kinds of suitable debris for habitat. Ducey and Newman (1995) recommended that Region B be reexamined to confirm the absence of herpetofauna in that area.

- **Region C:** This region is located along the southeast shoreline of Onondaga Lake between Onondaga Creek and Bloody Brook, and includes grass fields and a series of ponds (connected to the lake). Ducey and Newman (1995) found only a single turtle in this region, but speculated that the region probably supports one or two species of snakes. The authors concluded that they would not expect to find large numbers of herpetofauna in this region because of the lack of suitable cover.

- **Region D:** This region is located along the southwest shoreline of Onondaga Lake between Onondaga Creek and Tributary 5A, and includes extensive stands of *Phragmites australis* (common reed) along the shoreline, small forested areas, and broad grassy areas. Ducey and Newman (1995) noted that they did not investigate this region as extensively as Regions A, B, and C. They found moderate numbers of snakes, but no frogs or salamanders.

- **Region E:** This region is located along the western shoreline of Onondaga Lake between Tributary 5A and Ninemile Creek, and includes elevated Solvay wastebeds (with cliffs) and parking lots for the fairgrounds in the upper areas. Ducey and Newman (1995) found no herpetofauna in the elevated areas, but a small snake population was found on the lake shoreline. They concluded that the soil layers appeared to be too thin throughout most of this region to support adequate invertebrate prey populations or to provide ample subterranean tunnels for herpetofauna.

Ducey and Newman (1995) compared the results of their 1994 survey around Onondaga Lake with previous results from elsewhere in central New York State. They concluded that the herpetofauna around the lake was generally depauperate, and were surprised by the absence of some common species. They found that the seven amphibian and six reptilian species found around the lake were considerably fewer than the 19 amphibian and 15 reptilian species recorded for Onondaga County as a whole during 1990 to 1996 by NYSDEC (1997b).

Ducey (1997) conducted additional evaluations of the herpetofauna near Onondaga Lake between March 1995 and May 1997. For these studies, the author focused only on Region A along the northwest shoreline of the lake. The results confirmed that the Onondaga Lake littoral zone and shoreline supports no amphibian reproduction, in contrast to other moderately sized lakes in central New York. Five species of anurans and two species of salamanders utilized wetlands (not connected to the lake) and terrestrial habitats within 100 m of the lake. Red-spotted newts (*Notophthalmus viridescens*) were found in one of the unconnected wetlands at a density up to three orders-of-magnitude lower than found at other sites in central New York (Ducey, 1997).

Environmental factors that may be affecting herpetofaunal distribution include:

- High concentrations of ionic waste (chloride, sodium, and calcium ions) that may affect the physiological processes.
- Chemical contaminants.
- Effluent from the Metro sewage treatment plant.
- Poor habitat on the southern shores of the lake (Ducey and Newman, 1995).

Ducey et al. (2000) directly assessed the toxicity of water from the lake and wetlands on developing amphibian embryos. They found that water from connected wetlands and the lake has variable, but consistently negative, effects on amphibian development relative to controls. They hypothesized that there is a chemical interaction that affects amphibian embryos, because unfiltered Onondaga Lake water is highly toxic to embryos, while filtered water is not as toxic.

9.1.6 Terrestrial Plants

Various plant communities are found around Onondaga Lake (i.e., terrestrial, wetland, aquatic, or urban systems). The area around Onondaga Lake has been extensively modified, through development, for more than a century. The vegetation found on the wastebeds (Chapter 3) has been affected by activities at Honeywell facilities (i.e., disposal of Solvay wastes). Table A-1 of Appendix A lists characteristic flora of the ecological communities around Onondaga Lake.

9.1.7 Birds and Mammals

Tables 3-11 to 3-13 provide summaries of bird species observed around Onondaga Lake and Table 3-14 lists mammals potentially found in the area. Since there are no data available that could be used to correlate observed wildlife populations with contamination in the area, contaminant doses (estimated using food-web modeling) compared to TRVs (see Section 9.3) are the main measurement endpoint used to evaluate avian and mammalian assessment endpoints.

9.2 Benthic Macroinvertebrates/Sediment Effect Concentrations

The potential ecological risks posed to benthic macroinvertebrate communities by surface sediment in Onondaga Lake were evaluated using three kinds of information collected by Honeywell in 1992 and 2000 during the RI field investigation:

- Chemical concentrations in surface sediments.

- Sediment toxicity tests using the 10-day (1992) and 42/40-day (2000) tests, based on the amphipod *Hyalella azteca* and the midge *Chironomus tentans*.

- Evaluations of in situ benthic macroinvertebrate communities.

The three indicators used to evaluate surface sediment in Onondaga Lake (i.e., sediment chemical concentrations, sediment toxicity tests, and benthic community evaluations) are used to provide a weight-of-evidence approach regarding the risk of toxicity posed by sediment throughout the lake in Chapter 10. That is, the independent information provided by each of the three indicators will be combined to provide an integrated assessment of sediment quality that would not be possible using any single indicator (USEPA, 1997b). For example, sediment chemical concentrations provide information on which chemicals are elevated above sediment effect concentrations (SECs) or consensus probable effect concentrations (PECs) and may pose an ecological risk, but they do not provide the information to conclusively determine if those chemicals are sufficiently bioavailable to pose a risk. The sediment toxicity tests provide information on whether particular sediments are toxic to the test organisms, but they do not identify the toxic components of the sediment or determine whether the observed toxicity is sufficient to result in adverse effects to the resident benthic macroinvertebrate communities. Finally, alterations of benthic communities provide information on whether sediment may be adversely affecting resident organisms, but they do not identify the causative agents. The use of a weight-of-evidence approach to evaluate potential risks to benthic macroinvertebrates is consistent with USEPA guidance and recommendations (USEPA, 1997a,b).

The 114 stations sampled in Onondaga Lake and the eight tributary stations sampled around the perimeter of the lake for sediment evaluations in 1992 are presented on Figure 7-2 in Chapter 7. Sixty-six lake stations were sampled for sediment chemistry, sediment toxicity, and benthic macroinvertebrate communities. Thirteen lake stations were sampled only for sediment toxicity and sediment chemistry. Thirty-five lake stations were sampled only for sediment chemistry. The eight tributary stations were sampled only for benthic macroinvertebrate communities. The 15 stations sampled in Onondaga Lake for sediment evaluations in 2000 are presented on Figure 7-6 in Chapter 7.

In conducting the sediment evaluations, Otisco Lake was used as the reference lake for determining the significance of sediment toxicity and effects on benthic macroinvertebrate communities at stations in Onondaga Lake. In addition, NYSDEC suggested in May 1999 that stations deeper than 3 m may be depth-impacted (Larson, pers. comm., 1999b). However, in order to provide a greater areal coverage of the lake bottom, the evaluation is limited to the 4.5 m contour for the 1992 benthic data and the 5 m contour for the 2000 benthic data.

In the following sections, results of the sediment evaluations are described, and site-specific SECs and PECs are developed using the empirical information collected on chemical concentrations and sediment toxicity in Onondaga Lake in 1992 and 2000. The site-specific PECs are used in Chapter 10 for risk characterization.

9.2.1 Results of Sediment Toxicity Tests

Results of sediment toxicity tests for each station sampled in Onondaga Lake in 1992 and 2000 and the reference lake (i.e., Otisco Lake) are presented in Tables 9-1 and 9-2, respectively.

Toxicity results from each station in Onondaga Lake were compared statistically with a single reference station in Otisco Lake. Separate evaluations were conducted for the data collected in 1992 and 2000. Paired comparisons between results for each station in Onondaga Lake and the results for Otisco Lake were made using the Washington State Department of Ecology SEDQUAL program (WSDE, 2001).

SEDQUAL performs statistical comparisons among test, reference, and control stations to identify stations exhibiting adverse effects. In SEDQUAL, test data may be compared to either reference data or control data. As stated above, in this BERA, Onondaga Lake stations were compared to single reference stations in Otisco Lake. Records are distinguished as reference/control data or test data by the sample use code. Statistical and data analysis features in SEDQUAL include:

- Wilks-Shapiro test for normality.
- Levene's test for equality of variances.
- Student's t-statistic, approximate t-statistic, Mann-Whitney, and rankits.
- User-specified reference station when a survey has more than one reference.
- Comparison of reference or control data to numeric performance standards.
- Optional use of negative control instead of reference if a survey has no reference stations or reference stations fail to meet a performance standard.

9.2.1.1 1992 Sediment Toxicity Results

Survival for the 10-day amphipod and chironomid toxicity tests was relatively high (i.e., >80 percent) at most stations in Onondaga Lake, indicating that lethal toxicity was not widespread throughout the lake (Figure 9-2). The chironomid test exhibited a greater range of survival than did the amphipod test.

Biomass also exhibited a greater range for the chironomid test than for the amphipod test (Figure 9-2). Amphipod biomass was lower than control values (i.e., <100 percent) at approximately half of the stations (47 percent) in Onondaga Lake, whereas chironomid biomass was less than control values at approximately one-third of the stations.

The greater variation in the responses of the chironomid test compared to the amphipod test may be due to the fact that the chironomids generally live in more direct contact with sediments than do the amphipods. While the amphipods live primarily on the sediment surface, the chironomids burrow into the sediments where they reside in cases (American Society for Testing and Materials [ASTM], 1993).

Values of mean survival for the five stations sampled in Otisco Lake were high for both the amphipod and chironomid tests (90 and 97 percent, respectively). Values of mean biomass for the amphipod and chironomid tests for the five stations sampled in Otisco Lake were similar to or greater than negative control values (i.e., 97 and 182 percent of control values, respectively).

Results of the statistical comparisons of the sediment toxicity data collected in 1992 are presented in Table 9-1. In general, the chironomid test was found to be a more sensitive indicator of sediment toxicity than the amphipod test. For example, statistically significant ($P \leq 0.05$) reductions of amphipod survival and growth were found at 1 and 18 stations, respectively, whereas significant ($P \leq 0.05$) reductions of chironomid survival and growth were found at 35 and 15 stations, respectively. The different sensitivities of the two test species may be related to the life-history patterns described above, where the amphipods live primarily on the sediment surface, while the chironomids burrow more deeply into the sediments.

For the 79 stations sampled in Onondaga Lake in 1992, effects based on amphipod survival, amphipod biomass, chironomid survival, and chironomid biomass were identified at 1, 18, 35, and 15 stations, respectively. If the results of all four toxicity endpoints are combined, they jointly identify effects at 40 stations. Of these 40 stations, the chironomid test independently identifies effects at most (i.e., 36 of 40, or 90 percent) of the stations at which any kind of toxicity was found.

The spatial patterns of amphipod and chironomid toxicity are presented in Figure 9-3. Most amphipod toxicity was confined to a small area in the southwestern corner of the lake, along Wastebeds 1 through 8 and along the Honeywell lakeshore area near Harbor Brook and the East Flume.

Most chironomid toxicity was confined to the southern half of the lake (Figure 9-3), although toxicity was also found in two areas in the northern half of the lake (i.e., off Ninemile Creek and near Sawmill Creek). In the southern half of the lake, lethal chironomid toxicity was found in three general areas: off Tributary 5A, off Ley Creek, and in the southwestern corner of the lake (off Harbor Brook, the Metro outfall, and the East Flume).

9.2.1.2 2000 Sediment Toxicity Results

The results of the statistical comparisons of the 42-day sediment toxicity data collected in 2000 are presented in Table 9-2. The spatial patterns of amphipod and chironomid toxicity are presented in Figure 9-4. In general, the patterns of toxicity observed for both tests were similar.

For the amphipod test, lethal toxicity was found at six stations (Figure 9-4), including all of the shallow (i.e., <5 m water depth) nearshore stations from Tributary 5A to the East Flume (Stations S332, S337, S342, S344, and S365) and near the Metro outfall (Station S317). Mean survival at those six stations ranged from 9 to 59 percent, compared to a mean value of 88 percent found for Otisco Lake. Amphipod biomass was found to be impacted at Stations S317 and S323, while reproduction was affected at three locations, Stations S342, S344, and S365.

For the chironomid test, lethal toxicity was found at nine stations (Figure 9-4), including all five of the shallow (i.e., <5 m water depth) nearshore stations from Tributary 5A to the East Flume (i.e., Stations S332, S337, S342, S344, and S365), two stations off Ninemile Creek (Stations S302 and S303), as well as the stations off Ley Creek (Stations S320 and S323). Mean survival at those nine stations ranged from 0 to 46 percent, compared to a mean value of 83 percent found for Otisco Lake. In addition to the nine stations at which lethal toxicity was found for the chironomid test, sublethal toxicity was found at Station S317 off Onondaga Creek and at Station S372 along the northeastern shoreline of the lake. The sublethal toxicity at these stations was seen in reduced biomass (0.26 and 0.41 mg per individual, respectively) relative to Otisco Lake (0.73 mg per individual). Chironomid emergence was affected at five locations: Stations S332, S337, S342, S344, and S354.

9.2.1.3 Comparison of 1992 and 2000 Results

Overall, the results of the 2000 42-day chronic (long term) sediment toxicity tests (based on the top 15 cm of the sediment column) and the results of the 1992 10-day acute (short term) tests (based on the top 2 cm of the sediment column) confirmed that both sub-lethal (impaired growth and reproduction) and lethal impacts (survival) are occurring in the sediments of Onondaga Lake. That is, most sediment toxicity in Onondaga Lake is confined to the nearshore zone in the southern part of the lake between Tributary 5A and Ley Creek. By contrast, little toxicity is observed elsewhere in the lake, including the deeper parts of the entire lake and the lake's eastern shore.

9.2.2 Results of Benthic Macroinvertebrate Community Evaluations

In the following sections, benthic macroinvertebrate communities in Onondaga and Otisco Lakes are compared with respect to general lakewide characteristics and station-specific characteristics.

9.2.2.1 Lakewide Comparisons of Benthic Communities in Onondaga Lake

The major characteristics of the benthic macroinvertebrate communities found in 1992 at various water depths in Onondaga and Otisco Lakes are presented in Figure 9-5. Communities were assessed in keeping with NYSDEC's evaluation (Larson, pers. comm., 1999b), because depth can substantially influence the characteristics of benthic communities (Resh and McElravy, 1993). In most cases, taxa richness and abundances of major taxa tended to decline with increasing depth in both lakes, underlining the importance of stratifying benthic communities by depth.

In Figure 9-6, the oligochaete/chironomid abundance ratios for benthic communities at various depths in Onondaga and Otisco Lakes are compared. This ratio tends to increase with increasing depth in eutrophic lakes, as more stress-tolerant oligochaetes replace chironomids at deeper depths (Wiederholm, 1980). The oligochaete/chironomid abundance ratios in Onondaga Lake exhibited the same increasing trend with increasing depth. By contrast, the oligochaete/chironomid abundance ratios in Otisco Lake showed the opposite trend (i.e., decreasing ratios with increasing depth), indicating that conditions in the deeper parts of that lake were not as stressful to benthic organisms as conditions in the deeper parts of Onondaga Lake.

9.2.2.2 Station-Specific Comparisons of Benthic Communities in Onondaga Lake

Benthic macroinvertebrate communities at stations in Onondaga Lake were compared with the communities found in Otisco Lake using several methods. Separate evaluations were conducted for the data collected in 1992 and 2000. Evaluation methods used were based on NYSDEC's methodology (NYSDEC, 1994a, 2002c) and recommendations from other benthic ecological peer review panels (PTI, 1993c; WSDE, 1996). NYSDEC (Larson, pers. comm., 1999c) and PTI (1993c), recommended that if major taxa abundance is used in the analysis additional endpoints should be measured to increase overall sensitivity. They also recommended that more than one benthic endpoint should be used to assess adverse benthic effects. Species richness and total abundance should be considered for inclusion with major taxa abundance as primary benthic endpoints.

For direct statistical comparisons univariate statistical tests (i.e., Student's t-statistic) were performed first to compare the study area and reference conditions. Subsequent analysis was more exploratory to determine the cause of the observed patterns and used multivariate techniques. Classification analysis is a multivariate technique recommended for evaluating benthic macroinvertebrate communities in the Great Lakes by the International Joint Commission (IJC, 1988). The key attributes of the approach are that it provides an integrative evaluation of all benthic taxa and has the power to detect relatively subtle patterns (IJC, 1988).

The primary method used to evaluate benthic communities was based on comparing various benthic metrics between each test station in Onondaga Lake and the appropriate reference stations in Otisco Lake. The metrics were those recommended by NYSDEC (1994a and Larson, pers. comm., 1999b) and included the following:

- **Taxa Richness:** The total number of individual taxa in a sample. The term taxa instead of species is used, as the organisms in this study are not always identified to the species level.

- **Dominance Index:** The percent composition of the three most abundant taxa.

- **Abundance of Indicator Species:** The number of non-chironomidae/ oligochaete (NCO) taxa.

- **Species Diversity:** A measure of the distribution of individuals among the taxa observed.

- **Percent Model Affinity (PMA):** A measure of similarity to a model non-impacted lake community based on percent abundance in six major groupings:

  - 20 percent Oligochaeta.
  - 15 percent Mollusca.
  - 15 percent Crustacea.
  - 20 percent non-chironomid Insecta.
  - 20 percent Chironomidae.
  - 10 percent other.

For the 1992 and 2000 benthic data, the first four metrics values for each test station were compared with the values found at the reference station using the t-test in the manner described earlier for the sediment toxicity results. For PMA, the results for each test station were compared using the model described by NYSDEC (Larson, pers. comm., 1999c).

In addition, for the 2000 benthic data, the benthic metrics were compared statistically between stations in Onondaga Lake and Otisco Lake, even though the benthic community at the shallow station in Otisco Lake was found to be dominated by a nonnative invasive species (i.e., the zebra mussel). Because few organisms were found at most of the deeper stations in Onondaga Lake, due to low levels of dissolved oxygen below the thermocline, statistical comparisons were not conducted for depths greater than 5 m.

Metrics Analysis – 1992

Five macroinvertebrate replicate samples were collected at each station, and the benthic metrics presented herein were computed by replicate. The average values of the five replicates for each metric were determined and presented for each station. The following is a discussion of how each metric was computed and how they are used to assess the health of benthic invertebrate communities.

- **Taxa Richness:** Taxa richness was determined by counting the different number of taxa per replicate (e.g., if 5 taxa are observed in a replicate, the species richness is 5). The average of the five replicates was then computed. Data from each replicate were also pooled in order to determine the total number of taxa observed at each station. The total number of taxa collected at each station was used to conduct an impairment assessment.

  If the total taxa richness at a station exceeded 32, a station was considered to be non-impaired. Total taxa richness between 25 and 32 indicated slight impairment, between 14 and 24 indicated moderate impairment, and values between 0 and 13 signified that a station was considered to be severely impaired. Table 9-3 provides the benthic analysis assessment criteria ranges.

- **Dominance Index**: The dominance index was computed by determining the total percent composition of the three most abundant species. This was performed by first determining the three taxa with the highest individual abundance in a replicate. The percent composition of these three taxa was determined by dividing the abundance (the total number of individuals of these taxa) by the total number of all individuals in the replicate.

  If the dominance index at a station was less than 61, a station was considered to be non-impaired. Dominance indices between 61 and 75 indicated slight impairment, between 76 and 90 indicated moderate impairment, and values between 91 and 100 signified that a station was considered to be severely impaired. Table 9-3 provides the benthic analysis assessment criteria ranges.

- **Abundance of Indicator Species**: The abundance of indicator species was determined by enumerating the number of taxa within each replicate that are neither Oligochaeta or Chironomidae (NCO). In general, species of oligochaetes and chironomids are tolerant of pollution stress and therefore, are not species indicative of a healthy ecosystem. As described above, data from each replicate were also pooled in order to determine the total number of indicator species observed at each station. The total number of indicator species at each station was used to conduct an impairment assessment.

  If the abundance of indicator species or NCO at a station was greater than 15, a station was considered to be non-impaired. Total NCOs between 10 and 15 indicated slight impairment, between 5 and 9 indicated moderate impairment, and values between 0 and 4 signified that a station was considered to be severely impaired based on NYSDEC guidance (Larson, pers. comm., 1999b). Table 9-3 provides the benthic analysis assessment criteria ranges.

- **Species Diversity**: Species diversity was determined using the Shannon-Wiener function as follows (Krebs, 1977):

$$H = -[(p_1)(\log_2 p_1) + (p_2)(\log_2 p_2) + \ldots]$$

  where:

  H = index of species diversity; and
  p_i = the proportion of the sample belonging to the i^{th} taxa.

  If the Shannon-Wiener diversity index at a station was greater than 3.1, a station was considered to be non-impaired. Diversity indices between 2.1 and 3 indicated slight impairment, between 1.5 and 2 indicated moderate impairment, and values less than 1.5 signified that a station was considered to be severely impaired.

- **Percent Model Affinity**: Community composition was determined using the general method described by Bode et al. (1991), including the use of a model that has been determined by the NYSDEC (Larson, pers. comm., 1999c) to be suitable for a non-impacted lake, as described above.

  The percent contribution for each of the six major groups in a replicate was determined (adding up to 100 percent). For each group in the replicate, the *absolute* difference in percentage from the model value for that group was determined. The differences for each group were added per replicate. Per Bode et al. (1991), the total of the difference was multiplied by 0.5, and this number was subtracted from 100 to determine the PMA.

If the community composition or PMA at a station was greater than 64, a station was considered to be non-impaired. Model affinity between 50 and 64 indicated slight impairment, between 35 and 49 indicated moderate impairment, and values less than 35 signified that a station was considered to be severely impaired. Table 9-3 provides the benthic analysis assessment criteria ranges.

Overall Assessment

The values for each of the metrics for all stations are presented in Table 9-4, including 48 Onondaga Lake (S) stations, eight tributary (T) stations, and three Otisco Lake (OT) stations. Stations were divided into the following four categories based on the results of the five metrics:

- **Non-impaired** – The macroinvertebrate community is diverse. This level of water quality includes both pristine habitats and those receiving discharges that minimally alter the biota.

- **Slightly impaired** – The macroinvertebrate community is slightly, but not significantly altered from the pristine state.

- **Moderately impaired** – The macroinvertebrate community is altered to a large degree from the pristine state.

- **Severely impaired** – The macroinvertebrate community is limited to a few tolerant species, usually midges or worms. Often only one or two species are very abundant.

Table 9-5 presents the results of the impairment assessment for each station. The cumulative review of the five metrics (referred to as a multimetric approach) was used to coalesce the metrics into a single overall assessment of each station. The assessment determination for each station was made on the basis that three or more of the five metrics exhibited the same impairment category (of either non-impaired, slightly impaired, moderately impaired, or severely impaired). When less than three metrics exhibited the same impairment category, the results of all five metrics and professional judgment were used to characterize the station.

Following is a breakdown of Onondaga and Otisco Lake stations by impairment category. Tributary results are discussed in Section 9.2.2.3.

- **Non-impaired (n = 0)**: None

- **Slightly impaired (n = 12)**: Stations S26, S48, S53, S54, S67, S73, S76, S87, S100, S105, S110, and OT3.

- **Moderately impaired (n = 31)**: Stations S2, S11, S13, S14, S17, S18, S21, S34, S35, S37, S45, S46, S47, S61, S62, S72, S74, S75, S77, S82, S83, S92, S93, S94, S95, S104, S109, S111, S112, OT1, and OT2.

- **Severely impaired (n = 8)**: Stations S5, S7, S8, S22, S28, S29, S38, and S68.

Of the 51 (48 Onondaga Lake and 3 Otisco Lake) stations considered for further evaluation, none were found to be non-impaired, 12 stations were found to be slightly impaired, 31 stations were found to be moderately impaired, and 8 stations were found to be severely impaired.

Severely impaired stations are primarily located at the southern end of the lake (i.e., between Metro and Tributary 5A). One station (Station S68) considered to be severely impaired is located near Wastebeds 1 to 8. Moderately impaired stations are found throughout the lake.

Statistical and Classification Analysis (1992)

The results of the statistical analysis of four of the five benthic metrics requested by NYSDEC are summarized in Table 9-6 and Figures 9-7, 9-8, 9-9, 9-11, 9-12, 9-13, and 9-14. The results of the metrics analysis show that NCO richness and diversity were the most sensitive metrics, since they identified 48 and 28 stations as being impacted, respectively. By contrast, dominance was much less sensitive, identifying only 8 stations as being impacted. Taxa richness was intermediate in sensitivity, identifying 25 stations as impacted.

The patterns described above indicate that much of the littoral zone less than 4.5 m deep in Onondaga Lake is impacted. These patterns of impacted stations are correlated when one examines them in conjunction with the 1992 and 2000 sediment toxicity test results. As stated above, these tests indicate sub-lethal and lethal effects in nearshore sediments. The most useful and discriminating benthic metric appears to be taxa richness, which showed no depth-related bias and produced patterns similar to those based on sediment chemistry and sediment toxicity.

Using classification analysis and further segmenting the 4.5 m interval into 1.5 m and 4.5 m for easier review of the results, three benthic groups were identified in the 1.5 m depth stratum (Figure 9-13) and two benthic groups were identified in the 4.5-m depth stratum sampled in 1992 (Figure 9-14). For the 1.5 m depth stratum, based on similarity to stations from Otisco Lake, benthic communities from stations in Group A could be considered slightly altered; communities from Group B were considered moderately altered; and communities from Group C were considered to exhibit major alterations. For the 4.5 m depth, based on similarity to stations from Otisco Lake, benthic communities from stations in Group A were considered slightly altered, and communities from Group B were considered to exhibit major alterations.

The various benthic groups identified by the classification analysis are compared with respect to major community variables in Figure 9-15. In most cases, mean taxa richness and mean abundances of major benthic taxa for the minimally altered group were considerably greater than the respective mean values for the group exhibiting major alterations. In addition, values of taxa richness and amphipod abundance for the moderately altered group at 1.5 m depth were intermediate in magnitude. These patterns indicate that the major characteristics of benthic communities corresponded to the results of the classification analysis.

The spatial distribution of the various kinds of benthic effects described above is presented in Figure 9-16. Major alterations of benthic communities were found in two nearshore areas: off Tributary 5A and in the southwestern corner of the lake (off Harbor Brook, the Metro outfall, and the East Flume). Moderate alterations of benthic communities were found at most of the remaining nearshore stations in the southern part of the lake between Tributary 5A and Ley Creek.

Metrics Analysis – 2000

The five benthic metrics are summarized in Table 9-7 for the August 2000 data, including nine Onondaga Lake (S) stations and one Otisco Lake (OT) station. Onondaga Lake and Otisco Lake stations that were located in water depths greater than 5 m were excluded from the assessment (although NYSDEC's May 27, 1999 letter indicates stations deeper than 3 m were potentially depth-impacted, this report includes the stations at the 5 m depth to provide greater spatial coverage of the lake; thus, six of the 15 Onondaga Lake stations and one of the two Otisco Lake stations sampled in 2000 were excluded [Larson, 1999b, pers. comm.]). Analysis of the 1992 data on benthic communities excluded all stations below 4.5 m. However, no reference station (i.e., Stations OT-6 or OT-7) was shallower than 4.5 m in the 2000 sampling event. The Otisco Lake stations were at depths of 5 and 9 m, respectively. As such, the limit of exclusion was increased to 5 m in order to permit the use of a reference station for analysis.

The information presented below is a brief summary of the data analyses, as per the five matrices specified by NYSDEC. The reference value(s) are from Otisco Lake's Station OT-6.

- **Taxa Richness:** Values for all but one station (Station S372) in Onondaga Lake were lower than the reference value of 19, with values ranging from 8 to 16.

- **Dominance:** Values at seven of the nine Onondaga Lake stations were greater than the reference value of 77 percent. Dominance values ranged from 63 to 91 percent.

- **NCO Taxa:** Values for all Onondaga Lake stations were considerably lower than the reference value of 9, with values ranging from <1 to 4.

- **Community Composition (PMA):** Values at all of the Onondaga Lake stations were less than the reference value of 60, ranging from 22 to 54.

- **Species Diversity:** Values at seven of the nine Onondaga Lake stations were lower than the reference value of 2.4. Diversity at Onondaga Lake Stations S365 (2.5) and S372 (3.1) were higher than the reference value.

Table 9-8 presents the results of the impairment assessment for each station. As for the 1992 data, the cumulative review of the five metrics (referred to as a multimetric approach) was used to coalesce the metrics into a single overall assessment of each station. The assessment determination for each station was made on the basis that three or more of the five metrics exhibited the same impairment category (of either non-impaired, slightly impaired, moderately impaired, or severely impaired). When less than three metrics exhibited the same impairment category, the results of all five metrics and professional judgement were used to characterize the station. Following is a breakdown of stations by impairment category:

- **Non-impaired (n = 0):** None.

- **Slightly impaired (n = 3):** Stations S365, S372, and OT-6.

- **Moderately impaired (n = 6):** Stations S305, S323, S332, S337, S342, and S344.

- **Severely impaired (n = 1):** Stations S317.

Of the 10 (9 Onondaga Lake and 1 Otisco Lake) stations considered for further evaluation, none were found to be non-impaired; 3 stations were found to be slightly impaired; 6 stations were found to be moderately impaired; and 1 station was found to be severely impaired.

The severely impaired station (Station S317) is located in the southern end of the lake between the Metro outfall and the mouth of Onondaga Creek. Moderately impaired stations are found throughout the lake, clustered between Tributary 5A and Harbor Brook, and near the mouths of Ninemile Creek and Ley Creek.

Two of the three slightly impaired stations are located in Onondaga Lake: one (Station S365) is north of the mouth of Tributary 5A, and the other (Station S372) is in the northwestern portion of the lake. The one reference station found to be slightly impaired (Station OT-6) is located in Otisco Lake. This reference station (OT-6) differed considerably from seven of the nine Onondaga Lake stations, as it possessed a disproportionately high number of zebra mussels (*Dreissena polymorpha*). The zebra mussel comprised 28 percent of the total individuals for Station OT-6. In Onondaga Lake, the zebra mussel was rarely observed in 1992, but was found in abundance at two stations in 2000. At Stations S365 and S372, the zebra mussel comprised 25 and 7.5 percent of the benthic populations, respectively (based on the average of five replicates).

However, despite the presence of zebra mussels, an important difference between the benthic assemblages at Otisco and Onondaga Lakes are the numbers and percentages of NCO taxa to chironomidae and oligochaete taxa. At Station OT-6, approximately half of the total number of taxa (15 of 33 taxa), and half the taxa richness (9 of 19 taxa), were comprised of NCO taxa. Comparatively, in Onondaga Lake, the NCO to chironomidae and oligochaete ratio was much lower for eight of the nine stations. In fact, at seven of the eight Onondaga Lake stations, the taxa were comprised of 25 percent or less NCO taxa. This is important to acknowledge because, as stated earlier, chironomidae and oligochaete taxa are generally considered more pollution-tolerant than NCO taxa.

Stations S365 and S372 had the two highest diversity readings, at 2.5 and 3.1, respectively, and the two highest total number of taxa for Onondaga Lake. However, these stations also had high NCO taxa to chironomidae and oligochaete ratios for total taxa and taxa richness. The ratios of NCO taxa to chironomidae and oligochaete taxa for these two stations are the following:

- **Total Taxa:** Stations S365 (4 NCO of 26 total taxa) and S372 (6 NCO of 36 total taxa).

- **Taxa Richness:** Stations S365 (3 NCO of 14 total taxa) and S372 (4 NCO of 22 total taxa).

Thus, the high diversity of these two stations may not be indicative of a healthy environment, as a large majority of these taxa are pollution-tolerant.

Station S342 was the only Onondaga Lake station to exhibit somewhat similar NCO to chironomidae and oligochaete ratios as the reference station OT-6.

In the less than 5 m depth stratum in Onondaga Lake in 2000, the following general patterns were found:

- **Taxa Richness:** Values for all but one station in Onondaga Lake were lower than the reference value of 33 total taxa. Taxa richness at Station S372 (36 total taxa) off the northeastern shoreline of the lake exceeded the reference value.

- **NCO Taxa:** Values for all Onondaga Lake stations were considerably lower than the reference value of 15 total taxa, with 8 of the 9 stations having a value of less than half the reference value.

- **Species Diversity:** Values at most stations in Onondaga Lake were lower than the value of 2.4 for Otisco Lake. Diversity at Stations S365 (2.5) and S372 (3.7) were higher than the reference value.

- **Dominance Index:** Values at most stations in Onondaga Lake were greater than the reference value of 77 percent, most likely reflecting the dominance of oligochaetes and chironomids at most stations in Onondaga Lake.

- **Percent Model Affinity:** Values at most stations in Onondaga Lake were less than 30 percent with the zebra mussel included in the analysis with Otisco Lake having the highest PMA of 60 percent.

Statistical and Classification Analysis (2000)

The statistical analysis results for four of the five benthic metrics requested by NYSDEC are summarized in Table 9-9. Results of the metrics analysis show that NCO richness and species richness (total taxa) were the most sensitive metrics, since they identified nine and eight stations as being impacted, respectively. By contrast, dominance was much less sensitive, identifying no station as being impacted. Species diversity was intermediate in sensitivity, identifying five stations as impacted.

The patterns described above indicate that much of the littoral zone less than 4.5 m deep in Onondaga Lake is impacted. These effects are corroborated when one examines them in conjunction with the 1992 and 2000 sediment toxicity test results and 1992 benthic metrics. As stated above, these tests indicate sub-lethal and lethal effects in nearshore sediments.

For the nine shallow stations (i.e., 1.5 to 5 m) sampled in Onondaga Lake in 2000, one group of closely related stations was identified (Figure 9-17), consisting of the six shallow stations that extended from south of Tributary 5A to Ley Creek (i.e., Stations S317, S323, S332, S337, S342, and S344). The remaining shallow stations in Onondaga Lake did not cluster closely with other lake stations, most likely because they were from different parts of the lake: north of Tributary 5A (Station S365), off Ninemile Creek (Station S305), and off the northeastern shoreline (Station S372).

The reference station from Otisco Lake (Station OT6) showed little similarity to the stations from Onondaga Lake. Inspection of the taxonomic composition of the reference station showed that the benthic community at that station was dominated by the zebra mussel, which was not found in Otisco Lake during the RI sampling in 1992. That species accounted for nearly half (i.e., 45 percent) of the total number of organisms found at Station OT6 in 2000, at a density of 23,000 individuals/m^2.

In Onondaga Lake, the zebra mussel was rarely observed in 1992, but was found in abundance at two stations in 2000. At Station S365 (north of Tributary 5A) the mussel comprised 31 percent of the benthic community, at a density of 7,600 individuals/m^2. At Station S372, off the northeastern shoreline, the mussel comprised 12 percent of the benthos, at a density of 5,400 individuals/m^2. The large abundances of zebra mussels at those two stations was likely one reason that the stations were not similar to other stations in the lake, based on the results of the classification analysis.

9.2.2.3 Comparisons of Benthic Communities in Tributaries of Onondaga Lake

In 1992, the mouths of the eight tributaries were sampled for evaluation of their benthic community structure (see Chapter 7, Figure 7-2). Table 9-10 presents the results of the impairment assessment for each tributary station. The cumulative review of the five metrics was used to coalesce the metrics into a single overall assessment of each station, as was done for the lake. Following is a breakdown of stations by impairment category:

- **Non-impaired (n = 2)**: Stations T11 (Bloody Brook) and T15 (Sawmill Creek).
- **Slightly impaired (n = 0)**: None.
- **Moderately impaired (n = 2):** Stations T13 (Ninemile Creek) and T7 (East Flume).
- **Severely impaired (n = 4):** Stations T1 (Harbor Brook), T3 (Onondaga Creek), T5 (Ley Creek), and T9 (Tributary 5A).

The benthic macroinvertebrate communities near the mouths of the tributaries to Onondaga Lake were compared using the same methods of classification analysis described above for benthic communities in the lake. The analysis identified three groups of tributaries based on abundances of benthic macroinvertebrates (Figure 9-18), as follows:

- Group A included the four largest tributaries (Harbor Brook, Onondaga Creek, Ley Creek, and Ninemile Creek).
- Group B included the two small tributaries on the western shoreline of the lake (the East Flume and Tributary 5A).
- Group C included the two small tributaries on the eastern shoreline of the lake (Bloody Brook and Sawmill Creek).

The major characteristics of the benthic macroinvertebrate communities near the mouths of the eight tributaries are presented in Figure 9-1. The following major differences were found among the three groups of tributaries identified in the classification analysis:

- **Taxa richness** – This variable was highest in Group C, intermediate in magnitude in Group A, and generally lowest in Group B.
- **Oligochaetes** – The highest oligochaete densities were found in Group A and Tributary 5A, whereas densities in most of the other tributaries were uniformly lower.

- **Chironomids** – The highest chironomid densities were found in Group C, the East Flume, and Ninemile Creek, whereas densities in the other tributaries were highly variable, but generally much lower.

- **Amphipods** – The highest amphipod densities were found in Group C and Ninemile Creek, whereas densities in the other tributaries were uniformly lower and no amphipods were found in the East Flume and Tributary 5A.

These results indicate that the major characteristics of benthic communities were generally similar within the three groups of tributaries identified by classification analysis and generally different among the three groups. Although tributary size and shoreline location may have been partly responsible for the patterns identified by the classification analysis, the tributary groupings may also have been influenced by stressors such as chemical toxicity and organic enrichment.

The high taxa richness in Group C suggests that communities in those tributaries (i.e., Bloody Brook and Sawmill Creek) are minimally altered. High taxa richness indicates that many less-abundant species inhabit those tributaries. In many cases, the less-abundant benthic species tend to be more sensitive to stressors than the more-abundant species.

In contrast with the patterns described above for benthic communities in Group C, the relatively low taxa richness in Group B suggests that communities in those tributaries (i.e., the East Flume and Tributary 5A) are altered to a much greater degree. However, the moderate densities of chironomids found in the East Flume suggest that communities in that tributary are less altered than communities in Tributary 5A.

The intermediate values of taxa richness found for Group A suggest that communities in those tributaries (i.e., Harbor Brook, Onondaga Creek, Ley Creek, and Ninemile Creek) are moderately altered. In addition, the high densities of oligochaetes in Group A suggest that communities in those tributaries may be substantially affected by chemical contamination, ionic waste, or organic enrichment. However, the relatively high densities of chironomids and amphipods, as well as the relatively low densities of oligochaetes in Ninemile Creek, indicate that communities in that tributary are much less altered than communities in the other tributaries from Group A.

However, if the results of the impairment assessment are examined, a slightly different understanding emerges when all benthic metrics are considered, as required by NYSDEC and suggested by several peer review panels. The Group C tributaries, Sawmill Creek and Bloody Brook, are non-impaired, while the majority of Group A tributaries (with the exception of Ninemile Creek but including Tributary 5A) are severely impaired, and Group B (without Tributary 5A but with Ninemile Creek) are moderately impaired. However, when the densities of chironomids, amphipods, and oligochaetes are examined (Figure 9-19), it can be seen that Tributary 5A resembles Harbor Brook, Ley Creek, and Onondaga Creek, while Ninemile Creek may be more altered than communities in the other Group A tributaries and more in line with the East Flume benthic communities.

However, NYSDEC (Larson, 1999b, pers. comm.) stated that, based on NYSDEC kick sampling of the tributaries to Onondaga Lake in 1989 and from 1994 to 1996, the data indicate that Bloody Brook and Sawmill Creek are at least moderately impaired. This raises concerns about using a lake model for the tributary mouths. More appropriate assessments are obtained based on the kick-sampling results (Larson, 1999b, pers. comm.), which indicate that Harbor Brook, Ley Creek, Bloody Brook, Ninemile Creek, and Sawmill Creek are moderately impacted and that Onondaga Creek, the East Flume, and Tributary 5A are severely impacted.

9.2.3 Comparison of Results of Sediment Toxicity Test and Benthic Macroinvertebrate Community Evaluations

In this section, the 1992 results of the sediment toxicity tests and benthic macroinvertebrate community evaluations for Onondaga Lake are compared to determine the extent to which they agree. Close agreement between these different kinds of indicators enhances confidence that the observed patterns of adverse biological effects are real and that they are likely the result of chemical toxicity.

9.2.3.1 Comparisons Based on Benthic Groups

The sediment toxicity results for stations in the various benthic groups identified by classification analysis are compared in Figures 9-20 and 9-21. In general, the toxicity results(mean survival and mean biomass) were closely related to the benthic groups in Figures 9-13 and 9-14, with mean survival and mean biomass for both amphipods and chironomids generally declining from groups based on minimally altered benthic communities (based on benthic metrics) to groups based on communities exhibiting major alterations (based on benthic metrics).

9.2.3.2 Comparisons Based on Adverse Effects

Comparisons of results of the 1992 effects designations (i.e., the presence or absence of adverse biological effects) based on the sediment toxicity tests and the benthic macroinvertebrate community evaluations are presented in Figure 9-22. The percentages of stations identified as having adverse effects varied for the five biological indicators as described below:

- The lowest values were found for amphipod survival and chironomid biomass (0.8 and 13 percent, respectively).

- The highest values were found for benthic community alterations and chironomid survival (43 and 29 percent, respectively).

These results indicate that chironomid survival and benthic community alterations were the most sensitive indicators of sediment toxicity. Chironomid survival would be expected to be a sensitive indicator because it is a response of an organism that burrows into the sediment. Benthic community alterations would also be expected to be a sensitive indicator because it incorporates chronic exposure and sublethal effects on resident organisms, as well as acute exposure and lethal effects on those organisms.

Most stations at which lethal sediment toxicity and major benthic community alterations were found are located in the nearshore zone between Tributary 5A and the Metro outfall (Figure 9-22). Although sublethal toxicity was found throughout most of the southern half of the lake (including areas far from shore), most of the widespread sublethal toxicity was based only on the chironomid test and may be due to factors such as substrate type.

Agreement between the effects designations based on sediment toxicity tests and benthic community alterations was very high. Of the 48 stations at which both kinds of indicators were evaluated, the two kinds of indicators agreed on effects designations in 28 cases (58 percent) and disagreed in 20 cases (37 percent). This level of agreement was significant ($P \leq 0.01$, binomial test) compared to an assumed level of random agreement of 50 percent.

Based on the 2000 toxicity test results of the nine stations that were evaluated for both toxicity and benthic metrics, eight stations showed toxic and benthic community impairment with six of eight stations being moderately to significantly impaired.

9.2.4 Development of Site-Specific Sediment Effect Concentrations and Consensus Probable Effect Concentrations

Sediment effect concentrations (SECs) and consensus probable effect concentrations (PECs) were derived (using the sediment chemistry and toxicity data collected in 1992 and 2000) to allow site-specific assessment of whether the sediment chemical concentrations found at various stations in the lake were potentially related to adverse biological effects.

The SECs and PECs were developed primarily using the 1992 toxicity test data because that data set contained a large number of stations (i.e., 79) that were distributed across broad ranges of sediment chemical of concern (COC) concentrations throughout the entire lake. Development of SECs using the smaller 2000 data set was performed to evaluate whether the chronic toxicity endpoints would provide a different outcome than the short-term tests performed in 1992. Onondaga Lake SECs and PECs were developed for all thirteen metals and 17 organic contaminant compounds identified as COCs in Chapter 6. In addition, total polychlorinated biphenyls (PCBs) and total PAHs were broken out into specific Aroclor components and individual polycyclic aromatic hydrocarbon (PAH) compounds to provide additional detail for the risk characterization in Chapter 10.

The information on benthic macroinvertebrate communities collected during 1992 was not used to develop the SECs or PECs, because it was found that benthic communities at every station in the lake are impaired to some degree; thus, it is not possible to calculate an SEC because these calculations require that a certain proportion of stations have no effects. The results of the benthic macroinvertebrate evaluations for Onondaga Lake were, therefore, used primarily to interpret the magnitude and significance of any potential sediment toxicity predicted on the basis of the SECs. Site-specific SECs were developed for Onondaga Lake using the apparent effects threshold (AET) approach, as well as calculation of effects range-low (ER-L), effects range-median (ER-M), probable effect levels (PEL), and threshold effects level (TEL) values. Consensus-based PECs for COCs in Onondaga Lake were developed following the methodology described in MacDonald et al. (2000) and Ingersoll et al. (2000) as the geometric mean of the site-specific SECs. These sediment guidelines, coupled with other site-specific information on potential risks to ecological receptors, may be used as one tool in the derivation of site-specific sediment cleanup criteria in the FS. USEPA (1997b) recently used AETs (in conjunction with other kinds of SECs) to evaluate the potential toxicity of sediments from over 21,000 stations throughout the US as part of the National Sediment Quality Survey. The AET approach has also been used by the Washington State Department of Ecology (WSDE) to develop promulgated state sediment standards for managing contaminated sediment in Puget Sound, Washington (WSDE, 1995).

Based on recent reviews of the method and development of the proposed Freshwater Sediment Guidelines by WSDE (1997), Ingersoll et al. (1996, 2000) indicated that SEC values based on dry weight organic chemical concentrations either outperform or are not significantly different than organic carbon-normalized data in sensitivity (i.e., false negatives) and efficiency (i.e., false positives). In this BERA, all SECs are developed based on sediment dry weight contaminant concentrations for both organic and inorganic contaminants.

9.2.4.1 Development of Apparent Effect Threshold Effect Levels

The AET for a given chemical is the sediment concentration above which a particular adverse biological effect (e.g., increased mortality or decreased biomass) is always statistically significant ($P \leq 0.05$) relative to appropriate reference conditions (WSDE, 1997). The objective of the AET approach is to identify concentrations of contaminants that are associated exclusively with sediments exhibiting statistically significant biological effects relative to reference sediments.

A detailed description of AET methodology is found in Michelsen and Shaw (1996). AETs can be developed for any kind of biological indicator that has corresponding information on sediment chemical concentrations.

In order to conduct an appropriate analysis, sites were grouped, or "matched," according to a sediment characteristic (e.g., grain size, total organic carbon [TOC], or water depth). Sediment type was utilized to assign the appropriate reference station to the Onondaga Lake stations. Station OT3 is primarily made up of sand, and the next least-impacted site, Station OT4, is primarily made up of fines. Michelsen and Shaw (1996) offers guidance for samples collected from multiple reference stations. Pair-wise comparisons are necessary because each site station needs to be handled separately; therefore, multiple comparison tests that compare the distribution of the data for all locations are not appropriate (Michelsen and Shaw, 1996).

For each chemical, AETs were developed for all four measures of sediment toxicity from the 1992 toxicity data (amphipod survival and biomass and chironomid survival and biomass) and six measures of toxicity from the 2000 data set evaluated during the RI (i.e., amphipod survival, biomass, and reproduction and chironomid survival, biomass, and emergence) in Tables 9-11 and 9-12. The 2000 AETs in Table 9-12 are provided for comparison purposes only and are not used to derive site-specific SECs.

The final AET for each COC was defined as the lowest of all four AETs (amphipod survival and growth and chironomid survival and growth) derived from the 1992 toxicity data set. AETs could not be determined for four organic compounds (i.e., pyrene, indeno(1,2,3-cd)pyrene, chrysene, and benz[a]anthracene) because the concentrations of those COCs were not found over a sufficiently large range.

9.2.4.2 Development of Other Site-Specific Sediment Effect Concentrations

Two commonly used approaches to developing SECs, other than the AET approach, were evaluated for site-specific application to Onondaga Lake. These approaches are currently used by the National Oceanic and Atmospheric Administration's (NOAA's) National Status and Trends Program to evaluate sediments nationwide; by Environment Canada (CCME, 1995); and by the State of Florida (MacDonald, 1994) to derive sediment quality guidelines.

One approach was developed by Long and Morgan (1990) and calculates two kinds of SECs for each chemical, the ER-L and ER-M. The second approach was developed by MacDonald et al. (1996) and also calculates two kinds of SECs for each chemical, the TEL and PEL. For both approaches, the two SECs represent a lower level (i.e., ER-L and TEL) below which adverse effects are not expected, and a higher level (i.e., ER-M and PEL) above which effects are likely to occur. The approaches of Long and Morgan (1990) and MacDonald et al. (1996) calculate SECs as follows:

- **ER-L:** 10th percentile of the concentration distribution for the effects data.
- **ER-M:** Median of the concentration distribution for the effects data.
- **TEL:** Geometric mean of the 15th percentile of the concentration distribution for the effects data and the median of the distribution for the no-effects data.
- **PEL:** Geometric mean of the ER-M and the 85th percentile of the concentration distribution for the no-effects data.

For both approaches, the effects distribution for each chemical is defined as those stations at which a biological effect is observed and the associated chemical concentration is greater than or equal to twice the mean concentration of the no-effect stations. In addition, MacDonald et al. (1996) stipulate that it is desirable for both the effects and no-effects distributions to include at least 20 data entries.

A major distinction between the various kinds of SECs is the manner in which effects and no-effects data are used. As shown by the definitions above, the ER-L/ER-M values are based only on effects data, whereas the TEL/PEL values are based on both the effects and no-effects data. As described previously in this section, AET values are based only on the no-effects data (i.e., "nonimpacted" stations).

For Onondaga Lake, the various kinds of SECs were developed primarily on the basis of the chironomid survival endpoint, which identified effects at 35 of the 79 stations sampled in 1992. None of the other endpoints identified effects at a sufficient number of stations to achieve the stipulation of MacDonald et al. (1996) that the effects and no-effects distributions should both have at least 20 data entries. The results of these calculations are presented for comparison in Table 9-13.

The various SECs were initially calculated on the basis of chironomid survival only and on the basis of any kind of toxic effect. Because the resulting SECs showed little differences, the subsequent analyses were conducted only using the survival endpoint.

9.2.4.3 Evaluation of Mercury Sediment Effect Concentrations

The results of the mercury SEC comparisons are presented in Table 9-13. To assess the accuracy with which the various sets of SECs identified the presence or absence of effects in Onondaga Lake in 1992, the following performance criteria were calculated using mercury as an example:

- **False Positives (Type I Error):** The percentage of stations predicted to have effects (i.e., based on exceedance of one or more of the SECs) that actually had no observed effects based on the chironomid survival results.

- **False Negatives (Type II Error):** The percentage of stations predicted to have no effects (i.e., based on lack of exceedance of any of the SECs) that actually had observed effects based the chironomid survival results.

- **Overall Accuracy:** The percentage of all samples that were correctly predicted to have effects, or not to have effects based on the SECs.

From a practical standpoint, a high percentage of false positives is undesirable for a set of SECs because a large number of stations predicted to have toxic sediments actually would not have such sediments. This could potentially result in remediation of areas where such activities are not warranted. By contrast, a high percentage of false negatives is undesirable because a large number of stations predicted not to have toxic sediments actually would have such sediments. This could potentially result in remedial actions not being selected for all areas where they are warranted. Ideally, therefore, a set of SECs should have relatively small percentages of both false positives and false negatives.

The major performance criteria patterns for mercury are as follows:

- **False Positives (Type I Error):** The AET for mercury had the lowest false positive error (14 percent), whereas values for the other SECs ranged from 33 percent (ER-M and PEL), 53 percent(ER-L), and 48 percent (TEL).

- **False Negatives (Type II Error):** The ER-L had the lowest false positive error (12 percent), whereas errors for the other SECs were 47 percent (ER-M and PEL), 48 percent (TEL), and 82 percent (AET).

- **Overall Accuracy:** The mercury ER-M/PEL had the highest degree of overall accuracy (66 percent), followed by the AET value of 65 percent, with the TEL at 56 percent and the ER-L at 51 percent.

Based on the results of the SEC evaluations described above, it can be concluded that no one of the methodologies employed accurately describe or predict threshold concentrations of toxicity in Onondaga Lake sediments, nor can any one methodology accurately attribute the toxicity observed to any single contaminant. These values cannot be absolute because of the exposure of organisms to a complex mixture of metals and other contaminants which make it difficult to attribute the toxicity to any particular contaminant. However, collective evaluation through a strength-of-evidence approach does provide useful information.

9.2.4.4 Evaluation of Sediment Effect Concentrations Based on the 2000 Data

As described previously, sediment toxicity data were collected at 15 stations in Onondaga Lake in 2000 primarily to compare the results of the 42-day amphipod and chironomid toxicity tests with the 1992 results of the 10-day amphipod and chironomid toxicity tests. As shown in Section 9.2.1.3, the results of the 42-day tests were similar to those of the 10-day tests with respect to the areas of the lake in which sediment toxicity was present or absent.

In this section, an evaluation is conducted to determine whether SECs based on the 2000 data are similar to those developed using the 1992 data. However, because the 2000 data were collected at a relatively small number of stations, they will be considered for use in risk characterization in a qualitative manner. In addition, because the concentration ranges of the organic chemicals measured at the 15 stations were not evenly distributed between high, medium, and low values (as opposed to mercury), it was only possible to develop meaningful AETs for metals. Organic AETs were calculated (Table 9-12), but are provided only for qualitative comparison to the 1992 AET values.

The results of the 1992/2000 SEC comparisons are presented in Table 9-14 for seven metals. There was no consistent pattern with respect to one kind of SEC being greater than the other. The two kinds of AETs generally agreed, within a factor of two, for all of the metals; therefore, agreement between the two sets of AETs can be considered relatively good. For example, Long et al. (1995) considered agreement among various kinds of SECs to be close when they agreed within a factor of three.

9.2.4.5 Development of Consensus Based Probable Effect Concentrations

Consensus-based probable effect concentrations (PECs) for COCs in Onondaga Lake were developed to support an assessment to sediment-dwelling organisms and follow the methodology described in MacDonald et al. (2000) and Ingersoll et al. (2000). The PECs are the geometric mean of the AET, PEL, TEL, ER-M, and ER-L SECs. In addition, the PECs:

- Provide a unifying synthesis of site-specific effects concentrations.
- Reflect causal rather than correlative effects.
- Account for the effects of sediment COCs.

The PECs do not consider the potential for:

- Bioaccumulation in aquatic species.
- Potential effects that could occur throughout the food web as a result of bioaccumulation.
- Synergistic or antagonistic effects of chemical mixes in the sediment.

Onondaga Lake PECs were developed for all compounds identified as COCs (see Chapter 6) based on the 1992 data and are presented in Table 9-13.

9.2.5 Acid-Volatile Sulfide

Based on the concentrations of acid-volatile sulfide (AVS) observed throughout Onondaga Lake during the 1992 and 2000 RI sampling, the bioavailability of divalent metals such as cadmium, copper, lead, mercury, nickel, silver and zinc should be limited during the summer months in anoxic sediments. Because AVS binds with metals, it reduces their bioavailability (DiToro et al., 1990, 1992). When the molar ratio of simultaneously extracted metals (SEM) to AVS is less than or equal to 1, toxicity due to the divalent metals is not predicted because a sufficient amount of AVS is present to bind with the total amount of SEM. However, when the SEM/AVS ratio is greater than 1, toxicity may occur, depending on the concentrations of SEM and the presence or absence of other factors that modify the bioavailability of metals (USEPA, 1994a, 1995a; Ankley et al., 1996; Berry et al., 1996). Uncertainties related to use of the SEM/AVS ratio are discussed in Chapter 10.

As shown in Figure 9-23, AVS concentrations throughout most of the deeper parts of Onondaga Lake in 1992 were very high (>2,000 mg/kg), reflecting the hypereutrophic condition of the lake. By contrast, concentrations in most of the shallow nearshore areas of the lake were less than 500 mg/kg. The

SEM/AVS ratios at most stations in the lake were less than or equal to 1 during both 1992 and 2000 (Figure 9-24), largely reflecting the high concentrations of AVS found throughout most of the lake. The SEM/AVS ratios were greater than 1 at 13 stations in the shallowest parts of the lake in 1992, indicating that divalent metals could cause sediment toxicity at those stations. However, most of those stations were characterized by coarse-grained sediments with low concentrations of both SEM and AVS. Therefore, SEM/AVS ratios greater than 1 at those stations would not necessarily result in sediment toxicity. In fact, sediment toxicity was observed at only 3 of those 13 stations during the RI.

The theoretical wisdom indicates that there should not be any methylmercury in the sediments of the hypolimnion if there is excess AVS. As indicated on Figure 9-24, there is (or should be) enough AVS to complex the divalent metals and that should bind the inorganic mercury so it is not available to form methylmercury. There is sufficient literature to support the contention that AVS will bind inorganic mercury and make it unavailable for methylation. On the other hand, Long et al. (1998) suggests this process does not always occur under field conditions. Methylmercury formation in aquatic systems is influenced by a wide variety of environmental factors. The efficiency of microbial mercury methylation generally depends on factors such as microbial activity and the concentrations of bioavailable mercury, which in turn are influenced by temperature, pH, redox potential, and the presence of inorganic and organic complexing agents. Earlier studies noted that mercury methylation is inhibited by high sulfide levels in soils, sediments, and bacterial cultures (Fagerstrom and Jernelov, 1971; Bisogni and Lawrence, 1975; Jacobs and Keeney, 1974; Talmi and Mesmer, 1975). It was speculated that in the presence of sulfide, Hg forms insoluble HgS, which is not readily available for methylation under anaerobic conditions (Fagerstrom and Jernelov, 1971; Gillespie, 1972). However, current studies have reported that the solubility of Hg is actually increased in the presence of excess sulfide, most likely due to the formation of soluble complexes (Gognon et al, 1997; Benoit et al, 1998; Bloom et al., 1999). Recently, the work of Benoit et al (1998, 1999a, 1999b) shows that sulfide affects the bioavailability of mercury by controlling mercury speciation, and suggests that the bioavailability of mercury in sediments is determined by the concentration of neutral dissolved mercury complexes such as HgS^0, which may readily diffuse across bacterial cell membranes.

It is also important to note that the use of AVS to predict non-toxic sediment is less conservative than the use of sediment quality guidelines. Long et al. (1998) found that AVS resulted in a 19 percent false negative rate when they evaluated 77 samples from five marine locations. The authors reported that the use of the NOAA ER-Ls resulted in no false negatives. AVS may also be a poor predictor of bioaccumulation of metals (Ankley, 1996). Work by Howard and Evans (1993) in stratified lakes in Canada points toward another issue: seasonal changes of AVS concentrations in eutrophic lakes where significant temporal and spatial changes in AVS concentrations occur and bioavailability of divalent metals may increase strongly during times when the lake sediments are oxidized. Since no temporal variation in AVS levels was determined in this BERA, the usefulness of these data to predict the lack of year-round bioavailability and toxicity is questionable.

9.3 Effects Characterization for Terrestrial and Aquatic Vertebrates

9.3.1 Selection of Measures of Effects

For the selection of TRVs in this assessment, a comprehensive literature search of laboratory and field studies was conducted on the toxicity of COCs to terrestrial and aquatic vertebrates. Using the Ovid search engine, a variety of databases were searched for references containing toxicity information, including the following:

- TOXLINE.
- TOXNET (including the Aquatic Information Retrieval Database [AQUIRE]).
- USEPA's and US Army Corps of Engineers' (USACE's) Environmental Residue Effects Database (ERED).
- National Library of Medicine (NLM) MEDLINE.

Secondary sources that were used to identify studies that may have been overlooked in the database searches included the following:

- US Fish and Wildlife Service Contaminant Hazard Reviews.
- Agency for Toxic Substances Disease Registry (ATSDR) documents.
- USEPA Great Lakes Water Quality Initiative documents.
- Jarvinen and Ankley database (1999).

A number of criteria were considered in order to evaluate the appropriateness of a particular study for inclusion in the database used for this BERA. First of all, doses should be quantified and reported. An appropriate study design, including the use of adequate sample size and an appropriate negative control group, should be included in the design. Appropriate statistical analyses should be conducted and the statistical significance of the results reported. The remainder of this section describes the rationale that was used to select TRVs for the representative receptors.

Some studies examine toxicity endpoints (such as lethality, growth, and reproduction) that are thought to have greater potential for adverse effects on populations of organisms than toxicity endpoints evaluated in other studies. Other studies examine toxicity endpoints, such as behavior, disease, cell structure, or biochemical changes, that affect individual organisms but may not result in adverse effects at the population level. For example, toxic effects such as enzyme induction may or may not result in adverse effects to individual animals or populations. This BERA prefers TRVs from studies that examine the effects of COCs on growth or reproduction, as these endpoints typically present the greatest risk to the viability of the individual organism and, therefore, survival of the population. Thus, these are considered to be the endpoints of greatest concern relative to the stated assessment endpoints.

Because of the persistence of contaminants in Onondaga Lake, the exposure of ecological receptors is expected to be long-term. Some reproductive effects of contaminants are typically seen after long-term exposure, or in offspring of exposed individuals. Therefore, studies of chronic exposure were used to select TRVs for this risk assessment.

Dose-response studies compare the response of organisms exposed to a range of doses to that of a control group. Ideally, doses that are below and above the threshold level that causes adverse effects are examined. Toxicity endpoints determined in dose-response and other studies include:

- **No observed adverse effect level (NOAEL):** The highest exposure level shown to be without adverse effect in organisms exposed to a range of doses. NOAELs may be expressed as dietary doses (e.g., mg COC consumed/kg body weight per day [-d]), as concentrations in external media (e.g., mg COC/kg food), or as concentrations in tissue of the exposed organisms (e.g., mg chemical/kg egg).

- **Lowest observed adverse effect level (LOAEL):** The lowest exposure level shown to produce adverse effect in organisms exposed to a range of doses. LOAELs may also be expressed as dietary doses (e.g., mg COC consumed/kg body weight-d), as concentrations in external media (e.g., mg COC/kg food), or as concentrations in tissue of the exposed organisms (e.g., mg chemical/kg egg).

- **LD_{50}:** The lethal dose that results in the death of 50 percent of the exposed organisms. Expressed in units of dose (e.g., mg COC administered/kg body weight of test organism-d).

- **LC_{50}:** The lethal concentration in some external media (e.g. food, water, or sediment) that results in the death of 50 percent of the exposed organisms. Expressed in units of concentration (e.g., mg COC/kg wet weight [ww] food).

- **ED_{50}:** The effective dose that results in a sublethal effect in 50 percent of the exposed organisms (mg/kg-d).

- **EC_{50}:** The effective concentration in some external media that results in a sublethal effect in 50 percent of the exposed organisms (mg/kg).

- **Critical body residue (CBR):** The concentration in the organism (e.g., whole body, liver, or egg) that is associated with an adverse effect (mg COC/kg ww tissue).

- **EL-effect:** The effect level that results in an adverse effect in organisms exposed to a single dose, rather than a range of doses. Expressed in units of dose (mg/kg-d) or concentration (mg/kg).

- **EL-no effect:** The effect level that does not result in an adverse effect in organisms exposed to a single dose, rather than a range of doses. Expressed in units of dose (mg/kg-d) or concentration (mg/kg).

Most USEPA risk assessments typically estimate risk by comparing the exposure of receptors of concern to TRVs that are based on NOAELs. TRVs for this BERA were developed on the basis of both NOAELs and LOAELs to provide perspective on the range of potential effects relative to measured or modeled exposures.

Differences in the feeding behavior of aquatic and terrestrial organisms determine the type of toxicity endpoints that are most easily measured and most useful in assessing risk. For example, the dose consumed in food is more easily measured for terrestrial animals than for aquatic organisms since uneaten food can be difficult to collect and quantify in an aqueous environment. Therefore, for aquatic organisms, toxicity endpoints are more often expressed as concentrations in external media (e.g., water) or as accumulated concentrations in the tissue of the exposed organism (also called a "body burden"). In some studies, doses are administered via gavage, intraperitoneal injection into an adult, or injection into a fish or bird egg.

Where appropriate studies are available, TRVs for this BERA were selected on the basis of the most likely route of exposure, which for fish are expressed as CBRs (e.g., mg/kg whole body weight and mg/kg lipid in eggs or whole body) and for wildlife receptors (i.e., birds and mammals) are expressed as daily dietary doses (e.g., mg/kg whole body weight-d).

9.3.1.1 Methodology Used to Derive Toxicity Reference Values

The literature on toxic effects of COCs to animals includes studies conducted solely in the laboratory, as well as studies including a field component. Each type of study has advantages and disadvantages for the purpose of deriving TRVs for a risk assessment. For example, a controlled laboratory study can be designed to test the effect of a contaminant on the test species in the absence of the effects of other co-occurring contaminants. This is an advantage, since greater confidence can be placed in the conclusion that observed effects are related to exposure to the test compound. However, laboratory studies are often conducted on species that are easily maintained in the laboratory, rather than on wildlife species. Therefore, laboratory studies may have the disadvantage of being conducted on species that are less closely related to a particular receptor. This not a great disadvantage to the risk assessment, since the assessment endpoints evaluate feeding groups, as represented by individual receptor models. Field studies have the advantage that organisms are exposed to a more realistic mixture of contaminants than, for example, laboratory tests that expose organisms to a specific form of a contaminant (e.g., methylmercury chloride or Aroclor 1254). Field studies have the disadvantage that organisms are usually exposed to other contaminants, and observed effects may not be attributable solely to exposure to a specific contaminant.

Field studies were used in this BERA when they were available for species in the same taxonomic family as the receptor of concern and examined relevant sensitive endpoints, such as reproductive effects. When appropriate field studies were not available for a test species in the same taxonomic family as the receptor species of concern, laboratory studies or field studies on less-closely-related species were used to establish TRVs for the receptor species. The general methodology described in the following paragraphs was used to derive TRVs.

When appropriate chronic-exposure toxicity studies on the effects of contaminants on lethality, growth, or reproduction were not available for the species examined for a particular assessment endpoint, studies on other species were used to develop TRVs. In general, few receptor-specific studies were available. Therefore, avian TRVs were developed for application to avian receptors and mammalian TRVs were developed for application to mammalian receptors.

The general methodology used to develop LOAEL and NOAEL TRVs for this BERA is described below:

- If an appropriate LOAEL was unavailable for a phylogenetically similar species (e.g. within the same taxonomic family), the assessment used a study conducted on another species, preferably one that was closely related to the receptor of concern. Whenever several studies were available, professional judgment was used to select the most appropriate LOAEL. Interspecies uncertainty factors, which account for potential differences in sensitivity between a test species and a receptor, were not used in the development of the final TRVs for the risk assessment.

- In the absence of an appropriate NOAEL, an appropriate LOAEL may be divided by a conversion factor of 10 to estimate a NOAEL. The LOAEL to NOAEL conversion is similar to USEPA's derivation of human health reference dose (RfD) values, where LOAEL studies are adjusted by a factor of 10 to estimate NOAEL values (Dourson and Stara, 1983).

- When calculating chronic dietary dose-based TRVs (e.g., mg/kg-d) from data for sub-chronic tests, the sub-chronic LOAEL or NOAEL values were divided by a conversion factor of 10 to estimate chronic TRVs. The use of a conversion factor of 10 is consistent with the methodology used to derive human health RfDs (Dourson and Stara, 1983).

  USEPA has not established a definitive line between sub-chronic and chronic exposures for ecological receptors. This BERA generally follows Sample et al. (1996), which considers 10 weeks to be the minimum time for chronic exposure of birds and one year for chronic exposure of mammals (based on half the life span of laboratory rodents). However, in addition to duration of exposure, the time when contaminant exposure occurs is critical. Reproduction is a particularly sensitive life stage, due to the stressed condition of the adults and the rapid growth and differentiation occurring within the embryo (Sample et al., 1996). For many species, contaminant exposure of a few days to as little as a few hours during gestation and embryo development may produce severe adverse effects. Because TRVs were selected to evaluate the potential for adverse effects on wildlife populations and impaired reproduction is likely to affect populations, contaminant exposures of less than one year or 10 weeks, but that occur during reproduction, were considered to represent chronic exposures.

- In cases where TRVs were available as a dietary concentration (e.g., mg contaminant per kg food), a daily dose for birds or mammals was calculated on the basis of standard estimates of food intake rates and body weights (e.g., Sample et al., 1996; USEPA, 1993b).

9.3.2 Fish

Eleven COCs, i.e., antimony, arsenic, chromium, mercury/methylmercury, selenium, vanadium, zinc, DDT and metabolites, endrin, total PCBs, and dioxins/furans, were selected for fish. Risk was characterized based on measured body burdens in whole fish, which were then compared to body burden-based TRVs. Due to the limited range of body burden studies available, one set of TRVs was selected to apply to all fish species (Table 9-15).

9.3.2.1 Antimony

Antimony is a naturally occurring metal that is used in various manufacturing processes. Acute oral exposure of humans and animals to high doses of antimony or antimony-containing compounds may cause gastrointestinal disorders (e.g., vomiting, diarrhea), respiratory difficulties, and death at extremely high doses (Young, 1992). Subchronic and chronic oral exposure may affect hematologic parameters.

Doe et al. (1987) examined the toxicity of antimony to rainbow trout (*Oncorhynchus mykis*) in a 30-day test. Trout fingerlings (1.2 g) were exposed to antimony potassium tartrate in water at concentrations of 8 and 16 mg/L over a 30-day period. Fingerlings exposed to the higher dosage showed a reduction in survival of 50 percent, while those at the lower dosage showed no survival effects. Tissue residues were 9.0 mg/kg ww for fingerlings that showed reduced survival and 5.0 mg/kg ww for those with no survival effects. These values were selected for a LOAEL and NOAEL of 9.0 and 5.0 mg/kg ww, respectively. Sublethal antimony effect levels, such as reproductive endpoints, are likely to be much lower than these values.

9.3.2.2 Arsenic

Arsenic occurs naturally as sulfides and as complex sulfides of iron, nickel, and cobalt. However, anthropogenic input exceeds the amount of arsenic occurring naturally by about a factor of three (Eisler, 1988a). Inorganic forms of arsenic are more toxic than organic forms, and trivalent are more toxic than pentavalent forms. Arsenic toxicity varies between species and the effects can be altered by physical, chemical, and biological conditions. Health effects may occur to the respiratory, gastrointestinal, cardiovascular, and hematopoietic systems, and may range from reversible effects to cancer and death.

Gilderhus (1966, as cited in ERED) exposed bluegills to weekly applications of sodium arsenite herbicide in an artificial pond. Arsenite, being trivalent, is considered to be more potent than the pentavalent congener. At tissue concentrations of 1.7 mg/kg, abnormal ovary and oocyte development were observed, and at tissue concentrations of 2.2 to 11.6 mg/kg, decreased weight gains were observed. The effect of the abnormal ovary and oocyte development and decreased weight gain on growth and reproduction was unclear.

Diminished growth and survival was reported in immature bluegills when total arsenic residues in muscle is more than 1.3 mg/kg ww, and more than 5 mg/kg ww in adult bluegills (National Research Council Canada [NRCC], 1978). Therefore, 1.3 mg/kg ww was selected as a LOAEL to protect sensitive life stages. Walsh et al. (1977) determined that whole-body arsenic concentrations above 0.5 mg/kg may be harmful to fish. Therefore, a NOAEL of 0.5 mg/kg was selected for arsenic.

9.3.2.3 Chromium

Chromium plays a role in glucose and cholesterol metabolism and is thus essential to humans and animals. However, animals given lethal doses of various chromium compounds have exhibited symptoms including hypoactivity, lacrimation, mydriasis, diarrhea, changes in body weight, pulmonary congestion, fluid in the stomach and intestine, erosion and discoloration of the gastrointestinal mucosa, diarrhea, and gastric ulcers (Daugherty, 1992).

Chromium toxicity was evaluated based on a study by Van der Putte et al. (1981). Rainbow trout (*Oncorhynchus mykis*) were exposed to hexavalent chromium in water concentrations ranging from 2 to 50 mg/L over a period of four days. Significant lethality was noted at exposure concentrations corresponding to body burdens greater than 7.8 mg/kg ww. No mortality was noted at exposures corresponding to 2.3 mg/kg ww. Because of the short duration of the study, an uncertainty factor of 0.1 was used to extrapolate from subchronic to chronic exposure, resulting in a chromium NOAEL of 0.23 mg/kg ww and a LOAEL of 0.78 mg/kg ww. Sublethal chromium effect levels, such as reproductive endpoints, are likely to be much lower than these values.

9.3.2.4 Mercury/Methylmercury

Methylmercury is the most hazardous mercury species, due to its high lipid solubility and ionic properties that allow it to penetrate the membranes of living organisms. Methylmercury adversely affects reproduction, growth, behavior, osmoregulation, and oxygen exchange in aquatic organisms. Most mercury in fish is methylmercury, as confirmed by the data from Onondaga Lake. Therefore, all mercury concentrations in fish were considered to be methylmercury. Methylmercury readily penetrates the blood-brain barrier, produces brain lesions, spinal cord degeneration, and central nervous system dysfunctions.

There is both field and laboratory evidence that diet is the most important route of fish exposure to methylmercury, as it contributes 90 percent or more of the methylmercury accumulated. The assimilation efficiency for uptake of dietary methylmercury in fish is approximately 65 to 80 percent or greater.

Reproductive endpoints are generally more sensitive than growth or survival endpoints, with embryos and the early developmental stages being the most sensitive. Mercury can be transferred from tissues of the adult female to the developing embryo. Sublethal and lethal effects on fish embryos are associated with mercury residues in eggs that are perhaps 1 to 10 percent of the residues associated with toxicity in adult fish. Mercury concentrations in intoxicated rainbow trout range between 4 and 30 mg/kg (whole body), while intoxicated embryos contain 0.07 to 0.1 mg/kg (Weiner and Spry, 1996).

The toxic concentration of mercury compounds can vary by an order-of-magnitude or more, depending on the exposure condition. For example, toxicity is greater at elevated temperatures (Armstrong, 1979) and at lower oxygen content (Sloof et al., 1991).

The effects on aquatic organisms due to interactions of mercury with cadmium, copper, selenium, and zinc were found to be dependent on exposure concentrations (Birge et al., 1979). The interaction of mercury and other trace elements (e.g., selenium and zinc) can be both less than additive (antagonistic) and greater than additive (synergistic), depending primarily on exposure concentrations and the form of mercury. Effects were generally antagonistic at lower exposure levels and synergistic at higher levels. Exposure to low concentrations of mercury may not result in mortality directly, but may retard growth, thereby increasing the risk of predation (NOAA, 1996).

No standards that would be protective of aquatic organisms have been established for mercury concentrations in fish tissue. The current Food and Drug Administration (FDA) action level for the protection of human health, based only on methylmercury in the edible flesh of fish and shellfish, is 1 mg/kg (USFDA, 1984).

Friedmann et al. (1996) studied concentrations frequently observed in North American lakes to investigate the effects of dietary methylmercury on growth, gonadal development, and plasma cortisol levels in juvenile walleye (*Stizostedion vitreum*) over a six-month period. Reduced testicular development and immune function were observed at whole-body concentrations of 0.25 mg/kg ww. Rainbow trout exposed to mercuric chloride for 400 to 528 days showed significant reduction in alevin survival (four-day post-hatch) and a significant increase in teratogenic effects at a concentration of 0.5 mg/kg ww in ovary tissue (Friedmann et al., 1996).

NOAA (2002) has summarized toxicity associated with mercury in tissues. Based on their review of peer-reviewed studies, NOAA developed a mercury NOAEL and LOAEL of 0.1 and 0.3 mg/kg ww, respectively, for use at the LCP National Priorities List (NPL) EPA Region 4 site in Brunswick, Georgia (Mehran, 2002, pers. comm.). These TRVs were also selected for use in this BERA.

9.3.2.5 Selenium

Selenium is beneficial or essential in amounts from trace to part-per-billion (ppb) concentrations for humans and some plants and animals, but can be toxic at higher concentrations. Selenium chemistry is complex, and its metabolism and degradation are significantly modified by interaction with such elements as heavy metals, agricultural chemicals, microorganisms, and a variety of physicochemical factors (Eisler, 1985).

Fish with high body burdens of selenium failed to reproduce and exhibited teratogenic deformities in Belews Lake, North Carolina, which received dissolved selenium in wastewater from a coal-fired electricity generating facility (Lemly, 1997). Low waterborne concentrations of selenium eliminated 16 of 20 fish species present in the lake, and rendered the adults of two species sterile (Cumbie and Van Horn, 1978; Lemly, 1985). In these fish, selenium levels were elevated in liver (up to 21.4 mg/kg ww) and other tissues; kidney, heart, liver, and gills showed altered histopathology; and there were changes in blood chemistry. The ovaries of fish from Belews Lake had numerous necrotic and ruptured egg follicles that may have contributed to the population extinctions (Sorensen et al., 1984).

A survey performed ten years after the selenium releases to Belews Lake were stopped found developmental abnormalities in young fish, indicating that selenium-induced teratogenesis and reproductive impairment were occurring, and that concentrations of selenium in benthic food organisms are sufficient to cause mortality in young bluegill and other centrarchids due to Winter Stress Syndrome (WSS) (Lemly, 1997). WSS occurs when sublethal effects (metabolic stress) due to selenium are present at the same time as the arrival of cold water temperatures in late autumn. Cold weather and the associated short photoperiod of winter programs the fish for reduced activity and food intake, and they do not respond to the metabolic stress with increased feeding. If exposure to selenium persists, stored body fat necessary for overwintering is used up, fitness drops, and death may result.

Another, more compelling, effect of selenium on fish in regard to reproduction was also noted by Lemly (1997). Absorption of selenium passed from parents to their offspring in eggs causes morphological abnormalities as the young develop, if the concentrations in eggs reach 15 to 20 mg/kg dw (Gillespie and Baumann, 1986; Woock et al., 1987; Coyle et al., 1993). Using a conversion factor of fish whole-body values multiplied by 3.3 to calculate egg concentrations, based on the work of Lemly (1996, 1997), the threshold whole-body concentrations are between 4.5 and 6.1 mg/kg dw. Given an average percent solids of about 24 percent in Onondaga Lake fish, this translates to roughly 1.1 to 1.5 mg/kg ww.[1] The lower end of this range (1.1. mg/kg ww) was selected as the LOAEL, and an uncertainty factor of 0.1 was applied to the LOAEL to derive a NOAEL of 0.11 mg/kg ww (0.45 mg/kg dw).

[1] Each individual fish sample analyzed as wet weight was converted to dry weight based on its percent solids.

9.3.2.6 Vanadium

Vanadium is a natural constituent of sediment and water, as well as being found in fuel oils and coal. In water, vanadium can exist in both soluble forms and as a precipitate. Vanadium from water can be taken up and accumulated by fish.

Hilton and Bettger (1988) studied the effects of vanadium on juvenile rainbow trout. Trout were exposed to concentrations of 10.2 and 1.2 mg/kg sodium orthovanadate in their diet. They found reduced feeding and body weight and digestive tract distress at a concentration of 0.41 mg/kg ww in fish carcasses. They also noted protruding abdomens and darkened skin coloration on these fish. Trout that received a dose of 1.2 mg/kg in their diet had carcass concentrations of 0.02 mg/kg ww and showed no effects. Based on this study, a LOAEL of 0.41 mg/kg ww was selected. As the NOAEL concentration was about 20 times lower than LOAEL concentration, an uncertainty factor of 0.1 was applied to the LOAEL to derive a NOAEL of 0.041 mg/kg ww to avoid an overly conservative estimate of risk.

9.3.2.7 Zinc

Zinc is present in the environment naturally, but high concentrations come from activities such as mining, steel production, coal burning, and burning of waste. The toxicity of zinc to aquatic organisms is influenced by many factors, such as the temperature, hardness, and pH of the water, and previous zinc exposure of the organisms. Several fish kills in recent years have been attributed to zinc from runoff and discharges from mining areas and smelters. However, the concentrations causing mortality were generally not well documented, and, in many cases, high levels of other metals were also present.

A study on the effects of zinc on the American flagfish (*Jordanella floridae*) by Spehar (1976) was used to derive zinc TRVs. In this study, flagfish were exposed to zinc sulfate ($ZnSO_4$) in water at concentrations ranging from 26 to 139 μg/L. Reduced growth of females was seen at a dose of 51 μg/L and was the most sensitive measure of zinc toxicity. No effects were seen at exposure to 26 μg/L. These exposures translated into tissue concentrations of 40 and 34 mg/kg ww, respectively. Therefore, 40 mg/kg ww was selected as a LOAEL and 34 mg/kg ww was selected as the NOAEL.

9.3.2.8 DDT and Metabolites

Dichlorodiphenyl trichloroethane (DDT) was used as a pesticide until it was banned in 1972 due to unacceptable risks to the environment and potential harm to human health. DDT was developed as the first of the modern insecticides early in World War II. It was initially used with great effect to combat malaria, typhus, and the other insect-borne human diseases among both military and civilian populations. DDT came into wide agricultural and commercial usage in this country in the late 1940s.

DDT is toxic to several fish species, with the greatest mortalities in the younger age groups. DDT-contaminated feed has caused massive mortalities of sac fry of brook, rainbow, and cutthroat (*Oncorhynchus clarki*) trout in hatcheries (Connell and Miller, 1984). Rainbow trout and coho salmon

(*Oncorhynchus kisutch*) have been similarly affected in DDT-contaminated lakes (Connell and Miller, 1984). The organochlorines accumulate in eggs and can lead to the death of fry as the yolk sac is absorbed (Connell and Miller, 1984).

The toxicity to fish of DDT and its metabolites was based on a reproductive study of brook trout by Macek (1968). In this investigation, yearling trout were exposed to DDT through their diets at three dose levels (0.5, 1, and 2 mg/kg-week) for 156 days, including five months prior to spawning, with fertilized eggs produced from the control (i.e., no DDT exposure) and 1 and 2 mg/kg-week doses. A significant reduction in mature egg production was noted at the highest dose level. Increased mortality in eggs and sac fry were significantly higher in all mating combinations that received either one or both gametes from a treated parent. Observations indicated that mortality of fry may be due to DDT being released from the yolk fat (i.e., fry feeding) during the period of its maximum utilization (15th week). Total residues in adults corresponded to the levels of exposure.

The 1 mg/kg-week dose was selected as the LOAEL, based upon fry mortality. The mean body burden of DDT and metabolites (DDE and DDD) of fish treated with 1 mg/kg-week at the end of the exposure period was 2.9 mg/kg ww. The mean concentration of DDT and metabolites in the control group was 0.6 mg/kg ww, which was selected as a NOAEL.

9.3.2.9 Dioxins/Furans

Dioxins and furans are byproducts of chemical manufacturing, the result of incomplete combustion of materials containing chlorine atoms and organic compounds, or formed during the disinfection of complex effluents (e.g., pulp and paper effluents) containing many organic constituents. These substances have been associated with a wide variety of toxic effects in animals, including acute toxicity, enzyme activation, tissue damage, developmental abnormalities, and cancer.

To assess toxicity, chlorinated dioxins and furans are classified at varying levels of potency of 2,3,7,8-TCDD (Eastern Research Group [ERG], 1998). Dioxin/furan toxicity to fish was evaluated using toxicity equivalents [TEQs] for fish taken from Van den Berg et al. (1998).

Laboratory and field studies on the effects of dioxin-like compounds TEQs on fish typically report concentrations of TEQs in fish eggs, rather than in the whole body, since eggs represent a more sensitive life stage. Comparison of effect levels, such as NOAELs or LOAELs, reported as wet weight concentrations in eggs to whole-body tissue concentrations in adult Onondaga Lake fish is complicated by the fact that eggs and whole-body adult fish tend to have different lipid contents and concentrations of lipophilic contaminants, such as TEQs.

However, if TEQs are assumed to partition equally into the lipid phase of the egg and into the lipids in the tissue of adult fish (Niimi, 1983), then lipid-normalized concentrations in fish eggs that are associated with adverse effects (μg TEQs/kg lipid in egg) can be compared to lipid-normalized tissue concentrations of TEQs in adult fish (μg TEQs/kg lipid in whole-body adult). Therefore, LOAEL and NOAEL TRVs were established for TEQs in fish on a lipid-normalized basis, so that measured whole-body concentrations of TEQs in fish can be compared to TRVs established from studies on fish eggs.

A study on lake trout by Walker et al. (1994) was selected for the dioxin/furan TRVs. In this study, significant early life stage mortality was observed in lake trout eggs with a concentration of 0.6 μg TEQs/kg lipid. This effect was not observed at a concentration of 0.29 μg TEQs/kg lipid. These results were similar to other studies performed by Walker et al. (1992) and Walker and Peterson (1994). The values from this study were selected as TRVs in this assessment for a LOAEL of 0.6 μg TEQs/kg lipid and a NOAEL of 0.29 μg TEQs/kg lipid. Because the experimental study was based on the concentration in the egg, rather than an estimated dose, a subchronic-to-chronic conversion factor was not applied.

9.3.2.10 Endrin

Endrin is a rodenticide used to control mice and voles, and an insecticide used on cotton, rice, and maize. Closely related to aldrin and dieldrin, endrin is the most toxic of the three in the aquatic environment (UNEP, 2002) and its metabolites are more toxic than endrin itself.

Jarvinen and Tyo (1978) studied the effects of chronic exposure of fathead minnows (*Pimephales promelas*) to endrin concentrations in the water or food (clams), or both, for 300 days encompassing reproduction. Tissue residues were analyzed at present intervals for first-generation fish, and were also determined for embryos, larvae at hatch, and 30-day progeny. Endrin in the food (0.63 ppm) significantly reduced survival of the fathead minnows, and fish exposed to both endrin sources had lower survival than those exposed to either source alone.

Endrin residues in embryos and larvae were highest and larval survival lowest for progeny of adults exposed to endrin in both food and water. Survival of 30-day progeny was significantly reduced at all test exposures (0.63 ppm in the food, water exposures of 0.14 and 0.25 ppb, and all combinations of food and water exposure). Reduced survival was observed in larvae with a tissue residue of 0.24 mg/kg ww. This value was selected as a LOAEL and a factor of 0.1 was used to derive a NOAEL for LOAEL and NOAEL values of 0.24 and 0.024 mg/kg ww, respectively.

9.3.2.11 Polychlorinated Biphenyls

PCBs are industrial compounds that were used in a broad range of commercial applications until their manufacture was banned in 1976 under the Toxic Substances Control Act (TSCA) (15 U.S.C. Sec. 2601 et seq.). The toxicity of PCBs has been shown to manifest itself in many different ways, among various species of animals. Typical responses to PCB exposure in animals include wasting syndrome, hepatotoxicity, immunotoxicity, neurotoxicity, reproductive and developmental effects, gastrointestinal effects, respiratory effects, dermal toxicity, and mutagenic and carcinogenic effects. Some of these effects are manifested through endocrine disruption. PCB exposure through diet and water have been reported to cause a number of deleterious effects in fish survival, growth, egg production, and hatching success, as well as survival and development of progeny (Defoe et al., 1978; Cleland et al., 1988; Fisher et al., 1994).

A study using the sheepshead minnow (*Cyprinondon variegatus*) by Hansen et al. (1974) was selected as the most appropriate study to derive PCBs TRVs. This study established a NOAEL of 1.9 mg PCBs/kg and a LOAEL of 9.3 mg PCBs/kg for the sheepshead minnow. This study was based on a flow-through bioassay of Aroclor 1254 on adult female fish. Fish were exposed for 28 days, and then egg production was induced. The eggs were fertilized and placed in PCB-free flowing seawater and observed for mortality.

Survival of fry to one week of age was 77 percent for eggs from adults from the 0.32 μg/L concentration in water treatment (average 9.3 mg/kg in tissue of females), as compared to 95 percent survival of fry from control adults and 97 percent survival of fry from adults from the NOAEL treatment (0.1 μg/L; average 1.9 mg/kg in tissue of females). A LOAEL of 9.3 mg/kg in tissue and a NOAEL of 1.9 mg/kg in tissue were selected for this BERA. Because the experimental study measured the actual concentration in fish tissue, rather than estimating the dose on the basis of the concentration in external media (e.g., food, water, or sediment, or injected dose), a subchronic-to-chronic conversion factor was not applied.

9.3.3 Amphibians and Reptiles

A worldwide decline in amphibian and reptile populations has caused great concern in the scientific community (Environment Canada, 2001). As environmental contaminants have been implicated as a possible cause of some declines, there has been a substantial increase in the amount of amphibian and reptile ecotoxicology research conducted over the last decade. Amphibians may be exposed to toxic compounds through several routes because of their semipermeable skin, the development of their eggs and gill-breathing larvae in the water, and their changing position in the food web from herbivorous tadpoles to carnivorous adults (Gutleb et al., 1999). Amphibians typically have both terrestrial and aquatic life stages during which they may be susceptible to the effects of environmental contaminants. In addition, amphibians are important food organisms for a large variety of fish, birds, and mammals, and can be of major ecological significance (Nebeker et al., 1995). Reptiles are long-lived, sedentary beings and therefore may be good "biomonitors" of their local environment.

Effects from contaminant exposure may vary depending on exposure route, the point in time of exposure during the life cycle, and the length and intensity of the exposure. The embryo is generally the most sensitive life stage (Pèrez-Coll and Herkovits, 1996), so high concentrations of contaminants/stressors during embryonic stages (generally spring) may have significant repercussions on amphibian populations.

Although herpetofaunal toxicity studies are increasing (e.g, see Environment Canada's Reptile and Amphibian Toxicity Literature database [Environment Canada, 2001] or California EPA's Exposure Factor and Toxicity database [CAL/EPA, 2002]), there are still many data gaps. Therefore, general water and sediment quality criteria values for aquatic organisms are the best available values for many compounds. Based on the limited herpetological toxicity data, a quantitative analysis of risk to amphibians and reptiles is not performed in this BERA. It is acknowledged, however, that contaminants may adversely affect herpetofauna. For example, Zoll et al. (1988) examined the genotoxicity and bioaccumulation of mercury in newts. They observed broken chromosomes and chromosome aberrations in blood smears from larvae exposed to mercuric chloride and methylmercuric chloride. Bioaccumulation ratios after 12 days were 600 for mercuric chloride and 1,200 for methylmercuric chloride. Metals, such as cadmium, may also be bioaccumulated. Larval salamanders exposed to cadmium in the water had tissue concentrations up to 63 times the water concentration and exhibited adverse growth effects (Nebeker et al., 1995).

9.3.4 Birds and Mammals

Twenty-eight COCs were selected to evaluate potential risk to wildlife receptors (see Table 6-2). This section discusses the toxicity of selected COCs and the development of the specific baseline assessment TRVs necessary to characterize risk for terrestrial vertebrates. Generally, reproductive endpoints were preferred for development of TRVs. In instances where no reproductive studies were available, or where studies with other endpoints were considered to be superior, based on professional judgment, non-reproductive endpoints were chosen. TRVs for non-reproductive endpoints were generally higher than for reproductive endpoints. Tables 9-16 and 9-17 summarize the TRVs selected for terrestrial wildlife.

The model used to assess the potential risks was based on a numerical comparison of the modeled exposure rate over the TRV to derive the hazard quotient (HQ), as follows:

$$HQ = \frac{EER}{TRV}$$

where:

HQ	=	hazard quotient or the ratio of the exposure and the TRV (unitless)
EER	=	estimated exposure rate determined at the mean and 95 percent UCL of the mean COC concentrations in Onondaga Lake (mg/kg body weight [bw] per day)
TRV	=	toxicity reference value for the no effects or lowest observed effects thresholds (mg/kg body weight per day)

Exposure rates were evaluated using the NOAEL and LOAEL to provide a range of ecological risk. Considerations of uncertainty in the TRV predictions are discussed in Chapter 11. The derivations of specific TRVs are described below.

9.3.4.1 Arsenic

Arsenic is a naturally occurring element that is used as a wood preservative and also in insecticides and herbicides. Effects of arsenic exposure in mammals include stomach upset and diarrhea. Large doses may cause low birth weight, fetal malformations, or death.

The toxicity of inorganic compounds containing arsenic depends on the valence or oxidation state of the arsenic as well as on the physical and chemical properties of the compound in which it occurs (Sample et al., 1996). Trivalent (As^{+3}) compounds such as arsenic trioxide (As_2O_3), arsenic trisulfide (As_2S_3), and sodium arsenite ($NaAsO_2$), are generally more toxic than pentavalent (As^{+5}) compounds such as arsenic pentoxide (As_2O_5), sodium arsenate (Na_2HAsO_4), and calcium arsenate [$Ca_3(AsO_4)_2$]. The relative toxicity of the trivalent and pentavalent forms may also be affected by factors such as water solubility; the more toxic compounds are generally more water soluble. This BERA evaluates the effects of the trivalent form of arsenic.

Arsenic was selected as a COC for one avian receptor, the tree swallow (*Tachycineta bicolor*), and four mammalian receptors the little brown bat (*Myotis lucifugus*), short-tailed shrew (*Blarina brevicauda*), mink (*Mustela vison*), and river otter (*Lutra canadensis*).

The avian TRVs for arsenic were based on a study by US Fish and Wildlife Service (USFWS, 1969) where copper acetoarsenite (44 percent As^{+3}) was fed to cowbirds (*Molothrus ater*) at four dose levels. Cowbirds at the two highest dose levels (675 and 225 ppm) experienced 100 percent mortality, while those in the two lower groups (75 and 25 ppm) experienced 20 percent and 0 percent mortality, respectively. Because the study considered exposure over seven months, the 75 ppm Paris green (33 mg/kg As^{+3}) and the 25 ppm Paris green (11 mg/kg As^{+3}) doses, equivalent to 7.38 mg/kg-d and 2.46 mg/kg-d, were considered to be chronic LOAELs and NOAELs, respectively.

The mammalian TRVs for arsenic were developed based on a study by Schroeder and Mitchener (1971). Mice were exposed to 5 ppm arsenite in drinking water over three generations. This concentration was associated with a decrease in litter size and is, therefore, considered a potential population level LOAEL. An increase in the male-to-female ratio of offspring was also observed. Assuming a drinking water intake rate of 0.0075 L/d for a 30 g mouse, a LOAEL of 1.26 mg/kg-day was derived. An uncertainty factor of 0.1 was applied to derive a NOAEL of 0.126 mg/kg-day.

9.3.4.2 Barium

Barium is a naturally occurring element common in carbonate-based soils and metamorphic parent materials. Barium is used industrially in the production of paints, bricks, tiles, glass, and rubber. Exposure to barium can cause high blood pressure, changes in the function and chemistry of the heart, decreased life span, and decreased body weight.

Barium was selected as a COC for two avian receptors, the tree swallow and mallard (*Anas platyrhynchos*), and two mammalian receptors, the little brown bat and short-tailed shrew.

The avian TRVs were based on a study by Johnson et al. (1960) where one-day old chicks were fed barium throughout the 4 week study period. While barium exposures up to 2,000 ppm produced no mortality, chicks in the 4,000 to 32,000 ppm groups experienced 5 percent to 100 percent mortality. Because 2,000 ppm was the highest nonlethal dose, this dose was considered to be a subchronic NOAEL.

The 4,000 ppm dose was considered to be a subchronic LOAEL. Chronic NOAELs and LOAELs were estimated by multiplying the subchronic NOAELs and LOAELs by a subchronic to chronic uncertainty factor of 0.1 for a body weight-normalized NOAEL of 20.8 mg/kg-d and a LOAEL of 41.7 mg/kg-d.

Toxicity and carcinogenicity studies of barium chloride dihydrate were conducted by administering the chemical to rats and mice in drinking water for 13 weeks and for two years (National Toxicology Program [NTP], 1994). In the chronic study, male and female rats (60 animals/dose group/sex) received drinking water containing 0, 500, 1,250, or 2,500 mg/L barium chloride dihydrate (equivalent to a dose of 0, 15, 45 and 75 mg/kg-day) for 104 weeks (males) or for 105 weeks (females).

Increased relative kidney weight was seen in the females at 2,500 ppm, indicating that it may be a chronic NOAEL or LOAEL for rats. When considered together with the results in the 13-week subchronic NTP (1994) study in rats, in which increased relative and absolute kidney weights were seen in female rats receiving 2,000 ppm barium in drinking water (115 mg Ba/kg-day) and kidney lesions at 4,000 ppm (180 mg Ba/kg-day), greater increases in relative and absolute kidney weights were seen in female rats. Increased relative kidney weight in females of the two-year study are suggestive of potential renal effects. Therefore, 75 mg Ba/kg-day was selected as a chronic LOAEL and 45 mg Ba/kg-day as the chronic NOAEL for mammals, based on renal effects (USEPA, 1999b).

9.3.4.3 Cadmium

Cadmium is used to manufacture batteries, pigments, metal coatings, and plastics. Inhalation of cadmium is carcinogenic, and rats have been shown to develop lung cancer after exposure (USEPA, 1987a). Ingestion can cause high blood pressure, iron-poor blood, liver disease, and nerve or brain damage. It has also been demonstrated that rats have fewer litters, and pups may have more birth defects than usual when exposed to cadmium orally (Sutou et al., 1980).

Cadmium was selected as a COC for two avian receptors, the mallard duck and tree swallow, and two mammalian receptors, the little brown bat and short-tailed shrew.

The avian TRVs were derived using a study by White and Finley (1978). Mallard ducks were exposed to 1.6, 15.2, and 210 ppm cadmium chloride. Mallards in the 210 ppm group produced significantly fewer eggs than those in the other groups. Because the study considered exposure over a period of 90 days, the 15.2 ppm cadmium dose was considered to be a chronic NOAEL and the 210 ppm dose was considered to be a chronic LOAEL. Adjusted for mallard body weight, these equal a NOAEL of 1.45 mg/kg-day and a LOAEL of 20 mg/kg-day.

For the mammalian TRV, a study by Sutou et al. (1980), in which rats were exposed to cadmium (as $CdCl_2$) at four dose levels (0, 0.1, 1, and 10 mg/kg-day) by oral gavage, through mating and gestation (six weeks), was selected. Adverse reproductive effects, including reduced fetal implantations, reduced fetal survivorship, and increased fetal resorptions were observed in the rats exposed to 10 mg/kg-day. Numbers of total implants and live fetuses in the 1 mg/kg-day decreased slightly, but there was no significant difference from the control. As the study was conducted during reproduction, exposures were considered chronic even though exposure lasted only six weeks. Therefore, 1 mg/kg-day dose was considered to the NOAEL and a dose of 10 mg/kg-day was considered the LOAEL TRV for the evaluation of risk to mammals.

9.3.4.4 Chromium

Chromium is a naturally occurring element found in rocks, animals, plants, and soil. Chromium compounds are used for chrome plating, the manufacture of dyes and pigments, leather tanning, and wood preserving. The metal chromium is used to make steel and other alloys. Inhalation of high levels of chromium may cause lung cancer. Ingesting large amounts may result in the development of skin ulcers, stomach upsets, and kidney and liver damage.

Chromium was selected as a COC for all five avian receptors (belted kingfisher [*Ceryle alcyon*], great blue heron [*Ardea herodias*], osprey [*Pandion haliaetus*], mallard, and tree swallow) and all four mammalian receptors (little brown bat, short-tailed shrew, mink and river otter).

To derive the avian TRVs, a study by Haseltine et al. (unpublished data, cited in Sample et al., 1996) was selected. Black ducks (*Anas rubripes*) were exposed to chromium(III) (as $CrK[SO_4]_2$) at two dose levels (10 and 50 ppm in food) for 10 months through reproduction. Duckling survival was reduced at the 50 ppm dose level, while no significant differences were observed at the 10 ppm dose level. Because the study considered exposure throughout a critical life stage (reproduction), the 50 ppm dose was considered to be a chronic LOAEL, and the 10 ppm dose was considered to be a chronic NOAEL. Assuming that the body weight of a mallard is 1.25 kg (Dunning, 1993) and the food consumption rate is 10 percent (Heinz et al., 1989) the NOAEL was determined to be 1 mg/kg-day and the LOAEL TRV was determined to be 5 mg/kg-day.

A study by MacKenzie et al. (1958) was used to derive the mammalian NOAEL. Rats were exposed to chromium(VI) (as $K_2Cr_2O_4$) at six dose levels in drinking water (0.45, 2.2, 4.5, 7.7, 11.2, and 25 ppm in water) for one year. Because no adverse effects were observed at any of the dose levels, the maximum dose (25 ppm chromium in water, or 3.28 mg/kg-day) was considered to be a chronic NOAEL. The assumptions used in TRV calculations included a body weight of 0.35 kg and water consumption rate of 0.046 L/day for rats.

The LOAEL TRV for exposure to chromium was based on a study by Steven et al. (1976, as cited in Sample et al. 1996). Rats were exposed daily to 134 and 1,000 ppm chromium(VI) in drinking water for three months. Increases in mortality were noted at 1,000 ppm, which was considered to be a subchronic LOAEL. A chronic LOAEL was estimated by multiplying the subchronic LOAEL by a subchronic-chronic uncertainty factor of 0.1. Based on the same body mass and intake rates used to derive the NOAEL, a LOAEL TRV of 13.14 mg/kg-day was derived.

9.3.4.5 Copper

Copper is used as component of some insecticides and fungicides and may also enter the environment through industrial activities. It is a naturally occurring element, but at high doses, it may reduce growth and result in mortality.

Copper was selected as a COC for two avian receptors (mallard, and tree swallow) and one mammalian receptors (little brown bat).

A study by Mehring et al. (1960) examining the effects of copper oxide on chicks for 10 weeks was used to derive avian TRVs. Chicks were fed one of 11 dose levels in their diet ranging from 36.8 to 1,150 ppm. While consumption of copper up to 570 ppm had no effect of growth of chicks, 749 ppm copper in the diet reduced growth by over 30 percent and produced 15 percent mortality. Because this study was 10 weeks in duration, the 570 and 749 ppm Cu doses were considered to be a chronic NOAEL and LOAEL, respectively. These doses adjusted for body weight and intake translated into a NOAEL of 47 mg/kg-d and a LOAEL of 61.7 mg/kg-d (Sample et al., 1996).

A study by Aulerich et al. (1982) on young mink fed supplemental copper (copper sulfate) at concentrations of 25, 50, 100, and 200 ppm in their diet was used to derive the mammalian TRVs. A concentration of 60.5 mg/kg was present in the base feed. Consumption of copper at all but the lowest does level increased the percentage mortality of mink kits. Because this study was approximately one year in duration and considered exposure during reproduction, the 25 ppm supplemental copper (85.5 ppm total copper) dose was considered to be a chronic NOAEL, equivalent to 11.7 mg/kg-day based on body weight (1 kg) and intake rate (137 g/d), and the 50 ppm supplemental copper (110.5 ppm total copper) dose was considered to be a chronic LOAEL, equivalent to15.14 mg/kg-d.

9.3.4.6 Lead

Lead is a metal that ranges from 0.1 to 10 ppm in ultramafic rocks and calcareous sediments (Kabata-Pendias and Pendias, 1992). Lead is used industrially in the production of batteries, ammunition, ceramics, and medical and scientific equipment. The toxic effects of lead on aquatic and terrestrial organisms are extremely varied and include mortality, reduced growth and reproductive output, blood chemistry alterations, lesions, and behavioral changes. However, some of these effects exhibit general trends in their toxic mechanism. Generally, lead inhibits the formation of heme, adversely affects blood chemistry, and accumulates at hematopoietic organs (Eisler, 1988b). At high concentrations, near levels causing mortality, marked changes to the central nervous system occur prior to death (Eisler, 1988b).

Lead was selected as a COC for three avian receptors, the tree swallow, belted kingfisher, and red-tailed hawk, and two mammalian receptors, the little brown bat, and short-tailed shrew.

The avian TRV was developed from a study by Edens et al. (1976). Japanese quail (*Coturnix coturnix japonica*) were exposed to lead acetate in feed at four dose levels (1, 10, 100, and 1,000 ppm in food) for 12 weeks, through reproduction. The 10 ppm lead concentration (11 ppm dry weight [dw]) resulted in no significant adverse reproductive effects, but even as little as 1 ppm of lead caused a marked decline in egg production. Exposure at 100 ppm (110 ppm dw) resulted in a reduction in hatching success by 28 percent. Assuming a body weight of 0.15 kg (Vos et al., 1971) and food consumption rate of 16.9 g dw/kg body weight-day (based on Nagy, 1987), a NOAEL of 1.18 mg/kg-day and LOAEL of 11.8 mg/kg-day were calculated.

The mammalian TRVs for lead were based on a laboratory study by Azar et al. (1973) of three generations of rats given doses of 10, 50, 100, 1,000, and 2,000 ppm lead acetate in their food. Lead exposures of 1,000 and 2,000 ppm resulted in reduced offspring weights and produced kidney damage in the young. Therefore, the 100 ppm dose (8 mg/kg-day) was considered to be a chronic NOAEL and the 1,000 ppm dose (80 mg/kg-day) was considered to be a chronic LOAEL.

9.3.4.7 Manganese

Manganese is used as component of some insecticides and fungicides and may also enter the environment through industrial activities. It is a naturally occurring element, but at high doses, it has been shown to cause reproductive effects and stimulate tumors (NIOSH, 2002).

Manganese was selected as a COC for one mammalian receptor, the little brown bat.

A study by Lasky et al. (1982) where rats were fed manganese oxide (Mn_3O_4) in their diet at three dose levels (350, 1,050, and 3,500 mg/kg supplemented manganese + 50 mg/kg manganese in base diet) for 224 days. Pregnancy percentage and fertility among rats consuming 3,550 ppm manganese in their diet was significantly reduced No effects were observed at lower manganese exposure levels. Therefore the 1,100 ppm Mn dose was considered to be a chronic NOAEL and the 3,550 ppm Mn dose was considered to be a chronic LOAEL, equivalent to doses of 88 and 284 mg/kg-day, respectively.

9.3.4.8 Mercury (Inorganic)

Mercury exists in the environment in different chemical forms. The predominant species in water, soil, and sediment is ionic or inorganic mercury (Hg^{2+}). Ionic mercury can exist in a free ionic form (as chlorides or hydroxides), but most is adsorbed or chemically bound to clays, sulfides, and/or organic matter.

The kidney is the major reservoir of inorganic mercury in birds and mammals. In renal tissue, mercury binds to metallothionein. Consequently, the major toxic effect of inorganic mercury is kidney damage – specifically, necrosis of the proximal tubular cells. Inorganic mercury, unlike the metallic or organic species, is incapable of crossing the blood-brain barrier and, therefore, does not exhibit neurotoxicity. Other systemic effects include gastrointestinal damage and cardiovascular effects. There is limited evidence that inorganic mercury may pose some reproductive toxicity.

Inorganic mercury was selected as a COC for all avian and mammalian wildlife receptors.

The evaluation of risk to birds from exposure to inorganic mercury was based on a study by Hill and Shaffner (1976). Japanese quail chicks were fed 2, 4, 8, 16, or 32 ppm mercury as mercuric chloride until one year of age. There were no significant effects on food consumption, growth rate, or body weight maintenance at any dose. Hatchability and eggshell thickness were also unaffected at 4 ppm (4.4 ppm dw), but egg hatching rates were depressed by 16.1 percent at 8 ppm (8.8 ppm dw). Assuming a body weight of 0.15 kg (Vos et al., 1971) and an ingestion rate of 0.169 kg dw/kg body weight-day (derived from Nagy, 1987), a NOAEL TRV of 0.45 mg/kg-day and a LOAEL TRV of 0.90 mg/kg-day were calculated. The actual TRVs may be lower because the study did not discuss hatchling survival, which is influenced by some forms of mercury (e.g., Heinz, 1974).

The evaluation of risk to mammals from exposure to inorganic mercury was based on a study by Aulerich et al. (1974). Mink were orally dosed with 10 ppm (40 ppm dw) mercuric chloride for six months over gestation. No reproductive effects were observed. This was, therefore, considered to be the NOAEL for exposure of mammals to inorganic mercury. Assuming a body weight for penned mink of 1 kg (USEPA, 1993b) and a food consumption rate of 0.548 kg dw/kg body weight-day (based on the observations of Bleavins and Aulerich, 1981), the NOAEL TRV was determined as 1 mg/kg-day. Only one dose level was examined in the study, so the LOAEL TRV was calculated by applying an uncertainty factor of 10 for a LOAEL of 10 mg/kg-day.

9.3.4.9 Methylmercury

Methylmercury (CH_3Hg^+; $[CH_3]_2Hg$) represents a small, but significant, fraction of total mercury (approximately 10 percent) in the water column because of its high toxicity and natural tendency to bioaccumulate in upper trophic level prey. Most of the mercury in fish is present as methylmercury, providing exposure pathways for piscivorous and semi-piscivorous animals. Mercury methylation also occurs in wetlands (see Chapter 6, Section 6.3.1), providing exposure pathways for terrestrial wildlife.

Methylmercury was selected as a COC for all avian and mammalian wildlife receptors.

Methylmercury in birds has been demonstrated to affect various organ systems, with embryos being more sensitive than adults (Eisler, 1987a). Toxic effects of methylmercury include decreased reproductive success, altered behavior, hepatic lesions, ataxia, weakness, muscular atrophy, and death. Reproductive effects of mercury in birds include reduced hatchability (due to increases in early mortality of embryos), eggshell thinning, reduced clutch size, increased numbers of eggs laid outside of the nest, aberrant behavior of hatchlings, and potential hearing impairment in juveniles. In some cases, overall reproductive success in birds has decreased as much as 35 to 50 percent due to dietary methylmercury exposure insufficient to cause obvious signs of intoxication in adults. The most sensitive indicator of exposure appears to be reproductive parameters, which were used to establish the methylmercury TRVs for this BERA.

The avian TRVs were based on a three-generation study by Heinz (1974, 1976a,b, 1979) on mallard ducks. Mallards were fed methylmercury dicyandiamide at a level of 0.5 mg/kg-dw (0.1 mg/kg-ww) in dry duck mash. Females fed methylmercury laid fewer eggs and produced fewer ducklings than control ducks, and laid a greater number of eggs outside of nest boxes. Those ducklings that survived were less responsive to taped maternal warning calls and were hypersensitive to fright stimulus. Based on a food intake rate of 128 g/kg body weight (as reported by Heinz, 1979) for the treated F1 and F2 females, this represents a LOAEL TRV of 0.064 mg/kg body weight-day. No long-term studies were identified as suitable for the derivation of a no-effects TRV for methylmercury exposure to birds. Therefore, an uncertainty factor of 0.10 was applied to the LOAEL to derive a NOAEL of 0.0064 mg/kg-day.

Other avian field and laboratory studies support the concentration range of the TRVs selected. For example, Barr (1986) made similar observations in a field study of the common loon (*Gavia immer*) in northwestern Ontario. Egg laying and territorial fidelity were both reduced where mean mercury concentrations in loon prey was 0.3 to 0.4 mg/kg-ww. Loons in these areas established few territories and none laid any more than a single egg. The eggs contained mercury concentrations as high as 1.4 mg/kg-ww. Around waters where mean mercury concentrations of prey exceeded 0.4 mg/kg-ww, the loons raised no progeny. Reproductive effects may extend beyond the embryo and may reduce the rate of juvenile survival.

Subchronic histologic, neurologic, and immunologic effects were observed in great egrets dosed with methylmercury chloride at 0.5 mg/kg-ww, corresponding to intakes of 0.135 to 0.048 mg/kg-day during a 14-week experiment (Spalding et al., 2000). Dietary concentrations of methylmercury that produced significant reproductive impairment were about 20 percent of those required to produce overt neurological effects in adult birds (Scheuhammer, 1995).

The toxicity of methylmercury to mammals was based on two mink studies by Wobeser et al. (1976) and Wren et al. (1987). In a two-year study by Wobeser et al. (1976), mink were exposed to methylmercury chloride in their diets for 93 days (subchronic) at concentrations ranging from 1.1 to 15 ppm ww. Histopathological evidence of injury was present in all mink exposed to methylmercury. Clinical signs of neurotoxicity (anorexia and ataxia) were manifested at an exposure concentration of 1.8 ppm ww, and resulted in increased mortality in mink fed doses of 1.8 ppm or higher. In accordance with the procedures applied by USEPA in the Great Lakes Water Quality Initiative (USEPA, 1995b), an uncertainty factor of 0.1 was applied to account for extrapolation from a subchronic to a chronic toxicity. Based on a body mass of 1 kg for a mink in captivity and an intake rate of 0.137 kg ww/kg body weight per day (Bleavins and Aulerich, 1981), a NOAEL of 0.015 and a LOAEL TRV of 0.025 mg/kg bw-day were calculated from this study.

However, a study by Wren et al. (1987) observed increased mortality in mink fed 1 ppm of methylmercury for 81 days. The dosage was decreased after that time period due to excessive mortality. Wren et al. (1987) attributed the increased mortality to a combination of methylmercury exposure and cold stress, as the mink were maintained in outdoor cages. Based on these results, a NOAEL using the 1.1 ppm dosage from the Wobeser et al. (1976) study was not considered to be protective, as increased mortality occurs at lower dosages when combined with natural stresses present in field conditions. Therefore, an uncertainty factor of 0.1 was applied to the LOAEL of 0.025 mg/kg body weight-day to derive a NOAEL of 0.0025 mg/kg bw-day.

A study by Charbonneau et al. (1976), in which domestic cats (*F. Domesticus*) were exposed to methylmercury in their diets at doses ranging from 3 to 176 μg/kg-day continuously for up to two years, was not used to derive TRVs. In that study, significant and irreversible neurological impacts were noted at 74 μg/kg-day, while no significant neurological manifestations attributable to mercury exposure were observed at 46 μg/kg-day (0.046 mg/kg-day). Although cats and minks are both in the order Carnivora, they are in separate taxonomic suborders. *F. Domesticus* belong to suborder Feliformia, while mink and otter belong to suborder Caniformia. Both mink and otter are known to be very sensitive to the availability and toxicity of mercury within their habitat (Wren et al., 1986).

9.3.4.10 Nickel

Nickel is found in nature as a component of silicate, sulfide, or, occasionally, arsenide ores, and is usually found as Ni^{2+} in aquatic systems. Chemical factors that can affect the form of nickel in aquatic systems include pH and the presence of organic and inorganic ligands (USEPA, 1986d).

Nickel was selected as a COC for two avian receptor, the mallard and tree swallow, and one mammalian receptor, the little brown bat.

Reproductive and developmental effects from exposure to nickel have been observed in animals and various nickel compounds have been tested for mutagenicity (USEPA, 1986d). These tests have demonstrated the ability of nickel compounds to produce genotoxic effects; however, the translation of these effects into actual mutations is still not clearly understood. There is evidence both in humans and animals for the carcinogenicity of nickel, at least in some forms.

The avian TRVs were selected based on a study on mallard ducklings by Cain and Pafford (1981). Ducklings were fed nickel sulfate at three dose levels (176, 774, and 1,069 ppm) for 90 days. Consumption of up to 774 ppm nickel in the diet did not increase mortality or reduce growth; however, the 1,069 ppm nickel diet reduced growth and resulted in 70 percent mortality. Because the study considered exposure over 90 days, the 774 ppm dose was considered to be a chronic NOAEL and the 1,069 ppm dose was considered to be a chronic LOAEL. To estimate daily nickel intake throughout the 90-day study period, food consumption of 45-day-old ducklings was calculated. Using the consumption rate of 100 g food/day for a 1 kg adult (Heinz et al.,1989), a NOAEL of 77.4 mg/kg-day and a LOAEL of 107 mg/kg-day were calculated.

A study by Ambrose et al. (1976) was used to derive the mammalian TRVs. Rats were given doses of 250, 500, or 1,000 ppm nickel sulfate hexahydrate in their diets over three generations. While 1,000 ppm nickel in the diet reduced offspring body weights, no adverse effects were observed at the other dose levels. Because this study considered exposures over multiple generations, the 500 ppm dose was considered to be a chronic no effect level and the 1,000 ppm dose was considered to be a chronic lowest effect level. Assuming a body weight of 0.35 kg and a food ingestion rate of 28 g food/day (Sample et al., 1996), a NOAEL of 40 mg/kg-day and a LOAEL of 80 mg/kg-day were calculated.

9.3.4.11 Selenium

Selenium is a non-metallic element common in sedimentary soils. It is predominantly found either as insoluble metallic selenides or as soluble oxygen complexes, the most common being selenite and selenate. Average background concentrations in the US range from less than 0.1 to 4 ppm, with a mean of 0.31 (Kabata-Pendias and Pendias, 1992). Although selenium is an essential nutrient, exposure to high concentrations have been shown to result in adverse health effects. *In vivo*, selenium replaces sulfur in *de novo* amino acid synthesis, yielding selenomethionine and, to a lesser extent, selenocystine.

In birds, selenomethionine has been shown to be more toxic that the readily dissociated selenate or selenite. However, a survey of the mammalian literature yielded lower NOAELs for selenate than for any of the organic selenium compounds. The primary targets of toxicity include the gastrointestinal tract, the pancreas, and the thymus, with secondary toxicities associated with the kidney, liver, and central nervous system. Selenium has also been shown to be a reproductive toxicant, causing reduced fertility and increased malformations in both birds and mammals.

Selenium was selected as a COC for the tree swallow, belted kingfisher, great blue heron, and osprey, and for all mammalian receptors.

The toxicity of selenomethionine to birds was evaluated based on the results of a study by Heinz et al. (1989). Mallard ducks were fed 0, 1, 2, 4, 8, or 16 ppm selenomethionine in their diet for 100 days prior to egg set. An additional treatment of 16 mg/kg-day selenocystine was also included in the study. Reproductive productivity was significantly reduced at 8 ppm, with no significant effects noted at 4 ppm. Based on an average body weight of 1 kg and a food intake rate of 110 g dw/day (Heinz et al., 1989), a NOAEL TRV of 0.4 mg/kg-day and a LOAEL TRV of 0.8 mg/kg-day were derived.

The evaluation of the impact of selenium exposure on the reproduction of mammalian receptors was based on a study by Rosenfield and Beath (1954) in which rats were exposed to three levels of potassium selenate (1.5, 2.5, and 7.5 ppm) in drinking water over two generations. The treatment group exposed to 2.5 ppm showed no significant difference with regards to reduction rate or number of young reared. However, the second-generation female progeny of this treatment group showed a 50 percent reduction in the number of young reared. In the 7.5 mg/L group, fertility, juvenile growth, and survival were reduced. Therefore, the no-effects TRV was determined based on a dose of 1.5 ppm. Assuming a water intake rate of 0.046 L/day (based on the scaling function of Calder and Braun, 1983) and an average body weight of 0.35 kg (USEPA, 1988), a NOAEL TRV of 0.20 mg/kg-day and a LOAEL TRV of 0.33 mg/kg-day were determined.

9.3.4.12 Thallium

Thallium is a metal that can be released into the environment from coal combustion, heavy metals smelting, refining processes, and rodenticides. As a metal it exists in trace amounts in the earth's crust, with soil concentrations in the US ranging from 0.02 to 2.8 ppm (Kabata-Pendias and Pendias, 1992). The effects of thallium exposure can include gastroenteritis, diarrhea, constipation, vomiting, abdominal pain, and hair loss.

Thallium was selected as a COC for the tree swallow, little brown bat, and short-tailed shrew.

No avian studies on the effects of thallium were available and therefore no avian TRVs were derived for thallium.

The evaluation of thallium toxicity in mammals was based on a study by Formigli et al. (1986) in which rats were exposed to 10 ppm thallium sulfate (0.74 mg/kg-day, as provided in the study) in water for 60 days. The study dosage resulted in reduced sperm motility, and was therefore considered to be an LOAEL. Although exposure was subchronic, 90 percent of the animals in the treatment group showed effects of thallium exposure. Other endpoints, such as hair loss peripheral and nervous system disorders, have been documented to occur at the same dose (continuous exposure to 10 ppm thallium per day) in long-term studies (Manzo et al,.1983). Therefore, no uncertainty factors were applied for a LOAEL of 0.74 mg/kg-day. A NOAEL was estimated by applying a tenfold level of uncertainty to the LOAEL to derive a value of 0.074 mg/kg-day.

9.3.4.13 Vanadium

Vanadium is a natural constituent of soils, as well as being found in fuel oils and coal. Vanadium is concentrated mainly in mafic rocks and shales, and the average concentrations for US soils ranges between 58 and 100 ppm (Kabata-Pendias and Pendias, 1992). The most common anthropogenic sources involve vanadium entering the environment when fuel oils are burned.

Vanadium was selected as a COC for the tree swallow, mallard, little brown bat, short-tailed shrew, mink, and river otter.

The avian TRVs were developed from a study by White and Dieter (1978), in which mallard ducks were fed 2.84, 10.36, and 110 ppm vanadyl sulfate in their food over a 12-week period. No effects were observed at any dose level. The maximum dose of 11.4 mg/kg-day (based on study body weights and ingestion rates) was considered to be the chronic NOAEL. A chronic LOAEL of 114 mg/kg-day was calculated by applying an uncertainty factor of 10 to the NOAEL.

The mammalian TRVs were developed based on a study by Domingo et al. (1986), in which rats were exposed to sodium metavanadate ($NaVO_3$) at dose levels of 5, 10, and 20 mg/kg-day (corresponding to 2.1, 4.2, and 8.4 mg/kg bw-day) by oral intubation. Males were exposed for 60 days prior to mating and females were exposed for 14 days prior to mating. Significant decreases were observed in the development of the pups at all dose levels. The lowest dose (2.1 mg/kg bw-day $NaVO_3$) was therefore considered to be the chronic LOAEL TRV. The chronic TRV for mammals was determined by applying a tenfold level of uncertainty, to yield a NOAEL of 0.21 mg/kg-day.

9.3.4.14 Zinc

Zinc is used in many commercial products, including coatings to prevent rust, dry cell batteries, and is mixed with other metals to make alloys like brass and bronze. Some zinc is released into the environment by natural processes, but most comes from activities such as mining, steel production, coal burning, and burning of waste.

Zinc was selected as a COC for all avian receptors with an aquatic component in their diet (i.e., belted kingfisher, great blue heron, osprey, mallard, and tree swallow), the little brown bat, and the short-tailed shrew.

The avian TRVs were based on a study on leghorn hens by Stahl et al. (1990). The hens were fed zinc sulfate in their diet at doses of 48, 228, and 2,028 ppm for a period of 44 weeks. While no adverse effects were observed among hens consuming 48 and 228 ppm zinc, egg hatchability was less than 20 percent of controls among hens consuming 2,028 ppm zinc. The 228 ppm dose (corresponding to 131 mg/kg-day, based on the hens used in the study) was considered a chronic no-effect level and the 2,028 ppm dose was considered a chronic lowest effect level. Based on the body weights and food intake rates provided in the study, daily intake rates of 14.5 and 131 mg/kg-day were derived as the NOAEL and LOAEL.

The mammalian TRVs were based on a study by Schlicker and Cox (1968), where rats were exposed to zinc (2,000 and 4,000 mg/kg dose levels) in diet during gestation. Rats fed the higher dose displayed increased rates of fetal resorption and reduced fetal growth rates, while no effects were observed at the 2,000 mg/kg dose rate. As exposure occurred during gestation (a critical life stage), the lower dose corresponding to 160 mg/kg-day (based on body weight and intake rate) was considered a chronic NOAEL, and the higher dose corresponding to 160 mg/kg-day was considered to be a chronic LOAEL.

9.3.4.15 Bis(2-ethylhexyl)phthalate

Bis(2-ethylhexyl)phthalate (BEHP) is a synthetic chemical used principally as a plasticizer (an additive to plastics to make them more flexible), and may constitute as much as 40 percent of some PVC products (ATSDR, 1993). It is also used to a lesser extent in inks, pesticides, cosmetics, and vacuum pump oil (Sittig, 1991).

BEHP was selected as a COC for the tree swallow.

The avian TRVS were based on a study of ringed doves by Peakall (1974). Ringed doves were fed a dose of 10 ppm bis(2-ethylhexyl)phthalate (BEHP) in their diet over four weeks. No significant reproductive effects were observed among doves fed BEHP. The 10 ppm dose, corresponding to 1.1 mg/kg-d was considered to be a chronic NOAEL, as the study considered exposure over a critical life stage. A LOAEL of 11 mg/kg-day was derived by multiplying the NOAEL by an uncertainty factor of 10.

9.3.4.16 Chlordane

Chlordane is a viscous liquid, colorless to amber, with a slight chlorine-like aromatic odor. It was historically applied directly to soil or foliage (e.g., corn, citrus, deciduous fruits and nuts, and vegetables) to control a variety of insect pests. Short-term exposure to elevated levels of chlordane may affect the central nervous system, including irritability, excess salivation, labored breathing, tremors, convulsions, deep depression, and also result in blood system effects such as anemia and certain types of leukemia. Long-term exposure to chlordane has the potential to cause damage to liver, kidneys, heart, lungs, spleen and adrenal glands, and cancer.

Chlordane was selected as a COC for the short-tailed shrew.

The mammalian chlordane TRVs where based on Khasawinah and Grutsch (1989), in which mice (80/sex/group) were given 0, 1, 5, or 12.5 ppm technical chlordane in their diet for 104 weeks, corresponding to average doses of 0, 0.15, 0.75, and 1.875 mg/kg-day, respectively. Hematology, biochemistry, urinalysis, organ weights, and pathology of major tissues and organs were assessed on all animals that died during the study and on all survivors at week 104. Exposure-related effects were restricted to the liver. Based on the increased incidence of hepatic necrosis over controls, 1 ppm chlordane (0.15 mg/kg-day) was selected as the NOAEL, and 5 ppm chlordane (0.75 mg/kg-day) was selected as the LOAEL.

9.3.4.17 DDT and Metabolites

Avian species are particularly sensitive to the effects of DDT and its metabolites, specifically with regard to impacts on reproduction (McEwen and Stephenson, 1979). Toxicological impacts attributed to DDT exposure include eggshell thinning, reduced clutch size, elevated embryo mortalities, high mortality at time of pipping, increased hatchling mortality, and late nesting and unusual nesting behavior. In eggshell thinning, the activity of Ca^{2+} ATP-ase systems in the shell gland are affected, thereby interfering with the deposition of calcium in the shell (Lundholm, 1987). Eggshell thinning of greater than 20 percent has been associated with decreased nesting success due to eggshell breakage (Anderson and Hickey, 1969). Because of the tendency of DDT to magnify in food chains, higher trophic level birds appear to be at greater risk for egg loss due to shell thinning.

Another well-defined effect of DDT exposure is inhibition of acetylcholinesterase (AchE) activity. Inhibition of this enzyme results in the accumulation of acetylcholine in the nerve synapses, resulting in disrupted nerve function. Chronic inhibition of 50 percent of brain AchE has been associated with mortality in birds (Ludke et al., 1975).

The effects of DDT on other receptor groups are not as clearly defined. Recent studies indicate that DDT may be an estrogenic mimic, resulting in adverse reproductive effects. Observed effects include feminization and increased female:male population ratios for some receptors. Other responses include histopathological changes, alterations in thyroid function and changes in the activity of various enzyme groups (Peakall, 1993). In addition to toxic effects, DDT and its metabolites can bioaccumulate.

DDT and its metabolites was selected as a COC for the tree swallow, belted kingfisher, great blue heron, osprey, red-tailed hawk, and mink.

The TRV used for the evaluation of toxic effects of DDT and its metabolites in birds was based on the results of Anderson et al. (1975), which is the same study used by the Great Lakes Water Quality Initiative (USEPA, 1995b). Anderson et al. (1975) studied the reproductive success of brown pelicans (*Pelecanus occidentalis*) off the coast of southern California from 1969 through 1974. Concentrations of DDT and its metabolites in northern anchovies (a main component of the brown pelicans' diet) and pelican eggs were monitored during the course of the five-year investigation. Over this time, total DDT (and metabolite) concentrations declined in the fish from 4.27 to 0.15 ppm ww (0.60 ppm dw). At the lowest prey concentration, the fledgling rate was still 30 percent below that needed to maintain a stable population (1.2 to 1.5 young per pair). Therefore, 0.15 ppm was considered as the LOAEL. An LOAEL TRV of 0.028 mg/kg body weight-day was derived for birds, based on a body mass of 3.5 kg for an adult brown pelican (Dunning, 1993), and a food ingestion rate of 0.66 kg/day (USEPA, 1995b). A NOAEL TRV of 0.0028 mg/kg-day was derived by applying an uncertainty factor of 0.1.

For mammals, the lowest DDT and metabolite TRVs were derived from a reproductive study by Fitzhugh (1948), in which rats were exposed to doses of 10, 50, 100, and 600 ppm in their food for two years. While consumption of 50 ppm or more DDT in the diet reduced the number of young produced, no adverse effects were observed at the 10 ppm DDT dose level. These doses correspond to LOAELs and NOAELs of 0.8 and 4.0 mg/kg-day, respectively. It should be noted that studies indicate that mustelids (the family that mink belong to) can rapidly degrade DDT (e.g., Roos et al., 2001); therefore, the mammalian TRVs may be conservative for mink.

9.3.4.18 Dichlorobenzenes

1,2-Dichlorobenzene is used mainly as a chemical intermediate for making agricultural chemicals, primarily herbicides. Other present and past uses include use as a solvent for waxes, gums, resins, wood preservatives, and paints; insecticide for termites and borers; in making dyes; as a coolant, deodorizer, and de-greaser. 1,4-Dichlorobenzene (p-DCB) is an organic solid of white crystals with a mothball-like odor. It is used mainly as an insecticidal fumigant against moths in clothes and as a deodorant for garbage and restrooms. It is also used as an insecticide and fungicide on crops, in the manufacture of other organic chemicals, and in plastics, dyes, pharmaceuticals. Dichlorobenzene is known to bioaccumulate because of rapid metabolic turnover in exposed organisms. The long-term exposure to dichlorobenzenes may result in damage to the liver, kidneys, and cellular components of the blood, and may cause anemia, skin lesions, and appetite loss. Some neurological effects have been linked to inhalation of dichlorobenzene.

Dichlorobenzenes were selected as a COC for the mallard and the tree swallow.

The toxicity of dichlorobenzene to birds was evaluated based on a feeding study by Hollingsworth et al. (1956) in which geese were exposed to 500 ppm *p*-dichlorobenzene in their diet for a duration of five weeks. An exposure dose based on the measured intake rate was estimated to be approximately 600 mg/kg-day. The toxicological impacts of this exposure included a general reduction in growth and a mortality rate of 30 percent. This was therefore considered to be a sub-chronic LOAEL TRV. A tenfold uncertainty factor was applied to convert this value into a chronic LOAEL of 60 mg/kg-day, and an additional uncertainty factor of 0.1 was applied to derive a chronic NOAEL value of 6 mg/kg-day.

9.3.4.19 Dieldrin

Dieldrin, a chlorinated insecticide, was widely used from the 1950s to the 1970s, for soil and seed treatment, to control mosquitos and tsetse flies, as a sheep dip, for wood treatment, and for mothproofing woolen products. Most uses of dieldrin were banned in 1975, and it is no longer produced in, or imported to, the US (ASTDR, 1998). Dieldrin's toxic effects include carcinogenicity, mutagenicity, neurotoxicity, teratogenicity, and reproductive impairment.

Dieldrin was selected as a COC for all mammalian receptors.

In mammals, dieldrin is rapidly absorbed from the gastrointestinal tract upon ingestion. It is then transported from the liver to various tissues in the body, including the brain, blood, liver, and adipose tissue. Toxicity appears to be related to the central nervous system, resulting in stimulation, hyperexcitability, hyperactivity, incoordination, and exaggerated body movement, ultimately leading to confusion, depression, and death (ASTDR, 1998). Dieldrin has been shown to cross the placental barrier, and for that reason has been studied for its teratogenic properties and reproductive effects.

In the study selected for deriving TRVs, rats were exposed to dietary concentrations of dieldrin ranging from 0.08 to 40 mg/kg for up to 336 days (Harr et al., 1970). The concentration of 0.31 ppm (0.018 mg/kg-day) was the lowest concentration that resulted in adverse reproductive effects, which included a reduction in pup survival and conception rate. The highest dose that did not produce any reproductive effects was 0.16 ppm (0.009 mg/kg-day). Therefore, a value of 0.018 mg/kg-day was selected as the LOAEL and 0.009 mg/kg-day was selected as the NOAEL.

9.3.4.20 Dioxins/Furans

Polychlorinated dibenzo-*p*-dioxins (PCDDs) are composed of a triple-ring structure consisting of two benzene rings connected to each other by two oxygen atoms. Depending on the number and position of chlorine substitution on the benzene rings, 75 chlorinated dioxin congeners are possible. The polychlorinated dibenzofuran (PCDF) molecule is also a triple-ring structure, with the two benzene rings connected to themselves by a single oxygen atom. In all, 135 chlorinated dibenzofuran congeners are possible.

Dioxins and furans are not produced intentionally, but are unavoidable byproducts of chemical manufacturing or the result of incomplete combustion of materials containing chlorine atoms and organic compounds. Dioxins and furans may also be formed during the disinfection of complex effluents (e.g., pulp and paper effluents) containing many organic constituents.

Dioxins and furans may be distributed throughout the environment via air, water, soil, and sediments. Dioxins and furans tend to be very insoluble in water, adsorb strongly onto soils, sediments, and airborne particulates, and bioaccumulate in biological tissues (Hutzinger et al., 1985). These substances have been associated with a wide variety of toxic effects in animals, including acute toxicity, enzyme activation, tissue damage, developmental abnormalities, and cancer.

Dioxins, like PCBs, are polychlorinated hydrocarbons (PCHs) and toxicity is believed to be mediated intracellularly by binding with the aryl hydrocarbon receptor (AhR). The resulting PCH-AhR complex moves into the cell nucleus, where it will bind to the DNA, and may alter the expression of a number of gene sequences. Many of the observed toxic effects of dioxins (and the coplanar PCBs) are attributable to specific alterations in gene expression.

The effects of tetrachlorodibenzo-*p*-dioxins (TCDDs) have been reviewed by Safe (1990) and Giesy et al. (1994). Dioxins are not generally acutely toxic to adult organisms, but their long-term accumulation is thought to be expressed chronically, and may ultimately result in death. Key effects are those causing reproductive dysfunction. The PCDDs and PCDFs are thought to cause alterations to developmental endocrine functions (thyroid and steroid hormones), as well as interference in vitamin production, which results in disruption of patterns of embryonic development at critical stages (Giesy et al., 1994). General population-level manifestations of dioxin exposure include adversely affected patterns of survival, reproduction, growth, and resistance to diseases (Eisler and Belisle, 1996). Poor reproductive efficiencies and adventive, opportunistic diseases are characteristic of wild animals in the exposed populations of the Great Lakes region (Giesy et al., 1994).

To assess toxicity, chlorinated dioxins and furans are classified at varying levels of potency of 2,3,7,8-TCDD (Eastern Research Group [ERG], 1998). These variations in potency are quantified based on receptor-specific TCDD toxicity equivalence factors (TEFs).

Dioxins/furans were selected as a COC for all receptors, except for the great blue heron.

A study by Nosek et al. (1992) was used to derive avian TRVs. Ring-neck pheasants (*Phasianus colchicus*) were exposed to 2,3,7,8-TCDD at three dose levels: 0.01, 0.1, and 1.0 µp/kg BW/week via weekly intraperitoneal injection (equivalent to 1.4×10^{-6}, 1.4×10^{-5}, and 1.4×10^{-4} mg/kg-day) for 10 weeks, through reproduction. No adverse effects on reproduction were observed at the two lower dose levels. Therefore, the highest dose level of 1.4×10^{-4} mg/kg-day was considered to be a chronic LOAEL TRV and the 1.4×10^{-5} mg/kg-day the NOAEL TRV for birds.

The mammalian TRV used for the evaluation of risk to mammals was based on a study by Murray et al. (1979), who exposed rats to TCDD at three dose levels (1×10^{-6}, 1×10^{-5}, and 1×10^{-4} mg/kg-day in food) for three generations. The 1×10^{-6} mg/kg-day dose resulted in no significant adverse effects, and was therefore considered to be a chronic NOAEL TRV for mammals. The 1×10^{-5} mg/kg-day dose resulted in an approximately 10 percent reduction in live births and was therefore applied as the LOAEL TRV.

9.3.4.21 Endrin

Endrin is an organochlorine insecticide which has been used since the 1950s against a wide range of agricultural pests, mostly on cotton but also on rice, sugar-cane, maize, and other crops. It is also used as a rodenticide.. Like other chlorinated hydrocarbon insecticides, endrin also affects the liver, and stimulation of enzyme systems involved in the metabolism of other chemicals (WHO, 1992).

Endrin was selected as a COC for the belted kingfisher.

A study by Fleming et al. (1982) on screech owls (*Otus asio*) fed 0.75 ppm endrin in their diet for 83 days was used to derive the avian TRVs. Egg production and hatching success were reduced among owls fed endrin. Because the study considered exposure throughout a critical life stage (reproduction), the normalized dose of 0.1 mg/kg-d was considered to be a chronic LOAEL. A chronic NOAEL was estimated by multiplying the chronic LOAEL by a LOAEL to NOAEL uncertainty factor of 0.1 to obtain a value of 0.01 mg/kg-d.

9.3.4.22 Hexachlorobenzene

Hexachlorobenzene was widely used as a pesticide and fungicide for onions and wheat and other grains until 1965. It was also used in the manufacture of fireworks, ammunition, electrodes, dye, and synthetic rubber, and as a wood preservative (Sitting, 1991; ATSDR, 1997). There are currently no commercial uses of hexachlorobenzene (ATSDR, 1997).

Hexachlorobenzene was selected as a COC for the mink, little brown bat, and short-tailed shrew.

The mammalian TRVs were based on a study of mink and European ferrets (*Mustela putorius furo*) by Bleavins et al. (1984) fed diets that contained hexachlorobenzene (HCB). Diets treated with 125 or 625 ppm HCB were lethal to the adults of both species. The cross-fostering of mink kits whelped by untreated dams to females fed 2.5 mg/kg HCB resulted in increased kit mortality when compared to untreated controls. The in utero exposure to HCB resulted in higher kit mortality than exposure via the dam's milk. This dose resulted in a LOAEL of 0.14 mg/kg-day and a factor or 0.1 was applied to yield a NOAEL of 0.014 mg/kg-day.

9.3.4.23 Hexachlorocyclohexanes

Hexachlorocyclohexanes are found in the organochlorine insecticide lindane. The solubility of hexachlorocyclohexane isomers in lipid are as follows: delta > gamma > alpha > beta. Lindane has not been produced in the US since 1977, although it is still imported into and formulated in the US. Former uses included insecticide on fruit and vegetable crops including greenhouse vegetables and forest crops including Christmas trees.

Hexachlorocyclohexanes were selected as a COC for the belted kingfisher, great blue heron, and osprey.

Jansen's (1996) study on the common quail (*Cotumix cotumix*) was used to derive avian TRVs. Lindane was dissolved in the drinking water of captive quail at doses of 1, 3, and 9 ppm, for seven days. Eggshell thickness, egg volume, egg mass, incubation time, hatchability, and embryo development were recorded prior to, during, and after the treatment. Egg production was not affected by exposure to lindane. Egg mass was reduced significantly and egg volume increased slightly at 3 ppm lindane. There was no significant eggshell thinning as a result of exposure to lindane. Fertility and hatchability were lower at 3 and 9 ppm of lindane and incubation period was slightly reduced and overall fecundity decreased as a result of lindane ingestion. Doses of 1 and 3 ppm were selected as the NOAEL and LOAEL, respectively. Although quail were only exposed for seven days, exposure occurred during a sensitive reproductive period and therefore no uncertainty factor was applied. Using a body weight of 0.15 kg, based on Vos et al. (1971), a food intake rate of 16.9 g dw/day (Nagy, 1987), a NOAEL of 0.11 mg/kg-day and a LOAEL of 0.34 mg/kg-day were derived.

9.3.4.24 Polychlorinated Biphenyls

PCBs are industrial compounds that were used in a broad range of commercial applications until their manufacture was banned in 1976 under the Toxic Substances Control Act (TSCA) (15 U.S.C. Sec. 2601 et seq.). They are complex chemicals consisting of ten different homolog classes (monochlorobiphenyls to decachlorobiphenyls) that are distinguished by the number of chlorine atoms bound to the biphenyl molecule. Among these ten homologs are 209 different PCB congeners, which reflect the different number and location (isomer) of the bound chlorine atoms. PCBs were manufactured in the US under several trade names, but they are best identified with the name Aroclor (manufactured by Monsanto). Different Aroclors, which consist of different mixtures of congeners, were given four-digit codes (e.g., Aroclors 1248, 1260); the last two digits usually indicate the chlorine content (by percent weight).

For reviews of PCB toxicology, see Seegal (1996), Tilson et al. (1990), Safe (1990, 1994), Kimbrough (1985), Silberhorn et al. (1990), Bolger (1993), and Delzell et al. (1994).

PCBs were selected as a COC for all receptors, with the exception of the red-tailed hawk.

The avian TRVs selected for this BERA are based on a 16-week study by Dahlgren et al. (1972) that examined the effects of Aroclor 1254 on pheasants. In this study, ring-neck pheasants were dosed once a week with either 12.5 or 50 mg/bird-week Aroclor 1254. No impact on chick growth, egg production, or survivability was reported at the lower dose; however, egg hatchability was slightly lower at this dose. Therefore, the 12.5 mg/bird-week dose, equal to a daily dose of 1.8 mg/kg-day, was considered a LOAEL TRV. A NOAEL was derived by multiplying the LOAEL by a factor of 0.1 for a value of 0.18 mg/kg-day.

For mammalian receptors, separate TRVs were derived for mustelid receptors (mink and otter) and other mammals (bat and shrew), as mustelids are known to be sensitive to PCBs. A multigenerational study on mink by Restum et al. (1998) was used to derive mustelid TRVs. Over a three-year period, captive mink were fed carp caught from Saginaw Bay (Lake Superior) that contained PCBs and other contaminants. These fish were mixed with clean fish to produce treatment diets of 0, 0.25, 0.5, and 1 ppm ww total PCBs. Continuous exposure to 0.25 ppm or more delayed the onset of estrus and lessened the whelping (birth) rate. It also resulted in significantly lower body weights of kits at six weeks. Based on these effects, the 0.25 ppm treatment was considered a LOAEL. The LOAEL TRV was calculated to be 0.034 mg/kg-day, based on a body mass of 1 kg and an intake rate of 0.137 kg ww/kg body weight per day (Bleavins and Aulerich, 1981). The NOAEL TRV was estimated by the application of an uncertainty factor of 0.1 for a value of 0.0034.

The TRVs for the little brown bat and short-tailed shrew were based on multigenerational study of rats by Linder et al. (1974). Mating pairs of rats and their offspring were fed Aroclor 1254 at concentrations of 5 and 20 ppm. Offspring of rats fed 20 ppm exhibited decreased litter size (reduction of 15 to 24 percent) in comparison to controls., while those fed 5 ppm did not significantly differ from controls. Using a body weight of 0.35 kg and a food intake rate of 28 g/day based on Sample et al. (1996), a NOAEL of 0.4 mg/kg-day and a LOAEL of 1.6 mg/kg-day were calculated.

9.3.4.25 Polycyclic Aromatic Hydrocarbons

Polycyclic aromatic hydrocarbons (PAHs) is the general term applied to a group of compounds comprised of several hundred organic substances with two or more benzene rings. They occur in the environment mainly as a result of incomplete combustion of organic matter (forest fires, internal combustion engines, wood stoves, coal, coke, etc.). They are major constituents of petroleum and its derivatives. In addition, wastewater treatment plant effluents and runoff from urban areas, particularly from roads, are known to contain significant quantities of PAHs. Inputs of PAHs in aquatic ecosystems may occur as a result of oil spills, forest fires and agricultural burning, leaching from waste disposal sites, and coal gasification (Eisler, 1987b; Neff, 1979). PAHs are also produced by natural processes at very low rates.

In aquatic environments, PAHs tend to form associations with suspended and deposited particulate matter (Eisler, 1987b). This sorption of PAHs to sediments is strongly correlated with the TOC content of sediments, which influences its bioavailability. In general, elevated levels of sediment-associated PAHs are found in the vicinity of urban areas. Exposure to PAHs may result in a wide range of effects on biological organisms. While some PAHs are known to be carcinogenic, others display little or no carcinogenic, mutagenic, or teratogenic activity (Neff, 1979). Many carcinogenic PAHs also exhibit teratogenic and mutagenic effects. Several PAHs exhibit low levels of toxicity to terrestrial life forms, yet are highly toxic to aquatic organisms (Eisler, 1987b). Although PAHs are taken up and accumulated in terrestrial and aquatic plants, fish, and invertebrates, bioconcentration is limited by metabolism and elimination in many species.

PAHs can interact with cells in two ways to cause toxic responses. They may bind reversibly to lipophilic sites in the cell and thereby interfere with several cellular processes. Alternatively, their metabolites may bind covalently to cellular structures, causing long-term damage.

PAHs were selected as COC for all wildlife receptors except the osprey. Studies on the toxicity of PAHs in birds, particularly with regard to impacts on reproduction, are rare. A study examining the embryotoxicity of an artificial mixture of 18 PAHs in chicken, turkey, domestic duck, and common eider (*Somateria mollissima*) eggs found that a dose of 2 mg/kg-egg increased mortality among the embryos of all four species, and mortality was also increased in the duck at 0.2 mg/kg-egg (Brunstrom et al., 1990). Hough et al. (1993) examined the effects of benzo[a]pyrene on pigeons. Three- to six-month-old pigeons were administered a dose of 10 mg/kg weekly for a period of five months. The treatment birds were reported to have suffered complete reproductive failure and an associated gross alteration in ovarian structure. This dose, which corresponds to a daily exposure of 1.43 mg/kg-day, was considered representative of an LOAEL for birds. To estimate the NOAEL TRV, a tenfold level of uncertainty was applied to the LOAEL TRV to derive an estimate of 0.143 mg/kg-day.

The evaluation of PAH toxicity to mammals was based on a study by Mackenzie and Angevine (1981) that examined the reproductive effects of benzo[a]pyrene on mice. Female mice were exposed to benzo[a]pyrene at doses of 10, 40, and 160 mg/kg-day through daily intubation. Treatment commenced on day 7 after the best estimated time of conception and continued through day 16 of gestation. The effects of exposure were followed for up to six months. Although the duration of exposure was short, it is considered to be a chronic study because exposure occurred during a critical life stage. Total sterility was observed in 97 percent of the mice exposed prenatally to 40 or 160 mg/kg benzo[a]pyrene. Fertility was markedly impaired in animals exposed *in utero* to 10 mg/kg benzo[a]pyrene. After six months on a breeding study female mice in this group gave birth to significantly fewer and smaller litters and male mice in this group impregnated 35 percent fewer females than controls. The 10 mg/kg-day treatment was therefore considered to be applicable as a LOAEL TRV. The estimation of the NOAEL TRV was based on the application of a tenfold level of uncertainty to the toxicity estimate to derive a value of 1 mg/kg-day.

9.3.4.26 Trichlorobenzenes

1,2,4-Trichlorobenzene is an aromatic, colorless, organic liquid that is used primarily as a dye carrier. It is also used to make herbicides and other organic chemicals, as a solvent, in wood preservatives, and, previously, as a soil treatment for termite control. Short-term effects of exposure to 1,2,4-trichlorobenzene may cause changes in the liver, kidneys, and adrenal glands, and long-term exposure may result in increased adrenal gland weights.

Trichlorobenzenes were selected as a COC for the tree swallow, mallard, little brown bat, and short-tailed shrew.

No avian studies were found on the toxicity of trichlorobenzenes; therefore, potential risks to birds from this COC will be addressed in the BERA only qualitatively.

The mammalian TRVs are based on a multigenerational rat study (Robinson et al., 1981). From birth, male and female rats (F0 generation) were dosed with 0, 25, 100, or 400 ppm of 1,2,4-trichlorobenzene in the drinking water. Similar procedures were performed with their offspring (F1 generation) and the F2 generation. Fertility, as measured by the conception rates of the females, of F0 and F1 generation rats was not affected by treatment. An LOAEL was derived from a significant increase (11 percent in males, 13 percent in females) in adrenal gland weights observed in the 400 ppm (53.6 mg/kg-day) groups of males and females of the F0 and F1 generations. The NOAEL was considered to be the 100 ppm (14.8 mg/kg-day) dose. Effects on the F2 generation were less than on the F0 and F1 generations.

9.3.4.27 Xylenes

Xylenes are clear liquids with a sweet odor that are used mainly as a solvent. Other uses include as a component of gasoline, and for the production of phthalate plasticizers, polyester fiber, film, and fabricated items. Short-term exposure may result in disturbances of cognitive abilities, balance, and coordination, while long-term exposure may cause damage to the central nervous system, liver, and kidneys.

Xylenes were selected as a COC for the mallard, tree swallow, and little brown bat.

No studies examining the toxicity of xylenes to birds were found; therefore, potential risks to birds from this COC will be addressed in the BERA only qualitatively.

The mammalian TRVs are based on a mouse study by Marks et al. (1982), where mice received doses of 0.5, 1.0, 2.1, 2.6, 3.1, or 4.1 mg/kg-d of mixed xylenes a xylene mixture (60 percent m-xylene, 9 percent o-xylene, 14 percent p-xylene and 17 percent ethylbenzene) from days 6 to 15 of their gestation period via oral gavage. Xylene exposure of 2.6 mg/kg/d or greater significantly reduced fetal weights and increased the incidence of fetal malformities. While the xylene exposures evaluated in this study were of a short duration, they occurred during a critical life stage. Therefore, the highest dose that produced no adverse effects, 2.1 mg/kg/d, was considered to be a chronic NOAEL and the 2.6 mg/kg/d dose level was considered to be a chronic LOAEL.