8. ANALYSIS OF ECOLOGICAL EXPOSURES (ERAGS STEP 6)

8.1 Chemical and Stressor Characterization

In this section, the distributions of chemicals of concern and stressors of concern (COCs and SOCs) in Onondaga Lake media are described. Most of the information used in this section to characterize the general distributions of COCs and SOCs in the lake was taken from the 1992 and 1999/2000 field surveys conducted by Honeywell, and additional data (e.g., NYSDEC fish data from 1992 to 2000 and 2002 wetland sediment/soil data) were used. Additional detail on the nature and extent of contamination can be found in Chapter 5 of the Onondaga Lake Remedial Investigation (RI) report (TAMS, 2002b).

8.1.1 Distribution of Chemicals and Stressors of Concern in Water

In this section, the distributions of COCs/SOCs in the water of Onondaga Lake and its tributaries are described. Detailed summary tables of COC/SOC concentrations in water are presented in Appendix B, with 1992 data presented in Tables B-1 through B-26 and 1999 data presented in Tables B-27 through B-31. The 1992 data set is more extensive than the supplemental sampling performed in 1999. In 1992, sampling was conducted monthly from April to November at one station in the northern basin and one station in the southern basin of the lake. The 1999 sampling was oriented toward collecting data for the Onondaga Lake Human Health Risk Assessment (HHRA) (TAMS, 2002a) and included sampling selected areas where people could be exposed to lake water. Samples were also collected monthly from September to December 1999 at the southern basin station (W1) and in September and October 1999 at the northern basin station (W2).

In addition, 1997 to 2001 data for eutrophic stressors from the Onondaga County Ambient Monitoring Program (Onondaga County Department of Water Environment Protection [OCDWEP], 2002a, Onondaga County Department of Drainage and Sanitation [OCDDS], 1998]) are presented in this BERA. This program is intended to monitor Onondaga Lake, its tributaries, and the Seneca River in order to evaluate the impacts of alterations and improvements to the Metropolitan Syracuse Sewage Treatment Plant (Metro) and the combined sewer overflows (CSOs) on water quality. The program includes:

- Onondaga Lake monitoring.
- Tributary monitoring.
- Storm event monitoring in Onondaga Lake, Onondaga Creek, Ninemile Creek, Harbor Brook, and Ley Creek.
- Seneca River monitoring.
- Macroinvertebrate sampling in Onondaga Creek, Harbor Brook, and Ley Creek.

These areas are to be monitored from 1999 until 2012. Improvements to Metro and the CSOs are to be implemented in a phased program through this time period under an Amended Consent Judgment (ACJ) entered into on January 20, 1998, with a final completion date for the improvement projects in the year 2012. These improvements are designed to reduce loading of wastewater-related pollutants (i.e., ammonia, phosphorus, solids, floatables, and bacteria) by improvements to the CSO, filtration, and monitoring systems.

Monitoring stations in Onondaga Lake are at the northern and southern deep basins and, for storm event sampling, in nearshore areas of the lake, including the mouth of Ninemile Creek, the mouth of Sawmill Creek, the mouth of Ley Creek, the mouth of Harbor Brook, off of the Metro outfall, in the northeast corner of the lake between Bloody Brook and Sawmill Creek, and the northwest corner of the lake near the lake outlet.

Data from the ambient water quality monitoring program from 1997 through the first quarter of 2001 are included in select figures in this chapter.

The lake monitoring includes:

- Biweekly profiles of field parameters (pH, temperature, dissolved oxygen [DO], specific conductance, oxidation-reduction potential, salinity, and conductivity) at 0.5 m depth intervals at the southern deep station for the entire monitoring period (April through November).

- Biweekly profiles of ammonia, total Kjeldahl nitrogen (TKN), nitrate (N), and organic nitrogen at 3 m intervals at the southern deep station for the entire period and in the winter when possible.

- Biweekly samples of total phosphorus (P) at 1 m depths at the southern deep station from June through September.

- Biweekly analysis of solids and organic and inorganic carbon at 6 m intervals in the southern deep station.

- Biweekly composite samples of total dissolved solids in the epilimnion and the hypolimnion at the southern deep station.

- Biweekly Secchi disk transparency measurements at the southern basin for the entire period and at nearshore stations from June through September.

- Also included are DO profiles at 0.5 m depth intervals at the northern and southern deep stations during fall mixing (to include other water quality parameters during mixing), as well as DO measurements taken at tributary mouths once during

mixing, and diurnal sampling for DO, pH, and temperature in Onondaga Lake upper waters at 3-hour intervals for a 24-hour period during warm weather and algal bloom conditions.

In 1992, surface water samples were collected from eleven tributaries in the Onondaga Lake area in order to aid in determining the nature and extent of contamination and contaminant loadings to the lake. Most metal COCs (other than mercury) were detected occasionally in tributaries and point sources. Cadmium was detected infrequently, while copper, lead, and zinc were frequently detected. Tributary 5A, Geddes Brook, and the East Flume are the three tributaries in which the majority of the metal COCs were more frequently detected.

Organic COCs were occasionally detected in Tributary 5A, the East Flume, Harbor Brook and Geddes Brook. With the exception of a single detection of toluene in Metro effluent, organic COCs were not detected in any other tributaries in 1992. The tributaries and Metro effluent were also sampled for several other SOCs (chloride, ammonia, and depleted DO). The 1992 surface water data were stratified into three flow regimes: base flow, high flow, and intermediate flow. Base-flow conditions were determined by examination of daily flow records for each tributary, and were generally set to low flows in the late summer and early fall. High-flow conditions were defined as the highest 10 percent of the average daily flows, and intermediate flows were defined as flows between base-flow and high-flow stages. Summary statistics for the 1992 tributary data for base-flow, intermediate-flow, and high-flow conditions, as well as available 1992 and 1999 lake data, are presented in Tables 8-1, 8-2, and 8-3.

8.1.1.1 Mercury

Lake Water

The concentrations of total mercury and methylmercury found in unfiltered water samples from the epilimnion and hypolimnion of Onondaga Lake from April to November 1992 and September to December 1999 are presented in Figures 8-1 and 8-2, respectively. A high-flow event in April 1992 accounted for the highest concentration of total mercury (29 ng/L), which was detected in the southern basin hypolimnion (15 m depth).Otherwise, total mercury concentrations increased from spring to fall in both the epilimnion and hypolimnion. The increase in the hypolimnion was greater than the increase in the epilimnion. Methylmercury concentrations in the hypolimnion increased substantially from spring to fall of 1992, rising from a lake average of 1.0 ng/L in May to a maximum of 12 ng/L in early October. Methylmercury concentrations in the epilimnion increased at a higher rate than total mercury concentrations over the course of the year, rising from 0.4 ng/L in May to 1.5 ng/L in November, as compared to the rise in total mercury from 3.7 ng/L to 7.4 ng/L.

The mixing of lake water during fall turnover resulted in mercury and methylmercury concentrations in the hypolimnion declining and concentrations in the epilimnion increasing, so that contaminants were fairly evenly distributed at various lake depths. The maximum and mean concentrations of mercury detected in

lake water in 1992 of 29 and 9.0 ng/L were above the NYSDEC wildlife water quality value of 2.6 ng/L dissolved concentration (Chapter 4, Table 4-4; Appendix B, Table B-4).

In 1999, mercury in lake water was also analyzed from September to December in the southern basin and in September and October in the northern basin (Appendix B, Table B-27). Sampling was also performed at nearshore locations around the lake (see Chapter 2, Figure 2-17 of the RI). The maximum and mean concentrations of mercury detected in lake water in 1999 of 103 and 11 ng/L were above the NYSDEC wildlife water quality value of 2.6 ng/L (Appendix B, Table B-27). The maximum value of 103 ng/L was detected at Station W55 near Harbor Brook. A comparison of the 1992 and 1999 data showed no consistent pattern of increases or decreases of total mercury or methylmercury seasonally or by depth, indicating that mercury concentrations have remained elevated in Onondaga Lake over the last decade.

Tributary Water and Metro Discharge

Mean concentrations of total mercury and methylmercury in tributary water and Metro effluent in 1992 are presented for base-flow, intermediate-flow, and high-flow conditions in Figures 8-3 and 8-4.

Under base-flow conditions, mean concentrations of total mercury were highest in Geddes Brook (100 ng/L), the East Flume (54 ng/L), Metro effluent (24 ng/L), and lower Ninemile Creek (13 ng/L), and were less than 10 ng/L elsewhere. Mean total mercury concentration in Ninemile Creek downstream of the Geddes Brook confluence (13 ng/L) was significantly higher than the mean concentration in upper Ninemile Creek (3.8 ng/L) under base-flow conditions. Mean concentrations of methylmercury were highest in the Metro effluent (2.1 ng/L), the East Flume (0.8 ng/L), Harbor Brook (0.4 ng/L), and the lake outlet (0.4 ng/L), and were less than 0.3 ng/L elsewhere.

Under intermediate-flow conditions, mean concentrations of total mercury were highest in Geddes Brook (154 ng/L), the East Flume (96 ng/L), Tributary 5A (36 ng/L), and lower Ninemile Creek (31 ng/L), and concentrations were less than 25 ng/L elsewhere. Mean concentrations of methylmercury under intermediate flow conditions were highest in the East Flume (1.6 ng/L), the lake outlet (1.2 ng/L), Metro effluent (1.0 ng/L), Harbor Brook (0.61 ng/L), Onondaga Creek (0.54 ng/L), Tributary 5A (0.53 ng/L), and Geddes Brook (0.50 ng/L), and less than 0.5 ng/L elsewhere.

Under high-flow conditions, mean concentrations of total mercury were highest in the East Flume (155 ng/L), Geddes Brook (145 ng/L), Tributary 5A (94 ng/L), Harbor Brook (41 ng/L), Metro effluent (39 ng/L), and lower Ninemile Creek (27 ng/L), and were less than 20 ng/L elsewhere. A single high-flow total mercury concentration of 47 ng/L was reported for Bloody Brook. Mean concentrations of methylmercury were highest in Harbor Brook (2.4 ng/L), Tributary 5A (1.5 ng/L), the East Flume (1.3 ng/L), Metro effluent (1.2 ng/L), Geddes Book (1.0 ng/L), Ley Creek (0.85 ng/L), and the lake outlet (0.71 ng/L). Concentrations were less than 0.5 ng/L elsewhere.

8.1.1.2 Other Metals

Lake Water

The concentrations of Target Analyte List (TAL) inorganics were measured in unfiltered water samples from the southern and northern basins of Onondaga Lake in 1992. Analytes that were also analyzed in tributary water are presented in Table 8-1, with the exception of calcium and sodium. A complete list of inorganics and their detection frequency in lake water is provided in Table D-1 in Appendix D. Chromium, lead, and nickel were also sampled at a limited number of locations in 1999. The following distributions were found for each metal in 1992:

- **Barium** – Detected in all samples (n=4), with lakewide maximum and mean concentrations of 77 and 73 µg/L, respectively. All values exceeded the USEPA Tier II water quality criterion of 3.9 µg/L.

- **Cadmium** – Detected in about 4 percent of the samples collected in the lake, with maximum and mean concentrations of 2.9 and 1.0 µg/L, respectively, in the southern basin, and 3.1 and 1.1 µg/L, respectively, in the northern basin. Maximum concentrations in both basins exceeded the USEPA ambient water quality criterion (AWQC) final chronic value (FCV) of 1.3 µg/L and the USEPA and NYSDEC chronic aquatic values of 2.8 and 2.6 µg/L, respectively.

- **Chromium** – Detected in about 15 percent of samples, with maximum and mean concentrations of 5.3 and 1.7 µg/L, respectively, in the southern basin, and 4.1 and 1.5 µg/L, respectively, in the northern basin. All values were below the USEPA and NYSDEC chronic aquatic value of 94.8 µg/L.

- **Copper** – Detected in approximately 32 percent of samples, with maximum and mean concentrations of 51 and 2.5 µg/L, respectively, in the southern basin, and 3.0 and 1.3 µg/L, respectively, in the northern basin. The maximum concentration detected in the southern basin exceeded all USEPA and NYSDEC water quality criteria ranging from 12 to 18 µg/L.

- **Lead** – Detected in about 31 percent of samples, with maximum and mean concentrations of 18 and 1.1 µg/L, respectively, in the southern basin, and 7.7 and 0.9 µg/L, respectively, in the northern basin. Maximum concentrations in the southern and northern basins exceeded USEPA AWQC and NYSDEC chronic water quality values, ranging from 3.5 to5.2 µg/L.

- **Manganese** – Detected in about 98 percent of samples, with lakewide maximum and mean concentrations of 880 and 189 µg/L, respectively. All values exceeded the USEPA Tier II water quality criteria of 80 µg/L.

- **Nickel** – Detected in approximately 12 percent of samples, with maximum and mean concentrations of 15 and 3.7 µg/L, respectively, in the southern basin, and 5.3 and 3.1 µg/L, respectively, in the northern basin. All values were below the USEPA and NYSDEC chronic aquatic value of 67 µg/L.

- **Zinc** – Detected in about 96 percent of samples, with maximum and mean concentrations of 143 and 19 µg/L, respectively, in the southern basin, and 80 and 16 µg/L, respectively, in the northern basin. The maximum concentration in the southern basin was above the NYSDEC chronic water quality value of 107 µg/L and the USEPA AWQC-FCV water quality value of 135 µg/L.

- **Cyanide** – Detected in one of four samples, with a concentration of 171 µg/L in one 6 m sample from the northern basin. This concentration was above the USEPA and NYSDEC chronic and acute water quality values of 5.2 µg/L and 22 µg/L, respectively.

Detection frequencies and contaminant concentrations were generally higher in the southern basin than the northern basin. Chromium and nickel were also detected in the 1999 sampling (Appendix B, Table B-28).

Tributary Water and Metro Discharge

Concentrations of metals other than mercury in tributary water and Metro effluent in 1992 are presented for the three flow regimes in Table 8-1. Mean concentrations for each tributary and Metro are shown in Figures 8-5, 8-6, and 8-7. Under base-flow conditions, the following metal COCs were detected:

- **Cadmium** – Cadmium was undetected in all tributaries.

- **Chromium** – Chromium was detected in five tributaries and the lake outlet. Maximum concentrations were highest in Tributary 5A (28 µg/L), followed by the lake outlet (18 µg/L), and East Flume (11 µg/L).

- **Copper** – Maximum concentrations in seven tributaries were highest in the East Flume (15 µg/L), Metro effluent (12 µg/L), Tributary 5A (10 µg/L), and Onondaga Creek (6.4 µg/L).

- **Lead** – Lead concentrations were detected in six tributaries, and the maximum concentrations were highest in the East Flume (11 µg/L) and Ley Creek (7.4 µg/L).

- **Nickel** – Nickel was detected in five tributaries and the lake outlet. Maximum concentrations were highest in Sawmill Creek (17 µg/L), the lake outlet (10 µg/L), and Ley Creek (9.5 µg/L).

- **Zinc** – Zinc was detected in eight tributaries, with maximum concentrations in the East Flume (196 µg/L), Tributary 5A (59 µg/L), Onondaga Creek (51 µg/L), and Metro Outfall (42 µg/L).

Under intermediate-flow conditions, the following distributions were found for each metal:

- **Cadmium** – Cadmium was only detected once, in Tributary 5A (2.4 µg/L).

- **Chromium** – Chromium was detected in six tributaries. Maximum concentrations were highest in Tributary 5A (119 µg/L) and lower Ninemile Creek (12 µg/L).

- **Copper** – Copper was detected in seven tributaries. Maximum concentrations were highest in Tributary 5A (30 µg/L) and the East Flume (18 µg/L).

- **Lead** – Lead was detected in seven tributaries and the lake outlet. Maximum concentrations were highest in the East Flume (29 µg/L) and lower Ninemile Creek (22 µg/L).

- **Nickel** – Nickel was detected in four tributaries and the lake outlet. Maximum concentrations were highest in Tributary 5A (372 µg/L), lower Ninemile Creek (93 µg/L), and the lake outlet (115 µg/L).

- **Zinc** – Zinc was detected in eight tributaries and the lake outlet. Maximum concentrations were highest in the East Flume (179 µg/L), Tributary 5A (117 µg/L), Ninemile Creek (88 µg/L), and Onondaga Creek (85 µg/L).

Under high-flow conditions, the following distributions were found for each metal:

- **Cadmium** – Cadmium was detected in Bloody Brook (17 µg/L), Tributary 5A (3.2 µg/L), and lower Ninemile Creek (2.1 µg/L).

- **Chromium** – Chromium was detected in six tributaries. Maximum concentrations were highest in Tributary 5A (560 µg/L) and Ley Creek (19 µg/L).

- **Copper** – Copper was detected in nine tributaries and the lake outlet. Maximum concentrations were highest in Tributary 5A (125 µg/L), Ley Creek (58 µg/L), and Harbor Brook (48 µg/L).

- **Lead** – Lead was detected in all tributaries and the lake outlet. Maximum concentrations were highest in Ley Creek (95 µg/L), Tributary 5A (55 µg/L), Harbor Brook (63 µg/L), and Bloody Brook (44 µg/L).

- **Nickel** – Nickel was detected in four tributaries and the lake outlet. Maximum concentrations were highest in Bloody Brook (17 µg/L), Harbor Brook (17 µg/L), and the Metro outfall (15 µg/L).

- **Zinc** – Zinc was detected in all tributaries and the lake outlet. Maximum concentrations were highest in Tributary 5A (259 µg/L), Bloody Brook (201 µg/L), Harbor Brook (188 µg/L), and Onondaga Creek (182 µg/L).

8.1.1.3 Benzene, Toluene, Ethylbenzene, and Xylenes

Lake Water

No benzene, toluene, ethylbenzene, and xylene (BTEX) compounds were detected in the water of Onondaga Lake in 1992 (Table 8-2).

In 1999, benzene and xylenes were detected in two nearshore areas, and toluene was detected at one nearshore area. The Willis Lakeshore area (Station W50) had the highest benzene concentration at 6.3 µg/L and toluene at 0.16 µg/L. This lake water sample was collected near the groundwater plume originating at the Honeywell Willis Avenue site. The second set of detections were in the lake near Harbor Brook (Station W55), with benzene detected at 0.11 µg/L and total xylenes at 0.33 µg/L. The maximum concentration of xylenes (0.5 µg/L) in the lake was detected near Wastebeds 1 through 8 at Station W53. Ethylbenzene was not detected in the 1999 water samples.

Tributary Water and Metro Discharge

Under base-flow conditions, the following detections of BTEX compounds were found in 1992:

- **Benzene** – Benzene was detected in Harbor Brook (1 to 1.7 µg/L), the East Flume (1.5 µg/L), and Tributary 5A (6.9 to 34 µg/L).

- **Toluene** – Toluene was detected in the Metro effluent (3.1 µg/L), Harbor Brook (1 to 2.6 µg/L), the East Flume (2.5 µg/L), Geddes Brook (5 µg/L), and Tributary 5A (1.1 to 4.2 µg/L).

- **Xylenes** – Xylenes were detected in Harbor Brook (2 to 3.6 µg/L), the East Flume (1.4 µg/L), and Tributary 5A (1.1 to 2.2 µg/L).

There were no detections of ethylbenzene under base-flow conditions in 1992.

There were no detections of benzene, toluene, and xylenes under intermediate-flow conditions in 1992, but ethylbenzene was detected once in Ley Creek (1.7 µg/L).

Under high-flow conditions, the following detections of BTEX compounds were found in 1992:

- **Benzene** – Benzene was only detected in Tributary 5A (60 µg/L). The Semet Residue Ponds and/or the Willis Avenue sites are likely sources of this benzene.

- **Toluene** – Toluene was detected in Tributary 5A (5.1 µg/L) and Harbor Brook (2.1 µg/L).

- **Xylenes** – Xylenes were detected in the East Flume (1.6 to 1.8 µg/L), Tributary 5A (1.1 to 2.4 µg/L), and Harbor Brook (1.7 µg/L).

There were no detections of ethylbenzene under high-flow conditions in 1992.

8.1.1.4 Chlorinated Benzenes

Lake Water

Dichlorobenzenes and trichlorobenzenes were detected in only 1 of 98 water samples collected from Onondaga Lake in 1992 (Table 8-3). That sample was collected from the southern basin. Monochlorobenzene was not detected in any of the 98 samples collected from the lake in 1992.

In 1999, chlorinated benzenes were detected at various locations in nearshore habitat. 1,4-Dichlorobenzene was detected in all samples except the one near the boat ramp in Liverpool. 1,2-Dichlorobenzene was detected in three of 11 samples and monochlorobenzene was detected in only two of 11 samples. For all three compounds, the highest concentrations were detected at the Willis Lakeshore Area (Station W50) at 3.4 µg/L (1,4-dichlorobenzene), 3.2 µg/L (1,2-dichlorobenzene), and 12 µg/L (monochlorobenzene).

Tributary Water and Metro Discharge

Under base-flow conditions, monochlorobenzene (7.6 µg/L), 1,2-dichlorobenzene (2 to10 µg/L), and 1,4-dichlorobenzene (4.6 to13 µg/L) were detected only in the East Flume. Under intermediate flow conditions, 1,4-dichlorobenzene (2.0 to 2.3 µg/L) and 1,2,4-trichlorobenzene (0.9 to1.1 µg/L) were detected only in the East Flume. Under high-flow conditions, monochlorobenzene was not detected in any of the tributaries. Dichlorobenzenes were detected in Harbor Brook (1 µg/L of 1,2-dichlorobenzene and 5.3 to 7.2 µg/L of 1,4-dichlorobenzene) and the East Flume (1 to 5.2 µg/L of 1,2-dichlorobenzene and 7.8 to 20 µg/L of 1,4-dichlorobenzene). Trichlorobenzenes were detected only in the East Flume (1.9 to 2.7 µg/L of 1,2,4-trichlorobenzene).

8.1.1.5 Bis(2-ethylhexyl)Phthalate

The only detections of bis(2-ethylhexyl)phthalate were found in the 1999 lake water samples. The detections were at sample Stations W1 and W2. The four samples analyzed had detections with a

maximum concentration of 10 µg/L at Station W2 at a depth of 6 m, with the other three detections of 2 µg/L at Stations W1 and W2 at depths of either 6 or 12 m.

8.1.1.6 Chloride

Lake Water

Chloride concentrations in the epilimnion and hypolimnion of Onondaga Lake from April to November 1992 are presented in Figure 8-8. During stratification, chloride levels increased in the hypolimnion and decreased in the epilimnion. In the epilimnion, chloride concentrations ranged from 414 to 489 mg/L. The minimum values of chloride concentrations in the epilimnion were found between late June and mid-September, during the period of thermal stratification, with fairly constant concentrations during the remainder of the year.

Chloride concentrations in the hypolimnion were generally greater than the values found in the epilimnion, with a range of 469 to 525 mg/L. In the hypolimnion, the minimum chloride concentration was found in late May. Concentrations then increased continuously during the remainder of stratification. The maximum value was detected in early October. The mean chloride concentration for all samples collected in Onondaga Lake (i.e., epilimnion and hypolimnion) was 485 mg/L.

Tributary Water and Metro Discharge

Concentrations of chloride under base-flow conditions ranged from 46.4 mg/L to 1,411 mg/L. Mean chloride concentrations, shown in Figure 8-9, were highest in lower Ninemile Creek (1,250 mg/L), Onondaga Creek (890 mg/L), and Geddes Brook (730 mg/L), and were less than 500 mg/L in the remaining tributaries. Under intermediate flow conditions, chloride concentrations in the tributaries ranged from 35 to 1,042 mg/L and mean concentrations were greatest in Geddes Brook (682 mg/L) and lower Ninemile Creek (664 mg/L). Under high-flow conditions, chloride concentrations ranged from 36 to 844 mg/L, with the highest mean values occurring in lower Ninemile Creek (670 mg/L) and the East Flume (613 mg/L).

Chloride concentrations and loading (calculated for the USGS Lakeland Station) in Ninemile Creek have been monitored by Honeywell on a quarterly basis since 1989.

8.1.1.7 Salinity

Lake Water

The major components of salinity in the lake are calcium, chloride, and sodium, and minor contributors to lake salinity are magnesium, potassium, bicarbonate (HCO_3-), and sulfate (SO_4) (Effler et al., 1996). Between 1968 and 1990, the volume-weighted salinity was between 2.5 and 3.5 parts per thousand (ppt) (Effler et al., 1996). Relative contributions of calcium, sodium, and chloride decreased markedly with the

closure of the soda ash/chlor-alkali facility in 1986, resulting in a drop in the typical salinity of 3.3 ppt in 1981 to 1.1 ppt today (Effler et al., 1996; Onondaga Lake Partnership [OLP], 2002). This value is still an order-of-magnitude greater than the average world river salinity (0.11 ppt) and is several times higher than salinity levels in Otisco Lake (0.25 ppt), whose drainage basin is also within the Limestone Belt of central New York State (Figure 3-8).The salinity level is still artificially high because Solvay waste constituents continue to be released to Onondaga Lake and some of its tributaries from the Solvay Wastebeds located along the lakeshore and Ninemile Creek.

High salinity has altered the biological diversity of Onondaga Lake. Species diversity was reported to be heavily reduced at salinity levels found in Onondaga Lake through 1985 (Remane and Scheiper, 1971). High levels of salinity influenced the phytoplankton, zooplankton, and macrophytes (see Chapter 9, Section 9.1). The reestablishment of some salinity-intolerant zooplankton populations in Onondaga Lake has occurred since the closure of the soda ash/chlor-alkali facility (e.g., Hairston et al., 1999).

The high salinity of Onondaga Lake also affects thermodynamic properties of the lake and the adjoining portions of the Seneca River. Density stratification in the lake has been altered by ion-contaminated inflows to the lake, so that the temperature of maximum density shifted about 0.8°C (1.4°F) lower during winter than previously; the depression has been about 0.3°C (0.5°F) lower since closure (Effler et al., 1996).

Tributary Water and Metro Discharge

Under base-flow conditions, calcium concentrations ranged from 79,600 to 580,000 µg/L, with mean concentrations greatest in lower Ninemile Creek (518,300 µg/L) followed by Geddes Brook (404,200 µg/L). Under intermediate-flow conditions, calcium concentrations ranged from 71,400 to 348,000 µg/L, with lower Ninemile Creek and Geddes Brook averaging 301,600 and 304,000 µg/L, respectively. Calcium concentrations decreased under high-flow conditions in the tributaries, ranging from 54,200 to 383,000 µg/L. Mean high-flow concentrations at lower Ninemile Creek (253,700 µg/L) and Geddes Brook (230,800 µg/L) are lower than the corresponding values under base-flow and intermediate-flow conditions.

8.1.1.8 Nitrogen and Phosphorus

Lake Water

Concentrations of total ammonia, free ammonia, and nitrate (NO_3) that are maintained in the epilimnion throughout productive months and the concentrations of nitrite (NO_2) that develop in the epilimnion are unusually high in Onondaga Lake, compared to concentrations reported for other stratifying lakes in the literature (Effler et al., 1996). The summed concentration of total ammonia and nitrate have continuously exceeded levels associated with limitation of phytoplankton growth, and concentrations of free ammonia and nitrite in the epilimnion routinely exceed the NYSDEC standard of 0.1 mg/L nitrite for warm-water fish to protect non-salmonid (as well as salmonid) fish.

The concentrations of nitrogen species in the epilimnion and hypolimnion of Onondaga Lake from April to November 1992 are presented in Figure 8-10. These figures show increased concentrations of nitrate and nitrite in the epilimnion during the summer and early fall and decreases in these compounds in the hypolimnion during this period. The lowest concentrations of ammonia were found in the epilimnion from July to August.

More recent sampling of stressors in lake water has been conducted by Onondaga County. Figure 8-11 shows ammonia concentrations in Onondaga Lake measured from 1997 to 2001 as compared to the NYSDEC chronic water quality standard. The standard for unionized ammonia (as NH_3) is dependent on pH and temperature for different classes and specifications (NYSDEC, 1998). The state standard was consistently exceeded in the epilimnion and hypolimnion.

Concentrations of nitrite in Onondaga Lake are shown in Figure 8-12 for all epilimnion depths (0 to 9 m). Data from 2000 and the first half of 2001 indicate an improvement in water quality during the first half of the year, with exceedances of the NYSDEC standard of 0.1 mg/L for warm-water fish seen later in the year (i.e., summer and fall).

Figure 8-13 presents concentrations of phosphorus in Onondaga Lake in 1992. The most noticeable trend was the increasing concentrations of phosphorus in the hypolimnion during the period of stratification. The decrease in total phosphorous during the summer in the epilimnion is consistent with Effler et al. (1996). The NYSDEC guidance value of 20 µg/L (NYSDEC, 1998) was exceeded in the hypolimnion during the summer months in 1992. From 1997 to 2001, phosphorus concentrations were also elevated in the epilimnion samples (3 m depth based on human exposure), with almost all measurements above the guidance value (Figure 8-14). However, exceedances of an aesthetic effects guidance value are considered to have minimal impact on fish and wildlife in the area.

Tributary Water and Metro Discharge

The only nutrient form of nitrogen or phosphorus that was measured in tributaries and point sources in 1992 was ammonia (phosphorus was not measured). Mean concentrations of ammonia in the tributaries and Metro effluent under the three flow conditions are depicted in Figure 8-15.

Under base-flow conditions, maximum ammonia concentrations were highest in the Metro effluent (14 mg/L) followed by East Flume (4.2 mg/L), the lake outlet (2.5 mg/L), and Geddes Brook (1.6 mg/L), and were less than 1 mg/L in the remaining tributaries. Under intermediate flow conditions, maximum ammonia concentrations were highest in the Metro effluent (22 mg/L), followed by the East Flume (5.1 mg/L) and the lake outlet (3.0 mg/L), and were less than 3 mg/L elsewhere. Under high-flow conditions, maximum ammonia concentrations were also highest in the Metro effluent (15 mg/L), followed by the East Flume (5.0 mg/L) and the lake outlet (3.0 mg/L), and were 2 mg/L or less in the remaining tributaries.

8.1.1.9 Sulfide

The concentrations of sulfide in the hypolimnion of Onondaga Lake from April to November 1992 are presented in Figure 8-16. At depths of both 14 and 18 m, sulfide concentrations became measurable in early June to mid-July with the onset of stratification, increased substantially throughout the summer to maximum concentrations in late September, and then declined at 18 m after fall turnover. Sulfide concentrations were also elevated in the hypolimnion from 1997 to 2001 (Figure 8-17), but only depths of 12 to 18 m were sampled during this period.

8.1.1.10 Dissolved Oxygen

Lake Water

The concentrations of DO in the epilimnion and hypolimnion of Onondaga Lake from April to November 1992 are presented in Figure 8-18 and are indicative of a highly eutrophic lake. The ionic waste discharge of the chlor-alkali facility exacerbated the lake's problem of limited oxygen resources through alteration of the system's density stratification/mixing regime (Effler et al., 1996). Density differences have been reduced, but not eliminated, following closure of the facility (Owens and Effler, 1996). Prolonged stratification extends the period of anoxia in the lake.

The depth of anoxia (no oxygen) is presented from April to November 1992 in Figure 8-18. In the epilimnion, DO concentrations declined from 14 mg/L in April to 6 mg/L in early October. In the hypolimnion, DO concentrations rapidly declined from 13 mg/L in April to less than 0.5 mg/L in mid-June after the onset of stratification. Anoxic conditions remained in the hypolimnion until fall turnover. The depth of anoxia became established at 18 m in early June with the onset of stratification, rose to a depth of 9 to 12 m during the period of stratification, and then disappeared after fall turnover.

Dissolved oxygen concentrations sampled in 1997 to 2001 by Onondaga County have regularly failed to meet the NYSDEC standard of 4 mg/L in Onondaga Lake (Figure 8-20). Low DO levels were seen primarily at deeper depths. The failure to meet the DO standard only occurred at depths of 3 m or less in the northern basin during late October 1997 and 1998, around the time of fall turnover. The DO standard was always met at depths less than 3 m in the southern basin. Samples from later than October are not available for years other than 1997 and 1998.

Tributary Water and Metro Discharge

Mean DO concentrations in the tributaries and Metro effluent are presented for base-flow, intermediate-flow, and high-flow conditions in Figure 8-19. Under base-flow conditions, mean DO concentrations were lowest in the East Flume (5.6 mg/L) and were higher than 7.2 mg/L in the remaining tributaries. Under intermediate-flow conditions, mean DO in the East Flume was 5.9 mg/L and was higher than 7.4 mg/L elsewhere. Under high-flow conditions, mean DO concentrations were again lowest at the East Flume (5.7 mg/L) and were higher than 8.6 mg/L in the other tributaries sampled.

8.1.1.11 Water Transparency

The Secchi disk depths measuring visibility in Onondaga Lake from April to November 1992 are presented in Figure 8-21. Secchi disk depth generally varied from 1 to 2 m between April and early October and then increased to almost 6 m by the end of November, following fall turnover.

8.1.2 Distribution of Chemicals and Stressors of Concern in Onondaga Lake Surface Sediments

In this section, the distribution of COCs/SOCs in the surface sediments of Onondaga Lake is described. Detailed summaries of COC/SOC concentrations in sediments and exceedances of sediment quality values are presented in Appendices E (NYSDEC screening values) and F (site-specific sediment effect concentrations [SECs] calculated by TAMS) of this report. Chapter 5 of the RI (TAMS, 2002b) provides a detailed characterization of lake sediment contamination (surface and subsurface).

The 1992 lake sediment sampling provided a characterization of contamination throughout the entire lake, whereas the 2000 (Phase 2A) sampling focused on areas with higher levels of contamination, such as the southwestern portion of the lake and the mouth of Ninemile Creek. Surface sediments collected in 1992 were collected from the top 2 cm of the sediment column; whereas in 2000, surface samples were collected from the top 15 cm of sediment. The 0 to 15 cm zone of surficial sediment represents a fuller range of potential exposure of macroinvertebrates (Larson, pers. comm., 1999a). Benthic macroinvertebrates, and in particular oligochaetes, are commonly known to turn over as much as the top 20 cm of sediment in lakes, re-exposing contaminated sediments or concentrating the contaminants from sediments and making them available to higher level trophic receptors.

8.1.2.1 Mercury

The concentrations of total mercury in the surface sediments of Onondaga Lake in 1992 and 2000 are presented in Figure 8-22. Mercury concentrations were elevated throughout most of the lake, with maximum and mean detected concentrations in 1992 of 69 mg/kg and 4.0 mg/kg. Most samples had concentrations of mercury greater than 2.0 mg/kg, with the highest concentrations of mercury (>10 mg/kg) found in the southwest corner of the lake between the East Flume and the Metro outfall. Mercury concentrations throughout most of the nearshore zone were less than 2.0 mg/kg, whereas concentrations in most of the deeper parts of the lake were between 2.0 and 5.0 mg/kg. Both maximum and mean values were above all sediment screening values (Chapter 4, Table 4-5).

The 2000 sampling effort focused on the southwestern portion of the lake and the area near the mouth of Ninemile Creek (Figure 8-22). The maximum concentration of 78 mg/kg was detected offshore from the East Flume outlet. In 2000, the highest levels of mercury were found between the East Flume and Harbor Brook, with high levels also detected near the mouth of Ninemile Creek. The mean detected concentration of the surface sediment samples was 8.0 mg/kg (compared to the 1992 mean of 4.0 mg/kg), which may result from the more directed sampling locations and the greater depth of the samples. Most samples

collected in 1992 and 2000 exceeded NYSDEC sediment screening values (Appendix E), with the majority of samples and the means exceeding the NYSDEC severe effect level (SEL) of 1.3 mg/kg.

Mercury associated with sediment particles may be available to the food chain for the following reasons:

- Particle-borne mercury serves to maintain dissolved-phase concentrations of mercury, both in the water column of the lake as well as in porewater in the sediments.

- Sediment mercury represents an important pathway for ecological exposure via benthic invertebrate activity on the lake bottom.

- Water column mercury, as maintained by the suspended particle load as well as by sediment resuspension, represents an alternate exposure route for fish via direct uptake through the gills.

- Resuspension may not be the only way in which the mercury reenters the water column. Dissolution at the sediment/water interface and porewater migration can also serve to release sediment-bound mercury to the water column.

8.1.2.2 Other Metals

The concentrations of metals other than mercury in the surface sediments of Onondaga Lake (0 to 2 cm for 1992 data and 0 to 15 cm for 2000 data) are presented in Figures 8-23 through 8-29 (figures are not provided for metals with limited data; e.g., antimony). The following distributions were found for each metal in surface sediments:

- **Antimony** – In 1992, antimony was analyzed in 19 sediment samples at the 0 to 2 cm interval. Detections ranged between 3.1 and 6.4 mg/kg, with a mean of 4.6 mg/kg. In 2000, antimony was detected in 44 of 85 sediment samples at the 0 to 15 cm interval, ranging between 0.3 and 5.4 mg/kg. Of these samples, all of the 1992 and eight of the 44 in 2000 exceeded the NYSDEC LEL of 2 mg/kg.

- **Arsenic** (Figure 8-23) – In 1992, the highest concentration of arsenic (11 mg/kg) was detected near the mouth of Tributary 5A. The mean lakewide concentration was 3 mg/kg. The 2000 sampling, concentrated along the southwestern shore of the lake, detected a maximum concentration of 47 mg/kg and a mean concentration of 6.1 mg/kg. The highest arsenic concentrations were identified near the Interstate 690 (I-690) storm drainage discharge location. Maximum values exceeded screening values, as did the 2000 mean.

- **Cadmium** (Figure 8-24) – Concentrations in 1992 throughout most of the nearshore zone of the lake were less than 2 mg/kg, whereas concentrations in most of the deeper parts of the lake were between 2 and 5 mg/kg. The highest concentrations of cadmium were found off Harbor Brook and Ley Creek and at several stations along the eastern and western shorelines. The maximum and mean concentrations of cadmium detected in the lake in 1992 were 14 and 2.5 mg/kg, respectively. In 2000, the maximum and mean concentrations detected were 15 and 2 mg/kg, with the highest values detected near the mouths of Ley Creek and the East Flume. Mean and maximum concentrations were above some of the sediment screening values.

- **Chromium** (Figure 8-25) – Concentrations in 1992 in the northern part of the lake generally were less than 50 mg/kg, whereas concentrations throughout most of the southern part of the lake were between 50 and 100 mg/kg. The highest concentrations of chromium (up to 1,990 mg/kg) were found off Harbor Brook, between Tributary 5A and the East Flume, and at stations off the western shoreline near Wastebeds 1 through 8. The mean 1992 concentration of chromium detected in the lake was 81 mg/kg. In 2000, more intensive sampling along the southwestern shore of the lake confirmed that the area between Tributary 5A and the East Flume had the highest concentrations of chromium in surface sediments of the lake (up to 4,180 mg/kg). The mean concentration of the 2000 samples was 225 mg/kg. Both maximum and mean concentrations for both sampling periods were above sediment screening values.

- **Copper** (Figure 8-26) – Concentrations in 1992 in the northern part of the lake generally were less than 50 mg/kg, whereas concentrations throughout most of the southern part of the lake were between 50 and 100 mg/kg. The highest concentrations of copper (up to 173 mg/kg) were found off Harbor Brook and Tributary 5A and the lakewide mean concentration was 44 mg/kg. The maximum concentration detected in 2000 was 366 mg/kg, with a mean concentration of 66 mg/kg. The highest concentrations were detected between Tributary 5A and the East Flume. Maximum and mean concentrations were above sediment screening values for both sampling periods.

- **Lead** (Figure 8-27) – Concentrations in 1992 in the northern part of the lake generally were less than 50 mg/kg, whereas concentrations throughout most of the southern part of the lake were between 50 and 100 mg/kg, yielding an overall mean concentration of 51 mg/kg in the lake. The highest concentrations of lead (up to 251 mg/kg) were along the shoreline between Tributary 5A and Ley Creek. In 2000, more focused sampling confirmed that the areas with the highest levels of contamination were located between Tributary 5A and Ley Creek. The maximum concentration of 750 mg/kg was located near the mouth of Harbor Brook, where

it may potentially enter Wetland SYW-19. The mean concentration detected in 2000 was 93 mg/kg. Both maximum and mean concentrations exceeded screening values.

- **Manganese** – In 1992, manganese was detected in all 19 sediment samples at the 0 to 2 cm interval and ranged between 93 and 508 mg/kg, with a mean of 278 mg/kg. In 2000, manganese was detected in all 85 sediment samples at the 0 to 15 cm interval, ranging between 107 and 1,190 mg/kg, with a mean of 334 mg/kg. Of these samples, only one of the 1992 and 11 of the 85 in 2000 exceeded the NYSDEC LEL of 460 mg/kg. All of these exceedances were located near the I-690 lakeshore area.

- **Nickel** (Figure 8-28) – Concentrations in 1992 throughout the nearshore zone of the lake were generally less than 20 mg/kg, whereas concentrations throughout the deeper parts of the lake generally were between 20 and 50 mg/kg. The highest concentrations of nickel (up to 650 mg/kg) were found between Tributary 5A and the East Flume. The mean concentration detected in 1992 was 28 mg/kg. The 2000 sampling showed a maximum concentration of 1,670 mg/kg, detected near the mouth of Tributary 5A, and a mean concentration of 84 mg/kg. Maximum and mean concentrations during both periods exceeded screening values.

- **Selenium** – In 1992, selenium was detected in 12 of the 19 sediment samples at the 0 to 2 cm interval and ranged between 0.3 and 1.1 mg/kg, with a mean of 0.5 mg/kg. In 2000, selenium was detected in 49 of 85 sediment samples at the 0 to 15 cm interval ranging between 0.5 and 5.9 mg/kg, with a mean of 1.6 mg/kg. There are no selenium sediment screening criteria. This COC was retained based on it being a COC in fish.

- **Silver** – In 1992, silver was detected in 14 of the 19 sediment samples at the 0 to 2 cm interval and ranged between 0.6 and 5.1 mg/kg, with a mean of 1.5 mg/kg. In 2000, silver was detected in 53 of 85 of the 0 to 15 cm sediment samples analyzed, ranging between 0.1 and 6.1 mg/kg with a mean of 1.4 mg/kg. Of these detections, only seven of the 1992 and 20 of the 2000 samples exceeded the NYSDEC LEL of 1 mg/kg. In 1992 the exceedances were scattered throughout the lake, while in 2000 all of the exceedances were located in the southern basin.

- **Vanadium** – In 1992, vanadium was detected in 14 of the 19 sediment samples at the 0 to 2 cm interval and ranged between 0.5 and 101 mg/kg, with a mean of 12.7 mg/kg. In 2000, silver was detected in all 85 of the 0 to 15 cm sediment samples analyzed ranging between 1.8 and 319 mg/kg, with a mean of 24 mg/kg. There are no sediment screening criteria. This COC was retained based on it being a COC in fish.

- **Zinc** (Figure 8-29) – Concentrations in 1992 throughout the northern nearshore zone of the lake generally were less than 120 mg/kg, whereas concentrations throughout the deeper areas of the lake were between 120 and 150 mg/kg in the northern part and between 150 and 270 mg/kg in the southern part of the lake. The highest concentrations of zinc (>200 mg/kg, maximum of 276 mg/kg) were found off Harbor Brook and Ley Creek, and at discrete stations off the western shoreline in the southern basin. In 2000, the maximum concentration of 421 mg/kg was detected nearshore between Tributary 5A and the East Flume. The mean concentration was 122 mg/kg. Maximum concentrations in 1992 and 2000 and the mean concentration in 1992 were above sediment screening values.

8.1.2.3 Benzene, Toluene, Ethylbenzene, and Xylenes Compounds

The concentrations of BTEX compounds in the surface sediments of Onondaga Lake in 1992 and 2000 are presented in Figures 8-30 through 8-33. Concentrations of all four of these compounds were less than 50 µg/kg throughout much of the lake, however, elevated concentrations of all four compounds were found along the southwestern shoreline of the lake. The highest concentrations (>1,000 µg/kg) of all four compounds were detected off Harbor Brook and the East Flume, with increased concentrations stretching along the southern shoreline until north of Ley Creek. The maximum detected concentrations of benzene, toluene, ethylbenzene, and xylene were 5,700, 4,200, 1,300, and 13,000 µg/kg, respectively, and mean concentrations of these compounds were 440, 150, 660, and 3,400 µg/kg, respectively.

Higher concentrations of BTEX compounds were detected in 2000. The maximum detected concentrations of BTEX were 42,000, 8,300, 71,000, and 330,000 µg/kg, respectively, and mean concentrations of these compounds were 2,600, 1,900, 2,900, and 24,000 µg/kg, respectively.

Concentrations detected in both 1992 and 2000 exceeded sediment screening values. Most samples exceeding screening values were collected from the southwestern shoreline of the lake.

8.1.2.4 Chlorinated Benzenes

The concentrations of chlorinated benzenes (i.e., monochlorobenzene, dichlorobenzenes, trichlorobenzenes, and hexachlorobenzene) in the surface sediments of Onondaga Lake in 1992 and 2000 are presented in Figures 8-34, 8-35, 8-36, and 8-37. Concentrations of all of these compounds in 1992 were less than 100 µg/kg throughout most of the lake. However, concentrations of all of these compounds were sharply elevated along the southwestern shoreline of the lake. The highest concentrations (>1,000 µg/kg) were found in the area between Harbor Brook and the I-690 outfalls, which are located at the approximate border of the Semet Residue Ponds and the Willis Avenue sites. Maximum detected concentrations of monochlorobenzene, dichlorobenzenes, trichlorobenzenes, and hexachlorobenzene in 1992 were 43,000, 22,800, 4,200, and 1,200 µg/kg, respectively, and mean concentrations of these compounds were 2,700, 2,400, 1,100, and 63 µg/kg, respectively.

Samples collected in 2000 exhibited higher concentrations of contaminants with maximum concentrations of monochlorobenzene, dichlorobenzenes, trichlorobenzenes, and hexachlorobenzene of 1,000,000, 239,000, 35,000, and 6,750 µg/kg, respectively, and mean concentrations of these compounds were 34,700, 16,500, 6,800, and 180 µg/kg, respectively. Concentrations detected in both 1992 and 2000 exceeded sediment screening values. Benthic acute screening criteria for monochlorobenzene and dichlorobenzenes were exceeded at several locations along the southwestern shore.

8.1.2.5 Total Polychlorinated Biphenyls

The concentrations of total PCBs in the surface sediments of Onondaga Lake in 1992 (0 to 2 cm) and 2000 (0 to 15 cm) are presented in Figure 8-38. Concentrations of PCBs were less than 500 µg/kg throughout most of the lake, but even sediment above 10 µg/kg exceeds sediment screening values. Elevated levels of PCBs were concentrated in the southern part of the lake. The highest concentrations (>500 µg/kg) in 1992 were found primarily in the nearshore zone between Tributary 5A and Ley Creek. The maximum detected concentration in 1992 was 2,100 µg/kg, with a mean concentration of 290 µg/kg.

In 2000, the maximum detected concentration was 21,000 µg/kg, with a mean concentration of 1,100 µg/kg. The maximum concentration was detected slightly offshore from the mouth of the East Flume. Mean and maximum concentrations of PCBs from both sampling events exceeded sediment screening values.

8.1.2.6 Polycyclic Aromatic Hydrocarbon Compounds

PAHs were divided into LPAHs (low molecular weight PAHs: fluorene, naphthalene and 2-methylnaphthalene) and HPAHs (high molecular weight PAHs: acenaphthene, acenaphthylene, anthracene, benz[a]anthracene, benzo[a]pyrene, benzo[b]fluoranthene, benzo[g,h,i]perylene, benzo[k]fluoranthene, chrysene, dibenz[a,h]anthracene, fluoranthene, indeno[1,2,3-cd]pyrene, phenanthrene, and pyrene), based on results of the principal component analysis (PCA) presented in the RI (see Appendix I and Chapter 6 of the RI) (TAMS, 2002b).

The concentrations of PAH compounds in the surface sediments of Onondaga Lake in 1992 and 2000 are presented in Figures 8-39 and 8-40. Concentrations of these compounds were less than 1,000 µg/kg throughout most of the lake. However, elevated concentrations of these compounds were found in the southern part of the lake. The highest concentrations of both LPAHs (>5,000 µg/kg) and HPAHs (>10,000 µg/kg) were found primarily in the nearshore zone between Tributary 5A and Ley Creek. The 2000 sampling also revealed elevated levels of LPAHs (>5,000 µg/kg) between Tributary 5A and Harbor Brook and along Wastebeds 1 through 8. As is true for LPAHs, the HPAHs are found at high concentrations (>10,000 µg/kg) throughout the southern basin. The highest concentrations occur off the Oil City shoreline region, as well as the Honeywell shoreline area. From 2 to 6 m, both areas remain as centers of contamination. Individual and total PAHs exceeded sediment screening values in 1992 and 2000. The maximum surface concentration of naphthalene (26,000,000 µg/kg at 0 to 5 cm) was detected at Station S435 near the Semet Residue Ponds site.

8.1.2.7 Dioxins and Furans

Dioxins and furans were analyzed in lake sediments in 2000 and are presented in Figure 8-41 as avian toxicity equivalence quotients (TEQs), which are more conservative than the corresponding mammalian TEQs. The maximum avian TEQ in surface sediments (0 to15 cm) of 524 ng/kg was found at Station S346 near the East Flume. The mean and minimum avian TEQs were 117 and 4.7 ng/kg, respectively. The maximum mammalian TEQ was also found at Station S346 at 165 ng/kg, and the mean and minimum mammalian TEQs were 44 and 1.4 ng/kg, respectively.

8.1.2.8 Other Sediment Chemicals of Concern

Six other organic compounds/groups of compounds were selected as sediment COCs for this BERA. The concentrations of these compounds in surface sediments are shown in Figures 8-42 to 8-47. The following distributions were found for each COC:

- **Phenol** (Figure 8-42) – In 1992, phenol was analyzed in 19 sediment samples from the 0 to 2 cm depth interval and was detected in two samples at a concentration of 45 µg/kg. In 2000, phenol was analyzed for in 84 surface samples and detected in 11 samples with a range of concentrations from 190 to 2,600 µg/kg. The maximum sediment concentration was found at Station S349 near the East Flume.

- **Dibenzofuran** (Figure 8-43) – In 1992, dibenzofuran was analyzed in 19 sediment samples from the 0 to 2 cm depth interval and was detected in five of those samples, with concentrations ranging from 18 to 1,300 µg/kg. In 2000, dibenzofuran was analyzed in 84 samples and detected in 18 samples, with a maximum concentration of 81,000 µg/kg at Station S313 near Harbor Brook.

- **DDT and Metabolites** (Figure 8-44) – In 1992, DDT and its metabolites were analyzed in 19 sediment samples from the 0 to 2 cm depth interval and were detected in four of those samples. The range of detections was from 7.4 to 47 µg/kg. In 2000, DDT and its metabolites were analyzed in 84 samples and detected in 54 samples, with a range of concentrations from 1.1 to 88 µg/kg. The maximum sediment concentration of DDT in the 0 to15 cm depth interval was found at Station S313 near Harbor Brook.

- **Chlordane** (Figure 8-45) – In 1992, chlordane was analyzed in 19 sediment samples from the 0 to 2 cm depth interval and was detected in one sample at a concentration of 5.1 µg/kg at Station S1 near the mouth of Harbor Brook. In 2000, chlordane was analyzed in 84 samples and detected in 26 samples, with a

range of concentrations from 1.1 to 50 µg/kg. The maximum sediment concentration of chlordane in the 0 to15 cm depth interval was found at Station S314 near Harbor Brook.

- **Dieldrin** (Figure 8-46) – In 1992, dieldrin was not detected in surface sediment samples. In 2000, dieldrin was analyzed in 84 samples and detected in 26 samples, with concentrations ranging from 1.3 to 36 µg/kg. The maximum sediment concentration of dieldrin in the 0 to 15 cm depth interval was found at Station S338 near Tributary 5A.

- **Heptachlor and Heptachlor Epoxide** (Figure 8-47) – In 1992, heptachlor and heptachlor epoxide were analyzed in 19 sediment samples from the 0 to 2 cm depth interval and detected in none of those samples. In 2000, heptachlor and heptachlor epoxide were analyzed in 84 samples and detected in 28 samples, with a range of concentrations from 0.8 to 52 µg/kg. The maximum sediment concentration of heptachlor and heptachlor epoxide in the 0 to15 cm depth interval was found at Station S314 near Harbor Brook.

8.1.2.9 Calcium and Oncolites

The concentrations of calcium carbonate in the surface sediments of Onondaga Lake in 1992 are presented in Figure 8-48. Quantities of oncolites were estimated by determining the volume (in mL) of oncolites retained by a sieve with mesh size of 2 mm (i.e., gravel size) for the sieved fraction of all 0.06-m^2 benthic macroinvertebrate samples. No oncolite survey was performed in 2000. The following distributions were found:

- **Calcium** – Concentrations of calcium carbonate were generally greater than 60 percent throughout most of the nearshore zone and were generally less than 60 percent throughout most of the deeper parts of the lake. The highest calcium carbonate concentrations (>80 percent) were found in the nearshore zone off Ley Creek and Tributary 5A, along much of the northwestern and northeastern shorelines, and at several stations off the eastern and western shorelines.

- **Oncolites** – The distribution of oncolites was determined only for the nearshore zone because they are not found in the deeper parts of the lake (possibly because the degree of calcite oversaturation is greater in the epilimnion than the hypolimnion). In general, the distribution of oncolites closely corresponded to the distribution of calcium carbonate, of which they are composed. The lowest concentrations (<100 mL/0.06 sq m) were found between Tributary 5A and Ley Creek and along sections of the eastern and western shorelines. The highest concentrations (>300 mL/0.06 sq m) were found along most of the northwestern and northeastern shorelines and in small areas off Ley Creek and Tributary 5A.

Sandy material was mixed in with the oncolites in much of the high concentration zone, while silts and soft fine-grained sediments composed mostly of calcium carbonate occurred primarily on the southwestern shoreline and in the area just north of the wastebeds.

8.1.2.10 Sediment Porewater Characterization

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

The samples in 2000 were sectioned at 4 cm intervals for the top 8 cm and then one more 4 cm interval was collected at depths ranging from 30 to 120 cm, for a total of three intervals per core. These samples were analyzed for total mercury, methylmercury, and other analytes in porewater and solids. Four of the 2000 cores (from Stations S305, S344, S402, and S405) were from sediments above the thermocline, and three (from Stations S303, S354, and S355) were from below the thermocline. Three replicate cores were obtained at each station.

Mercury

Figures 8-49 through 8-55 show the distribution of total mercury and methylmercury in dissolved (filtered) form in porewater from the seven locations sampled in 2000 at Stations S303, S305, S344, S354, S355, S402, and S405, respectively. They also show the concentration in overlying water. In all cases, the porewater concentration in the 0 to 4 cm interval was higher than in the overlying water. The highest concentrations of total mercury and methylmercury occurred at Stations S344 and S402, located offshore of the East Flume within an area that contains Solvay waste material. There was no consistent pattern of concentration in porewater with depth probably because other parameters (e.g., porewater pH and sulfide concentration) influence mercury concentration independent of depth. There was also a poor correlation between concentration in porewater and concentration in the solids. Concentrations of mercury in sediments at these locations ranged from less than 1 to 66 mg/kg.

8.1.3 Distribution of Chemicals and Stressors of Concern in Wetland Soils/Sediments

The distribution of COCs in the wetland soils/sediments is discussed in this section. Sediment cores were collected in 2000 from four Onondaga Lake wetlands, including two wetlands located at the north end of the lake (Wetland SYW-6) and the mouth of Ninemile Creek (Wetland SYW-10), and two southern wetlands located at the mouth of Harbor Brook (Wetland SYW-19), and the mouth of Ley Creek

(Wetland SYW-12). Four stations were sampled in each wetland at two depth intervals (0 to 15 and 15 to 30 cm). In addition, SYW-6 was sampled by NYSDEC/TAMS in 2002 at five additional locations. Summary statistics for BERA COCs and screening against appropriate sediment criteria and ecological soil screening values are summarized in Appendix D of this BERA and figures can be found in Chapter 5 of the RI. In this BERA, only the 0 to15 cm interval is used in evaluating risks to receptors.

8.1.3.1 Mercury

Detections of total mercury in the 0 to 15 cm interval ranged from 0.05 to 25 mg/kg, with the maximum concentration occurring at Station S385 in Wetland SYW-19. Average total mercury concentrations in the surface sediment layers was 6.3 mg/kg. Total mercury concentrations in Wetland SYW-19 were significantly higher than values reported for the other wetlands. This wetland area is part of Honeywell's Wastebed B/Harbor Brook site and will be further evaluated as part of the RI/FS for that site. About 86 percent of the reported total mercury concentrations exceeded the LEL of 0.15 mg/kg, and 48 percent exceeded the SEL of 1.3 mg/kg. All samples exceeded the earthworm toxicity and USEPA Region 4 soil screening value of 0.1 mg/kg, except two samples (from Stations S387 and S390) located in Wetland SYW-12. About 76 percent of mercury concentrations exceeded the phytotoxicity screening value of 0.3 mg/kg and no samples exceeded the microbial toxicity screening value of 30 mg/kg.

8.1.3.2 Other Metals

Twelve metals in addition to mercury (i.e. arsenic, cadmium, chromium, copper, cyanide, lead, nickel, selenium, silver, thallium, vanadium, and zinc) were selected as plant COCs based on phytotoxicity (Efroymson et al., 1997a) (Chapter 6, Table 6-1). Additional metals (i.e., antimony, barium, iron, and manganese) based on microbial toxicity (Efroymson et al., 1997b), earthworm toxicity (Efroymson et al., 1997b), and USEPA Region 4 (1999) soil screening values were also selected as soil COCs (Table 6-1).

- **Antimony:** Concentrations ranged from 0.17 to 2.2 mg/kg. The maximum concentration at Station 3 in Wetland SYW-6, sampled in 2002, exceeded the NYSDEC LEL of 2 mg/kg. No station exceeded the NYSDEC SEL of 25 mg/kg. No station exceeded the USEPA Region 4 and phytotoxicity soil values of 3.5 mg/kg and 5 mg/kg, respectively.

- **Arsenic:** Concentrations ranged from 0.5 to 18 mg/kg, with the maximum concentration occurring at Station S379 in Wetland SYW-10, which exceeded the NYSDEC LEL of 6 mg/kg, along with three other stations. No station exceeded the NYSDEC SEL of 33 mg/kg. The maximum concentration at Station S379 in Wetland SYW-10 also exceeded the phytotoxicity and USEPA Region 4 soil screening value of 10 mg/kg. No station exceeded the earthworm or the microbial toxicity values of 60 and 100 mg/kg, respectively.

- **Barium:** Concentrations ranged from 31.4 to 390 mg/kg. About 24 percent of the samples (5 of 21) exceeded the USEPA Region 4 soil screening value of 165 mg/kg. No station exceeded the phytotoxicity soil value of 500 mg/kg.

- **Cadmium:** Concentrations ranged from 0.14 to 14 mg/kg, with the maximum concentration occurring at Station S375 in Wetland SYW-6, which exceeded the NYSDEC SEL of 9 mg/kg. About 86 percent of cadmium concentrations exceeded the LEL of 0.6 mg/kg. About 48 percent of cadmium concentrations exceeded the USEPA Region 4 screening value of 1.6 mg/kg and about 19 percent exceeded the phytotoxicity screening value of 4 mg/kg. No station exceeded the earthworm or the microbial toxicity values of 20 mg/kg.

- **Chromium:** Chromium concentrations ranged from 11 to 154 mg/kg, with 19 stations exceeding the LEL of 26 mg/kg and two stations exceeding the SEL of 110 mg/kg. All stations exceeded the USEPA Region 4 and earthworm toxicity screening value of 0.4 mg/kg and the phytotoxicity and microbial toxicity values of 1 and 10 mg/kg, respectively.

- **Copper:** Concentrations ranged from 9.5 to 167 mg/kg, with 19 of 21 samples exceeding the LEL of 16 mg/kg and two samples exceeding the SEL of 110 mg/kg. About 52 percent of the samples (11 of 21) exceeded the USEPA Region 4 screening value of 40 mg/kg. About 38 percent (8 out of 21 samples) exceeded the earthworm toxicity value of 50 mg/kg and two samples exceeded the phytotoxicity and microbial toxicity value of 100 mg/kg.

- **Cyanide:** Cyanide had only one detection of 5.4 mg/kg, which occurred at Station S376 in Wetland SYW-6. This detection exceeded the USEPA Region 4 screening value of 5 mg/kg.

- **Iron:** Concentrations ranged from 3,290 to 24,000 mg/kg. Two samples (Station S379 in Wetland SYW-10 and Station 2 in SYW-6) exceeded the NYSDEC LEL of 20,000 mg/kg. No station exceeded the NYSDEC SEL of 40,000 mg/kg. All detected concentrations exceeded the USEPA Region 4 and the microbial toxicity soil value of 200 mg/kg.

- **Lead:** Concentrations ranged from 18 to 259 mg/kg. The maximum concentration was found at Station S385 in Wetland SYW-19. The lead LEL of 31 mg/kg and the SEL of 110 mg/kg were exceeded in 15 and 6 samples out of 21 samples, respectively. About 62 percent of the samples (13 of 21) exceeded the USEPA Region 4 and phytotoxicity soil value of 50 mg/kg. No station exceeded the microbial toxicity or earthworm toxicity values of 900 and 500 mg/kg, respectively.

- **Manganese:** Concentrations ranged from 163 to 488 mg/kg. The maximum concentration, detected at Station S381 in Wetland SYW-10, exceeded the NYSDEC LEL of 460 mg/kg. No station exceeded the NYSDEC SEL of 1,100 mg/kg. All detected concentrations exceeded the USEPA Region 4 and the microbial toxicity soil value of 100 mg/kg, while no detected concentration exceeded the phytotoxicity soil value of 500 mg/kg.

- **Nickel:** Concentrations ranged from 5.5 to 64 mg/kg. The maximum concentration was found at Station S375 in Wetland SYW-6, which exceeded both the LEL of 16 mg/kg and the SEL of 50 mg/kg. A majority of stations exceeded the LEL. Fourteen sediment samples exceeded the NYSDEC LEL and one sample exceeded the SEL. About 33 percent of the samples (7 of 21) exceeded the USEPA Region 4 and phytotoxicity soil value of 30 mg/kg. No station exceeded the microbial toxicity or earthworm toxicity values of 90 and 200 mg/kg, respectively.

- **Selenium:** Concentrations ranged from 0.9 to 2.5 mg/kg, with the maximum concentration occurring at Station S375 in Wetland SYW-6. Only soil screening values were available for selenium. All detected concentrations exceeded the USEPA Region 4 screening value of 0.81 mg/kg. About 86 percent, or six out of the seven detected concentrations, exceeded the phytotoxicity value of 1 mg/kg. No station exceeded the microbial toxicity or earthworm toxicity values of 100 and 70 mg/kg, respectively.

- **Silver:** Concentrations ranged from 0.2 to 2.7 mg/kg, with the maximum concentration occurring at Station S389 in SYW-12, which exceeded both the LEL of 1.1 mg/kg and the SEL of 2.2 mg/kg. Four samples exceeded the NYSDEC LEL and only one sample exceeded the SEL. The maximum concentration also exceeded the USEPA Region 4 and phytotoxicity soil value of 2 mg/kg, but not the microbial toxicity value of 50 mg/kg.

- **Thallium:** Concentrations ranged from 1 to 2.5 mg/kg, with the maximum concentration occurring at Station S379 in Wetland SYW-10. Only soil screening values were available for thallium. Two of three detected concentrations exceeded the USEPA Region 4 and phytotoxicity soil value of 1 mg/kg.

- **Vanadium:** Concentrations ranged from 3.4 to 30.6 mg/kg, with the maximum concentration occurring at Station S379 in Wetland SYW-10. Only soil screening values were available for vanadium. All samples exceeded the USEPA Region 4 and phytotoxicity soil value of 2 mg/kg and two samples exceeded the microbial toxicity value of 20 mg/kg.

- **Zinc:** Concentrations of zinc ranged from 34 to 510 mg/kg, with the maximum concentration found at Station S376 in Wetland SYW-6. Concentrations exceeded the LEL of 120 mg/kg and SEL of 270 mg/kg in eight and two samples, respectively. All but one sample exceeded the USEPA Region 4 and phytotoxicity soil value of 50 mg/kg. About 67 percent of the samples (14 of 21) exceeded the microbial toxicity value of 100 mg/kg and about 19 percent of the samples (four of 21) exceeded the earthworm toxicity value of 200 mg/kg.

8.1.3.3 Organic Contaminants

Thirteen organic COCs/COC groups (i.e., benzene, chlorobenzene, dichlorobenzenes [sum], trichlorobenzenes [sum], hexachlorobenzene, phenol, total PAHs, aldrin, chlordanes, DDT and metabolites, dieldrin, hexachlorocyclohexanes, and total PCBs) were selected in soils. Plant screening values were not available for organic compounds. Summary statistics of wetland soils/sediments are presented in Appendix H, Tables H-15 to H-19.

- **Benzene:** Concentrations of benzene ranging from 3.9 to 60 µg/kg were detected in three of 12 wetland samples. The maximum concentration was found at Station S384 in Wetland SYW-19. Only this sample exceeded the NYSDEC chronic benthic sediment criterion of 28 µg/gOC when normalized by its respective organic carbon content. This sample also exceeded the USEPA Region 4 soil screening value of 50 µg/kg dry weight.

- **Chlorobenzene** (i.e., monochlorobenzene): Concentrations of chlorobenzene ranging from 2 to 600 µg/kg were found in the southern wetlands. Chlorobenzene was detected in five of 12 samples, with the maximum concentration found at Station S384 in Wetland SYW-19. Only three of the five stations exceeded the NYSDEC chronic benthic sediment criterion of 3.5 µg/gOC when normalized by the sample-specific organic carbon content. These three samples also exceeded the USEPA Region 4 soil screening value of 50 µg/kg dry weight. No station exceeded the earthworm toxicity value of 40,000 µg/kg dry weight.

- **Dichlorobenzenes:** Elevated levels of dichlorobenzenes (sum) were found in the wetlands around Onondaga Lake. Concentrations ranged from 54 to 14,700 µg/kg, with the highest concentrations in the four stations in Wetland SYW-19. When the concentrations of the sum of dichlorobenzenes are normalized by their respective organic carbon content, all values in Wetland SYW-19 exceeded the NYSDEC chronic benthic criterion of 12 µg/gOC. All detected concentrations exceeded the USEPA Region 4 soil screening value of 10 µg/kg dry weight.

- **Trichlorobenzenes:** Concentrations of 1,2,4,-trichlorobenzene ranging from 200 to 6,600 µg/kg were found in the wetlands around Onondaga Lake, with the

highest concentration at Wetland SYW-19. 1,2,4,-Trichlorobenzene was detected in five of 21 samples collected. Only four of the five stations exceeded the NYSDEC chronic benthic criterion of 91 µg/gOC for total trichlorobenzenes when normalized by organic carbon content. All detected concentrations exceeded the USEPA Region 4 soil screening value of 10 µg/kg dry weight.

- **Hexachlorobenzene:** Concentrations of hexachlorobenzene ranging from 2.7 to 5,400 µg/kg were found in the wetlands around Onondaga Lake, with the highest concentration at Wetland SYW-19. Hexachlorobenzene was detected in 12 of 21 samples, with four of the 12 exceeding the NYSDEC wildlife bioaccumulation sediment criterion of 12 µg/gOC. All detected concentrations exceeded the USEPA Region 4 soil screening value of 2.5 µg/kg dry weight, but none exceeded the microbial toxicity value of 1,000 mg/kg.

- **Phenol:** Concentrations of phenol ranging from 89 to 2,800 µg/kg were found in the wetlands around Onondaga Lake, with the highest concentration at Wetland SYW-19. Phenol was detected in three of 21 samples, with all three detections exceeding the NYSDEC chronic benthic sediment criterion of 0.5 µg/gOC. All detected concentrations exceeded the USEPA Region 4 soil screening value of 50 µg/kg dry weight, but no station exceeded the phytotoxicity, microbial toxicity or earthworm toxicity values of 70, 100, and 30 mg/kg, respectively.

- **Polycyclic Aromatic Hydrocarbons:** Maximum concentrations for total PAHs occurred at Stations S384 and S385 in Wetland SYW-19. The organic carbon-normalized concentrations for benz(a)anthracene indicate that ten samples exceeded the NYSDEC chronic benthic sediment criterion of 12 µg/gOC. Total PAHs were detected in 19 of 21 samples, with all but two detections exceeding the USEPA Region 4 soil screening value of 1,000 µg/kg dry weight.

- **Aldrin:** Concentrations of aldrin detected in the wetlands ranged from 21 to 45 µg/kg. Aldrin was detected only in Wetland SYW-19, where it was found in three of four samples. All three detections exceeded the Ontario Ministry of the Environment (OME) LEL of 2 µg/kg and the NYSDEC wildlife bioaccumulation sediment criterion. These three detections also exceeded the USEPA Region 4 soil screening value of 2.5 µg/kg.

- **Chlordanes**: Concentrations of chlordanes ranged from 2.3 to 30 µg/kg, with the maximum concentrations detected in Wetland SYW-19. Chlordane was detected in six of 16 samples, with all detections exceeding the NYSDEC wildlife bioaccumulation sediment criterion of 0.006 µg/gOC. There are no soil screening values for chlordane.

- **Dieldrin:** Concentrations of dieldrin ranged from 2.6 to 24 µg/kg, with elevated values detected in Wetlands SYW-19 and SYW-12. Dieldrin was detected in seven of 16 samples, with two detections exceeding the NYSDEC chronic benthic sediment criterion of 9 µg/gOC. All seven detected concentrations exceeded the USEPA Region 4 soil screening value of 0.5 µg/kg dry weight.

- **DDT and metabolites**: Concentrations of DDT and metabolites ranged from 1.2 to 51 µg/kg, with the maximum concentrations detected in Wetland SYW-19. DDT and metabolites were detected in 14 of 16 samples with exceedances of the NYSDEC chronic benthic sediment criterion of 1 µg/gOC. About 64 percent of the samples (nine of 14 detections) exceeded the USEPA Region 4 soil screening value of 2.5 µg/kg dry weight.

- **Hexachlorocyclohexanes**: Concentrations of hexachlorocyclohexanes ranged from 1.7 to 10 µg/kg, with the maximum concentrations detected in Wetland SYW-19. Hexachlorocyclohexanes were detected in five of 16 samples, with all exceeding the USEPA sediment quality benchmark of 0.37 µg/gOC. All detected samples also exceeded the USEPA Region 4 soil screening value of 1 µg/kg dry weight.

- **Polychlorinated Biphenyls:** Concentrations in the wetlands around Onondaga Lake were highest in Wetlands SYW-19 and SYW-12, where Aroclors 1242, 1254, and 1260 were detected. Total PCBs ranged between 28 and 1,100 µg/kg. PCBs were detected in all samples analyzed, with all detections exceeding the NYSDEC wildlife bioaccumulation sediment criterion of 1.4 µg/gOC. All detected values also exceeded the USEPA Region 4 soil screening value of 20 µg/kg dry weight. No station exceeded the phytotoxicity value of 40 mg/kg.

8.1.4 Distribution of Chemicals and Stressors of Concern in the Dredge Spoils Area

This section discusses the distribution of COCs in the dredge spoils area. Summary statistics for the dredge spoils area data are presented in Appendix D, Table D-50 and Appendix H, Table H-20, and further information, including figures, can be found in Appendix G1 of the RI (TAMS, 2002b). Surface and subsurface soil samples were collected by Honeywell in 2000 from dredged material disposal basins (Basins 1 to 4) located north of the mouth of Ninemile Creek, along the shoreline, in order to aid in characterizing the nature and extent of contamination of the dredged material and the fill placed on top of the spoils.

With the possible exception of Basin 4, these basins contain material dredged from the Ninemile Creek delta from 1966 to 1968. Eight sampling stations in the four basin areas were sampled at surface (0 to 60 cm) and subsurface (> 60 cm) intervals. The highest concentrations of many contaminants were found below 60 cm, making ecological receptor contact with these dredge spoils unlikely.

8.1.4.1 Mercury

Total mercury was detected at elevated concentrations within the dredge spoils area. Detections of total mercury in the top 60 cm ranged from 0.05 to 4 mg/kg, with higher concentrations (up to about 100 mg/kg) occurring in Basins 1 to 3 at subsurface intervals, confirming that contaminated sediments from the lake had been disposed of in this area. About 57 percent of the detected values (four of seven) exceeded the USEPA Region 4 and earthworm toxicity soil value of 0.1 mg/kg. Two detected values exceeded the phytotoxicity value of 0.3 mg/kg and none exceeded the microbial toxicity value of 30 mg/kg.

8.1.4.2 Other Metals

Some of the other metal COCs found in the dredge spoils soils are discussed in this section. Only soil values were used to screen the dredge spoils data (Chapter 5, Table 5-6), as the dredge spoils area is not regularly inundated by water.

- **Antimony:** Antimony concentrations ranged from 0.24 to 1.3 mg/kg in the 0 to 60 cm interval. The highest concentrations were seen in deeper samples not considered in this assessment. All detected values exceeded the USEPA Region 4 and phytotoxicity soil values of 3.5 and 5 mg/kg, respectively.

- **Arsenic:** Arsenic concentrations ranged from 3.2 to 8.4 mg/kg in the 0 to 60 cm interval. Higher concentrations were detected in deeper samples not considered in this assessment. No station exceeded the USEPA Region 4 and phytotoxicity soil value of 10 mg/kg.

- **Barium:** Barium concentrations ranged from 66.6 to 78.2 mg/kg in the 0 to 60 cm interval. Higher concentrations were seen in deeper samples not considered in this assessment. No station exceeded the USEPA Region 4 and phytotoxicity soil values of 165 and 500 mg/kg, respectively.

- **Cadmium:** In the top 60 cm, cadmium was undetected in all samples. In subsurface samples, concentrations ranged from 0.07 to 4.3 mg/kg. The highest concentration occurred at a depth of 230 to 250 cm in Basin 2, which is deeper than the surface soils considered in this assessment. However, the maximum detected concentration of 4.3 mg/kg exceeds both the USEPA Region 4 and phytotoxicity soil values of 1.6 and 4 mg/kg, respectively.

- **Chromium:** Chromium concentrations ranged from 12 to 29 mg/kg in the 0 to 60 cm interval. All detected concentrations exceeded the USEPA Region 4 and earthworm toxicity soil value of 0.4 mg/kg, the phytotoxicity soil value of 1 mg/kg, and the microbial toxicity soil value of 10 mg/kg.

- **Copper:** Copper concentrations ranged from 11 to 26 mg/kg in the 0 to 60 cm interval. No station exceeded the USEPA Region 4 or earthworm toxicity soil values of 40 and 50 mg/kg, respectively. The highest concentrations were seen in deeper samples from Basin 3 that were not considered in this assessment.

- **Cyanide:** In the top 60 cm, cyanide was undetected in all samples. In deeper samples, concentrations ranged from 0.9 to 1.4 mg/kg, with higher subsurface concentrations in Basin 3. No station exceeded the USEPA Region 4 soil screening value of 5 mg/kg.

- **Iron:** Iron concentrations ranged from 9,260 to 15,900 mg/kg in the 0 to 60 cm interval. All detected concentrations exceeded the USEPA Region 4 and the microbial toxicity soil value of 200 mg/kg.

- **Lead:** Concentrations ranged from 4 to 14 mg/kg in the 0 to 60 cm interval, with elevated concentrations below 220 cm at Basin 2. No detected concentrations exceeded the USEPA Region 4 and phytotoxicity soil value of 50 mg/kg.

- **Manganese:** Manganese concentrations ranged from 246 to 354 mg/kg in the 0 to 60 cm interval. All detected concentrations exceeded the USEPA Region 4 and the microbial toxicity soil value of 100 mg/kg, but none exceeded the phytotoxicity soil value of 500 mg/kg.

- **Nickel:** Nickel concentrations ranged from 9.3 to 13.8 mg/kg in the 0 to 60 cm interval. No detected concentration exceeded the USEPA Region 4 and phytotoxicity soil value of 30 mg/kg.

- **Selenium:** Concentrations ranged from 0.5 to 1.4 mg/kg. The highest surface soil concentration was detected in the 6 to 18 cm interval. About 75 percent of the samples (six of eight) exceeded the USEPA Region 4 soil screening value of 0.81 mg/kg. Only three detected samples exceeded the phytotoxicity soil value of 1 mg/kg. No detected samples exceeded the microbial toxicity or earthworm toxicity values of 100 and 70 mg/kg, respectively.

- **Silver:** In the top 60 cm, silver was undetected in all samples. In subsurface samples, concentrations ranged from 0.12 to 0.88 mg/kg. The highest concentration occurred at a depth of 219 to 244 cm, which is deeper than the surface soils considered in this assessment. However, the highest concentration did not exceed the USEPA Region 4 and phytotoxicity soil value of 2 mg/kg.

- **Thallium:** In the top 60 cm, thallium was undetected in all samples. Thallium was detected only in the subsurface interval of 60 to 120 cm at Basin 3, with a value

of 0.8 mg/kg. This concentration did not exceed the USEPA Region 4 and phytotoxicity soil value of 1 mg/kg.

- **Vanadium:** In the top 60 cm, vanadium concentrations ranged from 13.2 to 28.6 mg/kg, with the higher concentration occurring deeper than the surface soils considered in this assessment. All detected samples in the top 60 cm exceeded the USEPA Region 4 and phytotoxicity soil value of 2 mg/kg. Only two samples exceeded the microbial toxicity soil value of 20 mg/kg.

- **Zinc:** In the top 60 cm, zinc concentrations ranged from 29 to 50 mg/kg, with the highest values occurring below 200 cm at Basin 2. One sample exceeded the USEPA Region 4 and phytotoxicity soil value of 50 mg/kg, and none exceeded the microbial toxicity and earthworm toxicity soil values of 100 and 200 mg/kg, respectively.

8.1.4.3 Organic Contaminants

Concentrations of organic COCs detected in dredge spoils were as follows (note that VOCs, including chlorobenzene and BTEX compounds, were not analyzed in these dredge spoils samples):

- **Dichlorobenzenes:** In the 0 to 60 cm interval, dichlorobenzenes had only one detection, at 51 µg/kg for 1,2-dichlorobenzene. This concentration exceeded the USEPA Region 4 soil screening value of 10 µg/kg.

- **Hexachlorobenzene:** In the 0 to 60 cm interval, hexachlorobenzene was only detected once, at a concentration of 410 µg/kg. Maximum concentrations were found in the 60 to 150 cm interval at Station S438 in Basin 2. The detected sample in the 0 to 60 cm interval exceeded the USEPA Region 4 soil screening value of 2.5 µg/kg.

- **Polycyclic Aromatic Hydrocarbons:** In the 0 to 60 cm interval, total PAH concentrations ranged from 38 to 1,541 µg/kg. The maximum detected sample exceeded the USEPA Region 4 soil screening value of 1,000 µg/kg. The highest concentration of total PAHs was 208,000 µg/kg, detected in the 180 to 210 cm interval in Basin 4. All the individual PAH COCs had at least two values exceeding the NYSDEC allowable soil concentration.

- **Aldrin:** Aldrin was analyzed in one sample at the 61 to 149 cm range at Station S438 and detected at a concentration of 1.2 µg/kg. This concentration did not exceed the USEPA Region 4 soil screening value of 2.5 µg/kg.

- **Dieldrin:** Dieldrin was analyzed in one sample at the 61 to 149 cm range at Station S438 and detected at a concentration of 3.8 µg/kg. This concentration exceeded the USEPA Region 4 soil screening value of 0.5 µg/kg.

- **Polychlorinated Biphenyls:** In the 0 to 60 cm interval, only two samples had detections of PCBs: Aroclors 1254 (11 µg/kg) and 1260 (14 µg/kg). These concentrations did not exceed the USEPA Region 4 soil screening value of 20 µg/kg. PCB concentrations were highest in Basin 3 below the depth where receptors would have contact with soils (i.e., greater than 60 cm).

8.1.5 Biological Tissue Characterization

Biota have been sampled in Onondaga Lake and its tributaries as part of the Onondaga Lake investigation by Honeywell/PTI/Exponent, as well as by NYSDEC for monitoring purposes. The plankton, benthic invertebrates, and fish data collected and analyzed are presented in the following section. Further information can be found in Chapter 5 of the RI (TAMS, 2002b).

8.1.5.1 Phytoplankton and Zooplankton

Zooplankton and phytoplankton were collected at two stations in Onondaga Lake in 1992 and were analyzed for methylmercury and total mercury. Total mercury and methylmercury were detected in all samples analyzed. These concentrations were converted to a wet-weight (ww) basis in the original data report (PTI, 1993b).

Methylmercury concentrations for phytoplankton, on a wet-weight basis, ranged from 4.3 to 39 µg/kg, and total mercury concentrations ranged from 85 to 300 µg/kg. Methylmercury concentrations for zooplankton, on a wet-weight basis, ranged from 21 to 184 µg/kg in combined zooplankton assemblages and 165 to 390 µg/kg in daphnids. Total mercury concentrations for zooplankton, on a wet-weight basis, ranged from 23 to 247 µg/kg in assemblages and 247 to 994 µg/kg in daphnids.

8.1.5.2 Benthic Macroinvertebrates

Total mercury and methylmercury were analyzed in benthic macroinvertebrates collected in Onondaga Lake in 1992 and 2000. Benthic organisms sampled in 1992 consisted of chironomids and amphipods. Benthic organisms sampled in 2000 consisted of chironomids, amphipods, and oligochaetes.

Total mercury was detected in all 1992 samples collected, with concentrations ranging from 268 to 2,500 µg/kg dry weight (dw). The maximum detected total mercury concentration in 1992 was found in a chironomid sample from Station S013, in the southeast corner of the lake between Onondaga Creek and Ley Creek, in a chironomid sample.

Methylmercury was detected in all 1992 samples collected, with concentrations ranging from 66 to 670 µg/kg dw (10 to 100 µg/kg ww). The maximum detected methylmercury concentration was found in an amphipod sample from Station S04 in the southwestern corner of the lake, near the mouth of Harbor Brook.

Total mercury was detected in all but one of the 2000 samples, with concentrations ranging from 187 to 53,200 µg/kg dw. Methylmercury was detected in 35 of 41 samples, with concentrations ranging from 17 to 2,500 µg/kg dw. The maximum concentrations for both mercury and methylmercury were detected at Station S406 in a chironomid sample, in the in-lake waste deposit between the East Flume and Harbor Brook. Total mercury concentrations were also elevated at Station S344 (35,500 µg/kg dw in an oligochaete sample) and at Station S404 (20,300 µg/kg dw in an oligochaete). These two stations are also in the vicinity of the East Flume and the Honeywell in-lake waste deposit.

8.1.5.3 Fish

Fish were collected from Onondaga Lake and its tributaries by Honeywell in 1992 and 2000. Data collected by NYSDEC between 1992 and 2000 are used in this BERA. All fish concentrations are given on a wet-weight basis.

Mercury and Other Metals

Methylmercury concentrations in fish receptors sampled by Honeywell and NYSDEC in Onondaga Lake and its tributaries in 1992 and 2000 ranged from 0.03 to 3.2 mg/kg ww. Mercury was detected in every fish analyzed. Summary of exposure concentrations by species can be found in Appendix H, Tables H-7 to H-14. The breakdown by species is as follows:

- Bluegill: 0.05 to 0.9 mg/kg.
- Gizzard shad: 0.07 to 0.4 mg/kg.
- Carp: 0.04 to 0.8 mg/kg.
- Channel catfish: 0.3 to 1 mg/kg.
- White perch: 0.2 to 2 mg/kg.
- Smallmouth bass: 0.3 to 1.7 mg/kg.
- Largemouth bass: 0.2 to 1.4 mg/kg (only mercury measured).
- Walleye: 0.3 to 3.2 mg/kg.

Since nearly all of the mercury in fish tissue consists of methylmercury, total mercury rather than methylmercury, was analyzed in the samples collected by Honeywell/Exponent in 2000.

Mercury, antimony, arsenic, chromium, selenium, vanadium, and zinc were the seven metals selected as COCs for fish receptors. Concentrations of metals other than mercury in fish are summarized below.

- **Antimony:** was detected in one catfish sample at a concentration of 1.8 mg/kg and in one white perch at a concentration of 2.1 mg/kg. It was undetected in the remaining species.

- **Arsenic:** was detected in bluegill, carp, and smallmouth bass. Concentrations ranged as follows:

  - Bluegill: 0.6 to 0.7 mg/kg.
  - Carp: 0.7 to 2 mg/kg.
  - Smallmouth bass: 1.1 to 1.8 mg/kg.

- **Chromium**: was detected in all receptor species in which it was analyzed. Concentrations ranged as follows:

  - Bluegill: 3 to 14 mg/kg.
  - Carp: 1.2 to 4.8 mg/kg.
  - Channel catfish: 1.3 mg/kg.
  - White perch: 0.6 mg/kg.
  - Smallmouth bass: 0.7 mg/kg.
  - Walleye: 0.7 mg/kg.

- **Selenium:** was detected in all receptor species in which it was analyzed, except the walleye. Concentrations ranged as follows:

  - Bluegill: 4.7 to 7.8 mg/kg.
  - Carp: 3.2 to 9 mg/kg.
  - Channel catfish: 4.8 to 5.7 mg/kg.
  - White perch: 3.5 mg/kg.
  - Smallmouth bass: 4.5 mg/kg.

- **Vanadium:** was detected in all receptor species in which it was analyzed, except for walleye and white perch. Concentrations ranged as follows:

  - Bluegill: 0.5 to 1.2 mg/kg.
  - Carp: 0.3 to 1 mg/kg.
  - Channel catfish: 0.8 to 1.1 mg/kg.
  - Smallmouth bass: 0.2 to 0.8 mg/kg.

- **Zinc:** was detected in all receptor species in which it was analyzed, except for walleye. Concentrations ranged as follows:

  - Bluegill: 35 to 108 mg/kg.

- Carp: 48 to 425 mg/kg.
- Channel catfish: 20 to 74 mg/kg.
- White perch: 17 mg/kg.
- Smallmouth bass: 35 to 56 mg/kg.

Organic COCs

DDT and metabolites, endrin, total PCBs, and dioxins/furans were selected as COCs in fish. Concentrations of these contaminants in receptor species are discussed below.

- **DDT and Metabolites**: was detected in all receptor species in which it was analyzed. Concentrations ranged as follows:

  - Bluegill: 11 to 28 µg/kg.
  - Carp: 15 to 300 µg/kg.
  - Channel catfish: 25 to 600 µg/kg.
  - White perch: 5 to 100 µg/kg.
  - Smallmouth bass: 2 to 240 µg/kg.
  - Largemouth bass: 2 to 84 µg/kg.
  - Walleye: 19 to 200 µg/kg.

- **Endrin:** was detected in all receptor species in which it was analyzed, except for largemouth bass. Concentrations ranged as follows:

  - Bluegill: 1.6 to 5.5 µg/kg.
  - Carp: 5.4 to 36 µg/kg.
  - Channel catfish: 6.2 to 46 µg/kg.
  - White perch: 12 µg/kg.
  - Smallmouth bass: 8.5 to 33 µg/kg.
  - Walleye: 6.5 µg/kg.

Total PCBs

While many fish were analyzed for PCBs by Honeywell in 1992, these data were not included in the BERA due to data quality issues (see BERA Chapter 11, Section 11.1.3). PCBs were detected in all receptor species analyzed by NYSDEC from 1992 to 2000 and by Honeywell in 2000. Concentrations ranged as follows:

- Bluegill: 300 to 875 µg/kg.
- Carp: 500 to 9,800 µg/kg.
- Channel catfish: 780 to 6,000 µg/kg.
- White perch: 370 to 3,800 µg/kg.

- Smallmouth bass: 210 to 11,000 µg/kg.
- Largemouth bass: 75 to 2,800 µg/kg.
- Walleye: 660 to 7,800 µg/kg.

Dioxins and Furans

Dioxin and furan data from fish collected by Honeywell/Exponent in 2000 and by NYSDEC in 1992, 1997, and 1999 were used in this BERA. TEQs were calculated based on risks to fish (van den Berg et al., 1998). Dioxins/furans were detected in all receptor species in which they were analyzed. Concentrations of dioxin/furan TEQs ranged as follows:

- Bluegill: 20 to 127 ng/kg-lipid.
- Carp: 34 to 1,055 ng/kg-lipid.
- Channel catfish: 38 to 286 ng/kg-lipid.
- White perch: 50 to 285 ng/kg-lipid.
- Smallmouth bass: 26 to 165 ng/kg-lipid.
- Largemouth bass: 146 to 393 ng/kg-lipid.

8.2 Exposure Assessment

The assumptions and models used to predict the potential exposure of plants, fish, and wildlife (i.e., mammals and birds) to COCs associated with Onondaga Lake are described in this section. Site-specific chemical data characterizing the distribution of COCs in prey items (i.e., fish) and modeling the distribution in other prey items (aquatic invertebrates, terrestrial invertebrates, and small mammals) are discussed here. Food-web models used to estimate exposure of wildlife receptors to COCs are described, along with receptor life history characteristics and exposure assumptions. Receptors discussed in this section are surrogates for all species that inhabit or may inhabit Onondaga Lake.

8.2.1 Definition of Assessment Units

For the risk analyses, three specific assessment units were considered to represent the various receptor habitats. Sampling data from Onondaga Lake were divided into subsets depending on the location and habitat characteristics in order to evaluate receptor exposure. The three assessment units were defined as follows:

Onondaga Lake – The lake was divided into two areas, the pelagic and littoral areas. The pelagic assessment area encompasses the lake water column from the surface to the thermocline. All biota inhabiting this region were considered part of the pelagic assessment area. The littoral assessment area was considered to be the nearshore habitat from the edge of the lake to the point where the water depth exceeded 2 m. Sediments within the assessment area were considered to be from the surface to 15 cm for the evaluation of both incidental sediment ingestion and modeling uptake by biotic prey items. All biota within both the water column and the sediments were included in the Onondaga Lake assessment unit.

Wetlands – The wetland assessment unit focused on wetlands with hydrological connections to the lake. Data were collected from four of the wetlands that exist along the lake, including wetland areas along the northwest shoreline of the lake (Wetland SYW-6), and at the mouths of Ninemile Creek (Wetland SYW-10), Harbor Brook (Wetland SYW-19), and Ley Creek (Wetland SYW-12). All wetland areas were evaluated separately due to the locations and characteristics of each area, which suggest that contaminant concentrations and associated risks are likely to differ. The two northern basin wetlands (SYW-6 and SYW-10), both of which are on the west side of the lake between Ninemile Creek and the lake outlet, are dominated by floodplain forest, emergent swamps, and emergent vegetation (see Chapter 3, Section 3.2.4.1). These wetlands are expected to more closely resemble each other due to the general similarity of the area in which they are located. Wetland SYW-10 is being further investigated by Honeywell and NYSDEC as part of the Geddes Brook/Ninemile Creek RI/FS.

The southern Wetland SYW-12 is dominated by emergent vegetation as it approaches the lake, and along the shore of the lake it is a combination of floodplain forest and emergent marsh. Wetland SYW-19 is dominated by reedgrass and is severely contaminated with Honeywell COCs (note that this wetland is also being investigated as part of Honeywell's Wastebed B/Harbor Brook site).

Wetland receptors include plants, carnivorous birds, and insectivorous mammals. Avian and mammalian receptors were considered to have incidental soil exposure via ingestion up to a depth of 15 cm, based on the depth of the surface soils sampled in 2000. Likewise, soils from 0 to 15 cm were used for modeling the concentrations of contaminants in mammalian prey, as small mammals were considered to have the greatest exposure to this depth profile.

Dredge Spoils Area – The upland region represented in this BERA includes areas adjacent to the northwest shore where dredge spoils were used as reclamation fill in the late 1960s. Receptors in this unit include plants, carnivorous birds, and insectivorous mammals. Avian and mammalian receptors were considered to have incidental soil exposure via ingestion down to a depth of 107 cm. Likewise, all surface soils collected, ranging from 0 to 107 cm, were used for modeling the concentrations in mammalian prey. Drinking water was assumed to come from Onondaga Lake.

All receptors modeled for this assessment were determined to have foraging ranges within the Onondaga Lake area and were considered to be closed populations. Therefore, data from surrounding sites were not used directly in this BERA. However, other sites surrounding Onondaga Lake may represent additional sources of potential exposure. These other sites are discussed in Chapter 2 and Appendix G of this BERA.

8.2.2 Calculation of Exposure Point Concentrations

This section presents the methodology that was employed to calculate exposure point concentrations (EPCs) for the COCs in each exposure area. For each data set (representing a single chemical in one medium in an exposure area), a 95 percent upper confidence limit (UCL) on the mean concentration was calculated and compared to the maximum detected concentration for that chemical. The lower of the 95

percent UCL and the maximum detected value was used as the exposure point concentration, as recommended by USEPA (USEPA, 1992b).

Given a data set with no non-detect values, calculation of the UCL is straightforward. Depending on whether the data are normally or lognormally distributed, the UCL can be calculated using the Student's t-statistic or the H-statistic, respectively (USEPA, 1992b). In the presence of non-detects, however, the calculation is more complicated. The mean and standard deviation of the data (or the log-transformed data for lognormally distributed data sets) must be estimated in order to calculate the UCL.

Step 1: Assign Values to Non-Detect Data Points

USEPA guidance (1989) specifies that one-half the detection limit be used for non-detected results (i.e., data qualified "U") by the laboratory. Although the procedure is straightforward for inorganics (metals and cyanide), the determination of the appropriate value to use as the detection limit for organics (volatiles and semivolatiles) is open to question. However, based on USEPA Region 2 direction, all EPCs used in this assessment were based on non-detected results for organic and inorganic compounds being assigned a value of half the laboratory-reported detection limit (i.e., one-half the "U" value).

Step 2: Determine Data Distribution Type (Normal or Lognormal)

In accordance with USEPA (1992b]), the type of data distribution exhibited by a compound of concern in a medium (specifically, normal or lognormal) was evaluated based on a calculation of the W-test statistic developed by Shapiro and Wilks (1965) for sample sets containing more than 10 and less than 50 samples. The W-test was applied to each COC in each medium. This test is designed to examine the likelihood that the underlying population is normally distributed based on a random sample set. See Gilbert (1987) for details of the calculation method.

Values for W lie between 0 and 1. The closer the W value is to 1.0, the more normally distributed the data set is. The W-statistic was calculated for the data using the non-detect substitutions described in Step 1 and the log-transform of these distributions. If the W for the log-transformed data was greater than the W of the untransformed data, the distribution was assumed to be lognormal. Conversely, if the W for the untransformed data was greater than the W of the log-transformed data, the distribution was assumed to be normal. Where the W-statistic for the transformed and untransformed data was identical (such as when only two samples were collected), the distribution was assumed to be lognormal for the purpose of calculating the UCL.

Step 3: Calculate Exposure Point Concentrations

The EPCs used in this risk assessment were the arithmetic mean and an upper-bound estimate based on the lower value of the maximum detected value and the 95 percent UCL on the mean (see Appendix H). The term "95 percent UCL" is used throughout the remainder of the BERA to represent the upper-bound

estimate. The UCL was calculated from the summary statistics, depending on the form of the distribution that best fits the data.

For normally distributed data sets, the UCL on the mean is calculated as:

$$UCL = \overline{X} + t\left(\frac{s}{\sqrt{n}}\right)$$

where:

$\overline{X}$ = arithmetic mean of the sample data set for the compound of concern.

s = sample standard deviation of the sample data set for the compound of concern.

t = the Student's t-statistic for the 95 percent confidence interval for a one-tailed distribution; the t-statistic is a function of the number of samples collected.

n = number of samples in the data set.

For lognormally distributed data sets, the UCL on the mean is calculated as:

$$UCL = \exp\left[\overline{X} + 0.50s^2 + \frac{Hs}{\sqrt{n-1}}\right]$$

where:

$\overline{X}$ = arithmetic average of the natural log-transformed data.

s^2 = variance of the log-transformed data.

s = sample standard deviation of the log-transformed data.

H = H-statistic; the H value differs from the t-value because the formula is designed to estimate the UCL on the basis of the log-transformed data. H is a function of the standard deviation of the log-transformed data and the number of samples in the data set. H was taken from a standard table of calculated values (Gilbert,

1987) or linearly interpolated between values given in the table where necessary.

n = the number of samples in the data set.

When the data set contains less than ten sample results, the EPC was the maximum detected concentration.

Field Duplicates

Field duplicates were averaged based on USEPA Region 2 protocols as follows:

When averaging a set of data:

- Do not use rejected values. Do not include them in the sample count used to calculate the average.
- If the parameter was not detected, use half the detection limit.

When averaging data from a duplicate and its original sample:

- Detect versus detect: Average the two values and combine the qualifiers.
- Detect versus non-detect: Use only the detect. Note that the other value was not detected. Do not use half the detection limit.
- Detect versus rejected value: Use the detect. Discard the rejected value. Note the rejected value was not used.
- Non-detect versus non-detect: Average the non-detect values using half the detection limit.

Chemical of Concern Groups

Non-detected values were treated as observations at one-half the detection limit, with the exception of contaminants representing a group of COCs, such as PCBs and PAHs. Concentrations of PCBs were calculated as follows:

- When two or more Aroclors were detected, total PCBs was calculated to be the sum of all detected Aroclors.
- If only one Aroclor was detected, total PCBs was calculated to be the sum of the detected Aroclor concentration, plus half the detection of one other Aroclor (note:

all Aroclors [except 1221, which was not detected] have the same detection limit, so it is irrelevant which Aroclor is used).

- If no Aroclors were detected, total PCBs was calculated to be the sum of half the detection limit of each of two Aroclors.

Concentrations of PAHs were calculated as the sum of all detected PAHs. The sum of all detected compounds in a group was also used to calculate EPCs for DDT and metabolites, dichlorobenzenes, trichlorobenzenes, chlordanes, heptachlor/heptachlor epoxide, endosulfans, hexachlorocyclohexanes, and dioxins/furans.

8.2.3 Exposure Characterization for Aquatic Plants and Invertebrates

Risks to aquatic macrophytes and phytoplankton could not be assessed by comparing the mean and 95 UCL concentrations of COCs measured in surface water to water quality standards, criteria, and guidance, as there are no standards that specifically address the risk to for aquatic plants. However, narrative water quality standards (6 NYCRR Part 703.2), which regulate physical parameters and aesthetic conditions that impair the best use of the surface water but may not be physically measurable, were used to qualitatively evaluate water quality effects on aquatic macrophytes and phytoplankton. The effects of SOCs on aquatic plants were evaluated using site-specific literature.

Risks to zooplankton and benthic invertebrates were assessed by comparing the mean and 95 UCL concentrations of COCs measured in surface water and sediments (as appropriate based on the receptor) to water and sediment quality criteria for aquatic organisms. Sediment toxicity to benthic invertebrates was also evaluated using the results of laboratory toxicity tests conducted with Onondaga Lake sediments, as discussed in Chapter 9. The effects of SOCs were evaluated using site-specific literature. Aquatic invertebrates were considered to be part of the Onondaga Lake assessment unit.

8.2.4 Exposure Characterization for Terrestrial Plants

Plants were evaluated separately for the wetlands and the dredge spoils assessment units, since the two areas have different plant communities (see Appendix A and Chapter 3, Figure 3-4). The wetland areas are vegetated with floodplain forest, emergent vegetation, or reedgrass, or a combination of these covertypes. Wetland samples were collected from four wetlands connected to the lake: SYW-6, SYW-10, SYW-12, and SYW-19 (see Chapter 3, Section 3.2.4.1). Trees and other plants have colonized the dredge spoils area.

The same COCs were evaluated for both habitats, although concentrations of contaminants varied somewhat between locations. All of the COCs identified in the screening assessment as potentially posing a risk to plants are natural constituents of soils (i.e., inorganics), but this is partially due to the lack of plant screening values for organic compounds. Soil contaminated with heavy metals can produce apparently normal plants that may be unsafe for human or animal consumption (Kabata-Pendias and Pendias, 1992).

The primary potential for adverse impact to plants or plant communities from a COC is related to its uptake availability through the roots. For uptake to occur, a COC must be water-soluble and capable of being transported symplastically across the Casparian strip. The screening concentrations used are nominal concentrations of a soluble form (i.e., a highly bioavailable form) of the chemical added to soil (Efroymson et al., 1987a). Most metals in natural soils and contaminants of waste sites are in not readily bioavailable forms; therefore, risk estimates are considered conservative, as discussed in Chapter 11, Uncertainty Analysis.

8.2.5 Exposure Characterization for Fish

One method used to assess potential threats to fish in this BERA was to compare concentrations of COCs identified in the screening against TRVs selected from the literature. Site-specific literature was evaluated to determine the effects of SOCs. Fish were considered to be part of the Onondaga Lake assessment unit.

Forage fish, planktivorous fish, omnivorous fish, and piscivorous fish from Onondaga Lake were analyzed for contaminants. An overview of the biology, habitat selection, and feeding habits of fish receptors sampled in the lake are discussed in this section. Fish species selected as receptors represent a variety of habitats, sources of food, longevity, and size, all of which are likely to contribute to the amount of contamination that they come into contact with and subsequently accumulate.

Fish may be exposed to contaminants via direct uptake from the water column, uptake from sediments, and through feeding. Fish that feed extensively on fish eggs (e.g., white perch) contribute to a closed contaminant transfer loop between fish eggs and fish in Onondaga Lake.

Species profiles of fish receptors discussed in this BERA are presented below. The information presented is taken primarily from Werner (1980), Smith (1985), and Scott and Crossman (1973) or is based on professional judgment.

8.2.5.1 Bluegill (*Lepomis macrochirus*)

Bluegill are found in warm shallow waters in ponds, lakes, and in slow-moving bodies of water where there is adequate vegetation or other shelter in summer. In the winter they may retreat to deeper, cooler water where they tend to remain in colonial groups. Spawning occurs in late spring to mid-summer in colonial-style nests that are sometimes relatively dense. Growth is rapid, and bluegill can grow up to 25 to 28 cm in length, with ages up to 11 years.

Bluegill feed throughout the water column during the day, primarily in the morning and afternoon. They are omnivorous and eat a wide variety of organisms and, at times, significant amounts of plant material. Young bluegills feed on rotifer and copepod nauplii, while larger individuals eat insects and other larger particles.

8.2.5.2 Gizzard Shad (*Dorosoma cepedianum*)

Gizzard shad is predominantly a quiet-water fish, although it has been collected it in swift streams in the Genessee drainage in New York. It can tolerate high turbidity and relatively high salinities of 33 to 34 ppt, but most often is found in clear water. It is often found near the surface, and its young are common around weed beds. The young school during their first year and are sometimes found well upstream in small streams.

Newly hatched gizzard shad feed on protozoans and other zooplankton. After a few weeks the diet changes to include phytoplankton and algae. The gizzard shad is essentially a filter feeder. Lake Erie populations of gizzard shad may reach six years of age (five is more common) and lengths slightly greater than 38 cm (Scott and Crossman, 1973).

8.2.5.3 Carp *(Cyprinus carpio)*

Carp inhabit lakes, ponds, and larger streams and are most abundant where there is dense aquatic vegetation. Carp spawn in the spring and early summer when temperatures reach about 17°C (63°F) (Scott and Crossman, 1973). A female will lay from 36,000 to over two million eggs of approximately 1 mm in diameter. These adhesive eggs are deposited randomly in shallow waters, generally over aquatic vegetation. Growth is rapid after hatching, which occurs in about three to six days.

Carp are omnivorous bottom feeders and feed on filamentous algae and various benthic invertebrates, including snails, annelids, midge larvae, and crustaceans. In their feeding activities they often destroy vegetation by physically uprooting the plants, stirring up the bottom, and by making the water so turbid that sufficient light cannot reach the growing plants. Occasionally they may move up in the water column to feed on plant and animal materials. Carp may reach up to 18 cm in their first growing season and have been observed up to 86 kg in Lake Erie (Scott and Crossman, 1973). They are long-lived fish and 20-year-old fish are considered normal in North America.

8.2.5.4 Channel Catfish (*Ictalurus punctatus*)

The channel catfish occurs in larger streams, rivers, and lakes, where it is able to thrive in moderate currents over sandy to rocky substrate. It can tolerate relatively low oxygen levels and warm water, and has been found in waters with oxygen as low as 0.95 ppm and temperatures in excess of 32°C (90°F) (Smith, 1985). Channel catfish are not normally associated with aquatic vegetation as are other members of the catfish family that inhabit Onondaga Lake, such as brown bullhead (*Ameiurus nebulosus*) and yellow bullhead (*Ameiurus natalis*).

The channel catfish is a nocturnal feeder and depends heavily upon chemical senses to locate its food. The young feed largely on aquatic insects and other bottom-dwelling arthropods. When they reach about 100 mm standard length they become omnivorous, with fish making up a large part of their diet (Smith, 1985). Seeds and terrestrial animals, including birds, have been found in catfish stomachs. In southern ranges

channel catfish may reach ages over 20 years and sizes over 50 cm; however, in western Lake Erie, ages only extend to seven years, and larger fish are around 36 cm (Scott and Crossman, 1973).

8.2.5.5 White Perch (*Morone americana*)

White perch can tolerate a wide range of salinities from marine to fresh water. White perch have been known to migrate to shallows or to surface waters at night and offshore or to deeper waters during the day. They congregate in areas with DO of at least 6 mg/L (Seltzer-Hamilton, 1991) and are often found in rather turbid shallow areas where at times they form dense schools.

Young white perch feed heavily on small invertebrates, such as copepods, during their first two summers. *Gammarus*, chironomid larvae, and occasional *Cyathura* also become important foods as they grow (Smith, 1985). Fish eggs are an important food source in late spring and early summer (May through July). White perch more than 200 mm in length feed mostly on other fish (Setzler-Hamilton, 1991). Spawning takes place in the spring in waters between 14 and 16°C (58 and 60°F). Females have relatively large numbers (20,000 to 300,000) of small eggs (0.55 to 0.70 mm). White perch have an average life span of five to seven years and have been reported as old as 17 years. Growth rates vary widely and sizes over 33 cm and 3.8 kg have been documented (Scott and Crossman, 1973).

8.2.5.6 Smallmouth Bass (*Micropterus dolomieui*)

Smallmouth bass are found in streams and lakes. Although they tolerate a wide range of habitats, they tend to select cool waters with rocky or gravel substrate where there is shelter. Smallmouth bass spawn in May or early June. A mature female smallmouth bass may lay a total of 5,000 to 7,000 eggs in several nests that the males build and guard (Werner, 1980).

Juvenile smallmouth bass feed on plankton and invertebrates, switching to larger items as they grow. Smallmouth bass are opportunistic predators and feed on primarily on insects, crayfish, and fish, but will also feed on amphibians (e.g., tadpoles, frogs, and salamanders) and small animals, if available. In Lake Erie, smallmouth bass live up to nine years and reach sizes up to 38 cm (Scott and Crossman, 1973).

8.2.5.7 Largemouth Bass *(Micropterus salmoides)*

The largemouth bass is a relatively large, robust fish that has a tolerance for high temperatures and slight turbidity (Scott and Crossman, 1973). Largemouth bass occupy warm, weedy parts of lakes, ponds, and streams and show a low tolerance for low oxygen conditions. Largemouth bass mature at age five and spawn from late spring to mid-summer, in some cases as late as August. Male largemouth bass construct nests in sand and/or gravel substrates in areas of non-flowing clear water containing aquatic vegetation (Nack and Cook, 1986). Females produce 2,000 to 7,000 eggs per pound of body weight (Smith, 1985) and leave the nest after spawning.

Until they are about 5 cm in length, young-of-year (YOY) feed on plankton, insects and other invertebrates. As they get larger, their diet shifts to fish and other large items including almost anything that moves, including amphibians, reptiles and terrestrial species. Largemouth bass longer than 50 mm total length usually forage exclusively on fish. The largemouth bass represents a top predator in the aquatic food web, consuming primarily fish, such as gizzard shad, carp, bluntnose minnow (*Pimephales notatus*), golden shiner (*Notemigonus crysoleucas*), yellow perch (*Perca flavens*), pumpkinseed (*Lepomis gibbosus*), bluegill, and other largemouth bass (Scott and Crossman, 1973).

Largemouth bass take their food at the surface during morning and evening, in the water column during the day, and from the bottom at night. They feed by sight, often in schools, nearshore, and almost always close to vegetation. Feeding is restricted at water temperatures below 10°C (50°F) and decreases in winter and during spawning. Largemouth bass do not feed during spawning. In Lake Ontario largemouth bass live up to 13 years and reach sizes that average about 50 cm.

8.2.5.8 Walleye (*Stizostedion vitreum*)

Walleye occur in lakes and larger rivers and are active year-round. They generally swim near the bottom in loose aggregations and frequently move into shallows to feed at night. Spawning occurs in early spring over coarse gravel shoals in lakes, or over gravel and rocky bottoms in rivers or tributaries to the lakes the walleye inhabit. Eggs are 1.5 to 2 mm, with numbers from large females often as high as high as 500,000 (Scott and Crossman, 1973). An estimated 20 percent of the eggs survive to hatching under ideal conditions, and less than 5 percent is common under suboptimal conditions. To achieve a stable population only a small percent of those eggs surviving need to mature and reproduce (Werner, 1980).

Walleye are opportunistic predators. Upon hatching, walleye feed first on plankton crustaceans, but soon switch to insects and then to fry, including other walleye if food is scarce or there is crowding. By the time they are 8 cm long, walleye feed on fish and other larger items. Walleye live from 10 to 12 years in southern Canadian waters (including Lakes Erie and Ontario) and reach trophy sizes of between 12 to 19 pounds in Lake Ontario.

8.2.6 Exposure Characterization for Terrestrial Wildlife

Exposure of terrestrial wildlife to the COCs in Onondaga Lake was determined using a food-web modeling approach. Site-specific daily doses were estimated for each of the receptors based on their expected COC exposures resulting from modeled rates of contact with specific media. This approach allows for a direct comparison of exposure to toxicity in the characterization of the risk posed by the COC to receptor populations. Wildlife populations are defined as all individuals of a receptor species who may be exposed to COCs associated with Onondaga Lake water, sediment, soil, or biota.

Total exposure for receptor populations was determined through the summation of all pathways of exposure. It was assumed that the exposed receptor population is completely closed (i.e., no interactions with any population or location outside of the lake itself), and as such, dietary, drinking water, and

incidental sediment ingestion is derived from the appropriate assessment area of Onondaga Lake for their entire lifetime. Exposure to contaminated areas outside the Onondaga Lake area is discussed in the uncertainty section.

8.2.6.1 Food-Web Modeling

A deterministic risk assessment was performed to characterize risk to receptors from exposure to COCs. The exposure rate was predicted based on the mean and 95 percent UCL on the mean (or the maximum if less than the UCL) for COC concentrations measured in Onondaga Lake and was interpreted to be representative of exposed populations.

The general structure of the model used to estimate the exposure rate for a given contaminant by a wildlife receptor is as follows:

$$EED = \sum \left(IR_p \times [COC]_p + IR_w \times [COC]_w + IR_s \times [COC]_s \right)$$

where:

EED	=	estimated environmental dose (mg/kg body weight-day)
IR_p	=	receptor-specific prey intake rate (kg dry weight/kg body weight)
IR_w	=	receptor-specific water intake rate (L/kg body weight)
IR_s	=	receptor-specific incidental sediment intake rate (kg dry weight/kg body weight)
$[COC]_p$	=	COC concentrations in the receptors' prey (mg/kg dry weight)
$[COC]_w$	=	COC concentrations in the receptors' drinking water (mg/L)
$[COC]_s$	=	COC concentrations in the sediments or soils incidentally ingested (mg/kg dry weight)

Derivations of the parameters used to predict the exposure doses are discussed in the following sections.

8.2.6.2 Routes and Media of Exposure

The route of exposure is defined as the means by which a receptor may contact a contaminated medium. For this assessment, exposure was limited to ingestion because it was assumed to account for the majority of exposure to the COCs.

Based on the chemical properties of the COCs and the typical foraging behavior of the receptors, it was concluded that the primary routes of exposure of wildlife to COCs would be through: 1) ingestion of prey items (e.g., macroinvertebrates/insects, fish, small mammals), 2) ingestion of drinking water, and 3) the incidental ingestion of soil or sediment.

8.2.6.3 Wildlife Receptor Assessment Unit Association

Wildlife receptors were selected to represent species that inhabit or may inhabit Onondaga Lake. Birds selected were the tree swallow (*Tachycineta bicolor*), mallard duck (*Anas platyrhynchos*), belted kingfisher (*Ceryle alcyon*), great blue heron (*Ardea herodias*),osprey (*Pandion haliaetus*), and red-tailed hawk (*Buteo jamaicensis*). Mammals selected as receptors are the little brown bat (*Myotis lucifugus*), short-tailed shrew (*Blarina brevicauda*), mink (*Mustela vison*), and river otter (*Lutra canadensis*).

These receptors do not cover the entire range of species found around Onondaga Lake (see Chapter 3), but were selected to represent the species potentially at risk based on their exposure to specific prey items (e.g., piscivorous, insectivorous) and habitats associated with the lake. Receptors feeding on items with lower contaminant concentrations, such as herbivores (e.g., muskrat, deer mice), are at lower risk than receptors feeding on higher trophic level prey, and, therefore, this risk assessment is considered to be protective of them, as discussed in the preliminary conceptual model (Chapter 4, Section 4.1). The receptors associated with each of the assessment units (Section 8.2.1) are as follows:

- **Onondaga Lake – Pelagic Habitat**: The receptors expected to be at greatest risk in this habitat are those that forage within the water column of the open lake. There are no mammalian species indigenous to this region that utilize this habitat. However, the osprey does hunt in the pelagic zone, and therefore may be exposed to COCs in this region of Onondaga Lake. An intermediate case of exposure to the pelagic region was considered for the tree swallow and little brown bat, which feed predominantly on emergent insects. Prey items for these receptors were assumed to originate from anywhere within the entire lake. Other receptors in this unit include macrophytes, phytoplankton, zooplankton, and fish.

- **Littoral Habitat**: The receptors expected to be at greatest risk in this habitat are those that forage within the inshore zone of the lake and are dependent upon indigenous aquatic organisms as their primary food source. The terrestrial receptors considered most likely to be at risk are those that use the lake as a prey source, but are not expected to venture beyond the immediate shoreline in search of prey. The receptor species considered for this assessment unit are the mink, river otter, belted kingfisher, great blue heron, and mallard. Other receptors in this unit include macrophytes, benthic invertebrates, fish, and insectivorous birds and mammals.

- **Wetlands:** The receptors expected to be at greatest risk in this habitat are those that forage on insects and small mammals. These receptors are the short-tailed shrew (native insectivore) and red-tailed hawk (native top carnivore).

- **Dredge Spoils Area:** As in the wetlands habitat, the receptors expected to be at greatest risk in this habitat are those that forage on insects and small mammals. These receptors are the short-tailed shrew and red-tailed hawk.

8.2.6.4 Chemical of Concern Exposure from the Ingestion of Fish

Prey selection is a function of the receptor's size and method of hunting. Prey selection plays an important role in modeling exposure, because some prey sizes (e.g., large fish) may have higher concentrations of contaminants than others. For assessment purposes, prey selection was refined to provide greater confidence in the data sets gathered (i.e, increased sample size). The two parameters that often account for the much of the variation seen in contaminant concentrations in fish from a single location are species (feeding patterns, habitat) and age of the fish (longer period of bioaccumulation, change in feeding patterns). In general, the higher an organism is on the food chain and the older it is, the greater the concentration of bioaccumulative contaminants.

All the fish within the receptor prey size-selection range, for which COC concentration estimates were available, were used to predict the receptor's exposures. Fish consumed by wildlife receptors were divided into two size classes (3 to 18 cm and 18 to 60 cm) based on the available data and prey selection preferences of receptors.

Whole Fish and Fillet Data

Avian and mammalian receptors were assumed to consume whole fish. Therefore, ratios were developed to convert fillet concentrations to whole fish concentrations. Estimates of whole body COC concentrations were expressed on a dry weight basis in all exposure models to control for variations in water content between fish and between fish and sediment. The standardization also permitted direct application to ingestion rates that were determined in dry weight.

Small fish, such as bluegills, were generally analyzed as whole fish samples and contaminant body burdens could be used directly for ecological modeling. Omnivorous fish were analyzed as both whole fish and fillets. The majority of data available for piscivorous species is based on analyses of fish fillets.

Fish tissue concentrations used in this BERA were collected in 1992 and 2000 by Honeywell and between 1992 and 2000 by NYSDEC. Data from both Honeywell and NYSDEC were pooled into a single data set, which was then queried for information for evaluation of fish as receptors and size classes for use in food-web modeling (i.e., fish as prey of piscivorous receptors). NYSDEC has analyzed mainly fillets from species caught by anglers such as smallmouth bass, largemouth bass, white perch, and walleye, while

Honeywell's analyses were taken from a mix of species with various feeding habits. Therefore, Honeywell data were used to derive fillet to whole fish ratios.

Honeywell analyzed the fillet and remains from 11 fish (seven from Onondaga Lake and four from lower Ninemile Creek) in 2000 and used the data to develop regressions for estimating whole-fish concentrations from fillet concentrations. The 11 samples for which the fillet and remains were available consisted of two bluegill, two catfish, five carp, and two smallmouth bass. Although carp comprised about 45 percent of the fillet-remains data, it was only about 12 percent of the total Honeywell mercury samples and about 1 percent of the NYSDEC mercury samples. The regressions developed by Honeywell were considered inappropriate for use in this BERA based on the small number of samples used for the regression, the low correlation coefficients of some of the regression equations, and variability introduced by the use of several species with different lipid concentrations, feeding patterns/trophic levels, habitats, and other variables.

NYSDEC/TAMS developed conversion factors for contaminants with fillet and whole fish data for use in this BERA (Table 8-4). Whole-body concentrations were determined from the separate analyses of fillet and remains using the following formula:

$$[COC]_{\text{Whole Body}} = \frac{[COC]_{\text{fillet}} \times \text{Mass}_{\text{fillet}} + [COC]_{\text{Remain}} \times \text{Mass}_{\text{Remain}}}{\text{Mass}_{\text{fillet}} + \text{Mass}_{\text{Remain}}}$$

Fillet to whole fish conversion factors were used for mercury, total PCBs, DDT and metabolites, and dioxins/furans. A default value of one was used for other organic COCs, due to small sample sizes or low detection rates (e.g., hexachlorocyclohexane), and for all metals exclusive of mercury (e.g., arsenic, chromium, selenium, vanadium, and zinc), which had high levels of uncertainty associated with calculated ratios.

- **Mercury** – Honeywell fillet and whole-fish data (n = 22 and 11, respectively) yielded a ratio of 1.1. USEPA performed a national survey of mercury in fish in which a fillet to whole fish conversion factor of 0.7 was calculated (USEPA, 1999d). Scientists at ORNL also calculated a conversion factor of 0.7 for mercury (Bevelhimer et al., 1997). Based on the consensus for mercury found in the literature, a conversion factor of 0.7 was used to calculate the concentration of mercury in whole fish based on fillet samples (i.e., mercury in whole fish = mercury in fillet × 0.7).

- **PCBs** – PCBs tend to bioaccumulate in fatty (lipid-rich) tissues, resulting in higher PCB concentrations in whole fish than fillets, in contrast to mercury where retention of mercury in muscles and other tissues resulted in higher fillet concentrations. PCB concentrations were higher in whole fish than fillets. Although

the Honeywell PCB data set was much smaller than the NYSDEC data set (11 fillets versus 112 fillets), only the Honeywell data set was used to calculate ratios in order to compare similar species, as different species vary in lipid content and other parameters that influence total PCB concentration. Seventeen whole fish were used to calculate the ratio.

A wet-weight comparison resulted in a conversion ratio of 2.5 from fillet to whole-fish concentrations. This number was compared to the conversion used in the Hudson River PCBs site Baseline Ecological Risk Assessment (TAMS/USEPA, 2000) to confirm if it was representative. A ratio of 2.5 was obtained for the largemouth bass and a ratio of 1.5 was obtained for the brown bullhead in that assessment. The Onondaga Lake ratio was determined to correspond to other freshwater systems, and a ratio of 2.5 was applied to convert fillet concentrations to whole-fish concentrations.

- **DDT and Metabolites** – A ratio of 2.3 was calculated for DDT and metabolites based on 11 fillets and 17 whole fish collected by Honeywell. Species used for analyses were smallmouth bass, bluegill, carp, and catfish. The Honeywell regression equation used in the 2000 BERA (TAMS/USEPA) is not considered to be appropriate because concentrations of DDT in fish were generally less than 0.1 mg/kg and the regression equation overestimates DDT concentrations in that range.

- **Dioxins and Furans** – Ratios of 1.7 and 1.8 were calculated for dioxins and furans on a TEQ basis for avian and mammalian receptors, respectively. Eleven fillets and 18 whole fish samples were used to derive the ratios.

8.2.6.5 Chemical of Concern Exposure from the Ingestion of Terrestrial Prey

In estimating exposure rates for receptors consuming terrestrial prey items (i.e., mink and red-tailed hawk), COC concentrations were modeled based on available measured concentrations in wetland soils/sediments and surface soil concentrations (dry-weight basis). All non-detected values were considered observations at one-half the detection limit.

COC concentration in the prey items was predicted through the application of COC-specific transfer factors derived from ORNL guidance documents (Sample et al., 1998a,b) provided in Table 8-5. These factors are based on concomitant analyses of COC concentrations in both soil and appropriate biological tissues. COC concentrations ($[COC]_{prey}$) were modeled based on available estimates of soil concentrations ($[COC]_{soil}$) on a dry-weight basis. This modeling was accomplished through application of a COC-specific transfer factor ($TF_{soil \rightarrow prey}$) as follows:

$$[COC]_{prey} = [COC]_{soil} \times TF_{soil \rightarrow prey}$$

Prey were grouped into two classes based on the feeding patterns of receptors. The first class was soil invertebrates, which were represented in this assessment by earthworms and serve as prey for receptors, which were represented by the short-tailed shrew. The second class was small terrestrial mammals, which were defined for assessment purposes as any herbivorous, omnivorous, or insectivorous species (less than 2 kg in mass) that may be potential prey for receptors, which were represented by the red-tailed hawk and mink.

Concentration-dependent regressions or representative transfer factors were applied to calculate mean and 95 percent UCL contaminant concentrations in prey. Non-detected values were considered to be observations at one-half the detection limit. In contrast to the screening assessment, general regressions or median UFs were selected for the baseline assessment rather than the 95 percent upper prediction limit (UPL) or 90th percentile UF for both earthworm and small mammal models. The only exception to this procedure was methylmercury, for which no UFs were available, and therefore the conservative recommendation for mercury was applied.

Data from muskrats trapped in the vicinity of Geddes Brook (GB) and Ninemile Creek (NMC) between July 1998 and November 1998 were not used to develop transfer factors for mercury, PCBs, and dioxins and furans in small mammals. Prior to collection, NYSDEC eliminated the muskrat sampling effort from the GB/NMC field investigation (NYSDEC/NYSDOL, 1998) on the basis that it was inappropriate to use a herbivorous mammal to represent small mammals, inclusive of insectivores, based on differences in bioaccumulation related to feeding strategies.

In addition, the reliability of the transfer factors is questionable. The soil-muskrat transfer factors developed by Honeywell were based on seven muskrats from three locations, and six of the muskrats were collected from reference stations. The transfer factors have significant variability associated with them due to the small sample size and the narrow range of contaminant concentrations. In particular, the transfer factor for PCBs is considered to be particularly unreliable since levels of PCBs were below detection limits in muskrats and there was only one detection of PCBs (Aroclor 1254) in one of the co-located soil samples, resulting in the PCB uptake factor being based primarily on half the detection limit. Hence, transfer factors from the literature were considered to be more reliable than the Honeywell factors and were used in this assessment.

8.2.6.6 Chemical of Concern Exposure from the Ingestion of Benthos or Emergent Insects

COC exposure concentrations for receptors feeding upon emergent insects (i.e., tree swallow and little brown bat) and benthic macroinvertebrates (mallard duck) were modeled using a transfer factor method. The exception was mercury (methylmercury and ionic mercury) for which adequate measured invertebrate observations from the lake were available. Three amphipod samples were analyzed for PCBs in 1992, which was not sufficient to estimate PCB concentrations in macroinvertebrates. In addition, there were quantitation uncertainties associated with the Honeywell 1992 PCB analyses in biological tissue, as discussed in the uncertainty analysis (Chapter 11, Section 11.1.3).

In the transfer factor models, estimates of COC concentration in benthic larvae ($[COC]_{insects}$) were derived from measured concentrations in sediments ($[COC]_{sediment}$ on a dry-weight basis). Predicted COC concentrations were determined using available biota-sediment accumulation factor (BSAF) values (TF_{BSAF}) as predictive transfer coefficients, as follows:

$$[COC]_{insect} = [COC]_{sediment} \times TF_{BSAF}$$

BSAFs for metals were taken from recommendations from the Oak Ridge Reservation (US Department of Energy [USDOE], 1998). In contrast to the screening assessment, the degree of overestimation was minimized by using general, rather than conservative, recommendations.

All BSAFs for organics are from the US Army Corps of Engineers (USACE) BSAF database (USACE, 2002) or based on professional judgment. Freshwater invertebrate data were used from the USACE database, when available. The BSAFs from all freshwater invertebrate serving as prey for receptors in this assessment averaged for each contaminant to obtain contaminant-specific BSAFs. If no freshwater invertebrate data were available, saltwater invertebrate data were used. If data were not available for invertebrates, fish data were used. BSAFs for organic compounds remained the same between the screening and baseline assessment because average BSAF values were used.

Benthic invertebrate body burdens were derived by multiplying the BSAF directly by the sediment concentration for inorganic contaminants. The benthic invertebrate body burden for organic compounds was calculated as follows, based on McFarland (1998):

$$\text{Benthic invertebrate body burden} = \text{BSAF} \times \frac{\text{(sediment concentration mg/kg)}}{\text{(\% TOC sediment} \times \text{\% lipid)}}$$

The sediment TOC was assumed to be 1 percent, based on lake data. The average benthic invertebrate percent lipid value was assumed to be 2 percent, based on Lechich (1998), as Onondaga Lake invertebrate lipid data were limited to four chironomid and two (excluding a duplicate) amphipod samples.

Emergent insects were assumed to possess the same COC body burdens as the benthic larvae. All nondetected values were considered as observations at one-half the detection limit, except for groups of compounds as discussed previously. Values for TF_{BSAF} are provided in Table 8-5.

8.2.6.7 Chemical of Concern Concentrations in Water and Sediment/Soil (Incidental Ingestion)

No selection criteria were assumed for drinking water. Pooling all observations of whole water taken from the epilimnion provided an approximate water concentration in Onondaga Lake. The epilimnion data ranged from 0 (surface) to 3 m in depth and can be used by wildlife receptors as a source of drinking water.

Incidental sediment ingestion was confined to the littoral zone (water depth less than or equal to 2 m of standing water) and included all sediment analyses to a depth of 15 cm.

Soil ingestion for the receptors feeding on invertebrate and vertebrate terrestrial prey was based on soils collected from wetlands and the dredge spoils areas and included all observations to depths of 30 and 50 cm, respectively. Data are provided in Appendix I.

8.2.6.8 Food Ingestion Rates

Another component of the exposure assessment is the receptors' food ingestion rates (FIRs). The ingestion rate of an organism is a function of its energy requirements, the energy density (energy content) of its diet, and the efficiency of the organism's energy assimilation from the diet. Body weight estimates for all of the receptors were determined from literature reports. Mean body weights representative of populations indigenous to New York and the northeastern United States were preferred over other locations. Estimates of prey intake rates were based on the bioenergetic scaling relations of Nagy (1987) and expressed as field metabolic rates (FMR) (kcal/day) using data contained in USEPA (1993b). However, this is not the case for the short-tailed shrew and the little brown bat, since very small eutherian mammals were not represented in the data sets used for scaling equations. Therefore, literature sources were used to select FIRs for the shrew and bat, as discussed in the receptor profiles.

Energetic estimates were represented as lognormal distributions and determined as follows:

$$E_B = A(BW)^b = \begin{pmatrix} Birds : 2.601 \times (BW)^{0.640} \\ Mammals(non-herbivores) : 0.6167 \times (BW)^{0.862} \end{pmatrix}$$

$$IR = \frac{E_B}{BW \times E_{Prey} \times AE}$$

where:

E_B	=	estimated field metabolic rate for the receptor (kcal/day)
BW	=	receptor body weight (g)
A	=	intercept coefficient of the scaling regression (from Nagy [1987]; kcal/day)
b	=	independent variable coefficient of the scaling regression (from Nagy [1987]; unitless)

IR = estimated mean intake rate (kg dry weight/kg body weight-day)

AE = assimilation efficiency (percent)

E_{prey} = energy content of specific prey type (kcal/g dry weight).

8.2.6.9 Water and Incidental Sediment/Soil Ingestion Rates

To estimate ingestion rates of drinking water, the allometric relation between body size and water ingestion of Calder and Braun (1983) provided below was applied to all receptors as a point estimate.

$$WI(L/day) = \begin{pmatrix} Birds : 0.059 \times (BW)^{0.67} \\ Mammals : 0.099 \times (BW)^{0.90} \end{pmatrix}$$

Information for incidental soil or sediment ingestion is available only for the mallard (Beyer et al., 1994). Modeling for all other receptors was based on closely related species for which incidental soil/sediment ingestion data are available and/or professional judgment. All point estimate ingestion rates used in the analyses are included in Tables 8-6 and 8-7.

8.2.6.10 Chemical of Concern Speciation, Composition, and Bioavailability

Many of the COCs under consideration in this BERA are found in multiple chemical forms in the environment. The form of a chemical affects both its uptake rate and toxicity. Chemical analysis of abiotic media and prey tissue measures the total concentration of chemicals but not necessarily the amount biologically available to the receptors, which may be lower, but never greater. The assumption used in the food-web exposure model is that a specific COC in exposure media is as bioavailable as the form used in the toxicity studies on which the TRV is based. The assumption that COCs in the field are equally as bioavailable as chemicals in laboratory studies is retained in this BERA in the absence of adequate and consistent data on relative bioavailability.

This section discusses the considerations associated with speciation/composition and describes how they were reconciled for key contaminants. All relevant measured COC estimates in all media collected by Honeywell between 1992 and 2000 and by NYSDEC from 1992 to 2000, as described in the Onondaga Lake RI (TAMS, 2002b), were used in the exposure concentration estimates. Non-detected values were included at one-half the detection limit, except in the case of groups of compounds, as described previously.

Mercury

Organomercurial species, such as methylmercury, are more toxic to wildlife than inorganic forms of mercury. Mercury methylation occurs in aquatic habitats and in wetland habitats. Methylmercury, unlike inorganic and metallic mercury, is highly bioavailable and tends to bioaccumulate. In the higher-trophic-level aquatic prey, such as fish, methylmercury will concentrate to where it may comprise up to 95 percent of

the total mercury load for an individual fish. Therefore, receptors dependent on fish as prey were assessed based on exposure to methylmercury.

Results of the benthic invertebrate mercury and methylmercury analyses confirmed that methylmercury also accounts for a significant portion of mercury in invertebrates. Methylmercury averaged about 49 percent of the total mercury in amphipods, 16 percent of the total mercury in chironomids, 5 percent of the total mercury in oligochaetes, 53 percent of the total mercury in *Daphnia*, and 64 percent of the total mercury in other zooplankton for an average of 37 percent methylmercury in lake invertebrates (weighting all groups equally). Hence, for receptors reliant on benthic macroinvertebrates as prey, 37 percent of all mercury was assumed to be methylmercury and 63 percent to be inorganic mercury.

One percent of mercury in wetland areas was considered to be methylmercury, as discussed in Chapter 6, Section 6.3.1.1 of this report.

Total PCBs

PCBs were analyzed as specific Aroclors. Because Aroclors are themselves mixtures of PCB congeners, their composition may change over time. The risk from Aroclors was evaluated based on total PCB concentrations and defined as the sum of all Aroclors (within each sample) measured in the respective media.

Aroclor 1254/1260 (combined) was the predominant form of Aroclor detected in NYSDEC fish samples. Fish generally metabolize less chlorinated PCBs (mono and di), while retaining trichlorinated and higher congeners. The more highly chlorinated congeners are the most toxic to fish and wildlife. PCBs were only analyzed in three amphipod samples. Based on the limited data available, UFs were used to calculate PCB concentrations in aquatic and soil invertebrates rather than using the three samples to represent total PCB concentrations.

Dioxins/Furans

Dioxins and furans were assessed using the TEQ approach (Eastern Research Group [ERG], 1998). All chlorinated dioxins and furans were converted to TEQs for 2,3,7,8-tetrachlorodibenzo-*p*-dioxin (TCDD) using toxicity equivalence factors (TEFs) specific to either birds or mammals (van den Berg et al., 1998). The constituents were then summed within each sample and compared to the toxicity of 2,3,7,8-TCDD to evaluate risk. PCBs were not included in the TEQ analyses because they were analyzed as individual Aroclors and not as specific PCB congeners, some of which have TEQ factors available.

DDT and Metabolites

The toxicity of DDT, with regard to impacts on reproduction, is directly caused by the metabolite DDE, which is an intermediate in the catabolism of DDT and DDD. Therefore, the exposure of receptors to DDT

was determined based on the sum of DDE and all metabolite precursors (DDT and DDD) within each sample in the respective media.

Other Contaminants

Individual contaminants of the following groups of contaminants were summed together within each sample and exposure was based on the sum of constituents. The best available toxicological data for any compound within the group was used to derive TRVs.

- Total dichlorobenzenes.
- Total trichlorobenzenes.
- Total xylenes.
- Total PAHs.
- Total chlordanes.
- Hexachlorocyclohexanes.

All metals selected as COC for wildlife receptors other than mercury (i.e., antimony, arsenic [arsenite], barium, beryllium, cadmium, chromium [chromic], cobalt, lead, nickel, selenium, thallium, vanadium [vanadate], and zinc) were assessed based on the toxicity and bioavailability of the free ion in its most common oxidized form.

8.2.7 Life History Characteristics of Wildlife Receptor Species

Species-specific characteristics that were used in assessing chemical exposure through the food-web models are discussed in the following sections, along with the basis for their selection. Summaries of the avian and mammalian exposure factors can be found in Tables 8-6 and 8-7, respectively.

8.2.7.1 Tree Swallow (*Tachycineta bicolor*)

The tree swallow is a common perching songbird that breeds throughout the northern half of North America, where they are frequently found in association with bodies of water (Secord and McCarty, 1997). They prefer open areas in the vicinity of water, such as river valleys, lakes, marshes, flooded swamps, and beaver ponds in close proximity to decaying trees. However, they also use fields and meadows, if appropriate nesting sites are available and open water is nearby (Andrle and Carroll, 1988). The average weight of a female tree swallow in New York was reported to be 20.6 g (Secord and McCarty,1997), which was used in this assessment. This weight corresponds closely to the average spring weight reported for female tree swallows of 20.7 g (Robertson et al., 1992).

Tree swallows are insectivores that pursue flying insects on the wing, using abrupt turns, and sometimes converging in large numbers on insect swarms (Robertson et al., 1992). Food samples from tree swallows nesting along the Hudson River, New York consisted of 50 to 98 percent aquatic emergent insects (Secord and McCarty,1997), but they may supplement their diet with vegetation during cold spells (Robertson et

al., 1992). Tree swallows forage at heights of 0 to 50 m or more above ground, over open areas of water or ground that are sheltered from wind where flying insects accumulate. They feed from dawn to dusk, but most intensively between late morning and late afternoon during the breeding season (Robertson et al., 1992). Swallows, especially bank swallows but possibly others, have been observed nesting in the sides of Wastebeds 1 to 8 along Onondaga Lake. As tree swallows are considered to live near the lake, their diet was assumed to consist of 100 percent of aquatic insects.

The FIR (dry-weight basis) was estimated as 0.264 kg/kg-day based on a field metabolic rate of 875 kcal/kg body weight per day (based on Nagy, 1987). The daily drinking water intake rate (WIR) for tree swallows was estimated as 0.21 L/kg-day (based on Calder and Braun, 1983). Minimal contact with sediments is expected during feeding and grooming and, therefore, incidental sediment ingestion was set at zero.

Home-range size varies according to season and geographic area, but is between 0.1 and 0.2 km in New York (McCarty and Winkler, 1999). During the breeding season parents make 10 to 20 trips per hour to feed their nestlings (Quinney, 1986). Tree swallows nest in abandoned, excavated woodpecker holes, natural cavities in standing trees, or artificial nest boxes located in open fields or near water (Robertson et al., 1992). Most nests are spaced 10 to 15 m apart, but occasionally breeding pairs are found as close as 1 to 3 m apart.

Migrating tree swallows arrive in their northern breeding areas from February through April, with most arriving in March. Tree swallows in New York begin defending nest boxes and gathering nest material by late April, and commence egg laying by May (Secord and McCarty, 1997; Andrle and Carroll, 1988). Fall migration to wintering ranges occurs between July and September, with late August being the peak migration time (Robertson et al., 1992). A holdover population of tree swallows has been sighted regularly in the Syracuse area during the Audubon Christmas counts (Cornell University, 2001), indicating that some tree swallows remain in the Onondaga Lake area year-round and, therefore, exposure was set at 365 days per year.

8.2.7.2 Mallard (*Anas platyrhynchos*)

The mallard is one of the most common species of waterfowl in New York State (Bull, 1998). Mallards are dabbling ducks that forage by sifting through sediment in search of aquatic plants, seeds, and invertebrates (USEPA, 1993b). Although mallards are primarily herbivorous, females may switch to a diet with a larger component of invertebrates in spring in preparation for molting and egg-laying (Swanson et al., 1985; Heitmeyer, 1988). Ducklings also consume aquatic invertebrates almost exclusively during their period of rapid growth (Chura, 1961). Based on these studies, a female mallard was assumed to have a diet comprised of 50 percent aquatic invertebrates and 50 percent aquatic plants.

The average weight of female mallard in North America is 1,043 g, based on the weights of over 3,000 birds (Nelson and Martin, 1953). This weight corresponds to a field metabolic rate of 213 kcal/kg body weight per day (based on Nagy, 1987) and a consumption of 0.101 kg dry weight/kg body weight per day.

The drinking WIR was estimated at 0.058 L/kg body weight-day using Calder and Braun (1983). The sediment ingestion rate (SIR) was assumed to be 3.3 percent, based on the analyses of mallard scat (Beyer et al., 1994).

Female mallard home-range sizes vary from an average of 111 to 540 hectares, depending on habitat features such as size and distribution of available aquatic habitats (Dwyer et al., 1979; Kirby et al., 1985). Other factors shown to affect foraging range are gender, reproductive status, and population density (Dwyer et al. 1979; Kirby et al. 1985). The Onondaga Lake area was considered large enough to support a mallard population.

Migratory and resident mallards are found throughout New York State (Andrle and Carroll, 1988). Substantial numbers of mallards have been regularly documented in the Onondaga Lake area during the annual Christmas bird count (Cornell University, 2001). Based on these observations, mallards were considered to be year-round residents of Onondaga Lake.

8.2.7.3 Belted Kingfisher (*Ceryle alcyon*)

The belted kingfisher is found throughout much of North America (Bent, 1940). Although it typically inhabits areas around lakes, ponds, wooded creeks, rivers, bays, and estuaries, it is found in every ecozone in New York State (Andrle and Carroll, 1988). The belted kingfisher is an aquatic feeder and requires clear waters in order to see their prey (Davis, 1982; Salyer and Lagler, 1946). Kingfishers perch on a tree limb over a body of water while searching for prey and fish mainly at the surface of the water. The average body weight of an adult belted kingfisher selected for this assessment was 136 g based on a Pennsylvania population (Brooks and Davis, 1987).

Fish are the predominant prey of the belted kingfisher, as its name implies (Bent, 1940; USEPA, 1993b). However, diets can vary with prey availability and kingfishers may supplement their diets with aquatic macroinvertebrates, terrestrial prey, and/or plant material (Alexander, 1977). Fish are assumed to represent 100 percent of the total kingfisher diet in this assessment. Prey typically collected by the belted kingfisher range between 4 and 14 cm (Davis, 1982; Brooks and Davis, 1987), although they may consume fish up to 18 cm (Salyer and Lagler, 1946). Kingfishers appear to take prey in proportion to the relative abundance of each size (Davis, 1982). Fish less than 18 cm in length were used to model prey contaminant concentrations for the belted kingfisher.

The allometric equation of Nagy (1987) was used to estimate the bioenergetics for the belted kingfisher. Based on a daily metabolic field rate of 444 kcal/kg body weight per day, an average intake rate of 0.137 kg dry weight/kg body weight per day was determined. The drinking water intake was estimated at 0.114 L/kg body weight-day based on the algorithm of Calder and Braun (1983).

Incidental sediment ingestion during nest building and grooming was assumed to be 1 percent of total prey intake to account for soil ingestion during nest construction and nesting, as belted kingfishers construct their nests by excavating tunnels in embankments (Levine, 1988). Although the kingfisher hunts almost

exclusively within the pelagic zone, both the male and female dig the nesting burrow, using their bills as probes and their feet as shovels (Andrle and Carroll, 1988).

Home range is typically defined by length of shoreline defended by mated pairs (breeding territory) and feeding areas defended by solitary adults (non-breeding). Generally, breeding pairs defend a larger habitat than solitary individuals, although considerable overlap in size occurs. Kingfishers establish and defend summer territories for nesting and feeding. The foraging range of the kingfisher was reported to average between 0.4 and 2.2 km (Davis, 1982; Brooks and Davis, 1987). The Onondaga Lake foraging range was assumed to be 1 km based on the breeding Ohio population studied in Davis (1982). Resident kingfishers were considered to rely solely on the lake as their foraging habitat.

The timing and extent of migration appears to be related to the severity of the weather (Davis, 1982). The belted kingfisher is a hardy bird, and it remains as far north in fall and winter if it can find open water in which to catch a sufficient number of fish (Bent, 1940). Audubon Christmas counts in the Syracuse area have consistently recorded the belted kingfisher (Cornell University, 2001); therefore a year-round residency time was assumed. In addition, full exposure is considered appropriate because belted kingfishers are exposed to lake contaminants during sensitive reproduction and growth periods when their vulnerability is greatest (i.e., April to August).

8.2.7.4 Great Blue Heron (*Ardea herodias*)

The great blue heron is a wading bird that occurs in a variety of freshwater and marine habitats and breeds throughout much of North America (Bent, 1926). It is the largest member of the heron family in North America. Great blue herons may inhabit lakes, rivers, brackish marshes, lagoons, coastal wetlands, tidal flats, and sandbars, as well as occasional wet meadows and pastures (USEPA, 1993b). An average body weight for the female great blue heron of about 2,200 g was selected based on Dunning (1993).

The principal food of the great blue heron is fish of various kinds, but amphibians (e.g., frogs), snakes, small mammals, and aquatic and terrestrial invertebrates are also taken on occasion (Bent, 1926; Palmer, 1962). The great blue heron fishes by still hunting and stalking (Bent, 1926). Still hunting is the commonest method, where the heron stand motionless waiting for prey (primarily fish), which it captures striking swiftly with its bill (Eckert and Karalus, 1983). Great blue herons may also slowly wade in shallow water until it drives a fish out from a hiding place (Environment Canada, 2002). Fish make up 90 to 98 percent of the diet, with the rest consisting of crustaceans, insects, amphibians, reptiles, birds, and small mammals (Alexander, 1977; USEPA, 1993b). In this analysis, fish were assumed to comprise 100 percent of the dietary intake.

Great blue herons mainly eat fish 3 to 33 cm in length (Alexander, 1977), but may consume fish as large as 60 cm (Eckert and Karalus, 1983). Krebs (1974) found that smaller prey were selected more frequently because of greater abundance and less handling time. Although a greater number of small fish are eaten, the majority of the diet by weight consists of large fish. A proportion of two-thirds large fish (greater than 18 cm) and one-third small fish (less than or equal to 18 cm) was used to estimate fish consumption of the heron. Based on these assumptions and a field metabolic rate of 163 kcal/kg body weight per day, using

the bioenergetic algorithm of Nagy (1987), the daily FIR was estimated to be 0.0445 kg/kg (dry-weight basis).

The drinking WIR was estimated at 0.045 L/kg body weight-day based on the algorithm of Calder and Braun (1983). Data were not available on incidental SIR, which was assumed to be 1 percent, based on fishing techniques.

The average foraging ranges for the great blue heron in South Dakota ranged from an average of 3.1 km to a maximum distance flown of 24 km (Dowd and Flake, 1985). Foraging ranges of herons overlapped with mean densities of 2.3 birds/km and 3.6 birds/km observed at two separate locations (Dowd and Flake, 1985). An average home range of 3.1 km was assumed for this assessment. Based on the home range, the range overlap of individual birds, and the 16 km shoreline of Onondaga Lake, it was assumed that Onondaga Lake could support a small great blue heron population.

In New York State, the great blue heron can be both a seasonal migrant or a resident species throughout the year as long as open water persists (Bull, 1998). Results of the Audubon Christmas Bird Count show that the great blue heron is a regular winter resident in the Onondaga Lake area (Cornell University, 2001). Migrations in the northeast are highly dependent upon the severity of the winter season, primarily the degree of ice cover on feeding waters. During severe conditions (i.e., persistent cold and continuous ice cover), northeast populations will migrant south to portions of the Carolinas and Virginia. Fall migration in the Onondaga Lake population remains unclear given the tendency of this species to linger or reside in summer grounds during the winter period, and hence herons were assumed to be year-round residents.

8.2.7.5 Osprey (*Pandion haliaetus*)

The osprey is a large, powerful raptor that resembles an eagle, but its narrow wings are markedly angled when outspread and the structure of its feet and claws is so peculiar that it has been placed in a separate subfamily, the Pandioninae, of which it is the sole representative (Environment Canada, 2002). It is distributed throughout North America and found near both freshwater and saltwater environments. The average weight of an adult female is 1,568 g, while the males are slightly smaller averaging 1,403 g (Brown and Amadon, 1968).

Osprey are almost always associated with water, usually a river, lake, or the sea coast, although to reach some of these areas and during migration they may pass over large land areas (Brown and Amadon, 1968). Ospreys are skilled fishers and feed almost entirely on fish, although on occasion they may take other prey including birds (possibly wounded), frogs, and crustaceans (Brown and Amadon, 1968). On sighting prey, they hover briefly at a height of 10 to 30 m until the fish is in a suitable position. It then dives into the water, usually reappearing with a fish in its claws, which may be as large as two kilograms (Brown and Amadon, 1968; Environment Canada, 2002).

Ospreys fishing near a reservoir in Idaho consumed fish up to 41 cm in length, with the majority of prey 11 to 30 cm in length (Van Daele and Van Daele, 1982). The mean weight of fish taken by ospreys in

Chesapeake Bay was 237 g in 1975 and 157 g in 1985 (McLean, 1991). Based on these studies, the Onondaga Lake osprey was assumed to assume have10 percent of its fish consumption made up of fish less than or equal to 18 cm and 90 percent consisting of fish greater than 18 cm.

Based on a diet consisting entirely of fish and a field metabolic rate of 184 kcal/kg body weight per day, using the bioenergetic algorithm of Nagy (1987), the daily FIR was estimated to be 0.048 kg/kg (dry weight basis).

A drinking WIR of 0.051 L/kg body weight-day was estimated, based on free-living metabolic rate (Calder and Braun, 1983). No significant sediment ingestion was assumed for this species, as it has minimal contact with sediments during feeding and nesting.

Ospreys have been observed nesting in Clark Marsh (5.7 km from Onondaga Lake) on an annual basis (Clark, pers. comm., 2000,) and have been observed near the lake (Tango, 1993). The average foraging radius for ospreys ranges from 1.7 to 10 km (USEPA, 1993b). Therefore, Onondaga Lake was assumed to make up the majority of the source of food for some osprey.

Northern populations of ospreys migrate to warmer areas in the winter. Ospreys depart for their wintering grounds around the end of September and the spring migration reaches Onondaga County the first week of April (Purcell, 2001). This migratory pattern yields a typical residency time in New York of about half the year (183 days/year), but the osprey feeds at Onondaga Lake during sensitive periods of growth and reproduction (i.e., April to September). Therefore, a year-round residency time of 365 days per year was used to calculate osprey exposure.

8.2.7.6 Red-Tailed Hawk (*Buteo jamaicensis*)

The red-tailed hawk is one of the most widespread birds of prey in North America, with breeding populations distributed throughout most of the continent (Preston and Beane, 1993). They are highly mobile predators that often inhabit heterogeneous habitats (Preston, 1990). Adult females average 1,224 g, while males are smaller, averaging 1,028 g (Dunning, 1993).

The red-tailed hawk is classified as an avian carnivore with a diet consisting primarily of small mammals (about 70 percent), birds (about 18 percent), and reptiles (about 11 percent), with occasional amphibians, fish, and arthropods (Marti and Kochert, 1995). Its diet varies according to prey availability. For the purpose of this BERA, the potential risk due to exposure to COCs was modeled based on 100 percent small mammal consumption. Using the bioenergetic algorithm of Nagy (1987), a daily field metabolic rate of 246 kcal/kg body weight per day was estimated. This estimate yielded an FIR of 0.052 kg/kg-day dry weight for this species. Drinking WIR was estimated at 0.055 L/kg body weight-day, based on the algorithm of Calder and Braun (1983). An SIR of 1 percent was assumed based on professional judgment, because while some soil attached to prey may be ingested, the amount is assumed to be minimal.

The home range of the red-tailed hawk varies depending on topography, food availability, human activity, and season (Preston and Beane 1993). The average territory for the red-tailed hawk ranges from 60 to 1,770 hectares (USEPA, 1993b). Sample and Suter (1994) recommend using a home range of 233 hectares based on a study in Oregon by Janes (1984), which equals 2.3 sq km. This species has been noted to nest in the vicinity of Onondaga Lake during 1980 through 1985 (as reported by the NYSDEC's Breeding Bird Atlas Project). In 1981 and 1982, the New York State Breeding Bird Atlas Project noted nests with young and in 1983, recently fledged young were spotted near Onondaga Lake and its tributaries. Based on the area surrounding Onondaga Lake and observations, a small red-tailed hawk population is assumed to feed solely in the lake area. One resident population covering the entire area of the lake was modeled for this BERA.

Many red-tailed hawks breeding in northern regions migrate south. However, even in the harshest winters with extensive snow cover, some birds remain near their breeding territory year-round (Preston and Beane, 1993). The red-tailed hawk has been regularly spotted during the Audubon Christmas count in the Onondaga Lake area (Cornell University, 2001). Therefore, some individuals were assumed to have year-round residency (365 days per year) in the Onondaga Lake area.

8.2.7.7 Little Brown Bat (*Myotis lucifugus*)

The little brown bat is common throughout North America, including most of the United States and Canada. This insectivorous species is indigenous to New York State where it is considered a non-game species and is regulated by NYSDEC. Bats collected at the end of August at Ironville, New York had average weights of 8.8 g for females and 7.2 g for males (Davis and Hitchcock, 1965). These bats were collected at about their maximum weight, since July and August are spent in heavy feeding as bats build up their fat reserves before hibernation. The mean weight of female bats in the New York State Museum collection was 7.1 g, which was used to represent Onondaga Lake bats. These weights agree with the average adult weight of 6 to 8 g for little brown bats studied near Ithaca, NY (Wimbatt, 1945) and in New Jersey (McManus and Esher, 1971).

Little brown bats are nocturnal mammals that feed on insects primarily near bodies of water (Barbour and Davis, 1969). Foraging flights of little brown bats begin at dusk and last for 1.5 to 3 hours, with a second feeding period lasting for more variable periods of time, until dawn (Anthony et al., 1981). Fecal analysis revealed that little brown bats consume varied insect taxa, including Diptera, Lepidoptera, Coleoptera, Ephemeroptera, Hymenoptera, Trichoptera, and Neuroptera, typically 3 to 10 mm in length (Anthony and Kunz, 1977). Belwood and Fenton (1976) reported that according to fecal analysis, aquatic insects, primarily Diptera (chironomids) and Tricoptera (caddis flies), constituted about of the 95 percent of the adult diet at a site in northern New York, although Buchler (1976) observed that Ephemeroptera (mayflies) comprised the majority of the diet of his study population.

The amount of prey ingested during feeding varies by sex, age, and reproductive state. On average, pregnant bats ingested 2.5 g of prey, lactating females ingested 3.7 g, and juveniles ingested 1.8 g per feeding flight (Buchler, 1976). Digestion of ingested prey begins after the stomach is full and the bat has

returned to its colony. Transit time in the gut is rapid, and complete digestion and excretion of one stomach volume can take less than an hour for an active individual, allowing bats to fill their stomach two or more times each feeding period (Buchler, 1976). Little brown bats may consume between 30 and 100 percent of their body weight each night (Hoffman, 1999; Environment Canada, 2000; Snyder, 2002).

No field metabolic rate measurements have been made on very small, active eutherians. Therefore, information contained in the literature was used to estimate prey consumption rates. A consumption rate of 25 percent of the body weight (1.8 g/day) was selected to represent an average feeding rate for female bats, considering both their active and hibernating periods. The wet weight consumption rate was converted to dry weight using the a conversion rate of 1 kg dry weight to 4.5 kg wet weight based on the average wet weight to dry weight ratio for aquatic invertebrates for studies listed in Wildlife Exposure Factors Handbook (USEPA, 1993b). This value is within the range of conversion factors provided by Peters (1983). Food consumption on a dry-weight basis was therefore estimated as 0.102 kg dw/kg body weight per day based on the energy content of insect larvae taken from USEPA (1993b).

The WIR was estimated to be 0.162 L/kg body weight per day (Calder and Braun 1983). No SIR was assumed for this species, since bats capture smaller insect prey directly with the mouth in flight and use their body, tail, and wings to cup and direct larger prey into the mouth. All insect prey is masticated and devoured in flight.

A home range of 0.1 km was selected for the little brown bat based on observations of the distance traveled by a colony in New York for nightly feeding by Buchler (1976).

In response to colder temperatures and diminishing prey, little brown bats move from summer roosting and maternity colonies to winter hibernacula (i.e., hibernating shelters). Seasonal movements occur before and after hibernation. While not truly migration in definition, this movement results in temporary displacement of little brown bat populations from summer refuges/feeding areas and dispersal to other summer and winter refuges. The distance traveled by bats from New York and New England populations ranged from 8.7 to 105 km between summer and winter locations (Griffin, 1945; Davis and Hitchcock, 1965). In New York little brown bats were found to return to winter hibernacula from September to October (Davis and Hitchcock, 1965), with females leaving the hibernaculum earlier (April to mid-May) than males (mid-May to early June) to disperse or move to summer colonies. Although the little brown bat hibernates part of the year and may move from out of the Onondaga Lake vicinity, all food sources during the year (i.e., active feeding time and fat reserves used during hibernation) are assumed to be derived from Onondaga Lake. Reproduction and growth (the most sensitive time periods) also occur when the little brown bat is active at Onondaga Lake. Therefore, the little brown bat was treated as a year-round resident of Onondaga Lake and no temporal modifying factor was applied.

8.2.7.8 Short-Tailed Shrew (*Blarina brevicauda*)

The short-tailed shrew (*Blarina brevicauda*) is a small insectivorous mammal that ranges throughout the United States (George et al., 1986). Short-tailed shrews range from about 9.5 to 13 cm in length and weigh

12.5 to 22.5 g (Guilday, 1957). An average body weight of 15 g was used based on the average shrew weight in New Hampshire (Schlesinger and Potter, 1974).

Shrews are mainly insectivorous and carnivorous, but some eat seeds, nut meats, and probably other plant material (Nowak, 1997). Analyses of stomach contents of New York State shrews show that earthworms comprise the majority of the short-tailed shrew diet with slugs, snails, insect and miscellaneous animals contributing most of the remainder (Whitaker and Ferraro, 1963). For this assessment, the diet of the shrew was assumed to consist of 100 percent terrestrial invertebrates and modeled contaminant concentrations in earthworms were used to estimate body burdens of contaminants in prey.

The bioenergetic algorithm of Nagy (1987), does not include data for very small, very active eutherian mammals, such as the shrew. Since the field metabolic is strongly correlated with body size, it was considered inappropriate to use Nagy's equation to calculate a metabolic rate for shrews, as those data were not used to develop the equation. Shrews feed frequently and may consume more than their total body weight in food over a 24-hour period (Schmidt, 1994). In the laboratory, food consumption rates ranged from an average of 8 to 10 g/day (Buckner, 1964 and Barrett and Stueck, 1976; both cited in Sample and Suter, 1994). The higher average consumption rate of 10 g/day (two-thirds of the average body weight) was selected since field metabolic rates are likely to be higher than laboratory rates. This equals a daily consumption rate of 0.157 kg/kg-day on a dry-weight basis. Consumption rates do not consider increased food requirements in winter (about 40 percent greater) to maintain body temperatures in colder weather (Randolph, 1973), and therefore this consumption rate may underestimate Onondaga Lake food consumption.

The shrew has a high rate of evaporative water loss and an estimated WIR of 0.151 L/kg-day was calculated based on Calder and Braun (1983), although this equation also has more uncertainty associated with predictions for small sizes. Incidental soil ingestion was assumed to be 13 percent of food consumption (Talmage and Walton, 1993, as cited in Sample and Suter, 1994).

Short-tailed shrews are found in nearly all land habitats(Nowak, 1997). They construct runways in leaves, plant debris, snow, or the ground. Runways are usually in the top 10 cm of soil, but can be as deep as 50 cm (USEPA, 1993b). Home ranges for New York State shrews in the winter average 0.05 hectares, with maximum ranges of 0.10 to 0.22 hectares (Platt, 1976).

Shrews are active year-round and can be seen by day or night (Nowak, 1997). Five resident shrew year-round populations were modeled for this BERA, one living in each of the four wetland areas and the other in the dredge spoils area.

8.2.7.9 Mink (*Mustela vison*)

Mink are distributed throughout all of New York State and most of the United States and Canada (NYSDEC, 2000). They occupy wetland habitats including streams, lakes, rivers, and freshwater and saltwater wetlands. They prefer wetlands and riparian habitat with irregular shorelines, good cover (i.e.,

woods and shrub), and suitable den sites (Linscombe, et al. 1982; Allen, 1984), but in Sweden are most abundant near eutrophic lakes (Eagle and Whitman, 1987). Regardless of the type of habitat used, mink dens are always associated with water and typically are 5 to 100 m from a water body and mink can use several den sites within their home range. The most widely used den sites are bank burrows of other animals, particularly muskrats. Mink are reasonably tolerant of human disturbance/development as long as prey abundance is not affected (Allen, 1984).

Mink exhibit a pronounce sexual dimorphism in size with male 1.4 to 1.8 times heavier than females (Eagle and Whitman, 1987). An average body weight for mink of 600 g was used based on the average adult female weight provided in Mitchell (1961).

Mink are nocturnal in habit and opportunistic in diet. Although not totally restricted to wetland or wetland-associated habitats, the mink is dependent on aquatic organisms for much of the year (Allen, 1984). As a carnivore in an aquatic habitat, mink may concentrate environmental pollutants (Eagle and Whitman, 1987). They actively seek prey within their home range, and their diet varies according to season, prey availability, and habitat type (Allen, 1984). Mink feed primarily on small aquatic and terrestrial animals, although they can feed upon prey items larger than themselves, such as waterfowl and muskrats (Sealander, 1943). Common prey items include fish, frogs, crayfish, salamanders, clams, insects, muskrats, voles, and rabbits (USEPA, 1993b). Hunting in aquatic habitats occurs in shallow, nearshore areas where aquatic prey is captured and then moved to the shore prior to consumption (Doutt et al., 1977).

A Michigan riverine mink population fed mainly on fish (85 percent), catching fish ranging from 5 to 18 cm (Alexander, 1977). Fish, shellfish, and crayfish were the major food items of mink inhabiting coastal habitats of Alaska and British Columbia (Allen, 1984). A study in Idaho (Melquist et al., 1981) found fish occurring 59 percent of the time in the mink diet with unidentified cyprinids, ranging in length from 7 to 12 cm comprising the major type of fish eaten. A study of summer mink scat in Montezuma Marsh, a wetland in the Finger Lakes region of NY, found the diet to consist primarily of mammals (43 percent mammals), fish (27 percent), aquatic invertebrates (14 percent), and birds (9 percent) (Hamilton, 1940). The most abundant forage fish in Montezuma Marsh, the golden shiner (*Notemigonus crysoleucas*), comprised the greatest proportion of fish in the mink diet. Fish consumed from Montezuma Marsh were generally 8 to 11 cm, while mink belonging to a Montana riverine population feed mainly on brook stickleback *(Culaea incostans)* from about 4 to 6 cm (Gilbert and Nancekivell, 1982). Fish are also a common food item of mink during winter months (NYSDEC, 2000a).

Onondaga Lake mink were assumed to consume a diet consisting of 35 percent fish, 15 percent aquatic invertebrates, and 50 percent other food sources (e.g., mammals, waterfowl, amphibians). This dietary composition was selected to represent year-round exposure at Onondaga Lake. Small mammals were selected to represent "other" food sources, as no body burden data were available or modeled for birds or amphibians. Mink were assumed to feed on fish 18 cm or less in length. All fish in this size range were used to estimate fish contaminant concentrations. However, as the mink is an opportunistic feeder, prey selection often depends primarily on the abundance of fish or other prey species and secondarily on the size.

An average field metabolic rate for female mink was estimated as 255 kcal/kg body weight per day based on the algorithm of Nagy (1987). This estimate yields an intake rate of 0.064 kg dry weight/kg body weight-day, based on an assumption of 35 percent fish, 15 percent aquatic invertebrates (e.g., crayfish, beetles), and 50 percent mammals. This value is slightly higher than the daily feed consumption of caged female mink (0.05 kg dry weight/kg body weight-day; Bleavins and Aulerich, 1981), but mink in the wild are expected to have higher energy requirements than caged mink. The daily drinking WIR for the mink was estimated as 0.104 L/kg-day, using the allometric equation of Calder and Braun (1983).

Mink incidentally ingest a small quantity of vegetation and soil while feeding (Alexander, 1977; Sealander, 1943). Based upon the observations of soil and vegetation in mink stomachs (Hamilton, 1940), the percent ingestion of sediment during feeding and grooming was assumed to be approximately one percent of the diet.

Home ranges for the males are normally larger than those of females, particularly during the breeding season (Eagle and Whitman, 1987). During the breeding season, male home ranges may overlap those of several females. However, same-sex ranges never overlap (Eagle and Whitman, 1987). A female's home range, which includes both dens and foraging areas around waterways, may occupy from 1 to 2.8 km of shoreline, depending on food availability, age, gender, and season (Gerrell, 1970). Female mink have the smallest and most well-defined home range, while male ranges tend to be larger and less clearly defined. Mitchell (1961) reported home range sizes of 7.8 and 20.4 hectares for two female mink. A mean home range of 1.85 km of shoreline was selected for this assessment, based on Gerrell (1970). During daily activity periods mink move back and forth in a restricted core area that typically is less than 300 m in shoreline length (Gerrell, 1970, as cited in Allen, 1984). Dens have been reported to lie between 5 and 100 m from water (Melquist et al., 1981). This limited ranging behavior may preclude minks resident to Onondaga Lake from using any other major water body as a food source. Therefore, it was assumed in the food-web model that the mink's diet was derived entirely from the study area for the entire year.

Mink are active year-round and do not hibernate (Doutt et al., 1977; Alexander, 1977). They occupy and defend a resident territory throughout the year and do not migrate, with the exception of local territorial movements by adults and dispersal of sub-adults from resident populations (Allen, 1984). Populations within the study area of Onondaga Lake are year-round residents.

8.2.7.10 River Otter (*Lutra canadensis*)

The river otter is one of the larger members of the Mustelidae family. It is found throughout most of North America and is indigenous to New York State. It is morphologically adapted for land and water, and feeds almost exclusively on aquatic prey. Females are smaller than males with weights ranging from 5 to 15 kg at sexual maturity (Melquist and Dronkert, 1987). In New York State, the average weight is about 5.45 kg (NYSDEC, 2000a), which was selected for use in this assessment.

Fish comprise the majority of otter prey, but otter also commonly feed on crayfish, with aquatic invertebrates, amphibians, birds, mammals, and blueberries contributing a smaller percentage of the diet

(e.g., Hamilton, 1961; Knudsen and Hale, 1968; Serfass et al. 1990; Sheldon and Toll, 1964; Toweill, 1974). A diet of 90 percent fish (Newell et al., 1987) and 10 percent aquatic invertebrates was used to estimate dietary exposure of otter to contaminants. Prey for the river otter is generally fish with a reported size range between 2 and 50 cm (Melquist and Dronkert, 1987). Prey availability and catchability influence dietary composition. Common fish eaten by otter include both forage fish and game and pan fish, depending on the area (Tumilson and Shalaway, 1985). Few studies provided relative proportions of size distribution of fish consumed by otter, although work by Toweill (1974) clearly showed a preference for larger fish. Limited data provided in Alexander (1977) indicate that otter prefer feeding on larger fish, and hence a diet of two-thirds (67 percent) fish greater than 18 cm in length and one-third (33 percent) less than or equal to 18 cm in length was assumed.

An average field metabolic rate for a river otter of 5.45 kg was estimated as 188 kcal/kg body weight per day based on Nagy (1987). This estimate yields an intake rate of 0.044 kg dry weight/kg body weight-day, for a diet composed of 90 percent fish and 10 percent aquatic invertebrates. The daily drinking WIR for the river otter was estimated as 0.084 L/kg-day by using the allometric equation of Calder and Braun (1983). Incidental vegetation material occur commonly in the digestive tracts of otters (Toweill, 1974), and therefore the assumption of incidental soil ingestion was set at 1 percent of total daily food intake, as was done for the mink.

The shape and size of the otter home range varies by habitat type. Home ranges have been documented to range from 1 to 78 km (Melquist and Dronkert, 1987). Spinola et al. (undated) monitored otters released along the Genessee River in western New York. They found dispersal distances ranging from 1.5 to 22.5 km, with an average of 10 km for all otters and 9 km for female otters. Home ranges of otters have been shown to overlap extensively, both within and between genders (Erickson and McCullough, 1987). Average densities of otter range from one every 2 to 3 km (Erlinge 1967, 1968, as cited in Nowak, 1997) to one per 3.9 km of waterway (Melquist and Hornocker, 1983). As the shoreline of Onondaga Lake is approximately 18 km excluding the shoreline areas associated with tributaries (see Chapter 3, Section 3.2.1), the Onondaga Lake shoreline was considered adequate to support a small river otter population.

River otter are active year-round and do not hibernate (Doutt et al., 1977). River otters occupy and defend a resident territory throughout the year and do not migrate with the exception of local territorial movements by adults and dispersal of sub-adults from resident populations, and are therefore considered year-round residents.