APPENDIX 7

ECOLOGICAL RISK ASSESSMENT

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DRAFT BASELINE ECOLOGICAL RISK ASSESSMENT REPORT

MSGRP Northern Study Area
Industri-Plex Superfund Site
Operable Unit 2
Woburn, Massachusetts

Text, Figures,
Tables and Appendices

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EXECUTIVE SUMMARY

A Baseline Ecological Risk Assessment for the Northern Study Area of the Multiple Source Groundwater Response Plan (MSGRP) Remedial Investigation (RI) was conducted by Metcalf & Eddy, Inc. for USEPA Region I. For the risk assessment, the Northern Study Area (i.e., the study area) is defined as the area from the Industri-Plex Superfund Site south to Interstate 95/Route 128 and includes surface water, sediment, soil, and biota. The study area includes the Aberjona River, Halls Brook Holding Area (HBHA), and associated wetland areas located north of Interstate 95/Route 128 in North Woburn, Massachusetts. The southern boundary of the study area is the edge of the Wells G&H Interim Wellhead Protection Area (IWPA). The objective of the baseline ecological risk assessment is to determine whether contaminated media (surface water, sediment, soil, and biota) within the study area, remaining after the Industri-Plex Site soil remedy was completed, pose risks to ecological receptor populations present within the Northern Study Area.

The study area contains numerous commercial and light industrial businesses as well as a regional transportation center. The area was historically used for chemical and glue manufacturing which generated wastes containing residues of arsenic insecticides, benzene, toluene, and the metals chromium, copper, lead, and zinc. The hide wastes, consolidated into four hide piles and portions of a utility right-of-way, and contaminated soils were capped as part of the soil remedy for the Industri-Plex Superfund Site.

The field program for the study area was conducted between 1999 and 2004, and included the collection and analysis of surface water, sediment, sediment cores, surface soil, fish tissue, plant tissue, and sediment invertebrate samples. Chemical classes of concern included volatile organics (VOCs), semi-volatile organics (SVOCs), pesticides, PCBs, and inorganics. Twelve reference stations were also identified from which surface water, sediment, and fish tissue were collected.

Contaminants of potential concern (COPCs) in the study area, including VOCs, SVOCs, pesticide/PCBs, and inorganics, were identified via an effects-based screening involving the comparison of maximum contaminant concentrations in surface water and sediment to ecological benchmarks for these media. Screening-level maximum exposure models were used to select COPCs for avian and mammalian receptors. The screening process identified 12 COPCs for surface water and 52 COPCs in sediment. Fifteen COPCs (all inorganics) were identified as COPCs for wildlife receptors.

Receptor species were selected for exposure evaluation to represent various components of the food chain in the river/wetland ecosystem. Receptor species selected for the evaluation included muskrat, river otter, green heron, mallard, and short-tailed shrew. Additional indicators species/communities selected included fish populations and benthic invertebrates. The exposure estimates for each receptor species or community were evaluated on spatial scales representative of the home range of each receptor species. Shrew and muskrat, with small home ranges, and benthic invertebrates, which are mainly sedentary, were all evaluated for exposures on a station-by-station basis. River otter, green heron, and mallard, which have the largest foraging areas, were assessed on a site-wide scale. Fish populations were evaluated in two larger open water areas of the site, which included the HBHA Pond at the northern end of the study area and the HBHA wetland pond, immediately north of Mishuwam Road.

Receptors, endpoints and the corresponding risk summary are shown in Table ES-1. Each endpoint has associated with it a magnitude of risk and a degree of uncertainty. The magnitude of risk incorporates both the degree to which the endpoint was exceeded and also the proportion of the habitat affected. Since the endpoints were based on effects on populations, a reasonable probability of risk was determined to be present only when a risk was present through the majority of the organism's habitat. If the toxicity reference value based on a no observed adverse effects level (NOAEL TRV) was exceeded across most of the site, the contaminant was concluded to pose a low risk to populations. The highest risk was associated with contaminants that exceeded upper threshold effects levels using toxicity reference values based on lowest observed adverse effects levels (LOAEL TRVs), and was present throughout a majority of the indicator species' habitat within the study area. If high HQs were present in only a small proportion of the habitat for the selected indicator species, the magnitude of the overall risk to the population from exposure to the COPC was considered low.

Based on the analysis of the nine selected indicators/endpoints in the BERA, there were no indications of significant ecological risk associated with VOC, SVOC, and pesticide/PCB contamination within the study area, with the exception of potential risk to aquatic receptors from benzene in the deep water of HBHA Pond. Evidence suggests that there is high exposure to arsenic for semi-aquatic mammals, terrestrial mammals, bottom feeding fish, and small forage fish in the study area. The magnitude of the risk and uncertainty associated with the ecological effects for each receptor are discussed below.

Twelve COPCs were identified in the initial surface water screening, among the COPCs in surface water, benzene was detected in deep samples from HBHA pond. The average concentration in deep samples indicated potential risk based on Tier II benchmarks. SVOCs were infrequently detected and represent a low risk to receptors in surface water. Average inorganic COPC concentrations, including barium, cadmium, cobalt, manganese, silver, and zinc, exceeded surface water benchmarks (Table 45). Based on the low magnitude and frequency of the exceedances of National Ambient Water Quality Criteria (NAWQCs) and screening benchmarks, the risk to aquatic organisms in the study area from exposure to metals in surface water is low. Surface water screening indicated a possible risk from exposure to benzene in the pond, but the magnitude of the risk is low.

The USEPA chronic NAWQC for total arsenic is 340 μg/L and for dissolved arsenic is 150 μg/L (USEPA, 2002). The maximum dissolved arsenic concentration measured in surface water in the study area (120 μg/L) was below the NAWQC. The surface water concentrations are well below those expected to cause effects on fish. However, as described below, fish exposure to arsenic is high, and tissue level concentrations are slightly above values expected to be associated with adverse effects for species with feeding strategies that result in high exposure to sediment and ingestion of sediment-dwelling organisms.

Tissue concentrations of arsenic in fish collected in on-site ponds were elevated in comparison to reference locations. Risks were identified to sediment-associated species, primarily white sucker and brown bullhead, based on exceedances of tissue concentration that may be associated with harm. The magnitude of the exceedances were not high. Additional population characteristics were measured to evaluate the health of study area populations. The population data indicated impairment of fisheries, however, the relative influence of poor quality habitat conditions could not be distinguished from impacts associated with toxicity from contaminants. The tissue data provided evidence of potential ecological effects, although population data are inconclusive about the role of toxicity in impairing fish populations in the on-site ponds. The risks to fish were possibly underestimated based on the inability to discern any impacts from the exposure to toxic substances from impacts associated with the limited and poor overall habitat.

Among the other wildlife receptors, food chain modeling based on site-specific data indicated negligible risk to river otter, green heron, and mallard duck from exposure to COPCs in the study area. Omnivorous mammals, such as muskrat, that consume aquatic vegetation as a large portion of their diet have high exposure to inorganics, including arsenic. The dietary analysis indicates high exposure of muskrat to arsenic in HBHA Pond and upper portions of the HBHA wetland, with uncertain risk due to low confidence in dietary estimates and selection of reference toxicity values representing harm.

For northern short-tail shrew, among the inorganics with potential risk, only arsenic concentrations in soil corresponded to HQs greater than one in the average/LOAEL models at stations A6, BE-2, HB02-2, and HB03-3. Soil concentrations at stations A6, BE-2, HB02-2, and HB03 in shrew habitat exceeded this upper TEL value (160 mg/kg mg/kg). This indicates possible impacts to shrew or other small mammals due to exposure to arsenic in diet at stations (A6, BE-2, HB02-2, and HB03-3) bordering the HBHA pond and wetlands, although the uncertainty associated with this risk is high.

For the benthic invertebrates, comparison of sediment metals concentrations to effects-based sediment benchmarks indicated potential effects from arsenic, cadmium, chromium, copper, lead, mercury, and zinc. Toxicity testing results showed acute toxicity in laboratory test organisms exposed to HBHA Pond sediments. Toxicity and invertebrate community impacts throughout the study closely corresponded to the concentration of arsenic in sediments. Although the concentration of a number of metals co-varied with the elevated concentration of arsenic, both the toxicity results and the impairment in the invertebrate community structure were most consistently associated with high arsenic concentrations. The benthic invertebrate tissue data also add to the weight of evidence for the effects of arsenic, as the concentration of arsenic in invertebrate tissue exceeded ecological effects levels and were greatly elevated at station MC-06. In general, elevated concentrations of metals in invertebrate tissue corresponded to locations with high toxicity, but showed less association with concentrations of the same metals in downstream sediments. These results indicate that the toxicity and impairment to benthic invertebrates in HBHA Pond is likely related to the forms of metals in the sediment having higher toxicity and bioavailability than the same metals present in sediments downstream.

The risks to benthos were clearly identified. Risks increased with increasing concentration of arsenic in sediment, particularly when the amount of iron in the sediment, which ameliorates the effects of arsenic, was taken into account. Acute toxicity and impairment of benthic invertebrate communities were observed in the HBHA pond sediments, and less severe effects were observed in benthic communities downgradient in the HBHA wetland.

The results of the risk assessment are consistent with the site conceptual model and fate and transport information. Groundwater discharge of contaminants, including arsenic and VOCs, may be contributing to the acute toxicity of the sediments in HBHA Pond. Anomalously high concentrations of arsenic in invertebrate tissue in shallow areas of the pond could be attributed to higher availability of arsenic in the pond sediments. Poor habitat conditions, along with sediment contaminants, are likely to contribute to the highly impaired benthic invertebrate community in the pond, as well as the impaired fish populations in the pond. Downstream, as arsenic in surface water and sediment form relatively insoluble complexes with iron and other sediment constituents, the exposed receptors such as benthic invertebrates show lower toxic effects from exposure to contaminants in sediment.

A second risk assessment has been completed for the Aberjona River south of Interstate 95/Route 128 that includes environmental data collected from the Wells G&H Superfund Site to the Mystic Lakes. Collectively, the two risk assessments evaluate the environmental data collected along the entire river from the Industri-Plex Superfund Site in North Woburn to the Mystic Lakes. Section 7.0 of the comprehensive MSGRP RI Report merges, summarizes, and refines the two ecological risk assessments. The comprehensive RI Report documents all of the data collected along the Aberjona River and Halls Brook Holding Area from North Woburn to the Mystic Lakes, and further explains the nature and extent of contaminants and their fate and transport mechanisms.

TABLE ES-1
SUMMARY OF RECEPTOR RISKS AND COCS

INDUSTRI-PLEX SUPERFUND SITE

Assessment Endpoints	Receptor Species	Exposure Area	Risk Summary
Sustainability (survival, growth, reproduction) of local populations of omnivorous, semi-aquatic mammals	muskrat	- HBHA Pond, HBHA Wetland	Arsenic - risk above upper TEL, high uncertainty
Sustainability (survival, growth, reproduction) of local populations of small terrestrial mammals	northern short-tail shrew	A6, BE-2, HB02-2, HB03-3	Arsenic - risk above upper TEL, high uncertainty
Sustainability (survival, growth, reproduction) of local populations of bottom-feeding fish	white sucker & brown bullhead	- HBHA Pond - HBHA Wetland Pond	Arsenic - Low risk to populations based on tissue residue effects, uncertain risk based on population studies
Sustainability (survival, growth, reproduction) of local populations of small forage fish	pumpkinseed	- HBHA Wetland Pond	Arsenic - Low risk to populations, exceeding no-effect tissue residue effects levels, uncertain risk based on population studies
Sustainability (survival, growth, reproduction) of local populations of benthic invertebrates	benthic invertebrates	- HBHA Pond - HBHA Wetland	Exceedences of SELs for arsenic, cadmium, chromium, copper, lead, mercury, and zinc. Acute toxicity in deep water HBHA Pond. Severe impairment and chronic toxicity in shallow areas of HBHA Pond. Evidence of impairment at MC-10 and MC-11 in HBHA wetlands.

TEL - Threshold Effects Level
SEL - Severe Effects Level

APPENDIX 7A

BASELINE ECOLOGICAL RISK ASSESSMENT

MSGRP NORTHERN STUDY AREA

1.0 INTRODUCTION

This appendix to the comprehensive Remedial Investigation (RI), also known as the Multiple Source Groundwater Response Plan (MSGRP) RI, contains the baseline ecological risk assessment (BERA) conducted for the Northern Study Area (i.e., the study area). For the risk assessment, the Northern Study Area (i.e., the study area) is defined as the area from the Industri-Plex Superfund Site south to Interstate 95/Route 128 and includes surface water, sediment, soil and biota in areas adjacent to the Industri-Plex Superfund Site, south to Mishawum Road and Interstate 95/Route 128. The study area, located in North Woburn, Massachusetts, includes the Aberjona River, Halls Brook Holding Area (HBHA), and associated wetland areas north of Interstate 95/Route (Figures 1 and 2).

The East Drainage Ditch, Landfill Creek, and the New Boston Street Drainway, located within the boundaries of the study area, are not being assessed in the baseline ecological risk assessment because these drainage features were previously addressed by the Industri-Plex Remedy or through other state/local actions or investigations. Sections of the East Drainage Ditch and Landfill Creek that are outside the boundaries of the study area are being investigated under the auspices of the Massachusetts Department of Environmental Protection (MADEP) Bureau of Waste Site Cleanup (BWSC) and its cleanup regulations 310 CMR 40.0000 (the Massachusetts Contingency Plan [MCP]) as part of site investigations at the Olin Chemical Company facility in Wilmington, and under the MADEP Office of Solid Waste's regulations 310 CMR 19.000 through post-closure monitoring activities at the Woburn Landfill. These data include sediment samples with the designation "SD-ED-", and surface water samples collected for gauging stations SW-01-IP through SW-03-IP and SW-05-IP through SW-08-IP.

The purpose of the BERA is to determine whether contaminants present in surface water, sediment, and biota of the study area pose a current or potential future risk to environmental receptor populations. The BERA was prepared in the accordance with the following guidance documents:

Ecological Risk Assessment, Guidance for Superfund: Process for Designing and Conducting Ecological Risk Assessments, Interim Final (USEPA, 1997); and

Guidelines for Ecological Risk Assessment, Final (USEPA, 1998a).

The first step of the BERA is the screening-level problem formulations and ecological effects evaluation. Surface water, sediment, soil, and biota samples were collected within the study area including the Aberjona River, HBHA, and associated wetland areas north of Interstate 95/Route (Figures 3, 4, 5, and 6). Samples were also collected in similar habitats at 12 reference locations (Figure 2). During this screening-level effort, resources potentially at risk were identified, and chemicals of potential concern (COPCs) were selected and carried forward for detailed evaluation in the BERA. The screening involved the comparison of maximum detected contaminant concentrations from the media-specific study area samples to conservative ecological criteria or benchmarks.

1.1 SCREENING-LEVEL PROBLEM FORMULATION

Problem formulation, which is the first step in the risk assessment process, includes description of the environmental setting, descriptions of study area resources, review of data for study area contaminants, toxicity literature reviews for COPCs, identification of complete exposure pathways, selection of assessment and measurement endpoints, and preparation of a site conceptual model. General fate and transport characteristics of study area-related contaminants are discussed in Section 1.1.2 of this BERA. Detailed analysis of fate and transport is presented in Section 5.0 of the RI.

The initial Ecological Assessment Work Plan and field sampling was prepared for the Industri-Plex Site Remedial Trust (ISRT) by Menzie-Cura Associates (1999). The proposed approach in the 1999 workplan has been modified and incorporated into the BERA.

1.1.1 Environmental Setting

In this subsection, the ecological setting of the study area is defined through a discussion of the aquatic and terrestrial habitat types present within and adjacent to the study area. The potential for rare species to inhabit the study area is also addressed. Based on the early study area assessment, the workplan developed by ISRT, and input from USEPA, the following study area habitats are the focus of the ERA:

- On-site tributaries to Halls Brook Holding Area
- Halls Brook Holding Area (HBHA) Pond
- HBHA Wetlands
- Aberjona River upstream of HBHA
- Aberjona River downstream of HBHA
- Wetland/terrestrial buffer adjacent to HBHA Pond and wetlands

The ecological characterization of study area aquatic habitats was primarily based on field investigations conducted by Menzie Cura Associates, USEPA, and US Fish and Wildlife Service (USFWS); and recent site visits by USEPA and Metcalf & Eddy, to confirm study area conditions and collect additional soil and sediment samples. Fish species reported for different areas of the study area were primarily identified during field investigations conducted in June 1999 by USFWS. Sediment for toxicity testing and invertebrates and plants for tissue analysis were collected during June and July 1999. These samples were co-located with surface water samples and benthic macroinvertebrate community evaluation samples, taken at the same time as sediment chemistry samples from four reference and nine non-reference locations. These samples were supplemented by additional surface water and sediment sampling by EPA from 2000 to 2002.

The study area includes the surface water and wetland complex from Industri-Plex Superfund Site south to Route 128. The study area is bordered to the north, south, and west by mainly commercial and light industrial properties. Route 93 runs along its eastern border, and a commuter rail line crosses its western side. Most of the developed land is covered with buildings, asphalt, concrete, or other cover material. These upland areas do not provide significant ecological habitats and were not considered in the BERA. The wetland resources include reaches of Halls Brook, Halls Brook Holding Area River, and their associated wetlands.

The HBHA is a storm water management area constructed circa 1971 by filling-in portions of the former Mishawum Lake. The HBHA is approximately 4,200 feet long and 250 feet wide and is comprised mostly of vegetated wetlands with a few interspersed ponds. Located in the northern portion of the HBHA, the HBHA Pond represents the largest area of open water in the study area (approximately 4.6 acres). The HBHA Pond is a storm water retention basin, approximately 200 feet wide and 1,100 feet long. The HBHA Pond has a maximum depth of approximately 27 feet, which was observed at one location. However, the average depth along an approximate north/south centerline is approximately 14 feet (Ford, 2002). The banks of the pond are steeply sloped along the north and west and have a narrow vegetated zone, dominated by scrub/shrub and emergent vegetation. Woody vegetation overhangs the pond along the banks, interspersed with dense stands of *Phragmites* sp.

Tributaries to the pond include Halls Brook and two storm water culverts that discharge to the pond from drainage areas to the north and to the east. Halls Brook flows into HBHA Pond near the northwest corner. The headwaters of the brook are wetlands in Woburn, south of Route 128 and west of Main Street (Menzie Cura, 1999). The brook flows northerly, passing under Route 128 into North Woburn. Just north of Merrimac Street, the brook turns east, and flows through wetlands surrounded by residential and commercial areas of North Woburn. It flows east under New Boston Street, under the railroad tracks, and into the HBHA Pond.

A drainage channel runs north along the railroad tracks and discharges into Halls Brook east of the railroad tracks, just before Halls Brook discharges into the HBHA Pond. This channel, which is partially culverted, accepts drainage from the following sources:

- East Drainage Ditch – a drainage ditch which runs parallel to the railroad tracks, originating north of the Industri-Plex Site and merging into the New Boston Street Drainway at the border of the Industri-Plex Site. The East Drainage Ditch merges with Landfill Creek.
- Landfill Creek - a small brook that originates from wetlands adjacent to the recently capped Woburn Landfill, north of Halls Brook.
- New Boston Street Drainway – a mostly culverted network of storm sewers and drainage channels directing storm flows from New Boston Street and adjacent private properties to Halls Brook. A large portion of the New Boston Street Drainway was capped as part of the Industri-Plex Superfund Site Remedy. Prior to discharging into Halls Brook, the final approximate 700-foot section of the New Boston Street Drainway is an open channel,

The East Drainage Ditch, Landfill Creek, and the New Boston Street Drainway, located within the boundaries of the study area, are not being assessed in the baseline ecological risk assessment because these drainage features were previously addressed by the Industri-Plex Remedy or through other state/local actions or investigations. Sections of the East Drainage Ditch and Landfill Creek that are outside the boundaries of the study area are being investigated under the auspices of the MADEP BWSC and its cleanup regulations 310 CMR 40.0000 (the Massachusetts Contingency Plan [MCP]) as part of site investigations at the Olin Chemical Company facility in Wilmington, and under the MADEP Office of Solid Waste's regulations 310 CMR 19.000 through post-closure monitoring activities at the Woburn Landfill.

As mentioned above, there are two other storm water drainage systems that represent significant contributors of flow to the HBHA Pond. The first is the Atlantic Avenue Drainway, of which the flow originates at the North and upper South Ponds immediately north of the Industri-Plex Site. A portion of the flow from the North and South Ponds is diverted to the Created Wetland (a manmade wetland constructed as part of the Industri-plex remedy wetland mitigation and located immediately east of the Regional Transportation Center), which then flows under Atlantic Avenue, and merges with storm water runoff from the Regional Transportation Center detention basin effluent, and discharges at the northernmost end of the HBHA Pond. This channel is commonly referred to as the Atlantic Avenue Drainway.

The second drainage system discharges to HBHA Pond from the east. This drainage originates in a wetland system bordering Commerce Way, accepts storm discharges from commercial parking lots, and flows along the Boston Edison Company (BECO) Right of Way No. Nine. This drainway is commonly referred to as the BECO Drainway.

In addition to the surface water drainage to HBHA Pond, the hydrology and water quality are also influenced by groundwater discharge. As discussed in Section 3 of the RI, the general groundwater flow direction is southerly, mimicking the topography of the buried bedrock valley. Groundwater tends to flow from the west and east to the southeast and southwest towards the Aberjona River and HBHA. The groundwater elevation tends to decrease to the south consistent with decreasing bedrock elevations. In addition, groundwater elevations are greater to the west and east in comparison to the central portion of the study area, also consistent with bedrock elevations. Shallow groundwater flow is generally in the direction of the Aberjona River and associated tributaries, while groundwater flow at greater depths is parallel to the main buried valley.

The vertical gradients calculated at the study area vary slightly from downward to upward gradients depending on location, seasonal fluctuations, and precipitation patterns. The overall vertical gradients were evaluated and calculated for the shallow, intermediate, and deeper portions of the aquifer. Upward gradients were generally greater than the observed downward gradients. Based on these gradient calculations, the overburden aquifer generally discharges to HBHA Pond as well as other portions of the HBHA wetlands.

From the southern end of the HBHA Pond, a stream/wetland complex extends to Mishawum Road. The wetland consists of a channel, surrounded by emergent wetlands, and includes three larger areas of open water, south of HBHA Pond. Halls Brook flows from HBHA Pond for about 1,200 feet through an emergent wetland (dominated by *Phragmites*) and into the first small pond. Halls Brook flows from this small pond through a wooded swamp. This area is densely vegetated and provides overhanging woody vegetation along the banks, and fallen logs along the bank of the stream (Menzie-Cura, 1999). Halls Brook then flows into a second small pond which is surrounded by emergent *Phragmites* marsh. From the second pond, the main channel becomes indistinct, flowing through several channels in the emergent *Phragmites* marsh. The outer borders of this marsh are surrounded by a narrow band of scrub/shrub vegetation.

The largest of the open water areas is the third small pond, just north of, and visible from, Mishawum Road. The open water is surrounded by stands of emergent vegetation, dominated by *Phragmites* and cattails. The outlet of the pond discharges under Mishawum Road, and flows south to the confluence with the Aberjona River.

The Aberjona River flows into the study area from the east, and is conveyed south along the middle of Commerce Way. From Commerce Way, the river is directed west into an opern earth channel around a commercial area, and then abruptly turns south, parallel to the HBHA wetlands. The Aberjona River joins Halls Brook, just below Mishawum Road and flows south under Route 128. Upstream of the confluence, north of Mishawum Road, the Aberjona flows as a straight narrow stream of alternating runs and riffles within forested banks. In this reach, the channel of the Aberjona River parallels the lower portion of the HBHA wetlands, separated by a berm. The overhanging vegetation provide habitat and cover for fish and wildlife along this portion of the river.

A cart path is located on top of the berm along the east of HBHA Pond and most of HBHA Wetland. Soil samples collected by EPA (2002) at HB02-2 were collected along the edge of the cart path at the edge of the bordering dense stand of *Phragmites* which dominates the cover between the path and the wetland to the west.

The scrub/shrub vegetation similar to that along the banks of HBHA Pond, extends downstream to Mishawum Road, along the edge of the emergent marsh and bordering the berm. Common species observed included gray birch, (*Betula populifolia*), speckled alder (*Alnus rugosa*), common buckthorn (*Rhamnus cathartica*), elderberry (*Sambucus canadensis*), European buckthorn (*R. frangula*), big tooth aspen (*Populus grandidentata*), white pine (*Pinus strobus*), and russian olive (*Elaeagnus angustifolia*) (Menzie-Cura, 1999).

Waterfowl observed in the study area included mallard ducks and Canada Geese in the HBHA Pond and the open water of the downstream HBHA wetlands. Great Blue Heron have been observed in the open water of the pond just north of Mishawum Road and a green heron was observed in the open water of the wetland to the east, near Commerce Way. Evidence of muskrat in the stream downstream of HBHA Pond were observed during site visits in April 1999. White sucker and golden shiner were the most commonly observed fish species in both HBHA Pond and HBHA Wetland Pond 3. Brown bullhead, largemouth bass, and pumpkinseed were present in both ponds; carp were observed in HBHA Pond.

Based on correspondence with the Natural Heritage and Endangered Species Program (MNHESP, 2004, Appendix 7B.1) of the Massachusetts Division of Fisheries and Wildlife, there were no rare plants, animals, or exemplary natural communities known to occur in the study area downstream of the Industri-Plex Superfund Site.

1.1.2 Site Contaminants

1.1.2.1 Review of Data for Site Contaminants

Environmental data used in this hazard identification include surface water, sediment, soil, fish tissue, plant tissue, and invertebrate tissue samples collected during multiple sampling events between 1999 and 2004, at both study area and reference locations. The locations of the sediment sampling stations relative to reference stations are presented on Figure 2. Sampling locations for surface water, sediment, soil, and biota samples are presented on Figures 2 through

6, respectively. A complete discussion of the sampling activities conducted for the study area are contained in Section 2.0 of the MSGRP RI.

Surface water data collected prior to 1999 have not been quantitatively used in the risk assessment. These data are unlikely to represent current study area conditions and have been determined to be of insufficient quality for risk assessment purposes (e.g., data were not validated).

The following briefly describes the sampling data quantitatively evaluated in the baseline ecological risk assessment. Menzie-Cura conducted sediment, surface water, fish tissue, plant tissue, invertebrate tissue, and sediment toxicity sampling within the HBHA in 1999. Additional baseflow and storm event surface water sampling from the HBHA was conducted by Roux Associates in 2000/2001. Co-located baseflow and storm event surface water sampling, plus sampling within the Aberjona River, was conducted by TetraTech NUS in 2001/2002. In 2002/2003, USEPA conducted supplemental sediment sampling throughout the study area, including the collection of sediment core data from four locations. Soil sampling was conducted by USEPA at the same time to characterize residual contamination at the edges of the Boston Edison cap, and to determine whether periodic flooding of the HBHA had contaminated floodplain surface soils. A detailed reporting of these data can be found in Section 2 Appendices of the MSGRP RI.

1.1.2.2 Contaminant Fate and Transport

The contaminants under investigation are those that may have been transported to water bodies and wetlands (HBHA Pond, HBHA wetlands, and Aberjona River) on or near the study area via groundwater flow or surface water flow. Once in the water bodies, the contaminants may be taken up by vegetation or fauna, be deposited and remain adsorbed or complexed in the sediment, or be transported downstream. Fate and transport characteristics of study area-related contaminants are discussed in detail in Section 5 of the RI. The potential and evidence for study area-related contaminants to accumulate in the tissues of exposed organisms is addressed in the BERA.

1.1.3 Ecotoxicity and Potential Receptors

Based in the study area-specific information, evaluation of potential pathways, and inputs from reviewers during the development of the work plan, a number of receptors have identified for inclusion in the risk assessment. These include aquatic and semi-aquatic receptors of concern that are known to utilize habitat found within the study area. Those considered for assessment included: aquatic plants, benthic macroinvertebrates, warm water fish, waterfowl, piscivorus birds, aquatic mammals that feed on plants and macroinvertebrates, aquatic mammals that feed on fish, and small terrestrial mammals that inhabit the banks and borders of wetlands and waterways in the study area. The selection of specific receptors, based on complete exposure pathways, is discussed below. Plants growing in the sediments of study area waterbodies may take up COPCs in sediment pore water during water and nutrient uptake through their root surfaces. Free-floating aquatic plants, such as duckweed (*Lemna* spp.) and other emergent and submergent aquatic plants, may accumulate COPCs directly from surface water (Duxbury *et al.*, 1997).

Plant species are not utilized as indicators in this BERA because, for common species, only severe damage would be considered ecologically significant and no areas of visibly stressed vegetation were observed during field investigation work. In addition, no state-listed rare plant species are known to occur within the study area.

As a potential COPC source, plants may accumulate constituents and transfer them to herbivores. Detritus (dead plant and animal material) may also contain COPCs and be consumed by detritivores. Herbivores and detritivores may, in turn, become a source of COPC exposure for secondary consumers.

1.1.4 Complete Exposure Pathways

A complete exposure pathway exists if the ecological receptors have contact with the COPC in one or more medium and there is an exposure route (ingestion, dermal contact) to the receptor. Species groups most likely to receive potential exposures to study area COPCs are those whose activities frequently bring them into direct contact with sediment and surface water, that directly consume aquatic plants and/or detritus, or that feed upon species possessing one or both of these characteristics. These species groups are evaluated in this subsection to determine those potentially at risk of substantial exposure. This evaluation was used to determine the components of the aquatic and semi-aquatic food chain present in the study area, and those that may be most likely to receive potential exposures to study area COPCs. Species were selected as indicators for exposure evaluation to represent various components of the food chain in the river/wetland ecosystem.

1.1.4.1 Aquatic/Semi-aquatic Receptors. Aquatic invertebrates inhabiting study area waterbodies, such as amphipods, oligochaetes, crayfish, and the aquatic life stages of terrestrial insects, may be exposed to and accumulate COPCs in sediment and surface water. Benthic invertebrates, in particular, may have substantial exposure to COPCs in sediment. Exposure could result from direct contact with exposed outer membranes and respiratory surfaces, the direct ingestion of sediments during feeding activities, and the consumption of affected prey or detritus, depending upon the species' feeding habits. Many organisms, including fish, amphibians, reptiles, birds, and larger invertebrates, may be exposed to study area COPCs accumulated in the tissues of aquatic invertebrates.

As immature forms and adults, amphibians are potentially at risk of substantial exposure because of their close association with sediments and surface water. Most newts, toads, and salamanders are terrestrial hibernators, whereas most species of frogs hibernate under water in mud (DeGraaf and Rudis, 1983). Thus, frogs may be exposed to constituents in sediments during hibernation (although metabolism is greatly slowed) because of direct absorption through their relatively unprotected membranous skin. These organisms conduct considerable metabolic exchange directly through their skin (Schmidt-Nielsen, 1983). Salamanders, newts, toads, and frogs may consume earthworms, aquatic insects, and small fish or tadpoles (DeGraaf and Rudis, 1983). These prey may contain elevated levels of COPCs in their tissues. Amphibians may also ingest contaminated soil, sediment, and detritus during feeding activities.

Turtles and, to a lesser degree, snakes are also potentially at risk of substantial exposure. Turtles are mostly aquatic and spend considerable time on the bottom sediments of water bodies. Many snakes are sensitive to pollutants and have frequent contact with water, soil, or sediment (Hall, 1980; DeGraaf and Rudis, 1983). Turtles consume tadpoles, small fish, crustaceans, and some carrion (DeGraaf and Rudis, 1983). Semi-aquatic snakes also consume fish, frogs, aquatic insects, and salamanders, while more terrestrial species may consume large numbers of soil invertebrates, especially earthworms (DeGraaf and Rudis, 1983). These prey items may contain elevated levels of COPCs. Reptiles may also ingest affected soil, sediment, or detritus during feeding activities.

Although reptiles and amphibians are at risk of substantial exposure, there are limited data on the toxicological effects of COPCs on these organisms. In addition, available research focuses mainly on premetamorphic life stages with consideration of chemicals in the water column rather than in sediment. Little information is available to evaluate the effects exposure to sediment contaminants may have on adults. Therefore, reptiles and amphibians will not be used as indicator species in the BERA.

Fish, mammals, and birds inhabiting open water and wetland areas of the study area may also be exposed to COPCs via ingestion of contaminated tissue and/or abiotic media (*i.e.*, surface water and sediment), inhalation, or dermal contact. Of these pathways, greatest exposures are likely to be associated with the ingestion of contaminated tissue or direct ingestion of abiotic media.

Emergent marshes provide cover for juvenile fish (Moyle and Nichols, 1973; Scott and Crossman, 1973 *cited in* Stuber *et al.*, 1982a). Consequently, fish, at critical life stages, may be exposed to sediment and surface water COPCs in the emergent wetland areas of the study area.

Adult fish inhabiting the HBHA Pond or the HBHA Wetland Pond 3 may also be exposed to COPCs. Adult and juvenile fish may be exposed to COPCs through the consumption of a variety of prey items and abiotic media, or through absorption across gills or skin.

Warmwater fish species which may receive the greatest exposure to COPCs, and occur over large sections of the study area, are white sucker, brown bullhead, pumpkinseed, and largemouth bass. White sucker, which was commonly observed in both HBHA Pond and HBHA Wetland Pond 3, is primarily a bottom feeder. Preferred food items of white sucker include aquatic insect larvae, small mollusks, crustaceans, algae, and various terrestrial worms (Harlan *et al.*, 1987; McClan , 1978). Brown bullhead, which was also present in both ponds, is also an epibenthic fish, primarily consuming insect larvae, crustaceans, snails, small crayfish, worms, and small fish (Harlan *et al.*, 1987). Pumpkinseed were found in both ponds, and nesting along the shores of HBHA Pond. Pumpkinseed habitat includes littoral zones of lakes and ponds, and quiet vegetated pools of streams and small rivers. Pumpkinseeds forage among aquatic vegetation and in shallow sediments for invertebrates. They may also serve as prey for larger fish and piscivorus birds, such as herons. Largemouth bass were collected from both ponds. Juvenile largemouth bass feed on microcrustaceans, insects, and small fish, while adults feed primarily on fish and crayfish (Emig, 1966; Zweiacker and Summerfelt, 1974; Carlander, 1977 *in* Stuber *et al.*, 1982b).
Aquatic and semi-aquatic mammalian receptors at risk of substantial exposure to COPCs in sediment, surface water, and food items include muskrat, beaver, raccoon, and river otter. Muskrats may be exposed to COPCs in study area waterbodies through consumption of aquatic macrophytes, and to a lesser degree, through the consumption of animal matter. The roots and basal portions of aquatic plants make up most of the muskrat's diet, although shoots, bulbs, stems, and leaves are also eaten (Dozier *et al.*, 1950, 1953; Willner *et al.*, 1980; Svihla and Svihla, 1931 *cited in* USEPA, 1993d). Animals consumed by muskrats include crayfish, fish, frogs, turtles, young birds, and mollusks (Errington, 1939; Johnson, 1925; Willner *et al.*, 1980; Neves and Odum, 1989 *cited in* USEPA, 1993d).

Muskrats construct conical lodges or dig burrows in banks adjacent to aquatic habitats (Willner *et al.*, 1980 *cited in* Allen and Hoffman, 1984). Several studies summarized in USEPA (1993d) indicate that muskrats tend to remain in close proximity to their lodges or bank burrows. For example, one radiotelemetry study found that muskrats remained within 15 meters of their primary dwelling 50 percent of the time and only rarely traveled more than 150 meters from the dwelling (MacArthur, 1978 *cited in* USEPA, 1993d). Although habitat quality is variable, muskrats are likely present throughout the entire study area.

Beavers inhabit the Wells G&H 38-acre wetland, just to the south of the Industri-Plex Site. Although beaver may inhabit the study area, no beaver activity was observed during field investigations. Beavers are herbivores. Plants present in the study area that are likely consumed by beaver include willow, birch, and waterlily (Martin *et al.*, 1961). Lodges are typically constructed in quiet ponds, frequently behind dams. Lodges may also be built against banks, or bank dens with underwater openings may be used (Jones and Birney, 1988). Beavers carry sediments (mud) in their forefeet for the purpose of lodge and dam building (Novak, 1987).

Beaver and muskrats spend considerable time in contact with surface water and sediment. Hairless young may have considerable dermal exposure to sediment COPCs within lodges, and if sediment VOC concentrations were high, muskrat and beaver may potentially be impacted via inhalation of volatile COPCs within lodges. These species may also directly ingest sediment and surface water in the course of dam or lodge construction, and as they forage. Although inhalation and dermal absorption pathways are possibly complete for semi-aquatic mammalian receptors, these pathways are considered to be minor compared to dietary ingestion.

Raccoons are opportunistic omnivores (USEPA, 1993d) that will utilize marsh edges (Weller, 1981) as well as the stream and pond banks within the study area. Raccoon home ranges vary between 0.6 and 1.8 miles in diameter (180 to 1,630 acres) (Kaufmann, 1982; DeGraaf and Rudis, 1983). The percentage of time spent in the study area and the percentage of food obtained in the study area would substantially influence the degree of exposure to COPCs. Raccoons may ingest COPCs through the consumption of prey, ingestion of drinking water, and incidental ingestion of sediment.

River otter are medium-sized piscivorus mammals that would typically be found in riparian wetlands such as those present near the Hall's Brook and Aberjona River. Otters remain close to aquatic habitats such as lakes, streams, rivers, and wetlands. The average size of the home range of an adult otter is about 30 km. The diet of river otter is somewhat variable, but consists primarily of fish. Other prey, including crustaceans (crayfish), mollusks, amphibians (frogs and salamanders), reptiles (turtles and snakes), birds and small mammals may also be included. Since otter feed mainly on fish, they would be exposed to high concentrations of bioacummulative contaminants.

Of these aquatic and semi-aquatic mammals species, muskrat and beaver are likely to have the greatest exposure to COPCs due to greater contact with surface water and sediment. River otter may have higher exposures to bioaccumulative COPCs via the food chain. In addition, the home ranges of these aquatic species are contained within the boundaries of the study area, whereas the raccoon may spend a substantial percentage of time foraging outside of the boundaries of the study area.

Waterfowl and other aquatic bird species may also be exposed to COPCs. Species observed at the study area that may be exposed to COPCs include great blue heron, green heron, mallard, and Canada goose. These species, to varying degrees, may be exposed to COPCs through ingestion of plants, animals, detritus, sediment, and surface water. Based on several references summarized in USEPA (1993d), great blue heron primarily consume fish. The diet of the green heron is more mixed, consisting primarily of fish, crayfish, and aquatic insects (Martin *et al.*, 1961). Although both green heron and great blue heron have been observed within the study area, green herons are likely to spend a greater percentage of time within the study area than great blue herons (*i.e.*, the site use factor for individual green herons is greater).

Canada geese feed on grains, grass sprouts, and some aquatic vegetation (USEPA, 1993d). Mallards are surface feeding ducks that feed by dabbling and tipping up in shallow water, often filtering through soft mud for food (USEPA, 1993d). They feed primarily on seeds of aquatic plants and cultivated grains, although they also consume aquatic invertebrates, particularly during the breeding season (Jorde *et al.*, 1983; Swanson *et al.*, 1985 *cited in* USEPA, 1993d).

All four species have been observed foraging in the study area. Mallards may be present at the study area year round (Peterson, 1980). Green heron, great blue heron, and Canada geese may breed in eastern Massachusetts, but likely migrate in the fall. Of these bird species, exposure to COPCs is likely to be lowest for Canada geese. Canada geese primarily feed on vegetation, often in upland grassy areas, where the concentration of COPCs would be relatively low.

1.1.4.2 Terrestrial Receptors. Animals that inhabit the drier areas of the study area may also be exposed to COPCs in surficial sediment (*i.e.*, "soil" during drier periods) and surface water. In these sections of the study area, many organisms including mice, voles, shrews, upland birds, woodchucks, and skunks may be exposed to COPCs through the consumption of contaminated prey, incidental ingestion of soil, and consumption of dietary water. Shrews in particular may receive substantial exposure to COPCs due to diet, high ingestion rate (Morrison *et al.*, 1957 *cited in* USEPA, 1993d), and frequency of contact with surficial soils. Earthworms constitute a high proportion of the shrew diet (Whitaker and Ferraro, 1963; Hamilton, 1941 *cited in* USEPA, 1993d). Soil invertebrates, which are present in the drier portions of the study area, have significant direct contact with soil and may bioaccumulate COPCs (Beyer, 1990; Beyer and Stafford, 1993). Shrews may be exposed to COPCs in earthworm tissue and soil present in the gastrointestinal tract of earthworms. Shrews and other small mammals may accumulate COPCs and be consumed by higher-order predators such as raptors and owls.

1.1.5 Site Conceptual Model and Selected Receptor Species

The complete exposure pathways are summarized in the Site Conceptual Model (Figure 7). The conceptual model summarizes the release of contaminants from industrial and urban sources which have been transported through groundwater discharge, surface drainage, and sediment transport (secondary sources) to surface water, sediment, and riparian soil/sediment in the Aberjona River system. The primary receptors include organisms such as benthic invertebrates and aquatic plants directly exposed contaminants in sediment and surface water. The aquatic and semiaquatic receptors include organisms such as fish, predatory birds, waterfowl, semi-aquatic mammals, or terrestrial mammals exposed to riparian soil/sediment affected by sediment deposition of COPCs.

Based on the complete exposure pathways identified and the Site Conceptual Model, a group of indicator species or indicator communities were selected to evaluate risks associated with COPCs in the surface water, sediment, and biota of the study area. Receptors were also selected to be similar to those used in the BERA conducted for the Aberjona River Study area to the south which has similar habitats. The selected receptors include:

- Muskrat (*Ondatra zibethicus*), which has been observed in the study area, was chosen to represent the aquatic mammals inhabiting the study area. Muskrat and beaver may have a similar level of exposure. However, muskrats have been observed in the study area. Muskrats are primary consumers which feed on the basal portions and roots of aquatic vegetation. A small percentage of their diet may also consist of animals (*e.g.*, crayfish). Muskrats are important species in that they influence the species composition and density of vegetation within wetland areas, and they may heavily influence the percentage of open water. Muskrat eat-outs, as they are sometime referred to, create a mosaic of open water and vegetated areas that are valuable to waterfowl. Muskrats were observed in the wetlands downstream of HBHA Pond, and likely occur in all of the open water habitats within the study area.

- River Otter (*Lutra canadensis*) was selected to represent piscivorus mammals. This represents a mammalian predator feeding primarily on aquatic animals, particularly fish Its diet consists of species potentially with a high exposure to contaminants in surface water and sediments. River otter are typically be found close to lakes, wetland and streams. They prefer to forage in shallow water for slow-moving fish.

- Green heron (*Butorides striatus*) was selected to represent the semi-aquatic bird species inhabiting the study area. Green heron is an important top predator at the study area, feeding on crayfish, small fish, and rodents. It is also a migratory species. Green heron may occur at various locations throughout the study area.

- Mallard (*Anas platyrhynchos*) was selected to represent waterfowl within the study area, which constitute an important component of this aquatic system. The mallard was selected due to its common occurrence within the study area and because it may receive substantial exposure to sediments COPCs while filtering through soft mud. Mallards likely utilize all open water portions of the study area.

- Northern short-tailed shrew (*Blarina brevicauda*), a largely terrestrial species, was selected to represent the small mammal community that utilizes the drier areas of study area wetlands and upland areas that bordering the wetland habitats. Short-tailed shrews and other small mammals are a prey base for higher predators. Due to its small size, primarily animal diet, and high daily ingestion rate, shrews serve as a conservative indicator for other small mammal species inhabiting the study area.

- Largemouth bass (*Micropterus salmoides*) was selected to represent predatory fish in study area waterbodies. Largemouth bass, an important sport fish, feed primarily on fish and crayfish. They are at risk of substantial exposure to COPCs which bioaccumulate into the tissues of prey items. Fish sampling conducted in support of the BERA showed that largemouth bass inhabit the study area ponds.

- White sucker (*Catostomus commersoni*) was selected to represent epibenthic fish species present at the study area. White sucker feed on insect larvae, crustaceans, and worms via a method of foraging that involves substantial contact with sediments. White sucker composed a large portion of the fish community sampled in both HBHA Pond and HBHA Wetland Pond 3.

- Brown bullhead (*Ameiurus mebulosus*) is also an epibenthic fish present in the study area. Brown bullhead are bottom feeders that primarily consuming insect larvae, crustaceans, snails, small crayfish, worms, and small fish (Harlan *et al.*, 1987). Brown bullhead were collected in both study area ponds.

- Pumpkinseed (*Lepomis gibbosus*) was selected to represent a small foraging fish. Pumpkinseed habitat includes littoral zones of lakes and ponds, and quiet vegetated pools of streams and small rivers. Pumpkinseeds forage among aquatic vegetation and in shallow sediments for invertebrates and may be exposed to sediment COPCs both in foraging and during spawning in shallow hollows formed in sediment for nesting. These small fish serve as prey for larger predatory fish as well as piscivorus birds and mammals. Fish sampling conducted in support of the BERA showed that pumpkinseed were found throughout most of the study area.

- The benthic invertebrate community was also selected as an indicator group. Benthic invertebrates serve as a prey base for many aquatic and terrestrial species. Many benthic species also contribute to the breakdown of organic matter within an aquatic system.

Each of these indicator species or indicator communities may be exposed to substantial levels of contaminants through direct contact with and consumption of contaminated abiotic media or through the consumption of prey items that carry contaminant body burdens. The conceptual model shows the exposure pathways by which these species may be exposed to COPCs (Figure 7). This model allows evaluation of direct and indirect (food-chain) impacts on major components of the aquatic and semi-aquatic food chains in the study area.

2.0 SCREENING-LEVEL ECOLOGICAL EFFECTS EVALUATION

2.1 Sampling Locations

Surface water, sediment, and soil samples were collected within the Northern Study Area (Figures 3, 4, 5, and 6). Local and regional reference areas, representative of various study area habitats were collected from habitats outside of the main basin of the Aberjona River or in areas upgradient of the influence of the Wells G&H and Industri-Plex Superfund Sites (Figures 2 and 6). Reference stations are identified as stations 23 through 27, MC-01 through MC-04, MC-12, HB, and SA. Their locations are shown relative to the study area in Figure 2. A detailed discussion of the study area and reference sampling stations, sampling design, and analytical results are presented in Section 2 of the RI.

2.2 Surface Water Screening

Surface water samples were collected within the study area as part of three separate sampling efforts. Non-reference surface water sampling locations are shown on Figure 3. One set of samples were collected by Menzie-Cura Associates at a series of eight stations throughout the study area in June 1999; three stations within the HBHA Pond, four in the HBHA wetland, and one in the Aberjona River below the confluence with the HBHA wetland. A second set of samples were collected by Roux Associates between August 2000 and May 2001. Data utilized from this data set included samples from two stations, collected during both baseflow conditions and stormflow conditions. One station was located at the outlet of HBHA Pond (SW-04-IP) and one at the outlet of the HBHA wetland (SW-09-IP). The third set of data were collected by TetraTech NUS monthly between July 2001 and October 2002. Samples utilized from this data set included the monthly baseflow samples as well as six storm event samples collected in 2002. Data were utilized from one station at the outlet of HBHA Pond (SW-02-TT), one in the HBHA wetland (SW-04-TT), and one in the channel of the Aberjona River (AR) above the confluence of the HBHA wetland (SW-03-TT).

Eleven surface water samples were collected at ten reference locations (SW-23 through SW-27, MC-01 through MC-04, and MC-12). Two samples were collected at station MC-03 (Phillips Pond, one from shallow and one from deep water).

For surface water screening, all dates (including winter), both storm event and baseflow, and all depths in HBHA Pond were utilized. Only filtered samples were screened for metals, as described below. A comprehensive reporting of surface water data is provided in Section 2 of the RI. Data applicable to the ecological risk assessment are summarized in Appendix 7B.2.

Maximum detected levels of chemicals were compared to surface water quality criteria (Table 1). The sets of criteria used in the screening, in order of selection, consisted of:

- USEPA Ambient Water Quality Criteria (AWQC) (USEPA, 2002);
- USEPA Ecotox Thresholds for Surface Water based on Great Lakes Water Quality Initiative Tier II methodology (USEPA, 1996a), were used when a screening value from above was not available;
- Secondary Chronic Values (SCVs) for aquatic biota developed by Oak Ridge National Laboratory (Suter & Tsao, 1996), were used when a screening value from above was not available.

As presented in the AWQC document (USEPA, 2002), criteria for metals are based on a hardness value of 100 mg/L as $CaCO_3$ and are given as dissolved metals. When comparing metal concentrations to benchmarks, dissolved (filtered) concentrations were selected over total concentrations because dissolved concentrations correspond to NAWQC and represent the most bioavailable form of the metal; thus total concentrations of metals were not used for screening purposes. Equations presented in the AWQC document (USEPA, 2002) were used to adjust criteria to study area-specific conditions. The average hardness for all of the filtered surface water samples used in the screening (161 mg/L as $CaCO_3$) was used to adjust the AWQC criteria for hardness. Study area-wide hardness ranged from 28 to 973 mg/L for filtered samples. The majority of the hardness values observed were below 200 mg/L. The highest hardness values (four samples ranging from 527 to 973 mg/L) were observed in the deep water of HBHA Pond during summer stratification.

Frequency of detection was used as a screening tool for surface water. If a chemical was detected in less than or equal to 5% of all surface water samples, it was excluded from further consideration. However, infrequently detected analytes (<5% detection frequency) were further evaluated to determine whether or not they should be selected as COPCs and not eliminated solely on the basis of detection frequency. Reasons for retention of an infrequently detected chemical include (1) acute toxicity; (2) the potential for biomagnification and resultant toxic effects; (3) association with an area of habitat that is particularly important to fish or wildlife (e.g., a pond used by amphibians for reproduction, habitat meeting narrow spawning requirements, or an area with an important food source); or (4) a substantial presence within fish, crayfish, and/or plant tissues collected at the study area.

Ecological surface water screening criteria were unavailable for several chemicals. In all cases, chemicals lacking screening criteria but detected in a greater than 5% of all samples were included as COPCs in the BERA. Potassium, sodium, calcium, and magnesium were excluded from the screening process and eliminated as COPCs because they are nutrients and occur naturally at high concentrations.

Forty analytes were detected in surface water samples, and included VOCs, SVOCs, and inorganics (Table 1). Seven VOCs were detected, with detection frequencies ranging from 10% to 70%. One VOC was detected at concentrations exceeding its screening criterion (benzene) and one did not have a screening criterion (vinyl chloride), thus two VOCs were retained as COPCs. All maximum detected concentrations occurred at SW-MC-05 except for toluene which had a maximum detected value at SW-MC-06. Station SW-MC-05 was located at a deep pond location

(Sample collected at 10.8 ft) while SW-MC-06 was located at a shallow station in HBHA Pond (0-1 ft). Neither of the two VOCs selected as COPCs were detected at reference locations (Table 2).

Ten SVOCs were detected, with detection frequencies ranging from 3% to 33%. Two were eliminated based on detection frequency less than five percent (anthracene, and di-n-octylphthalate). Six were retained as COPCs; two exceeded screening criteria (benzoic acid, bis(2-ethylhexyl)phthalate) and four did not have screening criteria (cyclohexanone, fluoranthene, phenol, and pyrene). Four of the maximum detected concentrations of the COPCs occurred at SW-04-IP (the southern end of HBHA Pond) and six occurred at SW-09-IP (outlet of HBHA wetland). Among the six SVOCs selected as COPCs, only bis(2-ethylhexyl)phthalate was detected at reference locations, with its only detection at SW-MC-03.

No pesticides were detected in any of the surface water samples. No polychlorinated biphenyls (PCBs) were detected in any of the surface water samples.

Twenty-three inorganics were detected in surface water samples. Detection frequencies ranged from 6.7% to 100%. Seven inorganics were retained as COPCs; all were detected at concentrations exceeding screening criteria for dissolved concentrations (barium, cadmium, cobalt, iron, manganese, silver, and zinc). Locations of maximum detected concentrations were distributed among five surface water stations. Among the seven inorganics retained as COPCs, only barium, iron, manganese, and zinc were detected at reference locations.

Among the surface water contaminants eliminated from further consideration based on detection frequency alone, there were no special circumstances identified which would warrant inclusion in the BERA.

2.3 Sediment Screening

For the purpose of sediment screening, all data from the study area were combined since there are depositional areas at various locations throughout the study area where contaminants from upstream sources may have accumulated. Sampling locations included stations within the HBHA Pond, the stream channel and bordering wetlands in the HBHA wetland, BECO Drainway (stations BE-01 to BE-10), six stations along the upper reach of the Aberjona River above the confluence with the outlet from the HBHA wetlands (AR), and one station in the Aberjona River below the confluence with the HBHA wetlands (Figure 4). Data were collected in sampling rounds between June 1999 and July 2002. There were eight sediment samples collected in 1999 analyzed for organics (VOVs, SVOCs, pesticides, and PCBs). Inorganic data sets screened for sediments included 68 samples for most analytes and 51 samples for antimony and thallium.

The reference data set for the sediment samples included samples collected at 12 reference locations. For most analytes there were between 24 to 28 samples analyzed from reference samples. A comprehensive reporting of sediment data is provided in Section 2 of the RI; data applicable to the ecological risk assessment are summarized in Appendix 7B.3.

Maximum detected levels of chemicals were compared to sediment quality criteria (Table 3). The sets of criteria used in the screening, in order of selection, consisted of:

- USEPA Office of Solid Waste and Emergency Response Ecotox Thresholds (ETs) - Sediment Quality Criteria (SQCs), Sediment Quality Benchmarks (SQBs), or NOAA-Effects Range Low (ERLs) were used preferentially (USEPA, 1996a);

- Ontario Ministry of Environment and Energy (OMEE) Lowest Effects Levels (LELs) (Persuad *et al.*, 1993) were used when a screening value from above was not available;

- Oak Ridge National Laboratory (ORNL) Sediment Secondary Chronic Values (SCVs) (Jones *et al*., 1997) were used when a screening value from above was not available; and

- National Oceanic and Atmospheric Administration (NOAA) Threshold Effects Levels (TELs) (Buchman, 1999) were used when a screening value was not available in any of the above.

SQBs, SCVs, and SQCs, as presented in their respective documents, are based on a sediment organic carbon content of 1%. Although the actual organic carbon content of all sediment samples were greater than 1%, screening criterion were not adjusted upward in order to maintain a conservative screening process.

Frequency of detection was used as a screening tool for sediment. If a chemical was detected in less than or equal to 5% of all sediment samples, it was excluded from further consideration. However, infrequently detected analytes were further evaluated to determine whether or not they should be selected as COPCs and not eliminated solely on the basis of detection frequency. Reasons for retention of an infrequently detected chemical include (1) acute toxicity; (2) the potential for biomagnification and resultant toxic effects; (3) association with an area of habitat that is particularly important to fish or wildlife (e.g., habitat meeting narrow spawning requirements, or an area with an important food source); or (4) a substantial presence within fish, crayfish, and/or plant tissues collected at the study area.

Ecological sediment screening criteria were unavailable for several chemicals. In all cases, chemicals lacking screening criteria but detected in a greater than 5% of all samples were included initially as a COPC. Potassium, sodium, calcium, and magnesium were excluded from the screening process because they are nutrients and occur naturally at high concentrations.

Sixty-two analytes were detected in sediment samples, and included VOCs, SVOCs, pesticides, and inorganics (Tables 3 and 4). Thirteen VOCs were detected, with detection frequencies ranging from 12.5% to 100%. Seven VOCs were detected at concentrations exceeding screening criteria and were retained as COPCs: 1,1-dichloroethane, 2-butanone, acetone, benzene, carbon disulfide, m/p-xylene, and o-xylene. No screening value was available for vinyl chloride so it was also retained as a COPC. All maximum detections of VOCs occurred at MC-05 except for 1,1-dichloroethane, which had a maximum value detected at MC-07; both locations were at deep locations in HBHA Pond. Among the eight VOCs selected as COPCs, only 2-butanone, acetone, benzene, and carbon disulfide were detected at reference locations (Table 5); two maximum detections occurred at MC-02, one occurred at SD-25, and one occurred at SD-26.

Twenty-four SVOCs were detected, with detection frequencies ranging from 12.5% to 100%. Seventeen SVOCs were detected at concentrations exceeding screening criteria and were retained as COCs: 2-methylphenol, acenaphthene, acenaphthylene, anthracene, benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(g,h,i)perylene, benzo(k)fluoranthene, chrysene, dibenz(a,h)anthracene, fluoranthene, fluorene, indeno(1,2,3-cd)pyrene, naphthalene, phenanthrene, and pyrene. Three more were retained because they did not have screening values: carbazole, N-nitrosodiphenylamine, and phenol. Fifteen of the maximum detected concentrations of SVOCs selected as COPCs occurred in HBHA Pond, four occurred at MC-11 in the HBHA wetland pond, and one occurred in the Aberjona River channel below the confluence of the HBHA wetland (MC-13). Among the twenty SVOCs selected as COPCs, only 2-methylphenol, N-nitrosodiphenylamine, and phenol were not detected at reference locations (Table 5); fourteen maximum detected concentrations occurred at SD-24 and three occurred at SD-25.

Five pesticides were detected, with detection frequencies ranging from 12.5% to 37.5%. All five were detected at concentrations exceeding screening criteria and were retained as COPCs: 4,4'-DDD, 4,4'-DDE, 4,4'-DDT, alpha-chlordane, and gamma-chlordane. Three exceedances occurred in HBHA Pond, two occurred in the Aberjona River channel below the confluence of the HBHA wetland (MC-13). No PCB congeners were detected. All five pesticides selected as

COPCs were detected at the reference stations; two maximum detected concentrations occurred at MC-01, two occurred at SD-24, and one occurred at MC-03.

Twenty inorganics were detected in sediment samples. Thirteen inorganics were detected at concentrations exceeding screening criteria and were retained as COPCs: antimony, arsenic, cadmium, chromium, chromium VI, cobalt, copper, iron, lead, manganese, mercury, nickel, silver, thallium, vanadium, and zinc. Three more COPCs were retained, as they did not have screening values (barium, beryllium and selenium). Detection frequencies exceeded 90% for all analytes except antimony (73%), beryllium (76%), silver (81%), and thallium (61%). Locations of maximum detected concentrations were distributed among six individual stations. Five of the maximum detections (aluminum, antimony, chromium, selenium, silver and chromium VI) were within the Aberjona River channel above the confluence with the HBHA wetland. All metals selected as COPCs were detected at the reference stations, except chromium VI which was not analyzed (Table 5).

Among the sediment contaminants eliminated from further consideration based on detection frequency alone, there were no special circumstances identified which would warrant inclusion in the BERA.

In addition to the sediment data used for screening and for calculation of risk in the BERA, sediment core samples were collected at each of four locations (SC01 through SC04, Figure 4) throughout the HBHA wetland and analyzed for metals, hexavalent chromium, and TOC in February 2003. The samples collected at each location included one sample from each of the following depth intervals: 0 - 1 ft, 1 - 2 ft, 2 - 3 ft, and 3 - 4 ft. Metals concentrations from the surface sample (0-1 ft interval) were compared to sediment screening criteria (Table 3 of Appendix 7B.3). These core data were not combined with the other sediment data in the BERA as the depth interval (0-1 ft) was different than the rest of the sediment data (0 to 0.5 ft). The data are briefly discussed here to compare the concentrations in the surface sediment of the cores to the remainder of the study area sediment samples.

Among the metals in the sediment core samples, maximum concentrations of 10 analytes were measured above ecological benchmarks in one or more core sample, including arsenic, cadmium, chromium, cobalt, copper, iron, lead, manganese, mercury, and zinc. Each of these analytes were identified as COPCs in sediment in the BERA. Among the remaining metals, detected values of aluminum and nickel were below screening values in all of the cores. Antimony, beryllium, selenium, silver, and thallium were not detected in any of the core samples. There were no screening values for barium and vanadium.

The metals concentrations in the upper foot of the cores were within the range of concentrations observed in previous samples in the study area for each of the metals, with the exception of iron in SC02. The concentration of iron of 250,000 mg/kg in core SC02 slightly exceeded the study area maximum concentration of 233,000 mg/kg measured at HB02-11.

In conclusion, the metals concentrations in the sediment core data are consistent with sediment data collected in the Northern Study Area in support of the BERA. The BERA analysis regarding risk to ecological receptors was based on surface sediment samples (0 - 6 inches), as these represent the expected exposure point concentrations for ecological receptors. The surface data for metals (0 - 1 foot) collected in the sediment core data set were consistent with the data used to evaluate risk.

2.4 Soil Screening

Surface soil samples were collected within the study area at two stations (Figure 5). Ten soil samples were collected at the north end of HBHA Pond (station A6) and 13 samples were collected at a station along the east bank of the HBHA Wetland (station HB04). All of the 23 soil samples from the two stations were analyzed for inorganic chemicals. A comprehensive reporting of soil data is provided in Section 2 of the RI; data applicable to the ecological risk assessment are summarized in Appendix 7B.4.

Maximum detected levels of chemicals were compared to soil screening criteria (Table 6). The sets of criteria used in the screening consisted of the lowest of the following:

- USEPA Interim ECO-SSL - Ecological Soil Screening Levels, August 2003 (USEPA, 2003b-h);
- Oak Ridge National Laboratory (ORNL) Toxicological Benchmarks for Wildlife (Sample, Opresko, & Sutter, 1996);
- ORNL Toxicological Benchmarks for Screening Potential Effects on Terrestrial Plants (Efroymson, et al., 1997b);
- ORNL (Toxicological Benchmarks for Screening Potential Contaminants of Concern for Effects of Soil and Litter Invertebrates and Heterotrophic Process (Efroymson, Will, & Sutter, 1997a).

Frequency of detection was used as a screening tool for soil. If a chemical was detected in less than or equal to 5% of all soil samples, it was excluded from further consideration. However, infrequently detected analytes were further evaluated to determine whether or not they should be selected as COPCs and not eliminated solely on the basis of detection frequency. Reasons for retention of an infrequently detected chemical include: (1) acute toxicity; (2) the potential for biomagnification and resultant toxic effects; or (3) association with an area of habitat that is particularly important to wildlife.

Ecological soil screening criteria were available for chemicals except aluminum and iron. The toxicity of both of these metals is pH-dependent. As pH data was unavailable for soils at several of the stations, both of these metals were retained as COPCs at this screening step. Potassium, sodium, calcium, and magnesium were excluded from the screening process because they are nutrients and occur naturally at high concentrations.

Twenty COPCs were detected in soil samples and consisted of only inorganics (Table 6). Fifteen inorganics were detected at concentrations which exceeded screening benchmarks and were retained as COPCs, along with aluminum and iron: antimony, arsenic, barium, cadmium, chromium, chromium VI, copper, lead, manganese, mercury, selenium, silver, thallium, vanadium, and zinc. Detection frequencies exceeded 91% for all analytes except antimony (61%), beryllium (43%), cadmium (89%), silver (35%), thallium (83%), and chromium IV (65%). All exceedances occurred at location A6. Each of the COPCs identified in soil were also COPCs in sediment, with the exception of aluminum.

2.5 Chemicals of Potential Concern Summary

The screening by media resulted in the selection of 15 COPCs in surface water study area-wide. These included two VOCs, six SVOCs, and seven inorganics. Among the eight organic COPCs, five were retained due to a lack of a screening criterion. Benzene, benzoic acid, and bis(2-ethylhexyl)phthalate exceeded screening values in surface water. Among the seven inorganic COPCs, cadmium, iron, and zinc exceeded NAWQC values. Among the other four COPCs (barium, cobalt, manganese, and silver), there were no NAWQC values available for screening. Barium, cobalt, manganese, and silver exceeded Tier II values. Overall, five of the 15 surface water COPCs were retained because of a lack of screening benchmarks.

Fifty-two COPCs were selected in sediment including eight VOCs, 20 SVOCs, five pesticides/PCBs, and 19 inorganics (Table 4). Among the 52 COPCs in sediment, ten were selected because there was no available benchmarks against which they could be screened. In soils, beryllium, cobalt, and nickel were below screening values. All of the remaining 17 inorganics were above screening values and were retained as COPCs in soils.

3.0 SCREENING-LEVEL EXPOSURE ESTIMATE AND RISK CALCULATION

3.1 Exposure of Water Column Organisms (Fish and Invertebrates)

The exposure of surface water aquatic receptors (fish, aquatic invertebrates, and amphibian larvae) can be estimated by direct comparison to benchmarks of maximum measured concentrations in each habitat area. Among the 15 selected surface water COPCs, the maximum detected concentraions of ten COPCs exceeded corresponding benchmarks study area-wide; the remaining five COPCs were selected because no screening value was available. To provide more information on the locations of potential risks, surface water maximums were also screened for each habitat area. The results of these comparisons are discussed below. The maximum surface water concentrations (total, unfiltered) were also used in the food chain models to estimate exposure of wildlife species through surface water ingestion.

3.1.1 HBHA Pond

In HBHA Pond, the two VOC COPCs (benzene and vinyl chloride) were not detected in samples collected at shallow water stations (1-2 ft depths) (Table 7). Benzene was above the Tier II screening values in two deep samples collected in HBHA Pond in June 1999 at 9.8 and 10.8 feet deep, but was not detected in any other samples. There was no screening value for vinyl chloride, and it was detected in the same two samples (of 10 analyzed for VOCs) from the deep water of HBHA Pond at levels just above the detection limit.

Among the SVOC COPCs, only benzoic acid and bis(2-ethylhexyl)phthlate (BEHP) had available screening values. Benzoic acid was detected in three of nine samples study area-wide. The maximum concentration was located in the HBHA Pond outlet (SW-04-IP) and was the only sample above the screening value. BEHP was not detected in HBHA Pond samples, and the values for ½ the detection limit did not exceed screening values for any of samples within HBHA Pond. Among the other SVOCs in HBHA Pond that lacked screening values, the concentrations of both fluoranthene and pyrene were below detection limits. Based on this screening, among the SVOCs, only benzoic acid, cyclohexanone, and phenol were detected and are retained as COPCs in HBHA Pond.

Among the seven inorganic COPCs, maximum concentrations of barium, cadmium, cobalt, iron, manganese, silver, and zinc exceed screening values in HBHA Pond. These are retained as COPCs.

3.1.2 HBHA Wetland

Samples were collected in the HBHA wetland at stations MC-08 to MC-11, SW-04-TT, and SW-09-IP. The concentrations of both VOC COPCs were below detection limits. Among the SVOC COPCs, the concentrations of the two with screening values, only BEHP exceeded the screening levels. Among the inorganic COPCs, barium, iron, manganese, silver, and zinc exceeded screening values.

3.1.3 Aberjona River (AR upstream)

For surface water samples collected in the Aberjona River upstream of the HBHA wetland, only inorganics were analyzed. Barium, iron, and manganese exceeded surface water screening values; cadmium and silver were below detection limits, but one-half the maximum detection limit was above the screening value.

3.1.4 Aberjona River (AR downstream)

Surface water samples were collected in the Aberjona River downstream of the HBHA wetland at station MC-13. None of the organic COPCs were detected in these samples. Among the inorganic COPCs, barium, iron, and manganese exceeded screening values. Silver and cadmium were below detection limits, but one-half the maximum detection limit exceeded the screening values.

3.2 Exposure of Sediment-Dwelling Organisms (Benthic Invertebrates)

The exposure of sediment-dwelling organisms (benthic invertebrates) to sediment COPCs can be estimated by a direct comparison to benchmarks of maximum measured sediment concentrations. To provide more information on the locations of potential risks, sediment maximums were also screened for each habitat area. The results of these comparisons are presented in Table 8, and discussed below. The maximum sediment concentrations, applicable to each receptor, were also used in the food chain models to estimate exposure of wildlife species through incidental sediment ingestion.

Among the 62 analytes detected in sediment samples, maximum concentrations of 42 COPCs at non-reference locations exceeded screening criteria (Table 3), 10 were below screening values, and 10 had no screening value available. VOC concentrations were not measured in samples from AR or in the BE samples. All of the maximum VOC values, study area-wide, were observed in the deep samples from HBHA Pond. The concentrations measured in sediments of 1,1-dichloroethane, benzene, and xylene were below screening levels at all areas except the HBHA Pond deep (Table 8).

Among the 20 SVOCs detected, the majority (15) of the maximum detected values were observed in HBHA Pond. Concentrations of acenaphthene, fluorene and naphthalene were below screening levels at all areas except the HBHA Pond deep (Table 8). Among the five pesticides, maximum concentrations were observed in the HBHA Pond shallow sediment and at the downstream Aberjona River location (MC-13).

Among the 19 inorganics, ten of the maximum detected concentrations were observed in the HBHA wetland samples, four (arsenic, copper, lead, and zinc) in HBHA Pond samples, and five in Aberjona River upstream (AR) samples (antimony, chromium, selenium, silver, and chromium VI).

3.3 Estimates of Exposure through the Foodchain

Estimates of maximum exposures for wildlife were quantified for each of the selected receptor species. Dietary exposure models were used to make conservative estimates of exposure of each receptor species to each of the COPCs identified in the screening of sediment, surface water, and soil data from the study area against benchmarks. Doses were calculated by adding the ingested dose from drinking water, food items, and sediment that may be incidentally ingested during foraging. The input data and structure of the models are presented below.

3.3.1 Tissue Data for Wildlife Models

To assist in exposure estimation for the wildlife indicator species (muskrat, otter, heron, mallard, and shrew), fish, invertebrates, and plants were collected from the study area and analyzed to provide study area-specific estimates of concentrations of food items used in the dietary exposure models. These analyses were also conducted to support the evaluation of the fish and benthic invertebrate assessment endpoints, discussed further in sections 5.1 and 5.2. Analytical results are presented on a wet weight basis. Field methods, sampling locations, and analytical results for fish, invertebrates, and plants are discussed in detail in Section 2 of the RI. Surface water and sediment COPCs detected in plants, invertebrates, and fish are summarized in (Tables 9 to 11), respectively.

Plant Tissue. Samples were collected for plant tissue analysis at four locations (MC-06, MC-08, MC-09, and MC-11) in the study area and at two reference locations (MC-02 and MC-03, Phillips Pond and South Pond, respectively). Tissue samples were collected from several species of plants and were analyzed separately for stems and root portions of the plant. Plant tissue samples were analyzed for inorganics only. Locations of the plant tissues samples were paired with sediment sampling locations. A summary of the plant tissue analyses are presented in Table 9; a comprehensive reporting of plant tissue data is provided in Table 1 of Appendix 7B.5. For the screening-level exposure models, the maximum detected concentration among the applicable stations, or ½ the detection limit for non-detects, was used. In addition to utilization of these data in food-chain models, statistical analyses were performed on the data to evaluate if there were significant patterns in the concentration of inorganic COPCs in plant tissue between locations or among species sampled. This analysis and further discussion of the plant tissue data is presented in Section 5.1.

SVOCs were not measured in plant tissue samples. The dose of SVOCs in the maximum dietary models was estimated by multiplying the sediment concentration for the exposure area by an uptake factor. This uptake factor was derived from study area-specific data collected in the Aberjona River Study Area (EPA, 2003) and are presented in Table 2 of Appendix 7B.5. As

SVOCs were not measured in samples co-located with plant tissue samples, uptake factors for the Northern Study Area could not be calculated.

Invertebrate Tissue. Samples were collected for benthic macroinvertebrate tissue analysis at six study area locations (MC-06, MC-07, MC-08, MC-09, MC-11, and MC-13) and at four reference locations (MC-01, MC-02 and MC-03, and MC-04). Locations of the benthic macroinvertebrate tissues samples were paired with sediment sampling locations. Tissue samples were collected from several species of benthic macroinvertebrates and were analyzed for inorganics and SVOCs. Results of the invertebrate tissue analyses are presented in Table 10; a comprehensive reporting of invertebrate tissue data is provided in Table 1 of Appendix 7B.6. For the screening-level exposure models, the maximum detected concentration among the applicable stations, or ½ the maximum detection limit for non-detects, was used. In addition to utilization of these data in food-chain models, statistical analyses were performed on the data to evaluate if there were significant patterns in the concentration of COPCs in the invertebrate tissue between locations or among species sampled. This analysis and further discussion of the invertebrate tissue data is presented in Section 5.1.

Small Fish Tissue. Fish samples were collected for tissue analysis in HBHA Pond, HBHA Wetland, and at reference locations. USFWS conducted the sampling to both obtain samples for tissue analysis and assess the fish populations in the study area and reference locations. A copy of the USFWS report on the fish study is presented in Appendix 7B.7. Various fish species representing different trophic levels were collected. Pumpkinseed sunfish was selected as the small forage fish species and samples were analyzed whole, to represent fish tissue for piscivorus species in the food chain modeling. Results of the small fish tissue analyses are presented in Table 11, and raw data are presented in Table 1 of Appendix 7B.8. Tissue data for the other species were used for other endpoints and are presented and analyzed in Section 5.1.

For the screening-level exposure models, the maximum detected concentration among the applicable samples, or ½ the maximum detection limit for non-detects, was used. In addition to utilization of these data in food-chain models, statistical analyses were performed on the data to evaluate if there were significant patterns in the concentration of COPCs in the fish tissue between locations. This analysis and further discussion of the fish tissue data is presented in Section 5.1

3.3.2 Maximum Exposure Models for Wildlife Species. For muskrat, otter, heron, mallard, and shrew, the dose of each chemical that would be expected to be obtained from the ingestion of food (plant and/or animal) was estimated using the following equation:

$$Dose_{food} = FCR * C_{food} * ASUF * TSUF \qquad (1)$$

where,

$Dose_{food}$ = COPC ingested per day via food (mg COPC/kg body weight [wet]-day);

FCR = food consumption rate (kg food [wet]/kg body weight [wet]-day);

C_{food} = average or maximum COPC concentration in food (mg COPC/kg food [wet]);

ASUF = areal site use factor (unitless); and

TSUF = temporal site use factor (unitless).

In addition to the ingestion of COPCs accumulated in food items, receptors also may be exposed to chemicals through the ingestion of surface water. The following equation was used to calculate the dose of each chemical that each indicator species would be expected to obtain from the ingestion of surface water:

$$Dose_{water} = WCR * C_{water} * ASUF * TSUF \qquad (2)$$

where,

$Dose_{water}$ = COPC ingested per day via water (mg COPC/kg body weight [wet]-day);

WCR = surface water consumption rate (L of water/kg body weight [wet]-day);

C_{water} = average or maximum COPC concentration in surface water (mg COPC/L of water);

ASUF = areal site use factor (unitless); and

TSUF = temporal site use factor (unitless).

Receptors may also be exposed to COPCs through the ingestion of sediment while foraging. The following equation was used to estimate the dose of each COPC that each indicator species would be expected to obtain from the ingestion of sediment:

$$Dose_{sediment} = SCR * C_{sediment} * ASUF * TSUF \qquad (3)$$

where,

$Dose_{sediment}$ = COPC ingested per day via sediment (mg COPC/kg body weight [wet]-day);

SCR = sediment consumption rate (kg sediment [dry]/kg body weight [wet]-day);

$C_{sediment}$ = average or maximum COPC concentration in sediment (mg COPC/kg sediment [dry]);

ASUF = areal site use factor (unitless); and

TSUF = temporal site use factor (unitless).

Sediment ingestion rates were calculated by multiplying estimates of sediment ingestion found in the literature (expressed as a percentage of total food intake) by the food consumption rate. In cases where a species-specific sediment ingestion value was not available in the literature, a value from a species with similar foraging habits was used.

For the purposes of the maximum exposure models, an oral bioavailability factor of 1 was assumed for each chemical evaluated in the ingestion pathway. The use of a factor of 1 assumes that 100% of the chemical ingested in the diet is bioavailable and is utilized to be conservative.

Total COPC doses for muskrat, otter, heron, mallard, and shrew were calculated by summing doses via the ingestion of food, water, and sediment with the following equation:

$$Dose_{total} = Dose_{food} + Dose_{water} + Dose_{sediment} \qquad (4)$$

where,

$Dose_{total}$ = the total amount of COPC ingested per day (mg COPC/kg body weight [wet] - day).

Exposure parameters, values, and supporting citations for muskrat, otter, heron, mallard, and shrew are provided in Appendix 7C.1. Sets of surface water, sediment, plant, invertebrate, and small fish data used to estimate COPC exposures for muskrat, otter, heron, mallard, and shrew (Appendix 7C.2) are described below for the study area and for reference locations for each indicator species in the sections below (Table 13). Whether or not a particular station or sample was applicable to an indicator species was based on: (1) the likelihood that the indicator species may utilize the area covered by a station's samples; (2) the predominant habitat type covered by the sampling locations within the station; (3) the potential for the area to be utilized by terrestrial organisms (*i.e.*, shrews) during periods of drier weather; (4) the depth of the surface water; and (5) the impact of human disturbance on wildlife use. The selection process was conservative in that the likely intensity of use (based on habitat quality) was not heavily utilized in selecting applicable stations for each species.

Sampling stations consisted of groups of samples in similar habitats. However, the concept of a station deviates slightly from the typical concept of a sediment or surface water sampling station. In some locations within the Industri-Plex study area, the sample stations were defined by single GPS points and individual sediment samples co-located as close to this point as possible. However, in order to collect additional data for human health risk assessment, additional stations were subsequently added which expanded the concept of a station to include groups of samples collected in a similar habitat within a section of river or wetland. Sample groupings used for each media are listed in Tables 14 through 17. Sample locations are shown in Figures 2 through 6.

<u>Muskrat</u>

The home range for a muskrat is relatively small, and consequently, the risk evaluation for muskrat populations was conducted on a station by station basis. The maximum case scenario was calculated for all COPCs for the muskrat. Total dietary exposure to contaminants for muskrat are based on dietary exposure from ingestion of biota, and an additional small exposure to incidental sediment ingestion. Incidental ingestion of sediment (separate from the food consumption) is assumed to be equivalent to 3.3% of the total food ingestion. Dietary exposure for the muskrat was based on 90% plant tissue and 10% invertebrate tissue. Maximum station COPC concentrations in sediment were used to estimate incidental sediment ingestion. Exposure from surface water ingestion was based on the maximum COPC concentration in surface water for samples applied to the station (Table 15). Invertebrate tissue samples (10% of diet) collected in the study area were used to estimate maximum exposure concentrations (Table 16). Plant tissue concentrations (90% of diet) were estimated for each station from study area plant tissue data (Table 17). Since plant tissue was not analyzed for SVOCs or pesticides, the dietary doses from plant tissue were estimated using maximum station COPC concentrations in sediment multiplied by an uptake factor (Table 12).

Data used in the reference models included sediment data from all reference locations, except station MC-03 (Phillips Pond), since samples were taken at a depth of 9 to 13 feet (Table 14). All reference surface water samples, except samples collected at station MC-03, were used to calculate dietary exposure from water (Table 15). Samples from the three shallow reference locations were used to select the maximum reference invertebrate tissue concentration (Table 16). Plant tissue samples were collected at stations MC-02 and MC-03 (South Pond and Phillips Pond).

River Otter

River otter exposures were calculated for all samples collected within suitable habitat throughout the Industri-Plex Study area to compute a study area-wide scenario (Table 13), since the foraging ranges of this species is also relatively large. Home range for river otter are estimated at 30 km (18 miles) of shoreline (Melquist and Hornocker, 1983, in EPA 1993d). For the purpose of the maximum exposure estimate, it was assumed the river otter spends 100% of its time foraging in the study area. Stations with little open water (AR, BE, HB02-2, HB03-3, and HB04) were excluded from the river otter model for sediment ingestion since these do not represent typical foraging area for otter (Table 14). The maximum sediment COPC values were used to calculate incidental sediment exposure, and represented the highest detected COPC concentration from all selected sediment stations or was estimated as ½ the maximum detection limit. All surface water stations were used in the study area-wide model for estimating dietary ingestion of water (Table 15). Surface water samples included those collected at reference ponds and the Shawsheen River. All invertebrate data collected study area-wide were combined for the river otter evaluation (Table 16) in the study area (20% of diet). All small fish samples collected within the study area (Table 11) were used to estimate exposure for river otter fish ingestion (80% of diet). Since SVOCs were measured in benthic invertebrate tissue, but not in fish tissue, for SVOCs only, the dietary dose for otter is estimated using 100% invertebrates and no fish.

Data used in the reference models for river otter included sediment data and surface water data from all reference ponds and the Shawsheen River (Tables 14 and 15). The maximum concentrations among all small fish tissue data collected from Phillips Pond and South Pond were used for the reference otter model and all samples from South Pond and Phillips Pond (MC-02 and MC-03) were used to select the maximum reference invertebrate tissue concentration (Table 16).

Green Heron

Heron exposures were calculated for all samples collected within suitable habitat, throughout the Industri-Plex Study area to compute a study area-wide scenario, since the foraging ranges of this species is relatively large. Stations with little open water (AR, BE, HB02-2, HB03-3, and HB04) were excluded from the heron model for sediment ingestion, as well as stations with depths greater than three feet of water (Table 14). The maximum represented the highest detected COPC concentration from all selected sediment stations or was estimated as ½ the maximum detection limit. Total dietary exposure to contaminants for green heron are based on dietary exposure from ingestion of biota, and an additional small exposure to incidental sediment ingestion. Incidental ingestion of sediment (separate from the food consumption) is assumed to be equivalent to 1% of the total food ingestion. The maximum sediment COPC values were used to calculate incidental sediment exposure. Dietary exposure for the green heron was based on 55% plant tissue and 45% invertebrate tissue. All surface water stations were used in the study area-wide model for estimating dietary ingestion of water (Table 15). Samples included baseflow or storm event samples collected between April and October. All invertebrate data collected study area-wide, with the exception of station MC-07, were combined for the heron evaluation (Table 16). This sample was excluded as it was collected in deeper water (11.8 ft deep). The invertebrate tissue was used to estimate 55% of the heron diet. All small fish samples collected within the study area (Table 11) were used to estimate exposure for heron fish ingestion (45% of diet). Since SVOCs were measured in benthic invertebrate tissue, but not in fish tissue, for SVOCs only, the dietary dose for heron is based on 100% invertebrates and no fish.

Data used in the reference models for heron included sediment data from all reference locations, except station MC-03 (Phillips Pond), since samples were taken at a depth of 9 to 13 feet, and were too deep to represent incidental sediment ingestion (Table 14). The maximum concentrations among all small fish tissue data collected from South Pond and Phillips Pond were used for the reference heron model and all samples from the three shallow reference locations were used to select the maximum reference invertebrate tissue concentration (Table10). All reference surface water samples, except samples collected at station MC-03, were used to calculate dietary exposure from water (Table 15).

<u>Mallard</u>

The home range of mallards is large, and can range from 40 to 1,440 ha (96 to 3,556 acres) (USEPA, 1993d). Three exposure scenarios were evaluated for mallard: HBHA Pond, HBHA Wetland, and the study area-wide scenario. The surface water and sediment sampling stations used to estimate exposure concentrations for mallard are presented in Tables 14 and 15.

Total dietary exposure to contaminants for mallard are based on dietary exposure from ingestion of biota, and an additional small exposure to incidental sediment ingestion. Incidental ingestion of sediment (separate from the food consumption) is assumed to be equivalent to 3.3% of the total food ingestion. The maximum sediment COPC values were used to calculate incidental sediment exposure. Dietary exposure for the mallard was based on 33% plant tissue and 67% invertebrate tissue. Sediment samples with water depths less than three feet were used for estimation of incidental sediment ingestion for samples representing each of the three scenarios (Table 14). All invertebrate samples collected within the study area were used to estimate study area-wide exposure for mallard invertebrate ingestion, excluding the deep sample at MC-07 (Table 16). Data from MC-06 was used for HBHA Pond and three samples (MC-08, MC-09, and MC-11) were used to represent the HBHA wetland. All epilimnetic (shallow water) surface water stations were used in the study area-wide model for estimating dietary ingestion of water (Table 15). Samples included baseflow or storm event samples collected between April and October. A subset of these data were used to calculate surface water exposure for the HBHA Pond and wetland. Samples from four study area locations (MC-06, MC-08, MC-09, and MC-11) were used to estimate maximum exposures to plant tissue study area-wide for mallard (Table 17). Samples from MC-06 was used to represent HBHA Pond and samples MC-08, MC-09, and MC-11 were used for HBHA wetland. Since plant tissue was not analyzed for SVOCs or pesticides, the dietary doses from plant tissue were estimated using maximum station COPC concentrations in sediment multiplied by an uptake factor (Table 12).

Data used in the reference models included sediment data from all reference locations, except station MC-03 (Phillips Pond) since samples were taken at a depth of 9 to 13 feet (Table 14). All reference surface water samples, except samples collected at station MC-03, were used to calculate dietary exposure from water (Table 15). Samples from the three shallow reference locations were used to select the maximum reference invertebrate tissue concentration (Table 16). Reference plant tissue samples were collected at stations MC-02 and MC-03 (South Pond and Phillips Pond).

<u>Shrew</u>. The home range of the northern short-tailed shrew is small, on the order of less than one acre (USEPA, 1993d). Similar to muskrat, the risk evaluation for shrew populations was conducted on a station-by-station basis.

The majority of study area sampling stations were located in emergent wetland, pond, and river habitats where muskrat, mallard, and heron might be expected to occur, although habitat suitability varied widely. Fewer stations (A6, BE-1, BE-2, BE-4, HB02-2, HB03, and HB04) were relevant to the evaluation of shrew exposure (Table 13). Stations selected for shrews included upland habit and those in saturated areas with little standing water, that may be accessible to small mammals for foraging during periods of drier weather. Entirely aquatic stations, those with the balance of samples collected primarily in the center of the river channel or within impoundments, were not used to estimate COPC exposures to shrews.

In contrast to muskrat, heron, and mallard, study area-specific tissue data were not collected for the evaluation of COPC exposures to shrew. Study area-specific wetland sediment or soil data (Table 14) were used to estimate body burdens of prey for shrew. The concentration of COPCs in shrew prey (*i.e.*, earthworms) were estimated only for inorganic COPCs, as no sediment/soil data were collected for organics at any of the seven stations evaluated as shrew habitat. For the screening models, uptake factors for soil to earthworms are assumed to be 1.0. The concentration of COPCs in shrew prey (*i.e.*, earthworms) were estimated using the COPC concentrations in sediments, using the methods described above. For the maximum case models, earthworms were assumed to compose 87% of the shrew diet, with the remaining 13% attributed to incidental sediment ingestion.

Calculated earthworm COPC concentrations for each station used in the shrew model, based on maximum sediment COPC concentrations, are presented in Appendix 7C.3. Exposure from surface water ingestion was based on the maximum COPC concentration in surface water for samples applied to the station (Table 15). No surface water data were collected to represent upland stations A6 or BE, consequently, no dose from surface water is included in the total dose at these stations.

Total dose estimates for shrew at reference locations were calculated based on data from the three wetland reference locations, stations 24, HB, and SA. Sediment data from these three stations were pooled to estimate exposure at reference locations (Table 14) in order to have more data to calculate exposures than were available at each stations individually (only one sample each at stations HB and SA). Exposure from surface water ingestion for shrew was based on the COPC concentration in surface water for the one wetland station sampled (station 24, Table 15).

3.3.3 Maximum Exposure and Risk Estimate for Wildlife

Mammalian and avian toxicity reference values (TRVs) for COPCs were obtained from the literature (Appendix 7C.4). If available and appropriate, TRVs were selected which were associated with chronic exposures (*i.e.*, long duration exposures) and no adverse effects (NOAELs - no observed adverse effect levels), relating to reproduction or mortality. All TRVs for muskrat and shrew were based on laboratory tests with mammals. Few TRVs for SVOCs and VOCs based on other avian species were available for heron and mallard. The majority of the avian TRVs for pesticides/PCBs and metals were taken from studies with a variety of avian species. No adjustment factor was applied for interspecies extrapolations. It is sometimes recommended that the TRV be adjusted by a factor of 10 to account for inter-species extrapolations (Sample *et al.*, 1997). However, if the relative sensitivity of the two species is not known, this factor can add a large uncertainty, without much scientific basis. The uncertainty associated with TRVs is further discussed in Section 6.3.

For VOCs and SVOCs, laboratory tests reported in the literature were typically conducted for shorter periods of time than for pesticides/PCBs and metals. NOAELs or LOAELs (lowest observed adverse effect levels) associated with subchronic (or intermediate) exposures are generally reported for these chemical classes. When a suitable NOAEL was unavailable, LOAELs were used and adjusted downward with an uncertainty factor of 10. The LOAEL to NOAEL adjustment was the only calculation in which an uncertainty factor was used. No uncertainty factor was used to adjust subchronic NOAELs to chronic NOAELs.

In some cases, TRVs with endpoints relating to reproduction or mortality were not available in the literature. TRVs associated with other effects (systemic, hematological, carcinogenic, neurological, hepatic) are assumed to indirectly affect survival and/or reproductive capacity. Body weight scaling equations presented in Sample *et al*. (1996) and Opresko *et al*. (1994) were used to adjust test species TRVs to indicator species TRVs. Consistent with equations in Sample *et al.* (1996) no body scaling factors were used for avian species.

COPC daily dose estimates were compared to TRVs to evaluate the effect of exposure on indicator species. This comparison was quantified as follows:

$$\text{Hazard Quotient (HQ)} = \text{Dose COPC} / \text{TRV} \quad (8)$$

An HQ less than 1 indicates harm is unlikely, while an HQ greater than 1 suggests that a COPC is present at concentrations which may affect the survival or reproductive capacity of an exposed individual. The Hazard Index (HI), which is the sum of the HQs for a chemical class (VOCs, SVOCs, pesticide/PCBs, or inorganics), was also calculated for each indicator species. HQs and HIs for mammalian and avian indicator species are discussed below for maximum case, screening level exposure models. Model results are presented in Appendix 7C.5.

<u>Muskrat</u>

The results of the screening-level modeling for muskrat are shown in Appendix 7C.5 and Table 18. For the muskrat exposed to surface water and sediment, and ingesting plants and invertebrates, there were no NOAEL hazard quotients greater than 1 for any of the organic COPCs, with the exception of benzo(k)fluoranthere at reference locations and indeno(1,2,3-cd)pyrene at HB03-2. These results are limited to the extent that input data for some of the media was not available. The calculations for VOC exposure was based on surface water and sediment ingestion only, since no tissue concentrations of these COPCs were measured. For SVOCs and pesticides the exposures were based on surface water, sediment, and invertebrate tissue. As SVOCs and pesticides were not measured in plant tissue, the dietary doses from plant tissue were estimated using maximum station COPC concentration in sediment multiplied by an uptake factor. Chromium VI was only measured in sediment; dietary dose was estimated only on incidental sediment ingestion.

NOAEL HQs for the muskrat exceeded 1 for maximum doses for the majority of the inorganics at some or all of the stations modeled (Table18). There were no HQs above 1 at any station for beryllium, iron, mercury, nickel, silver, and chromium VI. HQs in the study area were equal to or below those at the reference location for barium, manganese, and thallium.

River otter

The results of the screening-level modeling for river otter are shown Appendix 7C.5 and Table 19. For the river otter exposed to surface water and sediment, and ingesting fish and invertebrates, there were no NOAEL hazard quotients greater than 1 for any of the organic COPCs. These results are limited to the extent that input data for some of the media was not available. The calculations for VOC and pesticide exposure was based on surface water and sediment ingestion only, since no tissue concentrations of these COPCs were measured. For SVOCs, the exposures for the majority of the stations were based on surface water, sediment, and invertebrate tissue only, as SVOCs were not measured in fish tissue. For SVOCs, the proportion of the diet for river otter was changed to 100% invertebrates (for which tissue data were available); there were no NOAEL HQs for SVOCs greater than 1. There is no evidence for potential risk to river otter from exposure to organic COPCs.

NOAEL HQs for the river otter exceeded 1 for maximum doses in the study area-wide model for arsenic (HQ = 2) and thallium (HQ = 3) (Table 19). In the reference model, the HQ was also equal to 3 for thallium.

Heron

The results of the screening-level modeling for heron are shown in Appendix 7C.5 and Table 20. For the heron exposed to surface water and sediment, and ingesting fish and invertebrates, there were no NOAEL hazard quotients greater than 1 for any of the organic COPCs. These results are limited to the extent that input data for some of the media was not available, and TRVs were available for only a few organic COPCs for avian species (Appendix 7C.4). For COPCs having no available avian reference values, risk was estimated based on mammalian TRVs. The uncertainty associated with this estimate is further discussed in Section 6.3. The calculations for VOC and pesticide exposure was based on surface water and sediment ingestion only, since no tissue concentrations of these COPCs were measured. For SVOCs, the exposures for the majority of the stations were based on surface water, sediment, and invertebrate tissue only, as SVOCs were not measured in fish tissue.

NOAEL HQs for the heron exceeded 1 for maximum doses in the study area-wide model for chromium, lead, mercury, thallium, and zinc (Table 20). In the reference model, HQs were equal to 3 for mercury (which was higher than the study area-wide HQ), and 3 for thallium (which was equal to the study area-wide HQ). HQs for all other COPCs and at reference locations were below 1 (Table 20).

<u>Mallard</u>

The results of the screening-level modeling for mallard are shown in Appendix 7C.5 and Table 21. For the mallard exposed to surface water and sediment, and ingesting plants and invertebrates, there were no NOAEL hazard quotients greater than 1 for any of the organic COPCs in the study area. The HQ for 4,4'-DDT was 2 at the reference locations. These results are limited to the extent that input data for some of the media was not available, and TRVs were available for only a few organic COPCs for avian species (Appendix 7C.4). For COPCs having no available avian reference values, risk was estimated based on mammalian TRVs. The uncertainty associated with this estimate is further discussed in Section 6.3. The calculations for VOC exposure was based on surface water and sediment ingestion only, since no tissue concentrations of these COPCs were measured. For SVOCs and pesticides, the exposures were based on surface water, sediment, invertebrate tissue, and estimated plant tissue concentrations. As SVOCs and pesticides were not measured in plant tissue, the dietary doses from plant tissue were estimated for eight SVOCs and all five pesticide COPCs using maximum station COPC concentrations in sediment multiplied by an uptake factor.

NOAEL HQs for the mallard exceeded 1 for maximum doses in the study area-wide model for aluminum, antimony, arsenic, chromium, lead, mercury, thallium, and zinc (Table 21). The same COPCs, with the exception of aluminum and antimony, had HQs in excess of 1 in HBHA Pond and HBHA wetland. In the reference model, HQs were equal to 2 for chromium, 4 for lead, 3 for mercury, 4 for thallium, and below 1 for all other COPCs (Table 21). The reference location HQ for thallium was equal to the HQ for study area-wide, HBHA Pond, and HBHA Wetland. This was due to utilizing a value for thallium in plant tissue equal to ½ the detection limit for all exposure areas. Plant tissue contributed the majority of the estimated dose of thallium at both reference and non-reference locations, resulting in similar estimates of dose of thallium in diet.

Shrew

The results of the screening-level modeling for shrew are shown in Appendix 7C.5 and Table 22. Exposure was calculated only for inorganic COPCs. For the shrew exposed to surface water and sediment, and ingesting earthworms as the sole prey item, there were no NOAEL hazard quotients greater than 1 for beryllium, cadmium, chromium VI, cobalt, iron, nickel, or silver. HQs in the study area were equal to or below those at the reference location for vanadium (Table 22).

4.0 BASELINE RISK ASSESSMENT PROBLEM FORMULATION

4.1 Refinement of COPCs

Screening consisted of comparing maximum study area media concentrations with conservative toxicologically based screening benchmarks, for sediment, water, and dietary doses to wildlife. In order to further refine the list of COPCs, additional components considered were comparisons to reference locations, frequency and magnitude of detection, and dietary considerations (USEPA, 2001).

4.1.1 COPCs in Surface Water

Among the preliminary COPCs identified in surface water, benzene and vinyl chloride were detected in the deep water of HBHA Pond only. The concentration of benzene exceeded the Tier

II value of 46 ug/L in two samples from the hypolimnion of HBHA Pond. Vinyl chloride was selected as a preliminary COPC in the screening as it was detected in HBHA Pond, and did not have a screening value. Vinyl chloride was also detected in two samples from deep water during summer stratification of HBHA Pond and was detected at concentrations of 1 and 3 ug/L, which were just above the detection limit of 1-2 ug/L. Both of these VOCs were selected as COPCs for the BERA, but are only further evaluated for risk posed to aquatic organisms (invertebrates and fish) in the hypolimnion of HBHA Pond (Table 23).

Among the SVOCs, four of the detected compounds were selected as preliminary COPCs in the screening because they lacked screening values. Two compounds with screening values that were exceeded in several samples were benzoic acid and bis(2-ethylhexyl)phthalate (BEHP). Benzoic acid exceeded the screening values in samples collected in HBHA Pond and HBHA wetland, and BEHP exceeded screening values in HBHA wetland. Both will be carried forward to assess risk to aquatic organisms. Among the other four SVOCs detected in surface water, fluoranthene and pyrene were detected in approximately 10% of the samples (Table 1) and were detected at very low concentrations (maximum of 0.6 ug/L). Based on the low frequency of detection and the low concentrations observed, these SVOCs are not carried forward in the BERA. Phenol was detected in a total of three out of 23 samples from HBHA Pond and HBHA wetlands. Cyclohexanone was detected in six out of 23 samples, with maximum detected concentrations of 290 ug/L in HBHA Pond and 120 ug/L in HBHA wetland. Based on these results phenol and cyclohexanone will be retained as COPCs in surface water and further evaluated for risk posed to aquatic organisms (invertebrates and fish).

Among the seven inorganics selected in surface water as preliminary COPCs, barium, cadmium, cobalt, manganese, silver, and zinc all exceeded dissolved surface water criteria. However, cobalt and zinc did not exceed criteria in the Aberjona River upstream (station AR) or in the Aberjona River downstream (station MC-13). With the exception of these locations, barium, cadmium, cobalt, manganese, silver, and zinc are carried forward in the BERA. Silver exceeded the Tier II chronic value in HBHA wetland, and it was below detection limits in HBHA Pond (shallow), AR, and AR downstream (MC-13). However, the detection limits exceeded the benchmarks, so silver was retained at these locations as well. Iron concentrations in surface water exceeded the NAWQC of 1,000 ug/L in all areas. One reference sample also exceeded the NAWQC for iron (SW-02, South Pond). Iron, however, is naturally occurring in surface water and a nutrient in organisms and typically poses little ecological risk. Iron will not be carried forward as a COPC in surface water in the BERA.

COPCs identified in surface water are similar to those identified in sediment. Potential exposures to COPCs are further evaluated for potential risk to receptors through food chain modeling and for aquatic exposure through evaluations of fish tissue concentrations of metals.

4.1.2 COPCs in Sediment

Table 8 summarizes the exceedances of sediment benchmarks by maximum sediment concentrations in each habitat area. Among the VOCs, the highest concentrations in sediments were observed in deep samples from HBHA Pond. In HBHA Pond, seven VOCs exceed benchmarks. No sediment screening value was available for vinyl chloride; since it was only detected in HBHA Pond deep, it is only carried forward as a COPC for sediment invertebrates in this exposure area (Table 24).

Although some observed pesticide concentrations were below reference values, all pesticides exceeding screening criteria were retained in the BERA, since these chemicals are bioaccumulative. In HBHA wetland 4,4'-DDD, 4,4'-DDE, 4,4'-DDT and alpha-chlordane were below screening levels. 4,4'-DDE exceeded screening values only in the downstream AR (MC-13) samples.

Reference concentrations of PAHs were high at SD-24. Using these values, none of the non-reference locations were dropped based on comparison to background. However, 2-methylphenol was not detected at any locations outside of the HBHA Pond deep samples, and both acenaphthene and naphthalene were below screening values at all locations except the deep samples in HBHA Pond. Carbazole is retained because there was no available screening benchmark.

The majority of the inorganics were retained for the BERA, for the evaluation of risk to sediment invertebrates. Among those COPCs having screening criteria, maximum values of antimony, arsenic, cadmium, chromium, copper, iron, lead, mercury, nickel, silver, and zinc exceeded their respective benchmarks for all sediment groups (Table 8). Cobalt was below screening levels for AR and BE samples. Maximum manganese concentrations were below screening values for HBHA Pond shallow samples and AR upstream. Barium, beryllium, selenium, thallium, and chromium IV were carried forward since they did not have screening values and consistently exceeded reference concentrations. Vanadium also did not have a screening value. However, all area maximums were below observed concentrations at reference locations, and vanadium is therefore not carried forward in the BERA.

4.1.3 COPCs in Soil

Two stations were classified as upland soils, rather than wetland sediment. The concentrations of inorganic chemicals the soils at these stations (A6 and HB04) were screened against conservative soil screening criteria (Table 6). Based on the screening, fifteen inorganics were retained as COPCs: antimony, arsenic, barium, cadmium, chromium, chromium VI, copper, lead, manganese, mercury, selenium, silver, thallium, vanadium, and zinc. The maximum concentrations of aluminum and iron were 6,530 mg/kg in sample A6-12 and 66,900 mg/kg in sample A6-08, respectively. Toxicity of aluminum and iron in soils are dependent on pH, and these compounds are generally not toxic to biota at pH values above 5.5. There were no soil pH data from these specific samples. The range of pH for five surface samples (colleted from 0-1 feet or 1-2 feet in depth) at station A6 ranged from 5.2 to 6.72. Two samples were below 5.5 with pH values of 5.21 and 5.44. These data indicate that at the majority of the stations, the soil conditions would be such that aluminum and iron would not be soluble and would have low potential for toxicity.

In addition, the typical range of aluminum in soils is from 1% to 30% (10,000 to 300,000 mg/kg) (USEPA, 2003b), and the observed maximum concentration of aluminum (6,530 mg/kg) is at the lower end of this range. The typical iron concentrations in soils range from 0.2% to 55% (20,000 to 550,000 mg/kg) (Bodek et al., 1988), and concentrations can vary significantly, even within localized areas, due to soil types and the presence of other sources. The observed maximum iron concentrations in soil (66,900 mg/kg) was also at the lower end of the typical range. Due to the relatively low potential toxicity of these metals, and the observed values falling within normal soil conditions, neither are carried forward as a COPC in soil.

4.1.4 COPCs for Wildlife Receptors

Among the mammalian and avian wildlife species there were no VOCs, SVOCs, or pesticides with NOAEL HQs greater than 1 in the dietary exposure models for study area exposure areas. The single exception was an HQ of 2 for muskrat from indeno(1,2,3-cd)pyrene in exposure area HB03-2. These results are limited in that VOCs and pesticides were not measured in plant, invertebrate, or fish tissue. SVOCs were measured in benthic invertebrate tissue, but not in fish tissue or plant tissue. For a sub-set of the SVOCs, plant tissue concentrations were estimated from sediment concentrations and an uptake factor. The concentration of chromium VI was measured only in sediment/soil data. No data on tissue concentrations of chromium VI were collected. For all receptor models, with the exception of shrew, the exposure to chromium VI was estimated based on incidental sediment ingestion only. For shrew, prey concentration of chromium VI was estimated from soil concentrations. These limitations of the data for the selection of COPCs for exposure of wildlife species are further discussed in the uncertainty section (6.3).

There were one or more inorganic COPCs that were identified in the food chain modeling to potentially pose a risk to wildlife species (Table 25). Among the inorganics, there were was no chronic TRV available for iron for the wildlife receptors. However, iron is naturally occurring in sediments, often in high concentrations. Since it is a nutrient in the diet, iron typically poses little ecological risk to wildlife. It is assumed that iron in sediments does not pose a risk to any of the five wildlife receptor species, and consequently is not further analyzed in the BERA.

The muskrat maximum exposure models showed NOAEL HQs greater than 1 for all of the inorganics, except beryllium, mercury, nickel, and silver. In addition, the NOAEL HQ values were higher for the reference area than for all station locations for barium and manganese. NOAEL HQ values in the study area for thallium were equal to those for the reference data. Based on these results barium, beryllium, manganese, mercury, nickel, silver, and thallium are not considered to pose a risk to muskrat from dietary exposures, and these chemicals are not carried forward in the BERA. A maximum exposure HQ of 2 for muskrat from indeno(1,2,3-cd)pyrene was observed in exposure area HB03-2. Only one of 10 exposure areas had an HQ greater than 1. As no other SVOCs exceeded benchmarks, the magnitude of the exceedance was very low, and occurred in only one area, the risk of exposure to SVOCs is assumed to be negligible. Indeno(1,2,3-cd)pyrene is not carried forward as a COPC.

The otter maximum exposure models showed NOAEL HQs greater than 1 for only arsenic (13) and thallium (3). The HQ for thallium was 3 for both the study area-wide model and for the reference model. Since the study area-wide model exposure was not higher than the reference model exposure for thallium, it is not carried forward as a COPC for otter.

The heron maximum exposure models showed NOAEL HQs greater than 1 for chromium, lead, mercury, thallium, and zinc. With the exception of thallium, each of these are retained as COPCs potentially posing a risk to heron. The HQ for thallium was 3 for both the study area-wide model and for the reference model. Since the study area-wide model exposure was not higher than the reference model exposure for thallium, it was not carried forward as a COPC for heron. Although the HQ for mercury for the reference model (3) was greater than the study area-wide model (2), mercury was further evaluated, since it can be biomagnified in the food chain.

The mallard maximum exposure models showed NOAEL HQs greater than 1 for aluminum (study area-wide), antimony, arsenic, chromium, lead, mercury, and zinc. Although the HQs for the maximum reference models exceeded 1 for chromium, lead, and mercury (2, 4, and 3, respectively), the HQs for the study area-wide model exceeded these values. Consequently, each of these was carried forward as a COPC for mallard. The HQ for thallium was 4 for both the study area-wide models and for the reference model. Since the study area-wide model exposure was not higher than the reference model exposures for thallium, it is not carried forward as a COPC for heron.

The shrew maximum exposure models showed NOAEL HQs greater than 1 at one or more station for aluminum, antimony, arsenic, barium, chromium, copper, lead, manganese, mercury, selenium, thallium, vanadium, and zinc. However, since the HQ values for vanadium in the reference wetland were greater than the HQs calculated for study area stations, it was not considered to pose a significant risk to shrew. The HQs for aluminum were at or just above the reference values. Toxicological information indicates that aluminum must be in a soluble form in order to be toxic to biota, and at normal soil pH (above 5.5), aluminum is typically not toxic. Based on these factors (high natural concentrations, minor exceedance of reference concentrations, and low toxicity to biota), aluminum was also excluded as a COPC for shrew. The remaining 11 inorganics carried forward as COPCs for shrew at one or more stations include: antimony, arsenic, barium, chromium, copper, lead, manganese, mercury, selenium, thallium, and zinc.

4.2 Ecotoxicity

An ecotoxicity literature review has been performed for COPCs carried forward in the BERA, and is discussed in the following subsections.

4.2.1 Volatile Organic Compounds

VOCs were detected in a limited number of sediment samples collected at the study area. In surface water, VOCs (benzene and vinyl chloride) were detected in the hypolimnion (deep water) of HBHA Pond only. VOCs are often not found within surficial sediment and surface water due to their tendency to volatilize into the air. At high concentrations, VOCs in surface water and sediment may impact aquatic receptors. These volatile compounds, when present at high concentrations, may also present an inhalation hazard to animals which inhabit confined areas (*e.g.*, burrows or lodges). VOCs do not bioaccumulate to any significant degree, and therefore, do not pose a risk to environmental receptors via trophic transfer. None of the VOCs were identified as COPCs for the wildlife food chain models.

Acetone. Using the EqP approach to develop a sediment criterion, Jones *et al.* (1997) calculated an SCV of 8.7 µg/kg for freshwater aquatic organisms, based on 1% sediment organic carbon content. The EqP-derived LCV for sediment was 3,000 µg/kg for fish and 9.1 µg/kg for freshwater daphnids, also based on 1% sediment organic carbon content (Jones *et al.*, 1997). In mammals, NTP (1991, *cited in* ATSDR, 1994a) reported a NOAEL of 1,700 mg/kg-day for mice. Data summarized in Jones *et al.* (1997) indicates that toxicity values based on the EqP approach likely overestimate exposure because acetone is a polar organic compound.

Benzene. The lowest chronic value (LCV) reported for daphnids in freshwater is >98,000 µg/L (Suter and Tsao, 1996). The estimated LCV for aquatic plants is approximately 525,000 µg/L. Using the equilibrium partitioning (EqP) approach to develop a sediment quality benchmark (SQB), EPA calculated an SQB of 57 µg/kg for freshwater aquatic organisms, based on 1% sediment organic carbon content (USEPA, 1996a). The EqP-derived secondary chronic value (SCV) for benzene (Jones *et al.*, 1997) is 160 µg/kg, and the lowest chronic value for daphnids was reported at >120,000 µg/kg (Jones *et al.*, 1997). Narwot and Staples (1979, *cited in* Sample *et al.*, 1996) reported LOAEL of 263.6 mg/kg-day with a reproductive endpoint for mice exposed to benzene via oral gavage during six days of gestation. Data summarized in Jones *et al.* (1990) for freshwater and saltwater studies were limited, but indicated that bioaccumulation of benzene was not significant, with a maximum BCF value of 225 in fresh water.

2-Butanone. As estimated by Suter and Tsao (1996), LCVs for daphnids and fish in freshwater are approximately 1,400,000 µg/L and 280,000 µg/L, respectively. Using the EqP approach to develop a sediment quality criterion, Jones *et al.* (1997) calculated an SCV of 270 µg/kg for freshwater aquatic organisms, based on 1% sediment organic carbon content. Like acetone, 2-butanone is a polar organic compound. Therefore, the SCV for 2-butanone may be lower than the level which would be associated with an impact to ecological receptors. Ralston *et al.* (1985, *cited in* ATSDR, 1992c) reported a NOAEL of 173 mg/kg-day with a neurological endpoint in rats orally exposed to 2-butanone for 13 weeks.

Carbon disulfide. In sediments, the EqP-derived SCV for carbon disulfide was 0.85 µg/kg (Jones *et al*., 1997), and the lowest chronic value for daphnids was reported as 230 µg/kg, and 8,800 µg/kg for fish based on 1% sediment organic carbon (Jones *et al*., 1997). In mammals, Hoffman and Klapperstuck (1990, *cited in* ATSDR, 1996) reported an LOAEL of 253 mg/kg-day with a cardiovascular endpoint for rats exposed to carbon disulfide for four weeks. The predicted rate of biodegradation of carbon disulfide in water is negligible compared with its rate of volatilization from surface water (ATSDR, 1996). Due to its low affinity for sorption to organic substances and its low organic carbon/water partition coefficient (log K_{oc}) of 1.79, very little carbon disulfide is likely to partition to or remain in sediment. Carbon disulfide is expected to have little or no tendency to bioaccumulate or biomagnify in biota due to its relatively low log K_{ow} value of 2.14 and rapid metabolism in most animals (Beauchamp *et al*., 1983).

1,1-Dichloroethane. Suter and Tsao (1996) report an SCV for 1,1-dichloroethane of 47 µg/L in surface water. Jones *et al.* (1997) report an SCV of 27 µg/kg in sediment, with an LCV for fish of 8,400 µg/kg based on 1% sediment organic carbon. NCI (1977 *cited in* ATSDR, 1990a) reported a NOAEL of 764 mg/kg-day with a systemic endpoint in rats exposed to 1,1-dichloroethane for 78 weeks. The Henry's law constant for l,l-dichloroethane (4.2×10^{-2} atm-m^3/mol) suggests that 1,1-dichloroethane should partition rapidly to the atmosphere (ATSDR, 1990). No information was found regarding partitioning of l,l-dichloroethane from the water column onto sediments. However, analogs of the compound (i.e., dichloromethane, trichloromethane, and l,l,l-trichloroethane) have not been found to concentrate selectively onto sediments (Dilling *et al*., 1975 *cited in* ATSDR, 1990a). The K_{oc} values for these compounds are similar to the K_{oc} for l,l-dichloroethane; therefore, adsorption onto sediments would not be considered significant for l,l-dichloroethane (USEPA, 1985b *cited in* ATSDR, 1990a). Based on the K_{ow}, (6.6), bioconcentration is not expected (ATSDR 1990a).

Vinyl Chloride. No toxicological studies were identified which described effects of vinyl chloride in aquatic systems. Feron *et al*. (1981, *cited in* Sample *et al*., 1996) reported a LOAEL of 1.7 mg/kg-day with a mortality endpoint in rats orally exposed. Biomagnification of vinyl chloride in terrestrial and aquatic food chains is improbable due to its high volatility and rapid metabolism by higher trophic level organisms, as well as its low log K_{ow} value of 1.62 (ATSDR, 1997).

Xylene. In surface water, the LCV reported for fish was estimated at 62,308 µg/L (Suter and Tsao, 1996). EPA used the EqP approach to develop an SQB of 0.025 mg/kg, based on 1% sediment organic carbon content (USEPA, 1996a). The EqP-derived SCV for xylene is 160 µg/kg (Jones, *et al*., 1997). Marks *et al*. (1982, *cited in* ATSDR, 1995c) reported a NOAEL of 2.1 mg/kg-day with a reproductive endpoint in mice exposed to xylene over their entire lifetime. Xylene is not expected to bioconcentrate in higher animal systems due to its low log K_{ow} value of 3.12 and rapid oxidation during metabolism (ATSDR, 1995c).

4.2.2 Semi-Volatile Organic Compounds (Polycyclic Aromatic Hydrocarbons)

SVOCs were detected in samples collected from the study area. BEHP and benzoic acid, cyclohexanone and phenol were identified as COPCs in surface water. PAHs were identified as COPCs in sediment, as well as 2-methylphenol, carbazole, N-nitrosodipnenylamine, and phenol.

Polycyclic Aromatic Hydrocarbons (PAHs). In aquatic environments, PAHs rapidly become adsorbed to organic and inorganic particulate materials and are deposited in sediments (Neff, 1985). Once adsorbed to sediment, PAHs have limited bioavailability to aquatic organisms (Neff, 1985). However, PAHs deposited in sediments can be toxic to benthic invertebrates. The Great Lakes Assessment and Remediation of Contaminated Sediments (ARCS) Program calculated a threshold effect concentration (TEC) of 3,553 µg/kg of total PAHs in sediment (*cited in* Jones *et al.*, 1997). The Ontario Ministry of the Environment (OMOE) calculated a lowest effects level (LEL) of 4,000 µg/kg of total PAHs in sediment based on a 1% sediment organic carbon content (*cited in* Jones *et al.*, 1997). In sediment toxicity tests with the tubificid *Limnodrilus hoffmeisteri*, Lotufo and Fleeger (1996) observed a median lethal phenanthrene level of 298 mg/kg (with a sediment organic carbon content of 0.7%). In the same study, pyrene levels up to 841 mg/kg were not acutely toxic. Decreases in tubificid reproduction were observed at much lower levels; IC_{25} s (concentration associated with a 25% inhibition in measured endpoint relative to the control) of 40.5 mg/kg and 59.1 mg/kg were reported for phenanthrene and pyrene, respectively.

Sediment-associated PAHs can be accumulated by bottom-dwelling invertebrates (Eisler, 1987b). Great Lakes sediments contaminated with elevated levels of PAHs were reported by Eadie *et al.* (1983 *cited in* Eisler, 1987b) to be the source of body burdens in bottom-dwelling invertebrates. Lake *et al.* (1985 *cited in* Eisler, 1987b) found that marine mussels (*Mytilus edulis*) and annelids (*Nereis virens*) exposed for 28 days to sediments heavily contaminated with PAHs accumulated up to 1,000 times more than controls.

Fish may be at risk from chronic exposure to PAHs. PAH contamination in sediments has been shown to be correlated with histopathological abnormalities at a number of sites (Baumann *et al.*, 1982; Malins *et al.*,1984 *cited in* Pastorak *et al.*,1994). Reductions in fish populations from acute exposures to areas of high PAH contamination is less likely; some fish species demonstrate avoidance of areas with high PAH contamination (North *et al.*, 1964; Rice, 1973 *cited in* Pastorok *et al.*, 1994).

The capacity to metabolize PAHs varies among organisms. Varanasi *et al.* (1985, *cited in* ATSDR, 1995b) ranked the extent of benzo(a)pyrene metabolism by aquatic organisms as follows: fish > shrimp > amphipod > crustaceans > mussels. One reason mussels are ranked last may be that they show no or limited mixed function oxidase (MFO) activity. MFO is an enzyme system responsible for the initiation of metabolism of various lipophilic organic compounds, including PAHs (Neff, 1985).

The primary effect of PAH exposure in mammalian laboratory species is tumor development (Eisler, 1987b). EPA has classified benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(k)fluoranthene, chrysene, dibenz(a,h)anthracene, and indeno(1,2,3-cd)pyrene as carcinogens (ATSDR, 1995b). Acenaphthylene, anthracene, benzo(g,h,i)perylene, fluoranthene, fluorene, phenanthrene, and pyrene are not classified as carcinogens (ATSDR, 1995b). Neal and Rigdon (1967, *cited in* ATSDR, 1995b) reported a NOAEL of 1.3 mg/kg-day with a cancer endpoint in mice for carcinogenic PAHs, and a NOAEL of 133.3 mg/kg-day with a reproductive endpoint in mice for noncarcinogenic PAHs. For the development of hazard quotients, carbazole was grouped with noncarcinogenic PAHs. EPA (USEPA, 1988b *cited in* ATSDR, 1995b) reported a NOAEL of 1,000 mg/kg-day with a reproductive endpoint for mice orally exposed to anthracene. EPA (1988b, *cited in* ATSDR, 1995b) reported a NOAEL of 125 mg/kg-day with a hepatic endpoint for mice orally exposed to fluoranthene; this value was also used as a surrogate value for fluorene.

In aquatic environments, exposure to ultraviolet light can photomodify some PAHs to products with increased polarity, water solubility, and toxicity compared to the parent compound (Duxbury *et al.*, 1997). Ireland *et al.* (1996) showed that the photoinduced toxicity of PAHs to the daphnid, *Ceriodaphnia dubia*, occurred frequently during low-flow conditions and wet weather runoff, and was reduced in turbid conditions. In studies on the marine amphipod, *Rhepoxynius abronius*, ultraviolet radiation exposure enhanced the toxicity of fluoranthene and pyrene in sediments, but did not affect the toxicity of acenaphthene and phenanthrene (Swartz *et al.*, 1997). Pelletier *et al.* (1997) found that the phototoxicity of individual PAHs (anthracene, fluoranthene, pyrene) to marine bivalves (*Mulinia lateralis*) and marine shrimp (*Mysidopsis bahia*) were 12 to >50,000 times that of conventional toxicity.

Bis(2-ethylhexyl)phthalate (BEHP). In surface water, the LCV reported for freshwater daphnids is 912 μg/L (Suter and Tsao, 1996). In sediment, Jones *et al*. (1997) used the EqP approach to calculate an SCV of 890,000 μg/kg based on 1% sediment organic carbon content. Lamb *et al*. (1987, *cited in* Sample *et al*., 1996) reported a LOAEL of 183.3 mg/kg with a reproductive endpoint in mice. Peakall, *et al*. (1974, *cited in* Sample *et al*., 1996) reported a NOAEL of 1.1 mg/kg-day with a reproductive effect in ringed doves. Bioaccumulation of BEHP in terrestrial and aquatic food chains may occur due to its moderate log K_{ow} value (7.6).

Benzoic Acid. The LCV reported for freshwater fish is 12,976 μg/L (Suter and Tsao, 1996). Benzoic acid appears to exhibit low to moderate toxicity in aquatic environments (WHO, 2000). Immobilization of freshwater daphnids has been demonstrated to be pH dependent, with a lower 24-h EC_{50} (102 mg/L) at acidic pH (WHO, 2000). For the freshwater fish golden ide (*Leuciscus idus*), a 48-h LC_{50} of 460 mg/L has been determined (WHO, 2000). Developmental effects have been found in frog *(Xenopus*) embryos at a concentration of 433 mg/L (96-h EC_{50} for malformation). In mammals, a four-generation study with rats reported no effects on life span, growth rate, or organ weights were reported after dosing with up to 1% in the diet (approximately 500 mg/kg body weight per day) (Kieckebusch and Lang, 1960 *cited in* World Health Organization (WHO), 2000). The WHO (2000) concluded that benzoic acid has a low to moderate potential to bioaccumulate, based in part on a low log K_{ow} of 1.87.

Carbazole. No toxicological studies were identified which described effects of carbazole in sediment. However, in surface water, Brooke (1991 *cited in* USEPA, 2004) reported an EC_{50} (concentration at which 50% of the individuals are affected) of 3,350 μg/L for the water flea (*Daphnia magna*) and LC_{50}s ranging from 930 ug/L to greater than 1,500 μg/L for the fathead minnow (*Pimephales promelas*). Carbazole bioconcentration factors reported for *Daphnia magna* and *Daphnia pulex* are 113.4 μg/L (Newsted and Giesy, 1987, *cited in* USEPA, 2004a) and 65 µg/L (Southworth, 1979 *cited in* USEPA, 2004a). Data on the toxicity of carbazole to wildlife are limited. Dermal treatment with benzo(a)carbazole at a NOAEL dose of 250 mg/kg-day resulted in significant reductions in maternal body weight gain and food consumption in pregnant Sprague-Dawley rats (Dutson *et al.,* 1997).

Cyclohexanone. No toxicological studies were identified which described effects of cyclohexanone in aquatic systems. Lijinsky and Kovatch (1986 *cited in* USEPA, 2004a) reported a NOAEL of 462 mg/kg-day with a weight loss endpoint in rats orally exposed to cyclohexanone in drinking water. Biomagnification of cyclohexanone in aquatic food chains is improbable due to its high volatility 5 mg/HG at 20 ^{o}C), as well as its low log K_{ow} value of 0.81 (Montgomery, 1996).

2-Methylphenol. In sediment, Jones *et al*. (1997) used the EqP approach to calculate an SCV for 2-methylphenol (also known as *o*-cresol) of 12 µg/kg based on 1% sediment organic carbon content, and reported an LCV for fish of 440 µg/kg and 1,200 µg/kg for daphnids. Lamb *et al*. (1987, *cited in* Sample *et al*., 1996) reported a LOAEL of 183.3 mg/kg-day with a reproductive endpoint in mice. Due to its low affinity for sorption to organic substances and its low organic carbon/water partition coefficient (log K_{oc}) of 1.70, very little 2-methylphenol is likely to partition to or remain in sediment (ATSDR, 1992d). 2-Methylphenol is expected to have little or no tendency to bioaccumulate or biomagnify in biota due to its relatively low log K_{ow} value of 1.96 (ATSDR, 1992d).

N-Nitrosodiphenylamine. No toxicological studies were identified which described effects of n-nitrosodiphenylamine in aquatic systems. NCI (1979 *cited in* ATSDR, 1993) reported a NOAEL of 150 mg/kg-day with a systemic endpoint in rats orally exposed to n-nitrosodiphenylamine. In the aquatic environment, n-nitrosodiphenylamine partitions from the water column to sediments and suspended particulate organic matter. It is subject to photolysis and biodegradation (ATSDR, 1993). The log K_{ow} (2.57) indicates a low potential for bioaccumulation (Barrows *et al.,*1980 *cited in* ATSDR, 1993). The relatively low bioconcentration potential suggest that biomagnification in the aquatic food chain is not a major environmental fate process (Barrows *et al.*, 1980 *cited in* ATSDR, 1993).

Phenol. Using the EqP approach, Jones *et al.* (1997) used chronic NAWQC criterion and a 1% sediment organic carbon content to calculate a criterion value of 31 μg/kg for phenol. Jones *et al.* (1997) also calculated a SCV of <57 μg/L for fish and 570 μg/L for freshwater daphnids. In surface water, the chronic toxicity value for phenol calculated by the Great Lakes Water Quality Initiative was reported at 110 mg/L (Suter and Tsao, 1996). In mammals, NCI (1980, *cited in* ATSDR, 1998) reported a NOAEL of 1,566 mg/kg-day with a reproductive endpoint for rats. Kishino and Kobayashi (1995, *cited in* USACE, 2004) reported a NOED for phenol of 66 mg/kg wet weight for whole body tissue with a mortality endpoint in goldfish (*Carassium auratus*). Since the pK_a of phenol is 9.68 at 20/C, it will exist in a partially dissociated state in water and moist soil; thus phenol is not expected to adsorb to sediment in the water column (HSDB, 1998, *cited in* ATSDR, 1998). Phenols react relatively rapidly in sunlit natural water via reaction with photochemically produced hydroxyl radicals and peroxyl radicals. Phenol is readily biodegradable in natural water, provided the concentration is not high enough to cause significant inhibition. Phenol is not expected to bioconcentrate significantly in aquatic organisms (ATSDR, 1998). Reported log bioconcentration factors (BCF) in fish for phenol include 0.28 for goldfish, (Kobayashi, *et al*., 1979 *cited in* ATSDR, 1998) and 1.3 for golden orfe (Freitag *et al*., 1984 *cited in* ATSDR, 1998).

4.2.3 Pesticides

Five pesticides were selected as COPCs in sediment samples collected in the study area. No PCBs were retained as COPCs. Pesticides are known to bioaccumulate, and thus may pose a risk to environmental receptors via trophic transfer, even at low sediment or surface water concentrations.

DDD, DDE, and DDT. DDD and DDE are breakdown products of DDT. When released in water, DDT primarily partitions into the sediment or adsorbs strongly to particulate matter in the water column, although some DDT may volatilize (ATSDR, 2002c). In freshwater, the reported LCV for DDT is 0.016 µg/L for daphnids, 0.3 µg/L for aquatic plants, and 0.73 µg/L for fish (Jones, *et al*., 1997). The fish LCV value for DDD is 1.69 µg/L (Jones, *et al*., 1997). In sediment, Jones *et al*. (1997) used the EqP approach for sediment to develop an SCV of 340 µg/L for DDT and 110 µg/L for DDD, based on 1% sediment organic carbon content. Jones, *et al*. (1997) also reported LCVs of 420 µg/kg DDT for freshwater daphnids, 19,000µg/k DDT for fish, and 17,000 µg/kg DDD for fish. Sediment effects range-low values (ERL) for DDT, DDD, DDE and total DDT are 1 µg/kg, 2 µg/kg, 2.2 µg/kg, and 1.58 µg/kg, respectively (Long and Morgan, 1990; Long *et al.,* 1995). Effects range-median (ERM) values for these same compounds are 7 µg/kg, 20 µg/kg, 27 µg/kg, and 46.1 µg/kg, respectively (Long and Morgan, 1990; Long *et al.*,1995).

LC_{50} values between 0.2 and 1,230 µg/L have been reported for aquatic invertebrates exposed to DDT, DDD, and DDE (USEPA, 1980c). Other 96-hr LC_{50}s, reported in Mayer and Ellersieck (1986), include 1 µg/L for the freshwater amphipod, *Gammarus lacustris*, and 4 µg/L for the isopod, *Asellus brevicaudus*, as well as 70 µg/L and 10 µg/L for mosquito larvae (*Culex fatigans* and *Anopheles albimanus*, respectively) and 7 µg/L stonefly (*Pteronarcys california*). The most sensitive freshwater invertebrate reported by Mayer and Ellersieck (1986) was the water flea, *Daphnia pulex*, with a 48-hr EC_{50} of 0.36 µg/L, based on immobilization.

DDT is absorbed by fish directly through the skin, and is also accumulated by invertebrates, which are prey for many fish species. A range of LC_{50} values from 2 µg/L to 21 µg/L are given for freshwater fish in Connell and Miller (1984). LC_{50} values for freshwater fish species are also presented in Mayer and Ellersieck (1986). The most sensitive species reported was largemouth bass (*Micropterus salmoides*), with a 96-hr LC_{50} of 1.5 µg/L. Other LC_{50}s reported by Mayer and Ellersieck (1986) were 4.9 µg/L, 5.0 µg/L, and 15 µg/L for bluegill sunfish (*L. macrochirus*), black bullhead (*Ictalurus melas*), and channel catfish (*Ictalurus punctatus*), respectively. Chronic effects have been observed at 0.74 µg/L in chronic life-cycle tests with fathead minnows (*P. promelas*) (USEPA, 1980c).

Median lethal dietary concentrations ranging from 651 mg/kg to 1,160 mg/kg have been reported for northern short-tailed shrews (*Blarina brevicauda*) exposed to DDT for up to 17 days via a corn oil diet (Blus, 1978). In studies reported in Klaassen *et al*. (1996), female rats given single DDT doses of 50 mg/kg showed estrogenic effects. Klaassen also reported an LD_{50} of 113 mg/kg for male rats fed DDT, and an LD_{50} of 880 mg/kg for rats fed DDE. At sufficiently high doses, DDT can kill organisms by interfering with sodium ion passages in the brain (Connell and Miller, 1984 *cited in* ATSDR, 2002c). Fitzhugh (1948, *cited in* Sample *et al*., 1996) reported a NOAEL of 0.80 mg/kg-day with a reproductive endpoint in rats orally exposed to DDT for two years. Tomatis *et al*. (1974, *cited in* ATSDR, 2002c) reported a LOAEL of 42.6 mg/kg-day with a cancer endpoint in mice orally exposed to DDD for 130 weeks. NCI (1978, *cited in* ATSDR, 2002c) reported a LOAEL of 27 mg/kg-day with a cancer endpoint for mice orally exposed to DDE.

Hudson *et al*. (1984) reported acute median lethal dosages for birds including LD_{50}s of >2,240 mg/kg for mallard ducks (*Anas platyrhynchos*) and 841 mg/kg for Japanese quail (*Coturnix coturnix*). Following chronic exposures to DDT dietary concentrations of 100 mg/kg, half of the exposed adult mallards died in about one year. DDE caused eggshell thinning in birds eating food contaminated with DDT and its breakdown products. Weimeyer *et al*. (1970) found 15% eggshell thinning in American kestrels (*Falco sparverius*) given a daily DDE dietary concentration of 3 mg/kg for less than 7 months. Stendell *et al*. (1989) fed pine voles (*Microtus pinetorum*) from pesticide-contaminated apple orchards to three captive American kestrels. The pine voles contained 48 mg/kg DDE, 3.5 mg/kg DDD, and 14.1 mg/kg DDT. One of the kestrels, which died at 31 days, contained 147 mg/kg DDE in the carcass (wet weight). Anderson, *et al*. (1975, *cited in* Sample, *et al*., 1996) reported a LOAEL for DDT of 0.03 mg/kg-day with a reproductive endpoint in brown pelicans orally exposed to DDT for five years.

High log K_{ow} values for DDT (6.91), DDE (6.51), and DDD (6.02) support empirical studies that these compounds have a high potential to bioaccumulate in tissue. This lipophilic property, combined with an extremely long half-life is responsible for its bioaccumulation in aquatic organisms and high bioconcentration in the food chain. Ford and Hill (1991) reported biomagnification of DDT, DDE, and DDD from soil and sediment to mosquito fish, a secondary consumer. Leblanc (1995) reported that the concentrations in plankton, invertebrates, fish, and fish-eating birds were 0.04 mg/kg, 0.3 mg/kg, 4.1 mg/kg, and 24 mg/kg, on a whole body basis. Evans *et al*. reported that DDE biomagnified 28.7 times in average concentrations from plankton to fish and 21 times from sediment to amphipods in Lake Michigan (Evans *et al.,* 1991 *cited in* ATSDR, 2002c).

Chlordane (*alpha* and *gamma*). The EPA chronic national ambient water quality criterion (NAWQC) for chlordane is 0.00043 µg/L (USEPA, 2002). Suter and Tsao (1996) calculated an LCV for fish and freshwater daphnids of 1.6 µg/L and 1.09 µg/L for non-daphnid invertebrates. In sediments, Jones *et al*. (1997) used the EqP approach to calculate a value of 2.8 mg/kg based on the NAWQC for chlordane and a 1% sediment organic carbon content. Jones *et al*. (1997) also used the EqP to calculate an SCV of 26 mg/kg for fish and freshwater daphnids, and 18 mg/Kg for non-daphnid invertebrates, based on 1% sediment organic carbon content (Jones *et al*., 1997). The World Health Organization (1984) reported a NOEAL of 4.58 mg/kg-day for α-chlordane and 4.60 mg/kg-day for γ-chlordane with reproductive endpoints in mice. Stickel *et al*. (1983, *cited in* Sample *et al*., 1996) reported a NOAEL of 2.14 mg/kg for both forms of chlordane with reproductive endpoints in red-winged blackbirds.

Chlordane is expected to bioconcentrate in higher animal systems due to the high log K_{ow} value (6.22 for mixed isomers) (ATSDR, 1994b). Chlordane bioconcentrates in fresh water fish with a bioconcentration factor of 18,500 in rainbow trout (ATSDR, 1994b). Chlordane is taken up by rooted aquatic vascular plants both from the water and from the sediment; the bioconcentration factor of chlordane in the submerged vascular plant, *Hydrillu verticillatu*, was 1.06 (Hinman and Klaine, 1992). Chlordane also bioconcentrates in plant roots from contaminated sediment and translocates into the shoots.

4.2.4 Inorganics

Metals which were detected in sediment and surface water, and retained as COPCs are discussed below.

Aluminum. The LCV for fish is 3,288 µg/L based on 28-day embryo-larval tests with the fathead minnow, *P. promelas* (USEPA, 1988a *cited in* Suter and Tsao, 1996). The lowest chronic value for daphnids was reported as 1,900 µg/L (McCauley *et al.*, 1986 *cited in* Suter and Tsao, 1996). An aluminum BCF of 268 has been reported for brook trout (*Salvelinus fontinalis*). BCFs for water fleas (*D. magna*) exposed to aluminum chloride ranged from 320 to 1,020 (Cleveland *et al.*, 1991 *cited in* ATSDR, 1999a; Havas, 1985 *cited in* USEPA, 2004a).

Availability and toxicity of aluminum is strongly influenced by the pH of the medium, due to the influence of pH on solubility. Soluble and toxic forms of aluminum are generally found in soils only at soil pH values less than 5.5 (USEPA, 2003b).

For mammals and birds, evidence suggests that the direct toxic potential of aluminum is low compared to that of many other inorganics; mammals and birds can effectively limit the absorption of aluminum and effectively excrete any excess (Scheuhammer, 1987). Bioavailabilty of aluminum is strongly influenced by the form ingested and also by the presence of other material in the digestive system. Significant accumulation in tissues of mice required dietary doses in excess of 200 mg/kg-day (Scheuhammer, 1987). Oral doses of 52 mg/kg-day aluminum in water over four months resulted in no chronic reproductive impairment in rats (Domingo *et al.*, 1987 *cited in* ATSDR, 1999a). Oral LD_{50} values for several animal species range from 380 mg/kg to 780 mg/kg (USEPA, 1985a). Ondreicka, *et al.* (1966, *cited in* Sample, *et al.*, 1996) reported a NOAEL of 52 mg/kg-day for a 39-day study administering oral doses of aluminum to mice. Carriere *et al.* (1986, *cited in* Sample, *et al.*, 1996) reported a NOAEL of 107.9 mg/kg-day with a reproductive endpoint for a four month study in which ringed doves were orally exposed to aluminum in their diet.

Antimony. LCVs for antimony exposure to fathead minnow, *P. promelas*, and daphnid, *D. magna*, of 1,600 and 5,400 µg/L, respectively, were reported by Kimball (no date, *cited in* Suter and Tsao, 1996). For freshwater algae (*Selenastrum capricornutum*), inhibition of the synthesis of chlorophyll-α was observed during antimony exposure of 610 µg/L (96-hour EC_{50}) (USEPA, 1978 *cited in* Suter and Tsao, 1996). Accumulation of antimony has been demonstrated in marine invertebrates (Amiard, 1973 *cited in* USEPA, 2004b).

Antimony can be toxic to mammals. Testing by Shroeder *et al.* (1968 *cited in* Sample *et al.*, 1996) showed that a chronic oral dose of 5 mg/L in drinking water caused a reduction in the median life span of female mice. A chronic NOAEL of 0.125 mg/kg-day was estimated from the chronic LOAEL for mice. A TRV for mammalian species (Eco-SSL) based on the geometric mean of the NOAEL values for reproduction and growth in a number of studies was calculated at 13.3 mg/kg-day (USEPA, 2003c).

Arsenic. The toxicity of arsenic depends on its form: trivalent arsenic [As (III)] leads to enzyme inhibition, while pentavalent arsenic [As (V)] probably acts by interfering with formation of ATP (uncoupling of oxidative phosphorylation) (Eisler, 1988a). Arsenic has been found to be carcinogenic, teratogenic, embryotoxic, and fetotoxic in laboratory species (NAS, 1980).

The USEPA chronic NAWQC for total arsenic is 340 µg/L (USEPA, 2002). Suter and Tsao (1996) reported a LCV of 2,962 µg/L for fish, 914.1 µg/L for daphnids, and 2,320 µg/L for aquatic plants. Sediment ERL and ERM values for arsenic are 8.2 mg/kg and 70 mg/kg, respectively (Long *et al.*, 1995). The OMEE LEL and Severe Effect Level (SEL) for arsenic are at 6 mg/kg and 33 mg/kg, respectively (Persaud *et al.*, 1993). Consensus-based threshold effect concentration (TEC) and probable effect concentration (PEC) for arsenic in sediment are 10 mg/kg and 33 mg/kg, respectively.

Reported LC_{50}s for freshwater invertebrates vary widely. Several of the values in this range include: a 96-hour As (V) LC_{50} for *D. magna* of 7,400 µg/L (Eisler,1988a); a 96-hour As (III)

LC_{50} for *D. pulex* of 1,300 µg/L (Eisler, 1988a); a 96-hour As (III) LC_{50} for *Pteronarcys californica* of 38,000 µg/L (Johnson and Finley, 1980 *cited in* Eisler, 1988a); and a 96-hour As (III) LC_{50} for *Simocephalus serrulatus* of 810 µg/L (USEPA, 1985a). Arsenic is mainly accumulated in the exoskeleton of invertebrates (ATSDR, 2000a).

Compared to invertebrates, freshwater fish exhibit a higher tolerance to arsenic during acute exposures. Some of the reported benchmarks for freshwater fish are: a 96-hour As (III) LC_{50} for flagfish, *Jordanella floridae*, of 14,400 µg/L (Lima *et al.*, 1984 *cited in* Eisler, 1988a); a 96-hour As (III) LC_{50} for fathead minnow, *P. promelas*, of 14,100 µg/L (Lima *et al.*, 1984 *cited in* Eisler, 1988a); a 96-hour As (III) LC_{50} for channel catfish, *Ictalurus punctatus*, of 25,900 µg/L (NAS, 1977 *cited in* Eisler, 1988a); and a 48-hour As (III) LC_{50} for spottail shiner, *Notropis hudsonius*, of 29,000 µg/L (NAS, 1977 *cited in* Eisler, 1988a). Gilderhus (1969, *cited in* USACE, 2004b) reported a NOED of 11.6 mg/kg wet weight in liver tissue with a growth endpoint in adult bluegill (*Lepomis macrochirus*). Barrows, *et al.* (1980, *cited in* USACE, 2004b) reported a NOED of 0.52 mg/kg wet weight in whole body tissue with a mortality endpoint in immature bluegill.

Eisler (1988a) reports that BCFs for arsenic in aquatic invertebrates and fish are relatively low. BCF values for As (III) in most aquatic invertebrates and fish were less than 17. For As (V), the BCFs were less than 6, and the maximum BCF for organoarsenicals was 9 (USEPA, 1985a *cited in* Eisler, 1988a). In fish, arsenic accumulates mainly in the liver (ATSDR, 2000a).

Toxicity to terrestrial receptors may vary greatly depending on the form of arsenic. A single oral dose of 1 to 4 grams of sodium arsenite was lethal to cattle (*Bos* spp.) (NRCC, 1978 *cited in* Eisler, 1988a). A single oral dose of 2.5 mg/kg to 7.5 mg/kg of arsenic acid was also acutely toxic to domestic goats, *Capra* spp. (NRCC, 1978 *cited in* Eisler, 1988a). A 50 mg to 150 mg dose of sodium arsenite was lethal to a domestic dog, *Canis familiaris* (NRCC, 1978 *cited in* Eisler, 1988a), and single oral doses of 39.4 mg/kg and 15.1 mg/kg of arsenic trioxide were associated with 96-hour LD_{50}s in mice, *Mus* sp. and rats, *Rattus* sp., respectively (NAS, 1977 *cited in* Eisler, 1988a). Byron *et al.* (1967, *cited in* ATSDR, 2000a) reported a NOAEL of 2 mg/kg-day with growth and development endpoints in rats which were orally exposed to As(III). Schroeder and Mitchner (1971, *cited in* Sample *et al.*, 1996) calculated a NOAEL of 1.2 mg/kg-day in three generations of mice exposed to food and water laced with arsenic.

Toxicity benchmarks for avian species, based on exposure to sodium arsenite, include: an acute oral LD_{50} of 47.6 mg/kg for California quail, *Callipepla californica* (Hudson *et al.*, 1984); an acute oral LD_{50} of 323 mg/kg for mallard, *Anas platyrhynchos* (Hudson *et al.*, 1984); and an acute oral LD_{50} of 389 mg/kg for ring-necked pheasant, *Phasianus colchicus*, (Hudson *et al.*, 1984). A NOAEL of 1.25 mg/kg-day was estimated in chickens after 56 days of exposure (Hermayer *et al.*, 1977 *cited in* NAS, 1980). The USFWS (1964, *cited in* Sample *et al.*, 1996) estimated a NOAEL of 5.14 mg/kg-day for mallards and a LOAEL of 12.84 mg/kg-day based on 128-day exposure to sodium arsenite in diet. Camardese *et al.* (1990) documented 30 mg arsenic/kg diet (or 3 mg/kg-day, at 100 mg food per day and 1 kg body weight) decreased growth of females over 10 week exposures.

Arsenic bioaccumulates in plants and animals, but does not appear to biomagnify between trophic levels (Eisler, 1994; Farag *et al.*, 1998 *cited in* ATSDR, 2000a). In aquatic systems, the most significant transfer occurs between water and algae at the base of the food chain. An extensive study of the factors affecting bioaccumulation of arsenic in Maryland streams found no evidence of biomagnification since arsenic concentrations decreased with increasing trophic level (Mason *et al.*, 2000 *cited in* ATSDR, 2000a). Another study showed no difference in arsenic levels among different species of exposed fish, which included herbivorous, insectivorus, and carnivorus species (Kidwell *et al.*,1995 *cited in* ATSDR, 2000a).

Barium. Barium readily forms insoluble carbonate and sulfate salts which have low toxicity, but soluble barium salts may be toxic (USEPA, 1985a). The Tier II SCV calculated by Suter and Tsao (1996) is 4.0 µg/L. In seawater, barium concentrations ranging from 0.1 mg/L to 0.9 mg/L have been shown to be toxic to mussel embryos (*Mytilus californianus*) (Spangenberg and Cherr, 1996).

BCFs for barium in marine animals, plankton, and brown algae are 100, 120, and 260, respectively (ATSDR, 1992b). Although there is some evidence that barium may bioconcentrate in certain terrestrial plants and aquatic freshwater organisms, the extent of plant uptake and the subsequent uptake by aquatic or terrestrial animals is not known (ATSDR, 1992b). Estimated soil-to-plant bioaccumulation factors (BAFs) are 0.015 to 0.15 (Bysshe, 1988).

Guidelines for the pollution classification of Great Lakes harbor sediments classify sediment barium concentrations of <20 mg/kg, 20-60 mg/kg, and >60 mg/kg as non-polluted, moderately polluted, and heavily polluted, respectively (Beyer, 1990). Eco-SSL values were derived by USEPA for barium for soil invertebrates and mammalian wildlife as 330 mg/kg dry weight soil and 1,000 mg/kg dry weight soil, respectively (USEPA, 2003d).

Oral LD_{50}s for barium (as barium carbonate) are reported as 418 mg/kg and 200 mg/kg for rats and mice, respectively (Sax and Lewis, 1989). Exposure of barium chloride to rats via water consumption over a 16-month period resulted in a NOAEL of 5.1 mg/kg-day for effects on growth and hypertension (Perry, 1983 *cited in* Sample *et al.*, 1996). A TRV for mammals (Eco-SSL) based on the geometric mean of the NOAEL values for reproduction and growth in rats and mice in a number of studies was calculated at 52 mg/kg-day (USEPA, 2003d). Sample *et al.* (1996) calculated a chronic NOAEL of 208 mg/kg-day for mortality in chickens based on a subchronic NOAEL reported by Johnson *et al.* (1960).

Beryllium. Reactions of beryllium in soil and in solution depend on the pH of the medium. At typical soil or sediment pH values between 4 to 8, beryllium complexes are highly insoluble and mobility is low. Eco-SSL values for beryllium were derived by USEPA (2003e) for soil invertebrates and mammalian wildlife as 35 mg/kg for mammals and 40 mg/kg for soil invertebrates.

LCVs for freshwater daphnids and plants are 5.3 and 100,000 µg/L, respectively (Suter and Tsao, 1996). Suter and Tsao also report a SCV of 120 µg/L. A final chronic value (FCV) of 5.1 µg/L was reported by USEPA (USEPA, 1996a). Bluegill sunfish have been shown to bioconcentrate beryllium (Barrows *et al.*, 1978 *cited in* USEPA, 2004b). A NOAEL for longevity and weight loss in rats of 0.66 mg/kg-day was observed by Schroeder and Mitchner (1975 *cited in* Sample *et al.*, 1996) in a study where rats were exposed to beryllium sulfate in drinking water over their lifetime.

A measured BCF of 19 was reported for beryllium in bluegill fish (USEPA 1980a, *cited in* ATSDR, 2002a). Other investigators have reported a BCF of 100 for freshwater and marine plants, vertebrates, and fish (Callahan, *et al.*, 1979 *cited in* ATSDR, 2002a). There is no evidence of biomagnification of beryllium within terrestrial or aquatic food chains (Fishbein, 1981 *cited in* ATSDR, 2002a).

Cadmium. The literature review of cadmium effects by Eisler (1985) concluded that freshwater organisms were the most sensitive biota. In the environment, cadmium is adsorbed to soil, but to a lesser extent than other metals. Leaching from soils is more likely to occur under conditions of low pH. Cadmium generally occurs as a divalent metal that is insoluble in water.

The USEPA chronic NAWQC for cadmium is 0.25 µg/L based on a hardness of 100 mg/L as $CaCO_3$ (USEPA, 2002). Suter and Tsao (1996) reported an LCV of 1.7 µg/L for fish, 0.15 µg/L for daphnids, and 100,000 µg/L for aquatic plants. Concentrations of 0.8 to 9.9 µg/L in water were lethal to several species of aquatic insects, crustaceans, and teleosts. Eisler (1985) also reported that cadmium concentrations ranging from 0.7 µg/L to 5.0 µg/L were associated with sublethal effects (decreased growth, inhibited reproduction, and population alterations) in these same groups. Cadmium has also been shown to be highly toxic to South African clawed frog (*Xenopus laevis*) embryos (Herkovits *et al.*, 1997). At the most sensitive embryonic stage, a concentration of 1 mg/L of cadmium resulted in arrested development in 100% of exposed frogs.

Mammals and birds are less sensitive to the biocidal properties of cadmium than freshwater biota (Eisler, 2000). Cadmium in mammals can bioaccumulate and interfere with zinc-containing enzymes, resulting in impairment of kidney function, reproduction, and growth (Scheuhammer, 1987). Cadmium does not accumulate in muscle tissue (Harrison and Klaverkamp 1990, *cited in* ATSDR, 1999b).

USEPA has derived Eco-SSL values for cadmium for plants, soil invertebrates, birds, and mammalian wildlife (USEPA, 2003f). These values ranged from 0.38 mg/kg in soil for mammals to 150 mg/kg dry weight in soils for invertebrates. Sutou *et al.* (1980, *cited in* Sample *et al.*, 1996) reported a NOAEL of 1 mg/kg-day with a reproductive endpoint based on a study conducted with rats orally exposed to cadmium in diet for six weeks. A TRV for mammalian species (Eco-SSL) based on the geometric mean of the NOAEL values for reproduction and growth in a number of studies was calculated at 1.86 mg/kg-day. White and Finley (1978, *cited in* Sample *et al.*, 1996) reported a NOAEL of 1.45 mg/kg-day with a reproductive endpoint in mallard ducks orally exposed to cadmium for 90 days during gestation. A TRV for avian species (Eco-SSL) based on the geometric mean of the NOAEL values for reproduction and growth in a number of studies was calculated at 1.47 mg/kg-day (USEPA, 2003f). Meteyer *et al.* (1988, *cited in* USACE, 2004) reported a LOED of 0.9 mg/kg wet weight in whole body tissue with a development endpoint in embryonic sheepshead minnow (*Cyprindon veriegatus*). Eaton (1974, *cited in* USACE, 2004) reported a NOED of 201 mg/kg wet weight in liver tissue with a mortality endpoint in bluegill (*L. macrochirus*) exposed to cadmium in water.

Chromium. The toxicity of chromium varies widely between organisms and is dependent on form. Chromium (III) has very low solubility and low reactivity resulting in low mobility in the environment and low toxicity in living organisms (Barnhart, 1997 *cited in* ATSDR, 2000b). Under oxidizing conditions Cr (VI), which is relatively soluble, mobile, and thus more toxic to living organisms, may be present (James *et al.,* 1997 *cited in* ATSDR, 2000b).

The USEPA chronic NAWQC for Cr (III) is 74 µg/L based on a hardness of 100 mg/L as $CaCO_3$ (USEPA, 2002). Suter and Taso (1996) report LCVs of 68.63 µg/L for fish, 0.15 µg/L for daphnids, and 2 µg/L for aquatic plants. Adverse effects of chromium to sensitive freshwater species have been documented at 10 µg/L of Cr (VI) and 30 µg/L of Cr (III) (Eisler, 1986a). For wildlife, adverse effects have been reported at 5.1 mg and 10.0 mg of Cr (VI) and Cr (III), respectively, per kilogram of diet (Eisler, 1986). These data support the generalization drawn by Eisler that Cr (VI) is more toxic to freshwater species and mammals than Cr (III). Anderson *et al.* (1997, *cited in* ATSDR, 2000b) reported a NOAEL of 9 mg/kg-day with a systemic endpoint in rats orally exposed to chromium for 20 weeks. Haseltine *et al.* (Unpublished data *cited in* Sample *et al.*, 1996) reported a NOAEL of 1 mg/kg-d with a reproductive endpoint in black ducks orally exposed to chromium for 10 months.

Exposure to Cr (VI) has been demonstrated to reduce growth rates in both freshwater algae and duckweed, and to affect the survival and fecundity of cladocerans (Eisler, 1986a). Some salts of chromium are carcinogenic in rats and Cr (VI) is a teratogen in hamsters (USEPA, 1985a). Mackenzie *et al.* (1958, *cited in* Sample *et al.*, 1996) reported a NOAEL of 3.28 mg/kg-day with an endpoint of body weight for rats orally exposed though water to Cr (VI) for one year.

Chromium has not been observed to biomagnify, and concentrations are usually highest at lower trophic levels (Eisler, 1986a). Chromium does not bioconcentrate in fish (USEPA 1980b, 1984b; Fishbein,1981; Schmidt and Andren,1984); the BCF for Cr (VI) in rainbow trout (*Salmo gairdneri*) is 1. The bioavailability of Cr (III) to freshwater invertebrates (*Daphnia pulex*) decreases when humic acid is available to complex with the free form of the metal, limiting its bioavailibility. Thus there is no little to no biomagnification of chromium along the aquatic food chain (Cary, 1982 *cited in* ATSDR, 2000b)

Because plants tend to store most chromium in the roots, with only minor translocation to aboveground tissue, biomagnification of chromium along the terrestrial food chain (soil-plant-animal) is not expected (Cary, 1982; WHO, 1988, *respectively cited in* ATSDR, 2000b).

Cobalt. Cobalt is an essential element that can be accumulated by plants and animals (USEPA, 1985a). However, excessive oral doses may result in toxicity. In the environment, cobalt is normally found as a divalent ion, and its solubility is regulated by pH (USEPA, 2003g). Mobility in aquatic systems is limited because cobalt adsorbs to clay minerals and hydrous oxides of iron, manganese, and aluminum in the clay fractions of sediments and soils (USEPA, 1985a). LCVs for cobalt have been reported as 290 µg/L for fish and 5.1 µg/L for daphnids (Suter and Tsao, 1996). Suter and Tsao also report a secondary chronic value of 23 µg/L. Estimated soil-to-plant BAFs range from 0.007 to 0.02 (Bysshe, 1988). Data indicate that cobalt does not biomagnify up the food chain (Barceloux, 1999; Evans *et al.*, 1988; *respectively cited in* ATSDR, 2001a).

Eco-SSL values were derived by USEPA (2003g) for plants (13 mg/kg dry weight in soil), birds (190 mg/kg dry weight in soil), and mammalian wildlife (240 mg/kg dry weight in soil). Nation *et al.* (1983, *cited in* ATSDR, 2001a) reported a NOAEL of 5 mg/kg-day with a reproductive endpoint for rats orally exposed to cobalt for 69 days. A TRV for mammals (Eco SSL) based on the geometric mean of the NOAEL values for reproduction and growth in rats and mice in a number of studies was calculated at 7.33 mg/kg-day (USEPA, 2003g). An avian TRV (NOAEL) was calculated as 7.61 mg/kg-day for cobalt (USEPA, 2003g).

Copper. The USEPA chronic NAWQC for copper is of 9.0 µg/L based on a hardness value of 100 mg/L as $CaCO_3$ (USEPA, 2002). The LCV for fish is 3.8 µg/L (Suter and Tsao, 1996). The lowest chronic values for daphnids, non-daphnid invertebrates, and aquatic plants were reported as 0.23 µg/L, 6.06 µg/L, and 1 µg/L respectively. Mean acute toxicity values for freshwater species range from 7.2 µg/L for the daphnid, *D. pulicaria,* to 10,200 µg/L for bluegill sunfish, *L. macrochirus* (USEPA, 1985a). Chronic toxicity values for freshwater species range from 3.9 µg/L for brook trout to 60.4 µg/L for northern pike (USEPA, 1985a).

Earthworms bioconcentrate copper and can be negatively affected via a decrease in growth, reproduction, or survival (Beyer, 1990). For the soil-dwelling collembolan, *Folsomia fimetaria,* Scotts-Fordsmand *et al.* (1997) reported a soil EC_{10} for reproduction of 38 mg/kg, and a soil EC_{10} between 509 mg/kg and 845 mg/kg for growth (depending on sex and developmental stage). Bysshe (1988) suggested that concentrations of copper in soils will generally kill plants before they can accumulate tissue concentrations that are toxic to grazing animals. Experimentation has shown that chronic exposure to dietary copper can impact both sheep and swine (USEPA, 1985a). Aulerich *et al.* (1982, *cited in* Sample *et al.*, 1996) determined a NOAEL for reproductive effects in mink of 11.7 mg/kg-day. Mehring *et al.* (1960, *cited in* Sample *et al.*, 1996) reported a NOAEL of 47 mg/kg-day with a mortality endpoint in chickens. Benoit (1975) reported a LOED of 13 mg/kg wet weight in whole body tissue with a growth endpoint for immature bluegill (*L. Macrochirus*) exposed to copper over a 22 month period. Lee *et al.* (1975, *cited in* USACE, 2004) reported a NOED of 57 mg/kg wet weight in liver tissue with a reproductive endpoint for juvenile bluegill exposed to copper in water.

Iron. The Ontario Ministry of the Environment (OMOE, 1993) reported a low-effects level (LEL) of 2% iron in sediment and a severe-effects level (SEL) of 4% in sediment. Iron is a vital nutrient for all living organisms because it is essential for multiple metabolic processes, including oxygen transport, DNA synthesis, and electron transport. No toxicological studies were identified which described effects of iron in aquatic systems. No toxicity reference values (TRVs) were identified for iron.

Lead. Lead is toxic to all phyla of aquatic biota (Wong *et al.*, 1978 *cited in* Eisler, 1988b). Based on a review of toxicity testing literature, Eisler (1988b) reported adverse effects to aquatic biota associated with lead concentrations ranging from 1 µg/L to 5.1 µg/L. The USEPA chronic NAWQC for lead is 2.5 µg/L, based on a hardness of 100 mg/kg as $CaCO_3$ (USEPA, 2002). The LCV for fish was reported as 18.88 µg/l. The LCV for daphnids, non-daphnid invertebrates, and aquatic plants were 12.26 µg/L, 25.46 µg/L, and 500 µg/L, respectively.

Lead is relatively immobile in soils. Uptake of lead by plants depends on soil characteristics and is favored by lower pH values and in soil with lower organic carbon content (USEPA, 2003h).

USEPA has derived Eco-SSL values for lead for plants, soil invertebrates, birds and mammalian wildlife. These values ranged from 16 mg/kg dry weight in soil for avian species to 1,700 mg/kg dry weight in soils for invertebrates. Adverse effects associated with lead in soil have been documented for terrestrial plants (Bysshe, 1988; Eisler, 1988b). Earthworms may bioaccumulate lead (Beyer, 1990; Roberts and Dorough, 1985), and high concentrations of lead may be toxic to earthworms, affecting both survival and rate of reproduction. Eisler (1988b) generalized that organolead compounds are more toxic than inorganic lead compounds, and that younger organisms are more susceptible than older organisms.

For domestic and laboratory animals, Eisler (1988b) reported that survival was reduced at acute oral doses of 5 mg/kg (rat), at chronic oral doses of 5 mg/kg-day (dog), and at dietary doses of 1.7 mg/kg-day (horse). Lead affects the kidneys, bone and cental nervous system in mammals and can have adverse effects on histopathology, neuropsychology, fetotoxicity, growth and reproduction (Eisler, 2000). In addition, lead may interfere with enzymes involved in cellular oxidative processes, and possibly affect the release of impulses at certain nerve endings (Locke and Thomas, 1996). The primary source of lead poisoning in wild waterfowl, and in large raptors that prey on waterfowl, has been the ingestion of shotgun pellets (Locke and Thomas, 1996).

Azar *et al.* (1973, *cited in* Sample *et al.*, 1996) reported a NOAEL of 8 mg/kg-day with a reproductive endpoint for rats orally exposed to lead acetate for three generations. The mammalian NOAEL TRV estimated as the Eco-SSL was based on reproduction, growth and survival endpoints was 4.7 mg/kg-day (USEPA, 2003h). Edens *et al.* (1976, *cited in* Sample *et al.*, 1996) reported a NOAEL of 1.13 mg/kg-day with a reproductive endpoint in Japanese quail orally exposed to lead acetate for 12 weeks. A TRV for avian species (Eco-SSL) based on NOAEL values for reproduction and growth and mortality in a number of studies was calculated at 1.63 mg/kg-day (USEPA, 2003h). Weber (1991, *cited in* USACE, 2004) reported a LOED of 0.451 mg/kg wet weight for whole body tissue with a behavioral endpoint in adult fathead minnows (*P. promelas*), which exhibited significant reduction in feeding rates.

Although the bioavailability of lead in soil to plants is limited because of the strong absorption of lead to soil organic matter, the bioavailability increases as the pH and the organic matter content of the soil are reduced. Plants and animals may bioconcentrate lead but biomagnification has not been detected (Eisler, 1988a). In aquatic organisms, lead concentrations are usually highest in benthic organisms and algae, and lowest in upper trophic level predators (e.g., carnivorous fish).

Manganese. Manganese is an essential nutrient for animals, important for growth and reproduction (NAS, 1980). Manganese toxicity can decrease with increasing water hardness (Davies, 1980; Lewis *et al.*, 1979) and can be affected by pH (Lewis *et al.*, 1979). The permanganate forms of manganese are more toxic than the manganous salts (Doudoroff and Katz, 1953). However, permanganates are not persistent in aquatic environments and they are rapidly converted to relatively nontoxic substances through the oxidation of organic materials (USEPA, 1985a). Most of the available toxicity information is for manganous salts. Antagonism with nickel toxicity has been reported, as well as synergistic effects with some other metals (Lewis *et al.*, 1979).

Daphnia spp. exhibited 16% reproductive impairment after three weeks of exposure to 4.1 mg/L (Biesinger and Christensen, 1982 *cited in* Suter and Tsao, 1996). The LCV for daphnids is <1,100 µg/L (Suter and Tsao, 1996). The LCVs for fish is <1,100 µg/L for daphnids (Suter and Tsao, 1996). Suter and Tsao also report a SCV of 120 µg/L.

As indicated, the permanganates are more acutely toxic than the manganous salts. In freshwater, eels (*Anguilla japonica*) survived exposure to more than 2,700 mg/L as manganese chloride for 50 hours, but were killed in approximately 8 hours when exposed to 4.1 mg/L manganese as permanganate (Doudoroff and Katz, 1953). Goldfish were killed in hard water in 12 to 18 hours when exposed to 3.5 mg/L manganese as permanganate (Doudoroff and Katz, 1953). The LC_{50} for *Orizias* sp., a freshwater fish, was 6,045 mg/L (as manganese chloride) (McKee and Wolf, 1963 *cited in* Lewis *et al.*, 1979). At a concentration of 300 mg/L, manganese was lethal to sticklebacks (*Gasterosteus aculeatus*) within 24 hours. Davies (1980) reported that the acute toxicity of manganese to fish decreases with increasing water hardness, as well as increasing fish size. The 96-hour LC_{50} for rainbow trout in soft water (hardness = 36 mg/L as $CaCO_3$) was 14.5 mg/L and the 144-hour LC_{50} was 5.7 mg/L (Davies, 1980). England and Cummings (1971 *cited in* Lewis *et al.*, 1979) reported a 96-hour LC_{50} in young rainbow trout of 16 mg/L.

For early life stages of brown trout (*Salmo trutta*), Stubblefield *et al.* (1997) reported 25% inhibition concentrations (IC_{25}s, based on combined endpoints of survival and body weight) of 4.67 mg/L, 5.59 mg/L, and 8.68 mg/L at hardness levels of approximately 30 mg/L, 150 mg/L, and 450 mg/L as $CaCO_3$, respectively. This work demonstrated an inverse relationship between water hardness and the toxicity of manganese to fish.

BCFs of 2, 0.33, and 5.3 were reported for green algae (*Chlorella* sp.), water flea (*D. magna*), and fathead minnow (*P. promelas*), respectively (Kwasnik *et al.*, 1978 *cited in* USEPA, 2004b). Litzke and Hubel (1993, *cited in* USEPA, 2004b) reported BCFs of 106 for common carp (*Cyprinus carpio*) and 98 for rainbow trout (*Oncorhynchus mykiss*).

Guidelines for the pollution classification of Great Lakes harbor sediments classify sediment manganese concentrations of <300 mg/kg, 300-600 mg/kg, and >600 mg/kg as non-polluted, moderately polluted, and heavily polluted, respectively (Beyer, 1990). The OMEE LEL and SEL are 469 mg/kg and 1,100 mg/kg, respectively (Persaud *et al.*, 1993).

Manganese is an essential mineral for birds and mammals (USEPA, 1985a). It is a cofactor for a number of enzymatic reactions (Klaassen *et al.*, 1996). Rats showed no adverse reproductive or growth effects at dietary levels of 4,990 mg/kg manganese and only growth was adversely affected at 9,980 mg/kg (NAS, 1980). Sheep had reduced feed intake at dietary concentrations of 9,000 mg/kg (Puls, 1988). Laskey *et al.*, (1982 *cited in* Sample *et al.*, 1996) reported a NOAEL of 88 mg/kg-day with a reproductive endpoint in rats orally exposed to manganese for 224 days through gestation. Chickens showed reduced growth and 52% mortality at dietary concentrations of 4,800 mg/kg (Heller and Penquite, 1945 *cited in* NAS, 1980). This dietary concentration is approximately equivalent to a daily dose of 600 mg/kg, based on a conversion factor for young chickens (1 mg/kg in the diet = 0.125 mg/kg-bw) in Lehman (1954). Laskey and Edens (1985, *cited in* Sample, 1996) reported a NOAEL of 997 mg/kg-day with a reproductive endpoint for Japanese quail orally exposed to manganese oxide for 75 days.

Maximum Tolerable Levels (MTLs) for dietary manganese recommended by NAS (1980) are 1,000 mg/kg for cattle (15 mg/kg-day) and sheep (40 mg/kg-day), 400 mg/kg (16 mg/kg-day) for swine, and 2,000 mg/kg (250 mg/kg-day) for poultry. Puls (1988) recommended a maximum manganese concentration of 0.05 mg/L (50 µg/L) in drinking water for livestock and poultry.

Mercury. Mercury is a mutagen, teratogen, and carcinogen, and causes embryocidal, cytochemical, and histopathological effects. Methylmercury can bioconcentrate in organisms and can biomagnify through food chains (Wolfe *et al.*, 1998; Eisler, 1987a).

The USEPA chronic NAWQC for mercury is 0.77 µg/L (USEPA, 2002). Suter and Tsao report an SCV of 1.30 µg/L. Lowest chronic values for inorganic (or total) mercury are <0.23 µg/L for fish (*P. promelas* through the embryo-larval stage) and 0.96 µg/L for daphnids (*D. magna* in flow-through life-cycle tests) (Call *et al.*, 1983; Biesinger *et al.*, 1982; respectively *cited in* Suter and Tsao, 1996). The transformation of inorganic mercury by anaerobic sediment microorganisms produces methylmercury (Wolfe *et al.*, 1998). Chronic values for methylmercury are reported as 0.52 µg/L for fish (brook trout in three-generation life-cycle test) and <0.04 µg/L for daphnids (McKim *et al.*, 1976; Biesinger *et al.*, 1982, *respectively cited in* Suter and Tsao, 1996).

As summarized in Sample *et al.* (1996), reproductive NOAELs for animals exposed to mercury in their diet include 1 mg/kg-day for mink exposed to mercuric chloride for 6 months (Aulerich *et al.*, 1974 *cited in* Sample *et al.*, 1996), 13.2 mg/kg-day for mice exposed to mercuric sulfide for 20 months (Revis *et al.*, 1989 *cited in* Sample *et al.*, 1996), and 0.032 mg/kg-day for rats exposed to methyl mercury chloride over 3 generations (Verschuuren *et al.*, 1976 *cited in* Sample *et al.*, 1996), 0.45 mg/kg-day for Japanese quail exposed to mercuric chloride for 1 year (Hill and Schaffner, 1976 *cited in* Sample *et al.*, 1996), and 0.064 mg/kg-day for mallards exposed to methyl mercury dicyandiamide for 3 generations (Heinz, 1979 *cited in* Sample *et al.*, 1996). Weiner *et al.* (1990) reported a NOEL of 0.135 mg/kg wet weight for whole body tissue with a growth endpoint for yellow perch (*Perca flavescens*). Panigrahi and Misra (1978) reported a NOED of 3 mg/kg wet weight with a mortality endpoint in liver tissue of climbing perch (*Anabas scandens*).

Nickel. The USEPA chronic NAWQC for nickel is 52 µg/L based on a hardness of 100 mg/L $CaCO_{3.}$ LCVs for fish, daphnids, non-daphnid invertebrates, and aquatic plants are <35 µg/L, <5 µg/L, 128.4 µg/L, and 5 µg/L, respectively (Suter and Tsao, 1996). Nickel is not significantly accumulated by aquatic organisms (USEPA, 1985a). Bysshe (1988) estimated a soil-to-plant BAF of 0.06 for nickel.

Rats fed 40 mg/kg-day of nickel sulfate hexahydrate in their food over three generations showed no effects on reproduction (Ambrose *et al.*, 1976 *cited in* Sample *et al.*, 1996). The NOAEL for mallards orally exposed to nickel sulfate for 90 days was 77.4 mg/kg-day (Cain and Pafford, 1981 *cited in* Sample *et al.*, 1996).

Selenium. The most common forms of selenium in aqueous conditions are the salts of selenic and selenious acids. Sodium selenate is one of the most mobile selenium compounds in the environment because of its high solubility and inability to adsorb onto soil particles (NAS 1976, *cited in* ATSDR, 2001b). In flow-through toxicity studies, selenium, as selenate, was found to reduce larval fathead minnow biomass at 108.1 µg/L (LOEC) and to impair algal and rotifer population growth rates at similar concentrations (Dobbs *et al.*, 1996). As reported in Suter and Tsao (1996), LCVs for fish, daphnids, and aquatic plants are 88.32 µg/L, 91.65 µg/L and 100 µg/L, respectively. The USEPA chronic NAWQC for selenium is 5.0 µg/L (USEPA, 2002).

Regardless of the original source, adverse environmental effects appear to result largely from transfer of selenium from lower to higher trophic levels (Riedel and Sanders, 1996). High bioconcentration and accumulation of selenium from water by numerous species of algae, fish, and invertebrates is well documented at levels of 0.015 µg/kg to 3.3 µg/kg (Eisler, 1987c). Game fish populations have suffered reproductive failure after bioaccumulation of selenium from concentrations of about 10 µg/L dissolved selenium (Riedel and Sanders, 1996). Mortality, gross malformations, and internal abnormalities of the young of several wetland bird species have been observed where high selenate concentrations exist (up to 350 µg/L) (Ohlendorf *et al.*, 1986; Ohlendorf *et al.*,1990 *cited in* Riedel and Sanders, 1996). In mammals, selenium is a essential trace element that shows evidence of toxicity at higher doses (Domingo, 1994).

Rosenfeld and Beath (1954, *cited in* Sample *et al.*, 1996) reported a NOAEL of 0.2 mg/kg-day with a reproductive endpoint for rats orally exposed to selenium in water for one year, through two generations. Heinz *et al.* (1989, *cited in* Sample *et al.*, 1996) reported a NOAEL of 0.4 mg/kg-d with a reporductive endpoint for mallard duck orally exposed to selenium in diet. Lemley (1982, *cited in* USACE, 2004) reported a NOED of 3 mg/kg wet weight in whole body tissue with a mortality endpoint for immature largemouth bass (*Mactopterus salmoides*), and a NOED of 0.12 wet weight whole body tissue with a mortality endpoint for juvenile bluegill (*L. Macrochirus*) exposed to selenium. Hermanut *et al.* (1992, *cited in* USACE, 2004) reported a NOED of 9.3 mg/kg wet weight with a mortality in liver tissue endpoint for adult bluegill (*L. Macrochirus*). Lemly (1985, *cited in* ATSDR, 2001b) has reported BCFs of 150–1,850 and BAFs of 1,746–3,975 for selenium in freshwater.

Silver. In fresh water, silver forms complex ions with chlorides, ammonium, and sulfates, forms soluble organic compounds such as the acetate and the tartrate, becomes adsorbed onto humic complexes and suspended particulates, and becomes incorporated into, or adsorbed onto, aquatic biota (Boyle 1968, *cited in* ATSDR, 1990b). In soils, silver complexes with inorganic chemicals and humic substances (Boyle, 1968). Since silver is toxic to soil microorganisms and inhibits bacterial enzymes (Domsch, 1984 *cited in* ATSDR, 1990b), biotransformation is not expected to be a significant process.

The secondary chronic value for silver is 0.36 µg/L (Suter and Tsao, 1996). The lowest chronic value for fish, daphnids, and aquatic plants is 0.12 µg/L, 2.6 µg/L, and 30 µg/L, respectively (Suter and Tsao, 1996). Acute toxicity values for freshwater invertebrates range from 0.25 µg/L for the water flea, *D. magna*, to 4,500 µg/L for the amphipod, *Gammarus pseudolimnaeus* (USEPA, 1985a). Chronic toxicity values ranging from 2.6 µg/L to 29 µg/L have been reported for *D. magna* (USEPA, 1985a). For sediment, the NOAA ERL value is 1 mg/kg (Long *et al.*, 1995). BCFs of 70 and 7 were measured for bluegill sunfish and largemouth bass, respectively, exposed to silver nitrate (Coleman and Cearley, 1974). Coleman and Cearley reported a NOED of 0.12 mg/kg wet weight in whole body tissue with a mortality endpoint for juvenile bluegill (*L. macrochirus*). Excess silver in the diets of mammals and birds has been reported to induce selenium, vitamin E, and copper deficiency symptoms (USEPA, 1985a). Walker (1972, *cited in* ATSDR, 1990b) reported a NOAEL of 181.2 mg/kg-day with a mortality endpoint in rats exposed to silver for 2 weeks.

Terhaar *et al.* (1977, *cited in* ATSDR, 1990b) studied bioconcentration (uptake from water) and bioaccumulation (uptake from food and water) of silver thiosulfate complexes in algae (*Scenedesmus* sp.), water flea (*D. magna*), mussels (*Ligumia* sp. and *Margaritifera* sp.), and fathead minnow (*P. promelas*) in 10-week exposures. Bioconcentration indices were 96-150 for algae, 12.2-26 for Daphnia, 5.9-8.5 for mussels, and 1.8-28 for fish. Bioaccumulation indices were 9-26 for Daphnia, 6.6-9.8 for mussels, and 4.0-6.2 for fish. These indices, which are based on measured wet weight concentrations in biota and nominal concentrations in water, indicate little potential for silver biomagnification.

Thallium. Toxicity test have shown that the acute toxicity of various thallium salts are independent of the anion, valence, and animal species (Stokinger, 1981; Aoyama, 1989 *cited in* Borges and Daugherty, 1994). The acute oral LD_{50}s of various thallium salts ranged between 15-

50 mg/kg body weight; death resulted from respiratory failure. LCVs for fish, daphnids, and plants are 57, 130, and 100 µg/L, respectively (Suter and Tsao, 1996). The secondary chronic value for thallium is 12 µg/L (Suter and Tsao, 1996).

Female rats given thallium sulfate in drinking water for 36 weeks at a dose of 1.4 mg/kg-day demonstrated a 21% increase in mortality (Manzo *et al.*, 1983 *cited in* Borges and Daugherty, 1994), and experienced alterations in motor and sensory action, damage to the sciatic myelin sheath, and peripheral nerve degradation, and loss of hair. Roll and Matthiaschk (1981, *cited in* Borges and Daugherty, 1994) studied developmental toxicity in rats and mice via gavage during gestation. Mice treated with 6 mg/kg-day of thallium chloride had a slight increase of fetal loss and a slight decrease in birth weight. Rats fed 3 mg/kg-day of thallium acetate also had a slight increase in fetal loss. The reproductive subchronic LOAEL for male rats orally exposed to thallium sulfate in drinking water for 60 days was 0.74 mg/kg-day (Formigli *et al.*, 1986 *cited in* Sample *et al.*, 1996). Thallium has been demonstrated to bioconcentrate in duckweed (*Lemna minor*) (Kwan and Smith, 1991; Kwan and Smith, 1988; *respectively cited in* USEPA, 2004a).

Thallium may bioconcentrate in organisms from water. One experimentally-measured BCF value for bluegill sunfish was 34 (Barrows *et al.* 1978, *cited in* ATSDR, 1992e). Thallium is also absorbed by plants from soil and thereby enters the terrestrial food chain (Cataldo and Wildung, 1983, *cited in* ATSDR, 1992e).

Vanadium. In fresh water, vanadium generally exists in solution as the vanadyl ion (V^{4+}) under reducing conditions and the vanadate ion (V^{5+}) under oxidizing conditions. The partitioning of vanadium between water and sediment is strongly influenced by the presence of particulates in the water. Both forms of vanadium bind strongly to mineral or biogenic surfaces by adsorption or complexing (Wehrli and Stumm 1989, *cited in* ATSDR, 1992f). The secondary chronic value for vanadium is 20 µg/L (Suter and Tsao, 1996). The lowest chronic value for fish, daphnids, and aquatic plants are 0.12 µg/L, 2.6 µg/L, and 30 µg/L, respectively (Suter and Tsao, 1996). Suter and Tsao (1996) report LCVs of 80 µg/L for fish and 1,900 µg/L for daphnids.

A vanadyl sulfate concentration of 5 mg/L-day in drinking water, plus a vanadium level of 3.2 mg/kg-day in the diet (4.1 mg/kg-day total) of mice, was reported to cause no adverse effects over a lifetime exposure period (Schroeder and Balassa, 1967 *cited in* Opresko, 1991). In similar lifetime studies, rats and mice exhibited no adverse effects when exposed to 5 µg/L vanadium (as vanadyl sulfate) in drinking water (Schroeder *et al.*, 1970 *cited in* Opresko, 1991; Schroeder and Mitchner, 1975 *cited in* Sample *et al*, 1996). The estimated dose levels were 0.7 mg/kg-day for rats and 0.9 mg/kg-day for mice.

Exposure of male and female rats before mating, and of female rats during gestation and lactation to sodium metavanadate by gavage (maximum of 8.4 mg/kg-d) did not induce any adverse effects on fertility, reproduction, or parturition; however, doses of 4.2 mg/kg-day significantly reduced pup size and body weight at birth and 21 days later (Domingo *et al.*, 1986 *cited in* Sample *et al*, 1996). Gavage doses of 8.4 mg/kg-d of sodium metavanadate to pregnant rats on gestation days 6-14 did not result in embryolethality, teratogenicity or significant visceral or skeletal abnormalities; however, there was an increase in facial and dorsal hemorrhages (Paternain *et al.*, 1987 *cited in* Opresko, 1991). In a two-generation study, altered lung collagen metabolism was seen in fetuses of adult rats receiving 2.8 mg/kg of sodium metavanadate in drinking water (Kowalska *et al.*, 1988 *cited in* Opresko, 1991). Domingo *et al.* (1986, *cited in* Sample *et al.*, 1996) also reported a NOAEL of 0.021 mg/kg-day calculated from a LOAEL with a reproductive endpoint for rats orally exposed sodium metavanadate for 60 days prior to and through gestation. In a study conducted with mallard ducks, individuals were exposed to vanadyl sulfate in their diet for 12 weeks. The NOAEL for mortality, body weight, and blood chemistry was 11.38 mg/kg-day (White and Dieter, 1978 *cited in* Sample *et al.*, 1996).

Zinc. Adverse effects of zinc exposure have been documented on the growth, reproduction, and survival of freshwater species of aquatic plants, invertebrates, and vertebrates at concentrations between 10 and 25 µg/L (Eisler, 1993). 96-Hour LC_{50} values for freshwater invertebrates range from 32 to 40,930 µg/L and from 66 to 40,900 µg/L for freshwater teleosts (Eisler, 1993). LCVs for fish, daphnids, non-daphnid invertebrates, and aquatic plants are 36.41, 46.73, >5,243, and 30 μg/L, respectively (Suter and Tsao, 1996). The USEPA chronic NAWQC for zinc is 120 μg/L based on a hardness of 100 mg/L as $CaCO_3$ (USEPA, 2002). BCF values ranged from 107 to 1,130 for insects and from 51 to 432 for freshwater fish (Eisler, 1993).

Varying concentrations of zinc may also affect sediment invertebrates. At a mine tailings site, populations of freshwater oligochaetes and leeches were reduced in numbers of individuals and numbers of taxa in areas where the concentration of zinc in sediment was >20 g/kg (Willis, 1985 *cited in* Eisler, 1993). In contrast, the NOAA ERL value for sediment, which reflects a level at which impacts are possible, is 150 mg/kg (Long *et al.*, 1995).

Reduced survival has been reported for terrestrial plants at soil concentrations of >100 mg/kg and for soil invertebrates from 470 mg/kg to 6,400 mg/kg, respectively (Eisler, 1993). Increased dietary zinc has also been shown to have adverse effects on poultry, avian wildlife, livestock and laboratory animals (Eisler, 1993). Schlicker and Cox (1968, *cited in* Sample *et al.*, 1996) reported a NOAEL of 160 mg/kg-day with a reproductive endpoint for rats exposed to zinc oxide for 16 days during gestation. Stahl *et al.* (1990, *cited in* Sample *et al.*, 1996) reported a NOAEL of 15 mg/kg-day with a reproductive endpoint for white leg-horn exposed to zinc sulfate for 44 weeks.

With respect to bioconcentration from soil by terrestrial plants, invertebrates, and mammals, BCFs of 0.4, 8, and 0.6, respectively, have been reported. Plant species do not concentrate zinc above the levels present in soil (Levine *et al.*, 1989 *cited in* ATSDR, 2003). Zinc can accumulate in freshwater animals at 51 to 1,130 times the concentration present in the water (USEPA 1987b, *cited in* ATSDR, 2003); bioconcentration is higher in crustaceans and bivalve species than in fish. Microcosm studies indicate that in general, zinc does not biomagnify through food chains (Biddinger and Gloss 1984; USEPA 1979; Hegstrom and West 1989, *cited in* ATSDR, 2003).

4.3 Assessment and Measurement Endpoints

Endpoints in the BERA define ecological attributes that are to be protected (assessment endpoints) and a measurable characteristic of those attributes (measurement endpoints) that can be used to gauge the degree of impact that has or may occur. Assessment endpoints most often relate to attributes of biological populations or communities. They contain an entity (*e.g.*, muskrat population) and an attribute of that entity (*e.g.*, survival rate). At hazardous waste sites, the entity in the assessment endpoint is typically an individual species or community, often referred to as an indicator species or indicator community, respectively. Measurement endpoints are related to the assessment endpoint, and are the effects that can be measured or observed (*e.g.*, toxicity in invertebrate bioassays). Measurement endpoints are most often used as surrogates for assessment endpoints since, in most cases, the assessment endpoint itself cannot be readily measured or observed. Criteria for the selection of assessment endpoints include; unambiguous operational definition, accessibility to prediction and measurement, susceptibility to the hazardous agent, biological relevance, and societal relevance (Suter, 1993).

Assessment and measurement endpoints for the BERA were defined as follows:

Assessment Endpoints	Measurement Endpoints
Wildlife receptors	
Sustainability (survival, growth, reproduction) of local populations of omnivorous, semi-aquatic mammals	• quantify the average daily exposures to COPCs in the muskrat via the consumption of plants, animal prey, dietary water, and sediment; compare these modeled exposures to published values which are indicative of potential impairment
Sustainability (survival, growth, reproduction) of local populations of piscivorus, semi-aquatic mammals	• quantify the average and maximum daily exposures to COPCs in the river otter via the consumption of animal prey, dietary water, and sediment; compare these modeled exposures to published values which are indicative of potential impairment
Sustainability (survival, growth, reproduction) of local populations of piscivorus birds	• quantify the average and maximum daily exposures to COPCs in the green heron via the consumption of animal prey, dietary water, and sediment; compare these modeled exposures to published values which are indicative of potential impairment

Sustainability (survival, growth, reproduction) of local populations of omnivorous waterfowl	• quantify the average and maximum daily exposures to COPCs in the mallard duck via the consumption of plants, animal prey, dietary water, and sediment; compare these modeled exposures to published values which are indicative of potential impairment
Sustainability (survival, growth, reproduction) of local populations of small terrestrial mammals	• quantify the average and maximum daily exposures to COPCs in the northern short-tailed shrew via the consumption of animal prey, dietary water, and sediment; compare these modeled exposures to published values which are indicative of potential impairment
Fish Receptors	
Sustainability (survival, growth, reproduction) of local populations of predatory fish	• compare tissue concentrations of COPCs measured in largemouth bass caught within the study area to published fish tissue benchmarks which are indicative of potential impairment • compare those same tissue concentrations to largemouth bass caught at reference locations • evaluation of population statistics for largemouth bass compared to reference locations
Sustainability (survival, growth, reproduction) of local populations of bottom-feeding fish	• compare tissue concentrations of COPCs measured in white suckers and bullhead caught within the study area to published fish tissue benchmarks which are indicative of potential impairment • compare those same tissue concentrations to white suckers and bullhead caught at reference locations
Sustainability (survival, growth, reproduction) of local populations of small forage fish	• compare tissue concentrations of COPCs measured in pumpkinseed sunfish caught within the study area to published fish tissue benchmarks which are indicative of potential impairment • compare those same tissue concentrations to pumpkinseed sunfish caught at reference locations

Benthic Invertebrate Community	
Sustainability (survival, growth, reproduction) of local populations of benthic invertebrates	• Compare the concentrations of COPCs measured in sediment samples collected in the study area and at reference locations to sediment benchmarks which are indicative of potential impairment • Compare toxicity of sediment samples collected in the study area to samples collected from reference locations using *Hyalella azteca* and *Chironomus tentans* laboratory bioassays • Compare the tissue concentrations of COPCs measured in benthic invertebrates in the study area to reference locations • Quantify the *in-situ* benthic community composition using sediment samples collected in the study area and at reference locations

For each of the individual indicator species, the assessment endpoint references an impact on survival, growth, or reproduction of the assessment population within the defined study area. Adverse effects on populations can be inferred from measures associated with impaired survival, growth, or reproduction. Some COPC exposures may be associated with sub-lethal effects which do not directly influence mortality or reproductive success. However, these sub-lethal effects may increase the probability of death or negatively influence reproduction by enhancing susceptibility to predation or parasitism, or weakening competitive ability. For this BERA, it is assumed that toxicity reference values representing sub-lethal and non-reproductive endpoints, may indirectly affect the survival or reproduction of the exposed individual, potentially leading to a reduction in study area populations. It is assumed that risk to the study area population exists if the threshold effects level is exceeded in the majority of the habitat of the receptor species.

5.0 ANALYSIS OF ECOLOGICAL EXPOSURES AND EFFECTS

5.1 Exposure Characterization

Exposure characterization describes the potential or actual contact of COPCs with receptors. For COPCs identified in study area media (surface water, sediment, and biota) contact is quantified as the amount of the chemical ingested, inhaled, or in direct contact with the body of the organism. Alternatively, for COPCs that may be internally absorbed, exposure is evaluated by the amount of the COPC in the target tissue. Exposure characterization for each of the receptor species is presented in the following subsections. Table 26 summarizes the organization of the data used to prepare the exposure estimates for each endpoint. Three different approaches were used in the exposure characterization.

- Wildlife species including muskrat, mallard, heron, otter, and shrew populations were evaluated using food-chain exposure models.

- Fish species exposures were based on evaluation of COPC tissue concentrations compared to concentrations in tissue of the same species collected at reference locations and comparison to tissue residue benchmarks for other freshwater species.

- Benthic Invertebrate community was evaluated based on four separate endpoints including: 1) comparisons of sediment concentrations to sediment effects benchmarks, 2) comparison of COPC concentrations in benthic invertebrate tissue to reference concentrations and tissue residue benchmarks, 3) evaluation community composition, and 4) results of sediment toxicity testing analyses.

Tissue Data

To assist in exposure estimation for wildlife indicator species (muskrat, heron, mallard, otter), fish, benthic invertebrates, and plants were collected from the study area and analyzed to determine contaminant concentrations in tissue. Plant and fish tissue were analyzed only for inorganics. Benthic invertebrate and fish tissue were analyzed for SVOCs and inorganics. These analyses were also conducted to support the evaluation of the fish and benthic invertebrate assessment endpoints. Analytical results are presented on a wet weight basis (data in Appendices 7B.5, 7B.6, and 7B.7). Methods and analytical results for the fisheries survey, which was conducted by USFWS in June 1999, are provided in Appendix 7B.7. Field methods, sampling locations, and analytical results of tissue for fish, benthic invertebrates, and plants are discussed in

Section 2.0 of this report. Surface water and sediment COPCs detected in plants, benthic invertebrates, and small fish are summarized in Tables 9 to 11, respectively.

Plant Tissue. Samples were collected for plant tissue analysis at four locations (MC-06, MC-08, MC-09, and MC-11) in the study area and at two reference locations (MC-02 and MC-03, Phillips Pond and South Pond, respectively). Tissue samples were collected from several species of plants and were analyzed separately for stems and root portions of the plant. Results of the plant tissue analyses are presented in Table 1 of Appendix 7B.5. Locations of the plant tissues samples were paired with sediment sampling locations.

In addition to utilization of these data in food-chain models, statistical analyses were performed on the data to evaluate if there were significant patterns in the concentration of metals in the plant tissues between locations, among species, and between the root and stem portions of the plants.

The matrix of plant data used in the analyses are shown in Appendix 7B.5. The frequency of detection of each metal was calculated for the 48 samples in Table 27. Since the frequency of detection was less than 10% of the samples for antimony, beryllium, cadmium, cobalt, mercury, nickel, selenium, silver, and thallium, these compounds were not carried further in the statistical analyses. They were dropped both because the low concentrations will result in a minor contribution to grazers in the food chain model and due to special statistical concerns utilizing data with very high frequency of non-detects (EPA, 2000a).

Pro UCL was used to test the normality or log-normality of the data distribution for each remaining compound (aluminum, arsenic, barium, chromium, copper, iron, lead, manganese, vanadium, and zinc). As none of the variables were normally or log-normally distributed, a non-parametric analysis of variance was conducted to evaluate if there were significant differences in the concentration of each metal between the study area and reference samples, among the five species sampled, and between the root and stem tissue.

Results of statistical analyses and histograms summarizing the data are presented in Appendix 7B.5. Concentrations of metals in plant tissues from all species pooled from study area samples (four stations) were compared to the two reference locations (pooled) using an appropriate statistical test to determine if the average value in one group (study area tissue samples) was statistically greater than the average value for another group, taking into account the amount of variability in the measurements for each group. The results are expressed that either the average values are not statistically significantly different ("ns" in Table 27) or expressed as the probability that the averages are different. The significance level selected was 0.05 which represents a 95% chance that the two values represent a statistically different sample averages. Simply stated, this says that there is a 95% chance the two numbers (averages from reference and non-reference) are really different, considering how much spread there is in the individual measurements. A test was run for each metal this way, individually. In Table 27, each test compares a set of mean values, and the probability that there are differences in the means are given. If the probability is less than 0.05 (greater than 95%), it is concluded that the values are different. For arsenic, copper, and zinc in the comparison of concentrations for study area samples versus reference, concentrations were higher ($p < 0.05$, meaning exceeding a 95% significance level) in study area tissue than in reference plants. The results for barium, manganese, and vanadium were also significant, however, the concentrations of these metals were higher ($p< 0.05$) in reference plant tissue.

There were statistically significant ($p< 0.05$) differences in tissue concentrations among the species present for arsenic, barium, manganese, and vanadium. For arsenic, the highest concentrations were found in cattails. For the majority of the metals evaluated, including aluminum, arsenic, chromium, copper, iron, lead, and vanadium, concentrations were higher in root tissue than in stem tissue (Table 27, Appendix 7B.5.4).

Cattail are preferred food for muskrat. Of the 42 samples collected, 18 samples were cattail plants, half of these stem/leaf samples and half root tissue. For all stations combined, the average arsenic concentration in cattail roots was nearly six times higher than in stem tissue (40.5 mg/kg vs 6.8 mg/kg). The highest arsenic concentration in plant tissue (240 mg/kg) was observed for cattail roots at station MC-08 in the HBHA wetland (sediment arsenic concentration was 594 mg/kg). No cattails were collected at station MC-06 in the HBHA Pond. The tissue concentrations at MC-06 was 39 mg/kg in roots and the stem tissue was 4.5 mg/kg with a co-collected sediment sample having an arsenic concentration of 273 mg/kg.

Invertebrate Tissue. Samples were collected for benthic macroinvertebrate tissue analysis at six study area locations (MC-06, MC-07, MC-08, MC-09, MC-11, and MC-13) and at four reference locations (MC-01, MC-02, MC-03, and MC-04). Three samples were collected at station MC-03 and two at station MC-02, for a total of 13 samples (Appendix 7B.6.1). Tissue samples were collected from several species of benthic macroinvertebrates and were analyzed for inorganics and SVOCs. Summary results of the invertebrate tissue analyses are presented in Table 10, and raw data are presented in Appendix 7B.6.1. Locations of the benthic macroinvertebrate tissue samples were paired with sediment sampling locations.

In addition to utilization of these data in food-chain models, statistical analyses were performed on the data to evaluate if there were significant patterns in the concentration of COPCs in the invertebrate tissue between locations or among species sampled.

The matrix of benthic macroinvertebrate data used in the analyses are shown in Appendix 7B.6. The frequency of detection of each SVOC or inorganic COPC was calculated for the 13 samples (Table 28). Since the frequency of detection was less than 10% of the samples for dibenz(a,h)anthracene, naphthalene, antimony, barium, beryllium, cadmium, cobalt, mercury, nickel, selenium, silver, and thallium, these compounds were not carried further in the statistical analyses. They were dropped due to special statistical concerns utilizing data with very high frequency of non-detects (EPA, 2000b). However, estimated values of ½ of the detection limit for samples with non-detected concentrations were used to estimate doses in food chain models for these COPCs..

Pro UCL was used to test the normality or log-normality of the data distribution for each remaining compound (Table 28). Analysis of variance (ANOVA) was performed for those compounds that were normally or log-normally distributed. For the four compounds whose distributions did not meet the assumptions of normality (untransformed or log-transformed), a non-parametric analysis of variance was conducted to evaluate if there were significant differences in the concentration of each metal or SVOC between the study area and reference samples, or among the species of invertebrates sampled (Appendix 7B.6).

Results of statistical analyses and histograms summarizing the data are presented in Appendix 7B.6. Only those COPCs with the highest frequency of detection (fluoranthene, pyrene, arsenic, and zinc) had statistically significant differences between either study area samples and reference samples or among species. Comparing concentrations of COPCs in macroinvertebrate tissues from all species pooled from study area samples (six stations) to the four reference locations (pooled), concentrations of fluoranthene, pyrene, arsenic, and zinc in study area tissue were statistically higher than tissue samples from reference locations ($p < 0.05$, Table 28).

There were no statistically significant ($p< 0.05$) differences in tissue concentrations among the species sampled for any COPC evaluated, with the exception of fluoranthene. The concentration of fluoranthene was higher in tissue samples from chironomids than in samples from odonates or from samples of chironomids and amphipods together. Histograms, comparing concentrations of the COPCs in invertebrate tissue among species indicated that all but two of the COPCs (acenaphthylene and manganese) had consistently higher concentrations of COPCs in chironomid samples than in samples containing odonates or chironomids and amphipods. However, these differences were not statistically significant, probably due to high variability in concentration and the small number of samples.

<u>Small Fish Tissue</u>. Samples were collected for fish tissue analysis at two study area locations (HBHA Pond and HBHA Pond No. 3) and at two reference locations (MC-02 and MC-03). HBHA Pond is the open water area of HB01, and HBHA Pond No. 3 is the open water area in

the HBHA wetland in the vicinity of HB03-2, and MC-11. Tissue samples were collected from several species of fish; however, only the data for small foraging fish (pumpkinseed) were analyzed for whole body tissue concentrations. These data were collected to use in food chain models as well as to evaluate exposure and potential effects in fish populations (discussed in Sections 5.1.2 and 5.2.2). A summary of results of the small fish tissue analyses are presented in Table 11, and raw data are presented in Table 1 of Appendix 7B.8. In addition to utilization of these data in food-chain models, statistical analyses were performed on the data to evaluate if there were significant patterns in the concentration of COPCs in the small fish tissue between reference and non-reference locations.

The frequency of detection of each inorganic COPC was calculated for the 20 samples (Table 29). The frequency of detection was less than 10% of the samples for all inorganics except arsenic, iron, selenium, and zinc. All inorganics with detection frequencies less than 10% were not carried further in the statistical analyses. They were dropped due to special statistical concerns utilizing data with very high frequency of non-detects (EPA, 2000). However, estimated values of ½ of the detection limit for samples with non-detected concentrations were used to estimate doses in food chain models for these COPCs.

Pro UCL was used to test the normality or log-normality of the data distribution for each remaining compound (Table 29). Analysis of variance (ANOVA) was performed for arsenic and zinc, as these both were log-normally distributed. For the other two compounds (iron and selenium) whose distributions did not meet the assumptions of normality (untransformed or log-transformed), a non-parametric analysis of variance was conducted to evaluate if there were significant differences in the concentration of each metal between the study area and reference samples (Appendix 7B.8).

Results of statistical analyses and histograms summarizing the data are presented in Appendix 7B.8. Comparing concentrations of COPCs in pumpkinseed tissue collected study area (10 samples) to the reference locations (10 samples), concentrations of arsenic and selenium in study area tissue were statistically higher than tissue samples from reference locations ($p < 0.05$, Table 29).

5.1.1 Exposure Estimation for Mammalian and Avian Species

For muskrat, otter, heron, mallard, and shrew, the dose of each chemical that would be expected to be obtained from the ingestion of food (plant and/or animal) was estimated using the equation presented in Section 3.3.2. Changes in exposure parameters for the average and upper confidence limit (UCL) case models from those used in the maximum models are presented below and in Appendices 7C.6 and 7C.10.

Sets of surface water, sediment, plant, benthic invertebrate, and small fish data used to estimate COPC exposures for muskrat, otter, heron, mallard, and shrew were generally the same as used in the maximum models (Table 26). For each receptor, two exposure models were calculated, an average case scenario and a UCL case scenario, representing the reasonable maximum exposure estimate. The average case scenario was a dietary exposure model based on mean concentrations of each COPC calculated for sediment, surface water, and animal or plant tissue, as appropriate for the receptor. An arithmetic mean of all of the samples collected within the foraging area of the species for each media (surface water, sediment, or animal tissue) was calculated. These mean values were used to calculate the total dose from dietary exposure in equation (4).

The maximum, or acute exposure case scenario, was modeled by calculating the upper confidence limit (UCL) providing 95% coverage to the population mean for each of the medium represented in the exposure model (sediment, surface water, and plant or animal tissue) used in the exposure estimate. The UCL of the average concentration is the value that, when calculated for an infinitely large randomly selected set of subsamples, will equal or exceed the true average 95% of the time. In risk assessments, the UCL is frequently used to represent the reasonable maximum exposure (RME) to occur at a site. USEPA requires the use of the UCL on the arithmetic mean concentration for the estimation of the RME risk in human health risk assessment (USEPA 1989; 1992b; and 1994). Therefore, whenever possible, the UCL has been calculated and used for the maximum exposure cases. The UCLs were calculated using EPA's program "ProUCL Statistical Software" (Version 3.0). The UCL values could be calculated by this program if four or more samples were available for summarization from a station or sample grouping. When less than four samples were available, the program was unable to calculate a UCL value, and the maximum sample concentration for the COPC was used. Also, if the UCL value was greater than the maximum detected concentration due to high variability of the data, the maximum detected concentration was used.

<u>Muskrat</u>

The home range for a muskrat is relatively small, and consequently, the risk evaluation for muskrat populations was conducted on a station-by-station basis. The average and UCL case scenario was calculated for the eleven inorganic COPCs indicated in Table 25 for the muskrat. These are the COPCs that exceeded maximum case exposure screening values.

Average and UCL station COPC concentration in sediment were used to estimate incidental sediment ingestion (3.3% of diet). Exposure from surface water ingestion was based on the COPC concentration in surface water for samples applied to the station. Invertebrate tissue samples (10% of diet) collected in the study area were used to estimate maximum exposure concentrations. Plant tissue concentrations (90% of diet) were estimated for each station from study area plant tissue. For stations AR, BE-1, BE-2, and BE-3 where no plant data were collected, plant tissue concentrations were estimated from uptake factors (Table 12) and average or UCL sediment concentrations for the station. Data groups utilized for the models are presented for each media in Tables 14 -17, and exposure point concentrations are presented in Appendices 7C.7 (UCL) and 7C.11 (average).

Data used in the reference models included sediment data from all reference locations, except station MC-03 (Phillips Pond), since samples were taken at a depth of 9 to 13 feet (Table 14). All reference surface water samples except samples collected at station MC-03 were used to calculate dietary exposure from water. Samples from the three shallow reference locations were used to select the maximum reference invertebrate tissue concentration. Reference plant tissue samples were collected at stations MC-02 and MC-03 (South Pond and Phillips Pond).

<u>River Otter</u>

Arsenic was the only COPC exceeding screening-level criteria for river otter and evaluated in average and UCL models. River otter exposure to arsenic was calculated for all samples collected within suitable habitat, throughout the Industri-Plex Study area to compute a study area-wide scenario since the foraging ranges of this species is also relatively large. Home range for river otter are estimated at 30 km (18 miles) of shoreline (Melquist and Hornocker, 1983, in EPA 1993d). For the purpose of the maximum exposure estimate, it was assumed the river otter spends 100% of its time foraging in the study area. Stations with little open water (AR, BE, HB02-2, HB03-3, and HB04) were excluded from the river otter model for sediment ingestion since these do not represent typical foraging area for otter. The average and UCL sediment arsenic values were used to calculate incidental sediment exposure (2% of diet). All surface water stations were used in the study area-wide model for estimating dietary ingestion of arsenic in water (Table 15). All invertebrate data collected study area-wide were combined for the river otter evaluation (Table 16) in the study area (20% of diet). All small fish samples collected within the study area (Table 11) were used to estimate arsenic exposure for river otter from ingestion of fish (80% of diet). Exposure point concentrations used for each media are presented in Appendices 7C.7 (UCL) and 7C.11 (average).

Data used in the reference models for river otter included sediment data and surface water data from all reference ponds and the Shawsheen River (station 27) (Tables 14 and 15). The average and UCL arsenic concentrations among all small fish tissue data collected from Phillips Pond and South Pond were used for the reference otter model and all samples from South Pond and Phillips Pond (MC-02 and MC-03) were used to select the maximum reference invertebrate tissue concentration (Table 16).

<u>Green Heron</u>

Four inorganic COPCs (chromium, lead, mercury, and zinc) exceeding screening-level criteria for heron, were evaluated in average and UCL models. Heron exposures were calculated for all samples collected within suitable habitat throughout the Industri-Plex study area to compute a "study area-wide" scenario since the foraging ranges of this species is relatively large. Green herons are known to defend feeding territories from other herons and are flexible, using a variety of freshwater habits within their range. Stations located in with little open water were excluded from the heron model, as well as stations with depths greater than three feet of water. COPC data from all selected sediment stations (Table14) were used to calculate incidental sediment exposure (1% of diet). All small fish data collected study area-wide were combined for the heron evaluation (Table 11) in the study area (45% of diet). Similarly, all benthic invertebrate samples collected within the study area (Table 10) were used to estimate exposure for heron invertebrate ingestion (55% of diet). All surface water stations were used in the study area-wide model for estimating dietary ingestion of water (Table 15). Both an average and UCL exposure case scenario were calculated for heron for the study area and the reference model.

Data used in the reference models for heron included sediment data from all reference locations, except station MC-03 (Phillips Pond), since samples were taken at a depth of 9 to 13 feet and were too deep to represent incidental sediment ingestion. All small fish data were collected from South Pond (MC-02) and Phillips Pond (MC-03). All reference surface water samples except samples collected at station MC-03 were used to calculate dietary exposure from water (Table 15).

Mallard

The home range of mallards is large, and can range from 40 to 1,440 ha (96 to 3,556 acres) (USEPA, 1993d). Three exposure scenarios were evaluated for mallard: HBHA Pond, HBHA Wetland, and the study area-wide scenario. COPCs carried forward from the screening-level effects evaluation for mallard included aluminum, antimony, arsenic, chromium, lead, mercury, and zinc.

Sediment samples with water depths less than three feet were used for estimation of incidental sediment ingestion (3.3% of diet) for mallard for samples representing each of the three scenarios (Table 14). Dietary exposure for mallard was based on 33% plant tissue and 67% invertebrates. All invertebrate samples collected within the study area were used to estimate study area-wide exposure for mallard invertebrate ingestion, excluding the deep sample at MC-07 (Table 16). Data from MC-06 was used for HBHA Pond and three samples (MC-08, MC-09, and MC-11) were used to represent the HBHA wetland. All epilimnetic (shallow water) surface water stations were used in the study area-wide model for estimating dietary ingestion of water (Table 15). Samples included baseflow or storm event samples collected between April and October. A subset of these data were used to calculate surface water exposure for the HBHA Pond and wetland. Samples from four study area locations (MC-06, MC-08, MC-09, and MC-11) were used to estimate maximum exposures to plant tissue study area-wide for mallard. Samples from MC-06 were used to represent HBHA Pond and samples MC-08, MC-09, and MC-11 were used for HBHA wetland.

Data used in the reference models included sediment data from all reference locations, except station MC-03 (Phillips Pond) since samples were taken at a depth of 9 to 13 feet (Table 14). All reference surface water samples except samples collected at station MC-03 were used to calculate dietary exposure from water (Table 15). Samples from the three shallow reference locations were used to select the maximum reference invertebrate tissue concentration (Tables16). Plant tissue samples were collected at stations MC-02 and MC-03 (South Pond and Phillips Pond).

Shrew

The home range of northern short-tailed shrew is small, on the order of less than one acre (USEPA, 1993d). Similar to muskrat, the risk evaluation for shrew populations was conducted on a station-by-station basis. COPCs carried forward from screening-level effects evaluation for shrew included antimony, arsenic, barium, chromium, copper, lead, manganese, mercury, selenium, thallium, and zinc.

Stations selected for calculation of exposures for shrews included upland habit and those in saturated areas with little standing water, that may be accessible to small mammals for foraging during periods of drier weather. In contrast to muskrat, heron, and mallard, study area-specific tissue data were not collected for the evaluation of COPC exposures to shrew. Study area-specific wetland sediment or soil data (Table 14) were used to estimate body burdens of prey for shrew. The concentration of COPCs in shrew prey (*i.e.*, earthworms) were estimated only for inorganic COPCs. For inorganic COPCs, regression equations relating contaminant concentrations in soil and earthworm tissue (Sample *et al.*, 1998) were used to estimate burdens of arsenic, chromium, copper, lead, manganese, mercury, nickel, selenium, and zinc in earthworms at the study area. A concentration factor for barium (0.36) (dry weight to dry weight), based on coupled analyses of soil and biota, were taken from Beyer and Stafford (1993). Uptake factors were not available for antimony or thallium. An uptake factor of 0.5 (dry weight to dry weight) was assumed to estimate the concentration of these inorganics in worm tissue.

Calculated earthworm COPC concentrations for each station used in the shrew model, based on average or UCL sediment COPC concentrations, are presented in Appendices 7C.8 (UCL) and 7C.12 (average). Exposure from surface water ingestion was based on the COPC concentration in surface water for samples applied to the station (Table 15). No surface water data were collected to represent upland stations A6 or BE, consequently, no dose from surface water is included in the total dose at these station. For the average and UCL case models, earthworms were assumed to compose 31% of the shrew diet, with the 13% attributed to incidental sediment ingestion, and the balance of the diet assumed to be from terrestrial sources, having a negligible body burden of soil-associated COPCs.

Total dose estimates for shrew at reference locations were calculated based on data from the three wetland reference locations, stations 24, HB, and SA. Sediment data from these three stations were pooled to estimate exposure at reference locations (Table 14) in order to have more data to calculate exposures than were available at each stations individually (only 1 sample each at stations HB and SA). Exposure from surface water ingestion for shrew was based on the COPC concentrations in surface water for the one wetland station sampled (station 24, Table 15).

5.1.2 Exposure Assessment for Fish

Risks to fish populations were evaluated by comparing inorganic COPC body burdens in four species of fish collected from the study areas to reference samples, as well as to tissue residue benchmarks from the literature. Species-specific average COPC concentrations for largemouth bass, white sucker, brown bullhead, and pumpkinseed sunfish were calculated for two study ponds (HBHA Pond and HBHA Pond No. 3) and two reference ponds (Philips Pond and South Pond). Separate averages were derived for carcass, fillet, and liver tissue in large fish (largemouth bass, brown bullhead, and white sucker) and whole body tissue in small fish (pumpkinseed sunfish) for individual species. Data from reference ponds were pooled. Data for small fish (whole body) were used in exposure models and are discussed above (section 5.1). Large fish data (carcass, liver, fillet) data are presented in Appendix 7B.8 and comparison of fish tissue to tissue residue benchmarks are presented in Section 5.2.2.

5.1.3 Exposure Assessment for Benthic Invertebrates

The exposure of benthic invertebrate community was quantified by direct comparison of COPC concentrations in sediment to benchmarks for benthic invertebrates, and also by comparison of COPC body burdens in benthic invertebrates collected from the study area to tissue concentrations from reference location (section 5.2.3). Further analysis of the potential effects of exposure of sediment dwelling invertebrate communities to contaminants was evaluated by comparing invertebrate tissue data to tissue residue benchmarks and conducting sediment toxicity testing at five reference and eight non-reference locations. The sediment toxicity testing results and an evaluation of benthic community composition data are presented in Section 5.2.3.

5.2 Ecological Effects Characterization

The results of the exposure analyses are presented in the following subsections, and evidence for existing and potential adverse effects on the receptor species is analyzed. The relationship of stressor (COPC) levels and ecological effects are evaluated. Evidence for adverse effects on receptors based on population and toxicity studies are presented.

5.2.1 Mammalian and Avian Indicator Species

Mammalian and avian TRVs for COPCs were obtained from the literature (Appendix 7C.4 and Appendix 7C.13). For the UCL case, a NOAELs (no observed adverse effect levels) TRV was used to calculate the HQ. For the average case, the hazard quotient (HQ) was calculated by dividing the estimated dose by the LOAEL (lowest observed effects level) TRV. If available and appropriate, TRVs were selected which were associated with chronic exposures (*i.e.*, long duration exposures) and relating to reproduction or mortality. All TRVs for muskrat, otter, and shrew were based on laboratory tests with mammals. The majority of the avian TRVs for metals were taken from studies with a variety of avian species. There were no avian TRVs available for antimony; so for the mallard evaluation, mammalian TRVs were applied. No adjustment factor was applied for this interspecies extrapolation. It is sometimes recommended that the TRV be adjusted by a factor of 10 to account for inter-species extrapolations (Sample *et al.*, 1997). However, if the relative sensitivity of the two species is not known, this factor can add a large uncertainty, without much scientific basis. The uncertainty associated with TRVs is further discussed in Section 6.3.

In some cases, TRVs with endpoints relating to reproduction or mortality were not available in the literature. TRVs associated with other effects (systemic, hematological, carcinogenic, neurological, hepatic) are assumed to indirectly affect survival and/or reproductive capacity. Body weight scaling equations presented in Sample *et al.* (1996) and Opresko *et al.* (1994) were used to adjust test species TRVs to indicator species TRVs. Consistent with equations in Sample *et al.* (1996), no body scaling factors were used for avian species.

COPC daily dose estimates were compared to TRVs to evaluate the effect of exposure on indicator species. This comparison was quantified as follows:

Hazard Quotient (HQ) = Dose COPC / TRV (8)

An HQ less than 1 indicates harm is unlikely, while an HQ greater than 1 suggests that a COPC is present at concentrations which may affect the survival or reproductive capacity of an exposed individual. HQs for mammalian and avian indicator species are discussed below. Model results are presented in Appendices 7C.9 and 7C.14.

Muskrat. For the UCL/NOAEL case, every station evaluated had two or more inorganic COPCs with HQs in excess of 1. For this lower effects level, HQs in excess of 1 are summarized in Table 30. HQs were greater than 1 for aluminum, antimony, arsenic, chromium, cobalt, copper, lead, selenium, vanadium, and zinc. HQs for arsenic were greater than 1 at all muskrat stations, except reference locations and BE-1. The highest observed HQs were for arsenic in HBHA wetland (HB02-1) and HBHA wetland pond (HB03-2) at 26 and 18, respectively. The compounds with UCL/NOAEL HQs greater than 1 at the reference sites for muskrat were aluminum, antimony, and vanadium.

For the average/LOAEL case, HQs were greater than 1 for only aluminum and arsenic (Table 31). HQs for arsenic were greater than 1 at all muskrat stationsexcept BE-1 and BE-3. The highest average/LOAEL was 5 at HB02-1. The highest observed HQ for aluminum was 4 at several locations. The average/NOAEL HQs also exceeded 1 (HQ = 3) for aluminum at reference locations. The site HQs for aluminum ranged from 2 to 4. For inorganic COPCs, exposure related to the ingestion of plant material generally dominated HQs (*i.e.*, the percent contribution to the HQ from invertebrates, sediment, and surface water was far less than for plants).

Muskrat modeling results indicate potential effects on populations due to elevated concentrations of inorganic COPCs in diet at levels potentially associated with harm.

River Otter. Arsenic was the only COPC exceeding screening-level criteria for river otter and carried forward to evaluate in average and UCL models. The HQ was less than 1 for the

UCL/NOAEL for site-wide exposure to dietary arsenic (Table 32). Exposures based on 95% UCL concentrations of arsenic in fish (80% of diet) and invertebrate tissue (20% of diet) were below a NOAEL TRV derived (rat, 2 years, growth). These results indicate that risks from dietary exposure of river otter to concentrations of COPCs in the site are below levels expected to cause ecological effects.

Green Heron. The UCL and average models evaluated the ecological effects of dietary exposure of green heron to chromium, lead, mercury, and zinc, site-wide. None of the HQs exceeded 1 for the UCL/NOAEL or the average/LOAEL for site-wide exposure to dietary exposures to chromium, lead, mercury, or zinc. For the heron model, concentrations for inorganics in fish tissue (45%) and invertebrate tissue (55%) contributed the major source of dietary exposure. Using average exposure concentrations and LOAEL TRVs, all HQs for heron were less than one. These results indicate dietary exposure of green heron to concentrations of COPCs in the site are below levels expected to cause ecological effects.

Mallard. The UCL and average models evaluated the ecological effects of dietary exposure of mallard to aluminum, antimony, arsenic, chromium, lead, mercury, and zinc, site-wide, and separately for HBHA Pond and the remainder of HBHA wetlands (Table 33). For UCL/NOAEL models, HQs were greater than 1 for chromium, lead, and zinc. The HQs for lead in the study area did not exceed those at the reference locations. Consequently, the risk from exposure of mallards from lead is not incrementally greater in the study area than at reference locations. The highest observed HQs were for zinc study area-wide and in HBHA wetland. For the average/LOAEL case, there were no HQs greater than 1 for any COPC.

For inorganic COPCs, exposure related to the ingestion of plant material generally dominated HQs (*i.e.*, the percent contribution to the HQ from invertebrates, sediment, and surface water was far less than for plants). Mallard modeling results indicate potential effects on populations due to elevated concentrations of chromium and zinc using UCL exposure concentrations and NOAEL

TRVs. However, there were no indications of potential effects on mallards using average exposure concentrations and LOAEL TRVs.

Shrew. The UCL and average models evaluated the ecological effects of dietary exposure of shrew to antimony, arsenic, barium, chromium, copper, lead, manganese, mercury, selenium, thallium, and zinc, at seven stations. For UCL/NOAEL models, HQs were greater than 1 for arsenic at all locations, with stations BE-1 and HB04 with HQs equal to reference locations (Table 34). The highest observed HQs for arsenic were observed in the upland edges of the HBHA wetlands at HB02-2 with an HQ of 92. Antimony and thallium HQs exceeded 1 for UCL/NOAEL models at A6 and HB02-2, and lead and mercury only at A6.

For the average/LOAEL case, HQs were greater than 1 for only arsenic (Table 35). Average case HQs for arsenic were greater than 1 at four of the seven shrew stations. The highest observed HQs for arsenic was 6 in HBHA wetland (HB02-2).

Shrew modeling results indicate potential effects due to elevated concentrations of arsenic in diet at levels potentially associated with harm.

5.2.2 Fish Species

The ecological effects evaluation for fish populations involved two components: analysis of fish tissue residues and fish population surveys.

5.2.2.1 Fish Tissue Analyses. COPC concentrations in fish tissues were evaluated in two ways. First, fish tissue samples were collected and analyzed for COPC concentrations and compared to reference area tissue concentrations. COPC concentrations in fillet, carcass, and liver tissue from large fish (largemouth bass, brown bullhead, and white sucker) from each individual study pond were compared to COPC concentrations in pooled reference samples. COPC concentrations in whole body tissue of small fish (pumpkinseed) from each individual study pond were compared to COPC concentrations in pooled reference samples. Although this evaluation does not directly address the ecological effects of COPCs, it demonstrates whether or not fish within the study area carry greater COPC body burdens than fish at reference stations.

Second, COPC tissue concentrations in fish collected from the study area were compared to tissue residue benchmarks reported in the literature for freshwater species in the families Ictaluridae, Cyprinidae, Esocidae, Percidae, and Centrarchidae. The Environmental Residue-Effects Database (ERED; USACE, 2004) contained entries for 7 of the 18 inorganic COPCs detected in carcass, fillet, or whole body tissue samples and 4 of the 18 inorganic COPCs detected in liver tissue. Although benchmarks were not available for all COPCs detected in fish, the benchmark comparisons still provided an indication of whether or not COPC concentrations in tissue are, in general, of a magnitude that may potentially be associated with harm to receptor populations. The benchmarks used for large fish carcass and fillet were identical to benchmarks used for pumpkinseed whole body, except selenium. For large fish, the selenium benchmark concentration was based on largemouth bass studies, while the selenium benchmark concentration for pumpkinseed was based on bluegill. Tissue-specific benchmarks were used to compare liver concentrations in large fish.

Eighteen COPCs were detected in fish tissue collected within the study area. Detected COPCs included aluminum, antimony, arsenic, barium, cadmium, chromium, cobalt, copper, iron, lead, manganese, mercury, nickel, selenium, silver, thallium, vanadium, and zinc. Beryllium was not detected in any fish tissue samples within the study area. Between the two study ponds, fish tissue from HBHA Pond No. 3 had 18 COPCs detected in fish tissue, while fish tissue from HBHA Pond had nine COPCs above detection levels.

Fish tissue data from the reference ponds were pooled. Sixteen COPCs were detected in fish tissue collected within the reference area. Detected COPCs included aluminum, arsenic, barium, cadmium, chromium, cobalt, copper, iron, lead, manganese, mercury, nickle, selenium, silver, vanadium, and zinc. Neither antimony, beryllium, nor thallium were detected in any fish tissue samples within reference areas.

There were several trends apparent from evaluation of study area and reference COPC tissue concentrations (Appendix 7B.8.3). These were as follows:

- COPCs in largemouth bass tissue were comparable between both study area ponds, although concentrations were slightly higher at HBHA Pond No. 3.
- COPCs in brown bullhead tissue were comparable between both study area ponds, although concentrations were slightly higher at HBHA Pond No. 3.
- COPCs in white sucker tissue were detected more frequently and at higher concentrations at HBHA Pond No.3 than HBHA Pond.
- COPCs in pumpkinseed tissue were comparable between both study area ponds, though concentrations were slightly higher at HBHA Pond No. 3.
- Fish from the study areas carry a slightly higher body burden of inorganic COPCs than fish from reference stations. Inorganics higher in tissue from fish in the study area included: aluminum, arsenic, copper, chromium, iron, cadmium, manganese, selenium, and zinc.
- The highest ratios of study area to reference data were consistently for arsenic. The highest ratio was 53.4 in white sucker liver at HBHA Pond No. 3.
- In pumpkinseed tissue, the concentrations of selenium and arsenic were statistically higher in the study area than in reference ponds (Table 29).
- White sucker tissue had the highest COPC detection frequency and the highest detected concentrations of all fish in both reference and study area ponds.

Tissue residue benchmarks and study area tissue concentrations for fish are presented in Appendix 7B.8.3. Arsenic was the only inorganic that had tissue residue values above corresponding tissue residue benchmarks and above reference tissue concentration. Aluminum and zinc concentrations were higher in tissue from study area fish, but there were no tissue residue benchmarks for these metals. Arsenic values in tissue are summarized separately in Table 36.

In largemouth bass tissue at HBHA Pond, lead (carcass, fillet), mercury (fillet), and silver (carcass, fillet) concentrations exceeded the tissue benchmark concentration, although lead and silver were both based on detection limits. The only benchmark values for liver exceeded at HBHA Pond for liver was chromium, however, these were based on detection limits. In largemouth bass tissue at HBHA Pond No.3, lead (carcass, fillet), and silver (carcass, fillet) concentrations exceeded the tissue benchmark concentration, although both lead and silver were based on detection limits. No COPCs exceeded both tissue benchmark concentrations and reference area concentrations for largemouth bass. No benchmark values for liver were exceeded at HBHA Pond No. 3. Tissue residue benchmark values for arsenic (average concentrations) were not exceeded in largemouth bass collected in the study area for any tissue fraction (carcass, liver, or fillet). The maximum carcase concentration for largemouth bass in HBHA Pond No. 3, ws slightly above the tissue benchmark (Table 36)

In brown bullhead tissue at HBHA Pond, arsenic (carcass), lead (carcass, fillet), and silver (carcass, fillet) tissue concentrations exceeded the tissue benchmark concentration, although lead and silver were based on detection limits. Arsenic exceeded both tissue benchmark concentrations and the reference area concentration. No benchmark values for liver were exceeded at HBHA Pond. In brown bullhead tissue from HBHA Pond No. 3, arsenic (carcass), lead (carcass, fillet), and silver (carcass, fillet) exceeded benchmark tissue concentrations, although silver and lead were based on detection limits. Arsenic exceeded both tissue benchmark concentrations and reference area concentrations and was approximately twice the benchmark concentration. No benchmark values for liver were exceeded at HBHA Pond No. 3.

In white sucker tissue at HBHA Pond, arsenic (carcass), lead (carcass, fillet), and silver (carcass, fillet) tissue concentrations exceeded the tissue benchmark concentration, although lead and silver were based on detection limits. Arsenic concentrations in carcass tissue were approximately twice the benchmark value. No benchmark values for liver were exceeded at HBHA Pond. In white sucker tissue at HBHA Pond No. 3, arsenic (carcass), lead (carcass), mercury (fillet), and silver (carcass, fillet) tissue concentrations exceeded tissue benchmark concentrations, although lead and silver were based on detection limits. Arsenic and lead exceeded both reference area concentrations and tissue whole body benchmark concentrations; arsenic concentrations were approximately twice the benchmark value. No benchmark values for liver were exceeded at HBHA Pond No. 3, except the maximum liver value.

Whole-body tissue residue benchmarks and study area tissue concentrations for small fish (pumpkinseed) are presented in Tables 13 and 14 of Appendix 7B.8.3. Study area tissue concentrations of arsenic, lead, and silver exceed the corresponding tissue benchmark concentrations at both HBHA Pond and HBHA Pond No. 3, although lead and silver were based on detection limits. At HBHA Pond and HBHA No.3, only arsenic (whole body) exceeded tissue benchmark concentrations and reference area concentrations.

Based on the fish tissue results, the only COPCs with tissue concentrations above available benchmarks were silver, lead, mercury, and arsenic. Among these, only arsenic values were above reference tissue concentrations and were based on detected values. Measured tissue values for carcass and fillets of large fish were compared to tissue residue benchmarks based on whole body literature values. This comparison can over-estimate risk. Consequently, for arsenic, Table 36 shows both the individual component tissue concentrations as well as the reconstructed whole body concentrations.

Fish tissue data indicate potential effects on brown bullhead, white sucker, and pumpkinseed in the study area due to elevated concentrations of arsenic in carcass or whole body tissue at levels potentially associated with harm from exposure to arsenic. Tissue residue benchmark values for arsenic were not exceeded in largemouth bass collected in the study area for any tissue fraction.

5.2.2.2 Fish Population Studies. The 1999 Industri-Plex Superfund Site Fisheries Survey (Appendix 7B.7) was conducted in order to document the fish community structure at two study area ponds (HBHA Pond and HBHA Pond No. 3) at the Industri-Plex Site. These locations were compared to two reference ponds (Philip's Pond and South Pond). Fish were collected by a stratified sampling program which included the use of boat electrofishing, gill nets, trot lines, and eel pots. The survey also included an evaluation of the habitat characteristics of each pond to help augment fish population statistics. All captured fish were placed in live wells, measured for length, weight, and assessed for gross histopathology. Fish tissue samples were also collected for body burden tissue analyses (results presented above). Catch Per Unit Effort (CPUE) was used as a measure of relative abundance for electrofishing runs in one reference pond and one study area pond (HBHA Pond and Phillips Pond). Shallow water and dense submergent vegetation interfered with the duration of runs at the other two ponds (HBHA Pond No. 3 and South Pond), consequently CPUE was not calculated.

A number of indices were used to examine the population health and structure of largemouth bass (*Micropterus salmoides*), the main predator within all ponds. A measure of relative weight (Wr) for largemouth bass was used to compare the condition of an individual fish at all ponds to populations in New Hampshire. A t-test was used to determine if the Wr of largemouth bass in the contaminated ponds differed significantly from a reference pond. To determine the potential and susceptibility of recreational angling, a measure of the relative and proportional stock density was calculated for largemouth bass using length measurements. In addition, a predator prey model was used to determine the stability of the fishery community by using a ratio of weight of predator and prey fish. Finally the habitat parameters were analyzed by using a littoral shoreline development index which was compared to the fish abundance data collected in the ponds (Appendix 7B.7).

Overall, eight fish species were observed during the sampling program at the Industri-Plex Study area including: American eel (*Anguilla rostrata*), bluegill (*Lepomis machrochirus*), brown bullhead (*Ameiurus nebulosus*), carp (*Cyprinus carpio*), golden shiner (*Notemigonus chrysoleucas*), largemouth bass (*Micropterus salmoides*), pumpkinseed (*Lepomis gibbosus*), and white sucker (*Catostomus commersoni*) (Table 37). Philips Pond had the greatest diversity and number of fish compared to the two study area ponds. South Pond had the lowest species diversity, but had a larger population of largemouth bass than the study area ponds. White sucker and golden shiner were the two most dominant species captured in the two study area ponds. The species composition within the two study area ponds was similar although largemouth bass was captured at a much lower density in HBHA Pond No. 3. The gross histopathology of all fish captured appeared normal, although a few fish had eroded fins and abrasions.

Due to its importance as a top predator in the food chain, largemouth bass were defined as the primary species of interest. Within the four ponds, only 77 largemouth bass were collected. Philips Pond was the only system in which a full range of largemouth bass size classes were collected (Table 38). The low sample size was due mostly to the lack of abundance of smaller juvenile bass in the other three ponds. However, the low count of juveniles may be attributed to a bias in the sample design, which deliberately targeted larger fish. This sampling bias towards larger fish may have resulted in an underestimate of smaller fish in all four ponds sampled. The lack of juvenile fish in all but Philips Pond could also be due to the fact that the habitat characteristics were not optimal for spawning or survival of juvenile fish. In only Philips Pond were greater than 30 individuals (largemouth bass) collected and only nine and three were captured in HBHA Pond and HBHA Pond No.3, respectively.

The relative weight of largemouth bass in the study area ponds appears to within the range of those for selected New Hampshire ponds (USFWS Report, Appendix 7B.7). Mean relative weight of fish in the Stock category was highest in HBHA Pond and lowest in South Pond (Table 38). The lack of smaller fish was evident when measurements of proportional and relative stock densities showed that Philips Pond had the highest overall values, suggesting a more balanced population relative to the other ponds sampled (Table 38). The poor quality of the bass fishery was also evident in that no fish were captured in the Memorable (fish length 510 – 629mm) and Trophy size (fish length 630mm+) categories. The predator-prey ratios between HBHA Pond and Philips Pond identified HBHA Pond as having a higher quality predator-prey ratio (Table 38).

The habitat characteristics of HBHA Pond No. 3 and South Pond were similar in that there was a preponderance of aquatic vegetation and both ponds are relatively shallow. These habitat features offer poor habitat for recreational species (i.e., poor over-wintering cover, poor spawning habitat). Philips Pond appeared to provide the best overall habitat for largemouth bass with a good proportion of littoral vegetation and deeper habitat required for overwintering. While bass stock production may be limited in all but Philips Pond, South Pond and the two study area ponds offer suitable rearing habitat for smaller more tolerant species such as golden shiner and white sucker. Historic dissolved oxygen levels HBHA Pond suggested that persistent low levels of dissolved oxygen and anoxic conditions may be one of the driving forces in preventing smaller individuals from being recruited to the population and may also cause die offs. Juvenile bass feed on insect larvae, plankton, and small crustaceans. Lack of invertebrate prey base may also contribute to poor bass population structure.

The survey suggests that HBHA Pond and HBHA Pond No.3 likely provide poor habitat for recreational species. The survey data also suggest that there are population differences between paired study areas and reference areas. However, there are many factors which can affect the population structure of freshwater communities. Reasonable hypotheses concerning the apparent lack of juvenile fish in all ponds include: poor spawning habitat, low DO, poor overwintering habitat, and lack of submerged aquatic vegetation. Effects of abiotic factors, including contaminant concentrations may also contribute to poor recruitment. Due to the low sample size and possible sampling bias, differentiation of factors contributing to of the differences between the two study area ponds and the reference ponds cannot be made.

5.2.3 Benthic Invertebrate Community

The ecological effects evaluation for the benthic invertebrate community involved four components: comparison of sediment COPC concentrations to sediment benchmarks, evaluation of benthic invertebrate tissue data, toxicity testing, and evaluation of benthic invertebrate community structure.

Sediment Benchmarks. The benthic invertebrate comparison utilizing sediment benchmarks was conducted by area for individual samples (Tables 1 to 16, Appendix 7B.9). The characterization of potential effects used both a lower threshold bench mark level and a higher effects benchmark. The lower effects benchmarks were the benchmarks utilized in the sediment screening (section 2.3) and included LELs, ERLs, SQC, and SQB benchmark values. In addition, TEC (threshold effects concentrations) from MacDonald et al. (2000) were used for lower effects toxicity benchmarks. These values represent concentrations at which effects are unlikely to occur. The higher effects benchmarks used included severe effect limits (SELs from OMOE, 1993) and probable effects concentrations (PECs from Mac Donald, et al., 2000). The higher effects benchmarks represent concentrations at which potential effects may occur. Among the organics, benchmarks were not available for vinyl chloride, carbazole, N-nitrosodiphenylamime, and phenol. Among the inorganics, benchmarks were not available for barium, beryllium, selenium, thallium, and chromium VI. Potential ecological effects of these compounds could not be evaluated from the comparison to sediment benchmarks.

Six samples were collected from the Aberjona River, upstream of the confluence with Hall's Brook (AR). Samples were analyzed for inorganics only. Among the inorganics, the majority of samples had concentrations of antimony, cobalt, manganese, and nickel below lower effects toxicity benchmarks, representing a negligible effect on invertebrates. Exposure to iron, lead, and silver poses a low risk to invertebrates, as these metals exceeded only the lower effects benchmarks. PECs or SELs were exceeded by several metals in the majority of samples at AR. The exceedances of upper effects-based benchmarks of arsenic, chromium, copper, mercury, and zinc in the majority of AR samples indicates a potential risk to benthic invertebrates in Aberjona River sediment.

One sample (MC-13) was collected from the Aberjona River, downstream of the confluence with Halls Brook. Among the organics, five SVOCs and total PAHs were above lower toxicity benchmarks, but below higher toxicity benchmarks. Risk from exposure to organics is low at MC-13. Exposure to antimony, nickel, and silver poses a low risk to invertebrates, as these metals exceeded only the lower effects benchmarks. PECs or SELs were exceeded by several metals. The exceedances of upper effects-based benchmarks for arsenic, cadmium, chromium, copper, iron, mercury, and zinc indicates a potential risk to benthic invertebrates in Aberjona River sediment.

Two samples were collected from area BE-1. Among the inorganics, the majority were below lower effects toxicity benchmarks, representing a negligible effect on invertebrates. Exposure to cadmium, copper, mercury, and silver poses a low risk to invertebrates, as these metals exceeded only the lower effects benchmarks. PECs or SELs were exceeded by only lead and zinc in one of the two samples in BE-1. The exceedances of upper effects-based benchmarks lead and zinc in one of the BE-1 samples indicates a potential risk to benthic invertebrates.

One sample was collected from the pond in area BE-3. Among the inorganics, several were below lower effects toxicity benchmarks, representing a negligible effect on invertebrates. Exposure to cadmium, chromium, copper, iron, mercury, nickel, and silver poses a low risk to invertebrates, as these metals exceeded only the lower effects benchmarks. PECs or SELs were exceeded by only arsenic, lead, and zinc in BE-1. The exceedances of upper effects-based benchmarks lead and zinc in one of the BE-1 samples indicates a potential risk to benthic invertebrates.

Four samples were collected from the channel of the BECO Drainway (area BE-2). Among the inorganics, cobalt and manganese were below lower effects toxicity benchmarks, representing a negligible effect on invertebrates. Exposure to antimony, cadmium, chromium, iron, mercury, nickel, and silver poses a low risk to invertebrates, as these metals exceeded only the lower effects benchmarks. PECs or SELs were exceeded by several metals in the majority of samples at BE-2. The exceedances of upper effects-based benchmarks of arsenic, copper, lead and zinc in the majority of BE-2 samples indicates a potential risk to benthic invertebrates in sediment.

Eleven samples were collected from the shallow depths (0 - 1 feet of overlying water) in HBHA Pond (HB01). One sample (SD-MC-06) was analyzed for organics. No upper effects-based benchmarks were exceeded for VOCs, SVOCs, or pesticides. Based on exceedances of lower effects toxicity benchmarks in one sample, carbon disulfide, 2-methylphenol, benzo(a)anthracene, and total PAHs present a low risk to benthic invertebrates in HB01. All eleven samples were analyzed for inorganics. Among the inorganics, the majority of samples had concentrations of cobalt, manganese, and nickel below lower effects toxicity benchmarks, representing a negligible effect on invertebrates. Exposure to antimony, iron, and silver poses a low risk to invertebrates, as these metals exceeded only the lower effects benchmarks. PECs or SELs were exceeded by several metals in the majority of samples at HB01. The exceedances of upper effects-based benchmarks of arsenic, cadmium, chromium, copper, lead, mercury, and zinc in the majority of HB01 samples indicates a potential risk to benthic invertebrates in HBHA Pond sediment.

Eleven samples were collected from HB02-1 (HBHA wetland, channel). One sample (SD-MC-08) was analyzed for organics. No upper effects-based benchmarks were exceeded for VOCs, SVOCs, or pesticides. Based on exceedances of lower effects toxicity benchmarks in one sample, carbon disulfide, 2-methylphenol, total PAHs, and seven individual PAHs present a low risk to benthic invertebrates in HB02-1. All eleven samples were analyzed for inorganics. Exposure to antimony, cobalt, manganese, mercury, nickel, and silver poses a low risk to invertebrates, as these metals exceeded only the lower effects benchmarks. PECs or SELs were exceeded by several metals in the majority of samples at HB02. The exceedances of upper effects-based benchmarks of arsenic, cadmium, chromium, copper, iron, lead, and zinc in the majority of HB02-1 samples indicates a potential risk to benthic invertebrates in HBHA wetland sediment.

Seven samples were collected from HB02-2 (HBHA emergent wetland). None of the samples were analyzed for organics. Among the inorganics, exposure to antimony, manganese, mercury, nickel, and silver poses a low risk to invertebrates, as these metals exceeded the lower effects benchmarks in the majority of the samples. Manganese and mercury concentrations exceeded upper effects benchmarks in one of the seven samples. PECs or SELs were exceeded by several metals in the majority of samples at HB02-2. The exceedances of upper effects-based benchmarks of arsenic, cadmium, chromium, copper, iron, lead, and zinc in the majority of HB02-2 samples indicates a potential risk to benthic invertebrates in HBHA wetland sediment.

Five samples were collected from HB03-1 (HBHA wetland). One sample (SD-MC-10) was analyzed for organics. No upper effects-based benchmarks were exceeded for VOCs, SVOCs, or pesticides. Based on exceedances of lower effects toxicity benchmarks for organics, only 2-methylphenol presents a low risk to benthic invertebrates in HB03-1. All five samples were analyzed for inorganics. Among the inorganics, the majority of samples had concentrations of cobalt, nickel, and silver below lower effects toxicity benchmarks, representing a negligible effect on invertebrates. Exposure to antimony, iron, and lead poses a low risk to invertebrates, as these metals exceeded only the lower effects benchmarks. Manganese concentrations exceeded upper effects benchmarks in two of the five samples. PECs or SELs were exceeded by several metals in the majority of samples at HB03-1. The exceedances of upper effects-based benchmarks of arsenic, cadmium, chromium, copper, mercury, and zinc in the majority of HB03-1 samples indicates a potential risk to benthic invertebrates in HBHA wetland sediment.

Six samples were collected from HB03-2 (HBHA wetland pond). One sample (SD-MC-11) was analyzed for organics. No effects-based benchmarks were exceeded for VOCs or pesticides. Based on exceedances of lower effects toxicity benchmarks for SVOCs, only 2-methylphenol, total PAHs, and 10 individual PAHs present a low risk to benthic invertebrates in HB03-2. All six samples were analyzed for inorganics. Among the inorganics, the majority of samples had concentrations of cobalt below lower effects toxicity benchmarks, representing a negligible effect on invertebrates. Exposure to antimony, nickel, and silver poses a low risk to invertebrates, as these metals exceeded only the lower effects benchmarks. PECs or SELs were exceeded by several metals in the majority of samples at HB03-2. The exceedances of upper effects-based benchmarks of arsenic, cadmium, chromium, copper, iron, manganese, mercury, and zinc in the majority of HB03-2 samples indicates a potential risk to benthic invertebrates in HBHA wetland pond sediment.

Eight samples were collected from HB03-3 (HBHA emergent wetland). None of the samples were analyzed for organics. Among the inorganics, the majority of samples had concentrations of cobalt below lower effects toxicity benchmarks, representing a negligible effect on invertebrates.

Exposure to antimony, iron, manganese, mercury, nickel, and silver poses a low risk to invertebrates, as these metals exceeded the lower effects benchmarks in the majority of the samples. Mercury concentrations exceeded upper effects benchmarks in two of the eight samples. PECs or SELs were exceeded by several metals in the majority of samples at HB03-3. The exceedances of upper effects-based benchmarks of arsenic, cadmium, chromium, copper, lead, and zinc in the majority of HB03-3 samples indicates a potential risk to benthic invertebrates in HBHA wetland sediment.

One sample (MC-09) was collected from the HBHA wetland between HB02 and HB03. Among the organics, one SVOC, 2-methylphenol, was above lower toxicity benchmarks. Risk from exposure to organics is low at MC-09. Exposure to antimony, cobalt, nickel, and silver poses a low risk to invertebrates, as these metals exceeded only the lower effects benchmarks. PECs or SELs were exceeded by several metals. The exceedances of upper effects-based benchmarks for arsenic, cadmium, chromium, copper, iron, manganese, mercury, and zinc indicates a potential risk to benthic invertebrates in this reach of HBHA wetland.

Two samples were collected in the deep water of HBHA Pond (SD-MC-05 and SD-MC-07). These were collected in 12 and 13 ft of water, respectively. Among the VOCs, carbon disulfide exceeded the lower toxicity benchmarks in both samples. In addition, benzene and xylene were above lower effects benchmarks in sample SD-MC-05, only. Among the SVOCs, 2-methylphenol and PAHs were above lower effects benchmarks, representing a low potential risk to invertebrates in deep water of HBHA Pond. Among the inorganics, both of the deep samples had concentrations of cobalt below lower effects toxicity benchmarks, representing a negligible effect on invertebrates. Exposure to antimony, manganese, nickel, and silver poses a low risk to invertebrates, as these metals exceeded the lower effects benchmarks in the majority of the samples. PECs or SELs were exceeded by several metals in both of the deep HBHA Pond samples. The exceedances of upper effects-based benchmarks of arsenic, cadmium, chromium, copper, iron, lead, mercury, and zinc indicates a potential risk to benthic invertebrates in HBHA wetland sediment.

Exceedances of lower effects benchmarks by VOC concentrations was observed in the sediments in deep portions of HBHA Pond only. PAHs posed a potential risk to benthic invertebrates above the lower effects benchmarks at HB02-1, HBHA Pond (shallow and deep), HB03-1 and AR downstream (MC-13). No total or individual PAHs values in sediment exceeded upper effects-based benchmarks (SELs or PECs) in any sample, indicating that study area-wide risk from exposure to PAHs is low. The exceedances of upper effects-based benchmarks by sediment concentrations of arsenic, cadmium, chromium, copper, lead, mercury, and zinc indicated a potential risk to benthic invertebrates in most of the sediments sampled in the study area, with the exception of areas BE-1 and BE-3.

PEC Quotients were calculated for the 13 locations used for benthic invertebrate toxicity and community evaluations (MC-01 through MC-13, Figure 6). PEC quotients are the ratio of the sediment concentration divided by the chemical-specific PEC (Table 18 of Appendix 7B.9). The seven metals identified as frequently exceeding SELs or PECs were used in calculating the mean PEC quotient. For each station, the likelihood of toxicity from the all of the selected metals was characterized by calculating the mean value PEC quotient value for all of the metals.

The highest PEC quotients for any COPC were for arsenic, reflecting sediment arsenic concentrations of 8 to 33 times the PEC in study area sediment samples. Zinc also showed high PEC quotients with values from 3 to 17. The mean PEC quotients ranged from 0.7 to 2.0 at the reference locations, indicating low probability of toxicity associated with metals in the sediment. The study area values for mean PEC quotient ranged from 4.9 at MC-13 to 13.8 at MC-07.

Acid Volatile Sulfide/Simultaneously Extracted Metals. Bioavailability of metals in sediment can significantly affect potential toxicity of metals in the sediment to benthic organisms. It is a common observation that similar concentrations of metals exhibit a wide range of effects on benthic organisms, depending on the properties of the sediments. Bioavailability of certain divalent metals (cadmium, copper, lead, mercury, nickel, and zinc) is also influenced by the amount of sulfide contained within the substrate. If the amount of acid-volatile sulfide (AVS) exceeds the amount of simultaneously extracted metals (SEM), then the divalent metals are unavailable for leaching from the substrate into pore water or the overlying water column. The comparison between SEM and AVS included calculating the amount of SEM and AVS in units of umol/g, subtracting the AVS value from the SEM value, and then normalizing this difference by the amount of organic carbon (expressed as a fraction) in the sediment (USEPA, 1999):

$$\text{Normalized Value (umol/g}_{OC}\text{)} = \frac{\text{(SEM-AVS)}}{f_{OC}}$$

where umol/g_{OC} is the concentration of the metal in micro moles per gram of organic carbon, and f_{OC} is the fraction of organic carbon in sediment.

If AVS or SEM parameters were qualified by the laboratory as U or UJ, a value of zero was used. If the normalized value is less than 130 umol/g_{OC}, then sediments are "unlikely to be toxic." If the normalized value is between 130 and 3,000 umol/$g_{OC,}$ then sediments are of "uncertain toxicity." If the normalized value exceeds 3,000 umol/$g_{OC,}$ then sediments are "likely to be toxic" (USEPA, 1999). A negative value indicates that AVS exceeds SEM, thus the divalent metals are unavailable for leaching into pore water or the overlying water column. AVS-SEM data also may vary seasonally, with AVS concentrations typically higher in the warmer seasons. Three rounds of AVS-SEM sampling occurred at the study area, August/September 1995, November 1997, and June 2001 (Table 39). A total of 8 study area locations and 26 reference locations were sampled in the summer sampling rounds; and no study area locations and two reference locations (SD-24-03-ME and SD-25-02-ME) were sampled in the November 1997 (winter) sampling round.

Normalized comparisons for data collected in summer (June through September), as presented in Table 39, indicate that at 50 percent (4 of 8) of study area samples, AVS exceeded SEM, thus divalent metals are likely to be biologically unavailable to benthos. AVS exceeded SEM in approximately 56 percent (15 of 26) of reference area sites. High AVS (indicative of anoxic sediment conditions) were observed in the deep pond samples at MC-05 and MC-07.

No sample locations exceed 3,000 umol/g_{OC} to indicate likely toxicity. The one sample collected at HBHA Pond Shallow (SD-MC-06, 60 umol/g_{OC}) had a normalized value between zero and 130 umol/g_{OC}, indicating unlikely toxicity. At HBHA Pond Deep, both samples (SD-MC-05 at -4,748 umol/g_{OC} and SD-MC-07 at -3,290 umol/g_{OC}) had normalized values below zero umol/g_{OC} indicating that divalent metals are biologically unavailable to benthos. At HBHA Wetland, two samples (SD-MC-08 at -749 umol/g_{OC} and SD-MC-09 at -475 umol/g_{OC}) had normalized values below zero umol/g_{OC} indicating that divalent metals are biologically unavailable, and two samples (SD-MC-10 at 393 umol/g_{OC} and SD-MC-11 at 591 umol/g_{OC}) had normalized values between 130 umol/g_{OC} and 3,000 umol/g_{OC} indicating uncertain toxicity. The one sample collected at SD-MC-13 (105 umol/g_{OC}) had a normalized value between zero and 130 umol/g_{OC}, indicating unlikely toxicity.

Because AVS values are typically lower in the winter months, one would expect the normalized values to be higher in the winter, indicating a higher bioavailability. However, winter data from the two reference locations does not support this expected trend because the normalized value decreased at both locations (-28 umol/g_{OC} at SD-24-03-ME in the winter compared to 29 umol/g_{OC} at SD-24-03-FW in the summer, and 15 umol/g_{OC} at SD-25-02-ME in the winter compared to 59 umol/g_{OC} at SD-25-02-FW in the summer). Therefore, no conclusions regarding bioavailability of divalent metals in the winter can be drawn given the small sample size of winter AVS/SEM data at the reference areas, and lack of winter AVS/SEM data from study area locations.

The AVS/SEM approach is used for evaluation of biological effects that predicted from SEM concentrations measured in the sediments. The data collected in the study area indicates that SEM levels are lower than would be expected to cause toxicity to benthic organisms from this group of metals. This analysis applies to the six SEM metals (copper, cadmium, nickel, lead, silver, and zinc). None of the samples analyzed showed likely toxicity from these divalent metals based on AVS/SEM data, and a few showed uncertain toxicity. Toxicity in field-collected sediments can be caused by other chemicals, and as such, AVS/SEM data is not a good predictor of toxicity of sediments with high concentrations of other COPCs including chromium and arsenic.

Tissue Residue Comparison. COPC concentrations in benthic invertebrate tissues were evaluated in two ways. First, invertebrate tissue samples were collected and analyzed for COPC concentrations and compared to reference area tissue concentrations. Statistical analyses of invertebrate tissue data are presented in Section 5.1. There were no significant differences in tissue concentrations among the species sampled for any inorganic COPC. For those metals with invertebrate tissue concentrations elevated in the study area (arsenic, cadmium, chromium, copper, lead, and zinc), only the concentration of arsenic and zinc were statistically higher in the study area versus reference locations. In addition, concentrations of fluoranthene and pyrene in study area tissue were statistically higher than tissue samples from reference locations ($p < 0.05$, Table 28).

COPC tissue concentrations in invertebrates collected from the study area were compared to tissue residue benchmarks reported in the literature for freshwater invertebrates species (Appendix 7B.6.4). The Environmental Residue-Effects Database (ERED; USACE, 2004) data entries for fluoranthene, pyrene, arsenic, chromium, copper, lead, and zinc in freshwater invertebrate tissue are summarized in Appendix 7B.6.4.

The highest concentrations of fluoranthene and pyrene in tissue of invertebrates in the study area were observed at MC-11. This location also had the highest concentrations of PAHs in sediment (including pyrene and fluoranthene) among those sampled for invertebrate tissue. The concentrations of fluoranthene measured in invertebrate samples in the study area (0.45 mg/kg maximum and 0.114 mg/kg average, Table 10 and Appendix 7B.6.1), are all below available tissue residue benchmarks for amphipods and oligochaetes. The concentrations of pyrene measured in invertebrate samples in the study area (0.31 mg/kg maximum and 0.084 mg/kg average, Table 10 and Appendix 7B.6.1), are all below available tissue residue benchmarks for amphipods and oligocheates. These data indicate that the exposure to some PAHs is higher in the study area than at reference locations, however, the potential for adverse effects based on tissue reside benchmarks is low.

The concentration of arsenic in invertebrate tissue was highest in the shallow water of HBHA Pond (26 mg/kg), and exceeded all available tissue residue benchmarks for invertebrates (NOED, mortalilty, stoneflies, and waterfleas). The average concentration of arsenic in invertebrate tissue in the study area was 9.4 mg/kg which was 20 times higher than the average at reference locations. The average tissue concentration study area-wide of 9.4 mg/kg was slightly below the highest available NOED of 9.8 mg/kg for *Daphnia magna* (Appendix 7B.6.4). These data indicate that the concentration of arsenic in invertebrate tissue is high, particularly in the shallow sediment of HBHA Pond, and these concentrations are above no-effects level benchmarks. The average concentrations study area-wide are also generally above no-effects level benchmarks. These results are limited in that there were no LOED invertebrate tissue benchmarks for arsenic.

The concentration of chromium in invertebrate tissue in the study area was highest in the shallow water of HBHA Pond and at MC-13 (16 mg/kg). The study area-wide average for chromium was 6.4 mg/kg and the only detected concentration at reference location was 1.2 mg/kg. The reference value exceeded both of the tissue residue benchmarks for oligochaetes (0.21 and 0.336 mg/kg). This indicates these benchmarks for oligochaetes are very conservative, and risk associated with exceedance is not incrementally greater than at reference locations. The other available benchmarks for chromium were from studies with stoneflies and ranged from 1.44 to 1.84 mg/kg (effective dose for 10% and 50%, respectively). The study area-wide average value for tissue chromium of 6.4 mg/kg exceed these benchmarks. The data indicate that tissue levels of chromium may be associated with effects on invertebrates, but the uncertainty associated with this evaluation is high.

The concentration of copper in invertebrate tissue in the study area was highest in HBHA Pond (26 mg/kg at MC-06 and 22 mg/kg at MC-07). This tissue residue bench marks show a wide range for copper. Data for midges and amphipods have NOED values ranging from 13 to 85.4

mg/kg, LOED ranging from 18 to 130 mg/kg and LD50 values ranging from 16 to 155 mg/kg. The average concentration in invertebrates sampled in the study area (16 mg/kg) falls below the effects levels for most invertebrate tests. The highest observed concentrations in HBHA Pond, however, exceed most benchmarks associated with effects (Appendix 7B.6.34). These data indicate that there is potential for adverse effects from exposure to copper based on tissue reside benchmarks in HBHA Pond.

The concentration of lead in invertebrate tissue in the study area was highest in the shallow water of HBHA Pond (13 mg/kg, chironomid). This value was below the majority of the all available tissue residue benchmarks for invertebrates (amphipods and midges, range 2.6 to 280 mg/kg). The average concentration of lead in invertebrate tissue in the study area of 4 mg/kg exceeded only a reproductive NOED for midges (Appendix 7B.6.4). These data indicate that the potential for adverse effects from exposure to lead based on tissue reside benchmarks is low.

For zinc, tissue residue effects level at the lowest effects level benchmark (LD25 for *Hyalella azteca* of 19.5 mg/kg) was exceeded at all study area stations, with the highest tissue concentrations observed in midge tissue from MC-06 (160 mg/kg). The lowest effects-based benchmark (LOED, reproduction) for *C. tentans* of 34 mg/kg was exceeded at MC-06, MC-08, and MC-13. However, other reported values for effects-based tissue residue levels (LD6, growth) were as high as 524 mg/kg for species of midges (*Chironomus riparius*), which was above any tissue level measured within the study area. These data indicate that there is potential for adverse effects from exposure to zinc based on tissue reside benchmarks in HBHA Pond, however, there is uncertainty associated with this evaluation.

Based on the invertebrate tissue results, potential for adverse effects for PAHs is low. There is potential for adverse effects from exposure to copper and zinc in sediments in shallow water of HBHA Pond based on tissue residue benchmarks. There is also potential for adverse effects from chromium, especially at MC-06 and MC-13, but the uncertainty associated with the evaluation is high. The tissue concentrations of arsenic in invertebrates in the study area, and particularly at

MC-06 in HBHA Pond, were high. The evaluation of tissue residue benchmarks indicates potential effects from exposure to arsenic, however, this evaluation was based on NOED values only.

Toxicity Testing. The sediment quality triad (SQT) approach was used to integrate data from chemical and physical analyses, whole-sediment laboratory toxicity tests and benthic community measures. The sampling included benthic community composition analyses at each station, as well as short-term and longer-term laboratory sediment toxicity tests for both the amphipod, *H. azteca* and the midge, *C. tentans*. The sediment toxicity tests are direct measures of sediment toxicity used to evaluate the potential effects of whole sediment on representative benthic macroinvertebrates.

Thirteen stations, eight in the study area, and five reference locations, were selected to represent a cross-section of habitat types throughout the study area. The triad sampling locations are shown on Figure 6. Description of the sediment characteristics at each of the sampling locations are presented in Table 40.

Sediment samples from the upper two inches were collected at each sampling stations. Short-term (10-day) toxicity tests were conducted at all 13 sampling locations with both *H. azteca* and *C. tentans*. For stations where the results of the short-term tests indicated that survival did not significantly differ from that in reference locations and controls sediment, long-term tests were conducted for these same two species. The sequential testing eliminated the need to run long-term tests for sediments in which toxicity had been demonstrated in short-term assays.

<u>Toxicity testing results</u>. Endpoints for each test were compared statistically to results in laboratory controls (artificial sediment) according to *Methods for Measuring the Toxicity and Bioacummulation of Sediment-associated Contaminants with Freshwater Invertebrates* (USEPA, 2000a). The purpose of the laboratory controls are to determine the health of the test organisms and validity of the tests. It is not an unusual result to have the growth, survival, or other

measurements of organism health perform differently on artificial sediment in laboratory controls as compared to natural sediments from reference locations. A summary of the toxicity endpoints from the sediment toxicity data are presented in Figures 8 to 21. The endpoints are expressed as a ratio of each study areas' endpoint (mean of 8 replicates) to the result observed for the corresponding field reference location. The selection of the corresponding reference location for each sample was based on the similarity of the habitat.

The 10-day toxicity tests for *H. azteca* showed short-term exposure to sediments resulted in reduced survival at both deep sediment stations in HBHA Pond (SD-MC-05 and SD-MC-07) compared to laboratory controls (Table 41) and to reference locations (Figure 9). The survival rates were 0% at SD-MC-05 for both the sample and its field duplicate (Appendix 7B.10). High ammonia concentrations in the overlying water reported by the laboratory may have influenced this result. The mean survival for SD-MC-07 (36%) was not statistically significantly different from the control, however, the sample met the project-specific criterion for severe toxicity (>50% mortality), and the control for this test group did not meet the test's acceptability criteria. The 10-day survival rates were greater than 90% for all other samples. The 10-day growth at SD-MC-05 and SD-MC-07 was not reported, as significant reduction in survival was noted.

The 10-day survival and growth was also significantly lower than laboratory controls for *C. tentans* at stations SD-MC-05 and SD-MC-07. In addition, there was reduced growth of *C. tentans* in the 10-day tests at SD-MC-11 (HBHA wetland pond) and at reference SD-MC-01 (Appendix 7B.11).

The 42-day chronic toxicity tests for *H. azteca* showed significant reduction in survival at 28, 35, and 42 days at station SD-MC-06 (HBHA Pond, shallow), as compared to laboratory controls (Table 41 and Appendix 7B.12). There was a reduction in growth of *H. azteca* observed at 42 days at stations SD-MC-10, SD-MC-11 and at reference locations SD-MC-01 and SD-MC-02, as compared to laboratory controls. For the reproduction endpoint, there was a decrease ($p < 0.05$) in neonates per female at station SD-MC-06 as compared to laboratory controls. Reproduction (neonates per female) was also low at SD-MC-11 compared to the stream reference (Figure 13) .

The life-cycle chronic toxicity tests for *C. tentans* showed significant reduction in survival (20-day) and growth at station SD-MC-06 as compared to laboratory controls (Table 41 and Appendix 7B.13). In addition, the percent of the midges emerged was lower than laboratory controls ($p<0.05$) and the mean number of days survived was lower for males (but not females) at station SD-MC-06. The long-term tests were not run on samples from stations SD-MC-05 and SD-MC-07, since both showed evidence of toxicity in the 10-day tests. None of the stations, other than SD-MC-06, showed evidence of significant impairment as compared to laboratory controls, based on the life-cycle endpoints for *C. tentans*. Twenty-day survival was low for *C. tentans* at SD-MC-10 (19%) and SD-MC-11 (17%) (Figures 14 and 15); although these values were low, they were not statistically significant in comparison to laboratory controls which also had low survival.

Based on the results of the toxicity testing, there is evidence of acute toxicity to benthic organisms at stations SD-MC-05 and SD-MC-07 in HBHA Pond. The data also present strong evidence of toxicity to invertebrates in at station SD-MC-06 in the shallow area of HBHA Pond. Evidence of toxicity was also observed for stations SD-MC-10 and SD-MC-11 in the HBHA wetlands, however the number of endpoints statistically significant from controls were limited. Toxicity data indicate potential chronic effects from exposure to sediments at these stations.

<u>Community Composition Results</u>. As part of the sediment quality triad sampling, samples were collected for identification and enumeration of benthic macroinvertebrates from each triad sampling location. A number of community indices were calculated to evaluate the community composition of the stations (Table 42).

Total abundance of organisms was highest among the depositional stream/wetland reference locations (Table 42). The number of different taxa (most identified to species level) observed at each station ranged from 1 to 34. The majority of stations had more than 10 taxa represented, with the exception of the deep pond stations (MC-05 and MC-07), the deep pond reference station (MC-03), two of the shallow pond stations (MC-06 and MC-11), and one of the depositional stream/wetland stations (MC-10). The diversity index values (Shannon-Weiner Index) at the deep pond study area stations, two of the shallow pond stations (MC-06 and MC-11), and one of the depositional stream/wetland stations (MC-10) were also subsequently low.

Most stations were dominated by either Oligochaeta (aquatic worms) or by Chironomidae (midges). Since all of the stations sampled were selected to represent depositional areas, high abundances of Oligochaetes and Chironomids were not unexpected, since these taxa are frequently found in fine sediment. However, communities composed of high proportions of Oligochaetes and Chironomids, with relatively low proportions of other taxa, are usually considered indicative of contaminated sediments (Canfield *et al*., 1994).

At the deep pond, at both stations MC-05 and MC-07, only one organism was found at each location. At the deep pond reference station, percent Chironomids was 33 percent with no Oligochaetes. Among the shallow pond stations, the percent of the community consisting of Oligochaetes plus Chironomids ranged from 68 to 99 percent at the study area locations, while only 58 percent at the shallow pond reference station. At the depositional stream/wetland stations in the study area, percent Oligochaetes plus Chironomids was between 50 and 97 percent while at the depositional stream/wetland reference stations percent Oligochaetes plus Chironomids was between 52 and 79 percent.

Pielou's evenness and the percent dominance calculations are both measures of the distribution of the individuals among all of the species present. Evenness is high (approaching 1.0) when the organisms present are evenly distributed among all species at a station. Conversely, percent dominance is a measure of the proportion of the individuals belonging to the most abundant species in the study area. High percent dominance and low evenness are usually indicative of an impaired habitat that allows the dominance of a few tolerant species (Plafkin *et al.*, 1989). All three shallow pond stations had low evenness (below the corresponding reference value). At the depositional stream/wetland locations, evenness values range from 0.636 to 1.0, which are similar to the values calculated for the depositional stream/wetland reference locations which range from 0.509 to 0.783. Evenness could not be calculated for the deep pond stations (MC-05 and MC-07) because of low total abundance and subsequent low diversity index values.

Among the deep pond stations, station MC-07 was dominated by a pollution tolerant Chaoborids, while MC-05 was dominated by moderately pollution sensitive Chironomids. The deep pond reference station was also dominated by pollution-tolerant Chaoborids. All shallow pond stations, as well as the shallow pond reference station, were dominated by pollution tolerant Oligochaetes. At the depositional stream/wetland stations, all study area stations were dominated by pollution tolerant Oligochaetes. Two of the reference stations were dominated by pollution-tolerant Chironomids or Oligochaetes, and one was dominated by moderately pollution-sensitive Isopods.

A Community Index (CI) was calculated as part of a weight of evidence approach to determine benthic community impairment. For the Community Index, each calculated index (total abundance, Shannon-Weiner Index, tolerance value, etc.) which showed impairment relative to the reference location was assigned value of 1 and totaled for each station. The highest Community Index values indicate a weight of evidence for impaired community characteristics. Within the deep pond, community index values for both study area stations indicated strong impairment compared to the reference location (CI=6). These stressed conditions may potentially be due to low oxygen, contaminants, or a combination of factors. Dissolved oxygen levels below 1.0 mg/L have been recorded at sample depths for both MC-05 (9 feet) and MC-07 (11.5 feet) during summer months. These low oxygen conditions are typical for deep water bodies in the summer when anoxic conditions naturally prevail below the thermocline. Among the shallow pond habitats, all three study area stations showed an equal degree of impairment compared to the reference area (CI=6). In shallow ponds, where dissolved oxygen levels at sampling depth tend to remain above 6 mg/L in the summer, oxygen is less likely to be a cause of stressed conditions. For the depositional stream/wetland reference stations, the lowest (poorest score) among the three reference stations was selected as the basis for comparison. Low community index values (measured against the selected reference area values) for MC-08 and MC-13 suggest that communities in these locations are under low stress (CI=0 to 2). Station MC-10 appears to be the most stressed of the depositional stream/wetland study areas (CI=4), showing the most indicators of benthic community impairment.

6.0 RISK CHARACTERIZATION

6.1 Risk Estimation

Risk estimation consists of integrating exposure profiles with exposure effects information and summarizing associated uncertainties. Several species or species groups were used to evaluate risks to ecological receptors in the Industri-Plex Study area. In the following text, each of the assessment endpoints is reviewed, results for measurement endpoints are provided, and the relationship between assessment and measurement endpoints is discussed, including the confidence in the relationships relative to accurately predicting risk and associated uncertainties. As applicable, the relationship between areas of contamination and the estimation of exposure effects are discussed.

Risks to wildlife receptors (muskrat, otter, heron, mallard, and shrew) were evaluated using the HQ approach, whereby daily dose, estimated from study area-specific data, was divided by a TRV. TRVs were based on a concentration that was not expected to cause an adverse effect (*i.e.*, a NOAEL), most often related to mortality or reproduction, in the exposed individual. The 95% UCL concentration in each media (sediment, water, and biota) was calculated to represent the reasonable maximum exposure, and the resulting dietary dose was divided by a NOAEL TRV to calculate the UCL/NOAEL or lower threshold effects level HQ. Any COPC with HQ values less than 1 for the lower effects level was considered to present a negligible risk to the receptor. Where the majority (>50%) of the study area or habitat area exceeded the lower threshold effects level, exposure to the COPC was identified as a potential risk to the assessment population. Secondly, an exposure scenario using the average case concentration for the COPC in each media (sediment, water, and biota) was used to calculate an average exposure scenario. The resulting dietary dose was divided by a LOAEL TRV. Using less conservative LOAEL-based TRVs and average-case exposure scenario, the upper bound of the threshold for adverse ecological effects can be estimated. The upper-bound TEL represents the media concentrations at which ecological impacts are predicted to occur. Where the average case scenario for exposure or the majority of the stations within the habitat area exceed the upper TELs, it is assumed that the COPC represents a risk to receptor populations.

6.1.1 Semi-aquatic Mammals. The assessment endpoint was:
Sustainability (survival, growth, reproduction) of local populations of omnivorous, semi-aquatic mammals.

Risks to muskrats, used to represent an omnivorous semi-aquatic mammal, were evaluated using the HQ approach, whereby daily dose, estimated from study area-specific data, was divided by a TRV. For the UCL/NOAEL case, every station evaluated had two or more inorganic COPCs with HQs in excess of 1. For this lower effects level, HQs were greater than 1 for aluminum, antimony, arsenic, chromium, cobalt, copper, lead, selenium, vanadium, and zinc. As discussed in section 5.2.1, the average/LOAEL HQs were less than 1 for all COPCs except aluminum and arsenic (Table 31). Among the stations used in the muskrat model, the greatest number of exeedences for metals was for stations AR and HB02-1.

The UCL/NOAEL HQs were below 1 at each station for cadmium. For antimony and vanadium the HQs at the reference stations were 3 and 8, respectively. As the exposures at study area stations for antimony and vanadium were equal to or below reference at most of the stations, the risk to muskrat is not incrementally greater than reference. Consequently the risks to cadmium, antimony, and vanadium are considered negligible. The HQ at station AR represents a low risk to muskrat in a limited area of the study area.

There were low risks at a limited number of stations for chromium, copper, cobalt, lead, selenium, and zinc. The magnitude of the HQs were low (6 or less) at a limited number of stations for each of these metals, with the exception of an HQ of 9 for zinc at station HB02-1. As none of these inorganics had HQs above 1 for the average/LOAEL case, and there were limited number of stations with HQs above 1 for the UCL/NOAEL case, the risk of dietary exposure of muskrat to these inorganics is low.

The only COPC other than arsenic with HQs above 1 for most of the stations in the UCL/NOAEL case was aluminum. Only three stations had HQs above the reference value of 3. The only location with an HQ greater than 2 was HB02-1. Confidence in the prediction of effects from exposure to aluminum from the dietary model are low. Availability and toxicity of aluminum is strongly influenced by pH of the media, due to the influence of pH on solubility. Soluble and toxic forms of aluminum are generally found in soils only at soil pH values less that 5.5 (USEPA, 2003b).

For mammals and birds, previous studies suggest that the direct toxic potential of aluminum is low compared to that of many other inorganics; mammals and birds can effectively limit the absorption of aluminum and effectively excrete any excess (Scheuhammer, 1987). Bioavailabilty of aluminum is strongly influenced by the form ingested and also by the presence of other material in the digestive system. This was based on a bioavailability factor of 100%, which likely overestimates aluminum exposure.

Based on the low confidence in the aluminum TRVs, the uncertainty associated with bioavailability, and the limited number of stations with HQs above the reference value, the confidence in aluminum exceeding TRVs as adequately representing risk to receptors is low.

The HQs in the UCL/NOAEL model were consistently the highest for arsenic (ranging from 2 to 26). Arsenic was the only COPC with average/LOAEL HQ values consistently above reference concentrations. For the average/LOAEL model, a TRV of 3.22 mg/kg-d dose of arsenic was used.   The other available LOAEL TRVs for muskrat ranged from 0.03 to 40.2 mg/kg-d. The highest TRV of 40 mg/kg-d was subchronic and based on a physiological endpoint, and was not selected as being representative of the exposure scenario. If a LOAEL TRV with a chronic exposure and growth endpoint of 24 mg/kg-d was used as a reference dose, none of the average HQ values at any station would be greater than 1. This indicates a high uncertainty in the estimation of risk to muskrats related to the toxicity reference values.

The toxicity of arsenic depends on its chemical speciation, occurring in various oxidation states and in organo-complex forms. The TRV is based on oral doses of sodium arsenite, which is likely to be more toxic than forms found in the muskrat diet in the study area. Due to these uncertainties, the confidence in the conclusion of risk to muskrat is reduced, although the HQ values indicate the exposures in diet are high, particularly in HBHA wetland.

Based on the endpoint for muskrat, the data indicate a low risk of ecological effects from exposure to chromium, copper, cobalt, lead, selenium, and zinc at some stations. The dietary models indicate potential of risk from dietary exposure to arsenic at all stations in the HBHA pond and wetland complex, as well as the Aberjona River upstream (AR) and downstream (MC-13), and at location BE-2, with a moderate level uncertainty associated with the potential risk.

6.1.2 Piscivorus Mammals. The assessment endpoint was:
Sustainability (survival, growth, reproduction) of local populations of piscivorus, semi-aquatic mammals.

Risks to river otter, used to represent a semi-aquatic piscivorous mammal, were evaluated using the HQ approach, whereby daily dose, estimated from study area-specific data, was divided by a TRV. The majority of the diet was based on consumption of small fish (pumpkinseed) caught in the study area.   For the UCL/NOAEL case, the study area-wide HQ for otter was less than 1. These results indicate that risks from dietary exposure of river otter to concentrations of COPCs in the study area are below levels expected to cause ecological effects.

6.1.3 Piscivorus Birds. The assessment endpoint was:
Sustainability (survival, growth, reproduction) of local populations of piscivorus birds.

Risks to green heron, used to represent piscivorous birds, were evaluated using the HQ approach, whereby daily dose, estimated from study area-specific data, was divided by a TRV. The majority of the diet was based on consumption of small fish (pumkinseed) caught in the study area. For both the UCL/NOAEL and average/LOAEL cases, the study area-wide HQs for heron were less than 1 for chromium, lead, mercury, and zinc. Based on the dietary exposure models, there is little evidence of significant ecological effects associated with contaminants on piscivous birds feeding in the study area.

6.1.4 Waterfowl. The assessment endpoint was:
Sustainability (survival, growth, reproduction) of local populations of omnivorous waterfowl.

Risks to mallard, used to represent omnivorous waterfowl, were evaluated using the HQ approach, whereby daily dose, estimated from study area-specific data, was divided by a TRV. The majority of the diet was based on consumption of invertebrate and plant tissue for which study area-specific data were available. For the UCL/NOAEL cases, the study area-wide HQs for mallard were greater than 1 for chromium, lead, and zinc. However lead values did not exceed those observed at reference locations, therefore, exposure to lead does not represent risk incrementally greater than reference. The HQs for chromium and zinc exceeded 1 in HBHA Pond, and HBHA wetland as well as in the study area-wide model. Concentrations of zinc in plant contributed to higher exposures to zinc in HBHA wetland than in HBHA Pond. For the average/LOAEL case, no HQs for mallard were greater than 1. Based on the dietary exposure models, there is evidence of potential risk to waterfowl exposed to chromium and zinc in the study area but the risk is low since there were no exceedances of average/LOAEL effects levels.

6.1.5 Small Terrestrial Mammals. The assessment endpoint was:

Sustainability (survival, growth, reproduction) of local populations of small terrestrial mammals.

Risks to northern short-tailed shrew, used to represent an insectivorous small terrestrial mammal, were evaluated using the HQ approach. A daily dose, estimated from study area-specific soil data, was used to estimate COPC concentration in diet. The daily dose was divided by a TRV. For the UCL/NOAEL case, the highest HQs observed for arsenic at A6, BE-2, HB02-2, and HB03-3. HQs for arsenic did not exceed reference values at BE-1 or HB04. Station A6, representing soils at the northern end of HBHA Pond, had HQs greater than 1 for antimony, arsenic, lead, mercury, and thallium. Antimony and thallium also exceeded HQs of 1 at HB02-2. The average/LOAEL HQs were less than 1 for all COPCs except arsenic (Table 35). HQ's for arsenic exceeded upper threshold effects levels at stations A6, BE-2, HB02-2, and HB03-3.

For the average/LOAEL model, a TRV of 1.5 mg/kg-d for arsenic was used. This LOAEL TRV for arsenic is based on a chronic (3 generation) exposure with reproductive endpoint in mouse (Table 5 in Appendix 7C.13). There is uncertainty associated in the selection of a TRV for shrew, as well as the form of arsenic in soil. The TRV is based on oral doses of sodium arsenite which is likely to be more toxic than forms found in the shrew diet in the study area. Due to these uncertainties, the confidence in the conclusion of risk to shrew is reduced, although the HQ values indicate the exposures in diet are high, particularly at HB02-2.

Based on the endpoint for shrew, the data indicate a low risk of ecological effects from exposure to antimony, lead, mercury, and thallium at A6. The dietary models indicate potential of high risk from dietary exposure to arsenic at stations A6, BE-2, HB02-2, and HB03-3. Overall, the exposure analysis indicates that survival or reproduction for shrew may be impaired in the study area due to exposure to inorganics in diet, but these results are associated with a moderate level of uncertainty.

6.1.6 Fish Receptors. The assessment endpoints for fish were:

Sustainability (survival, growth, reproduction) of local populations of predatory fish, bottom-feeding fish, and small foraging fish.

Comparisons of COPC concentrations in fish tissue (largemouth bass, white sucker, brown bullhead, and pumpkinseed) collected from the study area against COPC concentration in tissue collected from reference stations indicated that fish within the study area carry higher body burdens of inorganic COPCs than do fish from reference stations (Appendix 7B.8.3). Fish samples from the study area were collected in HBHA Pond and HBHA wetland. Fish species may migrate and forage throughout the study area, particularly the larger species. Similar concentrations of COPCs were found in the two collection areas, with the highest ratios of study area-to-reference COPC body burdens found in fish collected from HBHA Pond No. 3.

<u>Large mouth bass</u>: Tissue concentrations in largemouth bass indicated elevated concentrations of arsenic in HBHA Pond and HBHA Pond No. 3 as compared to reference areas. Although tissue data indicated elevated exposure of largemouth bass to arsenic concentrations, tissue residue benchmark values for arsenic were not exceeded in largemouth bass collected in the study area for any tissue fraction. Population statistics focusing on the health of the largemouth bass population demonstrated a lack of smaller juvenile bass in the study area ponds. However, the population and habitat data showed that poor recruitment of smaller size classes could be due to physical habitat characteristics of the pond. Consequently, the poor quality of the predatory fish population in the study area can not be attributed solely to contaminant concentrations. Overall, there is little evidence of significant ecological effects associated with contaminants on predatory fish in the study area.

<u>White sucker and brown bullhead:</u> The highest concentrations of arsenic were observed among tissue samples from the bottom-feeding fish in HBHA pond and HBHA wetland. Overall, the tissue data provide evidence of exposure of fish to arsenic at levels above which growth or survival may be impaired. The magnitude of the exceedance of the tissue benchmark (no-effects dose level) is low (the NOED is exceeded by a factor of 2). However, lowest observed effect

dose (LOED) values in bluegills in one study (whole body) ranged from 1.72 (reproduction, adult) to 11.6 mg/kg (growth, adult). The maximum concentration in carcass tissue of arsenic in brown bullhead tissue and white sucker were 1.4 mg/kg and 2.9 mg/kg, respectively. These tissue concentrations resulted in maximum HQs of 2 and 6, respectively. The data indicate that bottom-feeding fish in the study area carry body burdens of arsenic at levels associated with ecological effects.

Fish population studies did not provide strong evidence of impairment of bottom-dwelling fish in the study. Ecological effects related to contaminant toxicity on survival or reproduction were not distinguishable from other biotic or abiotic factors impairing fish populations within the study area.

Pumpkinseed Pumpkinseed represent populations of small foraging fish in the study area. Whole body tissue concentrations indicate exposure to arsenic at levels 6 to 8 times higher than conditions at the reference locations. The average arsenic tissue concentration in pumpkinseeds was 1.4 times the NOED for bluegills. Among the 10 pumpkinseed tissue samples, the highest value observed was three times higher than the NOED. A whole body LOED for reproduction in bluegills is reported by (Gilderhaus, 1966) at 1.72 mg/kg. The average pumpkinseed tissue concentration of 0.63 mg/kg and the maximum of 1.6 mg/kg in the study area are both less than the LOED. These data indicate an exposure to arsenic in the small foraging fish at levels above lowest effects concentrations.

Fish population studies did not provide strong evidence of impairment of small foraging fish in the assessment populations. Ecological effects related to contaminant toxicity on survival or reproduction in small foraging fish were not distinguishable from other biotic or abiotic factors impairing fish populations within the study area.

6.1.7 Benthic Invertebrate Community. The assessment endpoint for the benthic invertebrate community was:

Sustainability (survival, growth, reproduction) of local populations of benthic invertebrates.

The assessment endpoint for the benthic invertebrate community was not met. There were four separate lines of evidence evaluated. The weight of evidence from the benchmarks screening, invertebrate tissue concentrations, community composition, and toxicity testing are each discussed below.

Exceedance of lower effects benchmarks by VOC concentrations was observed in the sediments in deep portions of HBHA Pond only. PAHs posed a potential risk to benthic invertebrates above the lower effects benchmarks at HB02-1, HBHA Pond (shallow and deep), HB03-1, and AR downstream (MC-13). No total or individual PAHs values in sediment exceeded upper effects-based benchmarks (SELs or PECs) in any sample, indicating that study area-wide risk from exposure to PAHs is low. The exceedances of upper effects-based benchmarks by sediment concentrations of arsenic, cadmium, chromium, copper, lead, mercury, and zinc indicated a potential risk to benthic invertebrates in most of the sediments sampled in the study area, with the exception of areas BE-1 and BE-3.

Calculation of mean PEC quotients indicated the greatest risk to sediment invertebrates at station MC-07 with a mean quotient of 13.8. The stations with the next highest quotients were MC-05, MC-09, and MC-11. These data are consistent with the results of the toxicity testing which indicted the greatest toxicity effects at MC-05 and MC-07 and some toxicity effects at MC-11.

The overall results from the benchmark analysis indicate potential effects on benthic communities from inorganics, especially arsenic, cadmium, chromium, copper, lead, mercury, and zinc. Since the benchmarks used for each of these metals was the SEL (Persaud, *et al.*, 1993), or PEC (probable effects concentration, MacDonald et al., 2000), these represent contaminant levels that potentially eliminate most of the benthic organisms (Persaud, *et al.*, 1993). However, there is uncertainty in applying these screening criteria that do not account for bioavailability of metals under study area-specific conditions. Although the total concentration of metals in the sediment may be high, the ultimate or potential availability of the metal depends on the fraction of the contaminant that is not irreversibly sequestered or bound to the sediment matrix. Important factors that significantly affect the toxicity of metals and the ability of the organism to assimilate the available fraction include the concentration of acid volatile sulphide (AVS) and the presence of organic carbon in the sediment. As summarized in Section 5.2.3, AVS-SEM data collected in the study area indicated that AVS concentrations were high enough to reduce toxicity of divalent metals (cadmium, copper, lead, mercury, nickel, and zinc) at the majority of the stations sampled. Based on the AVS/SEM results, most of the sediments would be unlikely to be toxic due to divalent metals, except stations MC-10 and MC-11, which had values indicating uncertain toxicity. In order to investigate study area-specific toxicity of metals in sediment, further effects-based testing was conducted, focusing on areas of high metal contamination.

The second endpoint for evaluation of the effects on benthic invertebrates was toxicity testing. Overall, the weight of evidence from the toxicity testing supports the conclusion that there are adverse ecological effects on the composition of the benthic community associated with high concentrations of metals in the sediment. Based on the results of the toxicity testing, there is evidence of acute toxicity to benthic organisms at stations MC-05 and MC-07 in HBHA Pond. The data also present strong evidence of toxicity to invertebrates in at station MC-06 in the shallow area (1 ft standing water) of HBHA Pond. Evidence of toxicity was also observed for stations MC-10 and MC-11 in the HBHA wetlands, however, the number of endpoints statistically significant from controls were limited. Potential chronic effects from exposure to sediments at these stations may be present.

Community composition data indicated highly impaired benthic community with low abundance and diversity in the sediments of HBHA Pond in deep water. In the shallow area of HBHA Pond, the community indices showed evidence of impairment with limited number of taxa, low diversity, and high dominance of pollution-tolerant oligochaetes. Additional stations with community composition indicative of impairment included the pond-like stations (MC-11) in HBHA Pond

No. 3 and the HBHA wetland/stream location at MC-09 (HB02). Community statistics characterized stations MC-10 with moderate impairment and MC-13 with low impairment.

Two derived indicators of community impairment, the Toxicity Index (TI) and the Community Index (CI), were computed to summarize the weight of evidence for the benthic invertebrate endpoints in the study area (Table 43). The highest TI values were assigned to SD-05 and SD-07 where significant reduction in survival in short-term assays (10-day) was observed. Each of these had high community index values (6) indicating an impaired benthic community composition. Benthic community composition in the deep area of HBHA Pond may be influenced by low dissolved oxygen as well as sediment toxicity. However, the toxicity tests clearly indicate sediment toxicity in the lab, for which low dissolved oxygen is not responsible.

The relationship of contaminant concentrations to toxicity endpoint and to indicators of benthic community composition were evaluated by performing simple correlation analyses (Table 44). Variables were transformed prior to computing the Pearson correlation coefficients (Appendix 7B.15). Generally, growth endpoints were normally distributed and untransformed. Those variables with log-normal distribution were log transformed, including total abundance of invertebrates and concentrations of arsenic, TOC, total PAHs, and mercury. Percent data were arcsine squareroot transformed before computing correlations.

Among the metals exceeding effects-based sediment screening benchmarks (arsenic, cadmium, chromium, copper, lead, mercury, and zinc), the CI value was most highly correlated to arsenic, but also correlated with cadmium, chromium, copper, lead, mercury, and zinc, whereas the TI value did not correlate strongly with metals concentrations (Table 44). The highest correlation of individual benthic community measures to contaminant concentrations were high correlations of the percent chironomids and oligochaetes with arsenic, PAHs, chromium, copper, lead, and mercury concentrations. The highest correlations were seen with arsenic concentrations and toxicity endpoints. High concentrations of arsenic were significantly correlated to reduced growth of *C. tentans* (10-day), *H. azteca* growth (28-day), *H. azteca* survival (42-day), and *C.*

tentans emergence. The relationships with these endpoints were dramatically improved when the effects of high concentrations of iron in the sediment were taken into account. By computing a simple ratio of the concentration of arsenic to the concentration of iron in the sediment, the correlations of all of the variables with arsenic toxicity are dramatically improved, and several endpoints with weaker correlations become statistically significant (Table 44). The ratio of arsenic:iron represents the ameliorating effect of high iron concentrations on arsenic toxicity. Stations with high arsenic concentrations, but with high iron as well, have lower toxicity due to the effect of iron to bind arsenic in less toxic forms. High arsenic:iron ratios were associated with decreased community diversity, decreased community evenness, higher dominance ratios and higher proportion of the benthic community contributed by oligochaetes and chironomids. High arsenic:iron ratios were also highly correlated with survival, reproduction, and growth endpoints of both *C. tentans* and *H. azteca*. Scatterplots of percent toxicity endpoints with arsenic:iron ratios are presented in Appendix 7B.15.

The high arsenic:iron ratios and the higher observed toxicity at MC-05 and MC-07 in the deep water of HBHA Pond plays a major role in driving the observed correlations. If the results from these two stations, with the highest toxicity, are removed from the correlation (dropping the number of stations used from n=13 to n=11) the number of significant correlations decreases (Table 44 and Appendix 7B.15).

Among the other metals, copper (and to a lesser extent, lead) had multiple endpoints with which concentrations correlated to impairment of benthic communities (Table 44). These relationships are not as strong as those observed with arsenic. The relationship with copper and invertebrate endpoints may exist either because copper concentrations co-vary in depositional sediments with the arsenic concentrations, or alternatively, that copper contributes to toxicity observed in the samples. TOC-normalized AVS/SEM data collected in the study area indicates relatively low potential for divalent metal (such as copper) toxicity to invertebrates due to presence of AVS and TOC in sediment which can bind copper and other metals into relatively unavailable complexes.

The fourth measurement endpoint for the benthic invertebrate community involved the comparison of COPC burdens in invertebrate tissue collected from the study area to body burdens in reference samples. As discussed in Section 5.2.3, the invertebrate tissue data indicated that there were statistically higher concentrations of arsenic and zinc in invertebrates collected from the study area as compared to reference. Data from the ERED database showed that metals concentrations observed in tissue of invertebrates in the study area, exceeded effects-based benchmarks. There is potential for adverse effects from exposure to copper and zinc in sediments in shallow water of HBHA Pond, based on tissue residue benchmarks. There is also potential for adverse effects from chromium especially at MC-06 and MC-13, but the uncertainty associated with the evaluation is high. The tissue concentrations of arsenic in invertebrates in the study area, and particularly at MC-06 in HBHA Pond, were high.

For those metals with invertebrate tissue concentrations elevated in the study area (arsenic, cadmium, chromium, copper, lead, and zinc), the highest tissue levels of each were measured in organisms collected in HBHA Pond (Figures 22 to 26). Metals concentrations in invertebrate tissue at MC-06 indicate very high exposure to arsenic, lead, copper, chromium, and zinc, which do not correspond to the highest observed sediment concentrations of the same metals. These results indicate an increase in the bioavailability of arsenic and divalent metals in the pond, and particularly as compared to habitats downstream in the HBHA wetland sediments. These results are consistent with the variation seen in the toxicity and benthic community results that show lower toxicity and impairment to benthic invertebrates in downstream locations that had relatively high sediment concentrations of arsenic (MC-09, and MC-11), chromium (MC-13), copper (MC-09), lead (MC-13), and zinc (MC-08 and MC-09). The presence of higher total sediment concentrations of these metals in locations with lower evidence of biological effects, and lower concentrations in tissues, indicates that the bioavaiability and resulting toxicity are lower in these areas.

Consideration of the evidence among the four benthic invertebrate measurement endpoints indicates that there are impacts from inorganic contaminants on invertebrate communities within the study area. The comparison of sediment concentrations to effects-based benchmarks indicate that there are potential effects on benthic communities from inorganics, especially arsenic, cadmium, chromium, copper, lead, mercury, and zinc. The toxicity testing demonstrated adverse effects on survival and growth of *C. tentans* and *H. azteca* at some stations, and also adverse effects on community composition that were associated with high contaminant concentrations. Similarly, the invertebrate tissue data indicated elevated concentrations of contaminants, with concentrations above levels associated with known biological effects on some invertebrate species.

Evidence of significant toxicity and impairment of benthic invertebrate communities was documented at MC-05, MC-07, and MC-06 in HBHA Pond, which was associated with high concentration of arsenic, and divalent metals, including cadmium, chromium, copper, lead, mercury, and zinc. Study area-wide, the composition of benthic communities, as well as the growth and survival of *H. azteca* and *C. tentans* on sediments correlates most strongly to arsenic concentrations, and to a lesser degree to cadmium, chromium, copper, lead, mercury, and zinc. The relative contribution of arsenic versus the other divalent metals in contributing to ecological effects can not be determined from the toxicity studies alone, as the source of the toxicity is not known from toxicity tests. The benthic invertebrate tissue data also add to the weight of evidence for the effects of arsenic, as the concentration of arsenic in invertebrate tissue exceeded ecological effects levels and were greatly elevated at station MC-06. In general, elevated concentrations of metals in invertebrate tissue corresponded to locations with high toxicity, but showed less association with concentrations of the same metals in sediments. These results indicate that the toxicity and impairment to benthic invertebrates in HBHA Pond is possibly related to the forms of metals in the sediment having higher toxicity and bioavailability than the same metals present in sediments downstream.

6.2 Risk Description

The objective of risk description is to provide information important for interpreting risk results and to identify thresholds for adverse effects on assessment endpoints. The goal of the risk description is to identify the location and areal extent of existing contamination above a threshold for adverse effects.

The risk estimates based on UCL/NOAEL values represent the lower bound of the threshold for adverse ecological effects for each assessment endpoint in dietary models. In the preceding section, the COPCs were identified for each receptor for COPCs above the lower threshold for adverse effects. Heron is not considered here, since there was no indication of ecological effects associated with exposure to COPCs in the study area. Where stations both exceeded reference values and exceeded the lower threshold effects level (HQs > 1 using the NOAEL TRV), exposure to the COPC was identified as a potential risk to the population. The wildlife receptors with endpoints exceeding the lower threshold effects level (UCL/NOAEL HQs > 1), included: muskrat (antimony, arsenic, chromium, cobalt, copper, lead, selenium, and zinc), mallard (chromium and zinc), and shrew (antimony, arsenic, lead, mercury, and thallium). Evaluation of fish and benthic invertebrate risks are discussed separately, below.

Using less conservative LOAEL-based TRVs and an average-case exposure scenario, the upper bound of the threshold for adverse ecological effects can be estimated. A summary of toxicity studies and associated TRVs for muskrat, otter, shrew, and mallard are presented in Appendix 7C.13. The upper-bound TEL represents the media concentrations at which ecological impacts are predicted to occur. Where the average case scenario for exposure exceeds the upper TELs, it is assumed that the COPC represents an risk to receptor populations. The location and extent of the risk can be evaluated by identifying the stations with COPC concentrations above the TELs.

Species-specific dose-response relationships for the majority of inorganic COPCs are unavailable, and the combined effects of all COPCs on the health of individuals is unknown. However, an estimate of the TEL level for an individual COPC can be made by using the food chain models to calculate a concentration of each COPC through the main exposure routes (ingestion of sediment and plants) that corresponds to a toxicity reference value, resulting in an HQ of approximately 1.

Exposure to COPCs is the sum of the contribution from three media (food, sediment, and surface water). However for the muskrat, mallard, and shrew, 80 to 90% of the daily dose was derived from the ingestion of sediment or plants. Since the plant tissue concentrations can be derived from sediment concentrations using uptake factors, the models can be used to estimate a sediment concentration that corresponds to a daily dose resulting in an HQ of approximately 1 (using the LOAEL TRV). This provides an estimate of the COPC concentration in sediment that would be likely to correspond to a risk for each species if this concentration was distributed throughout the study area or foraging area. The assumption of uniform contamination to back-calculate effects levels has been used for the development of water-quality criteria (USEPA, 1995) and is used here in a similar fashion to evaluate the approximate adverse effects levels for each indicator species. Using the TELs, the number and location of stations with COPC concentrations exceeding these estimated thresholds can be evaluated, to assist in the characterization of the risk to ecological receptors across the study area.

For muskrat, arsenic had HQs in the average/LOAEL model above 1 at all stations except BE-1 and BE-3. Using the food-chain model for muskrat, the upper TEL for arsenic was calculated to be 152 mg/kg arsenic in sediment with a TRV of 3.22 mg/kg-d and over 1200 mg/kg in sediment with a TRV of 24 mg/kg-d. The majority of the dietary exposure results from the ingestion of plant (88% of dose). Using a TRV of 3.22 mg/kg-d, the plant tissue concentration corresponding to an HQ of 1 is 8 mg/kg and using a TRV of 24 mg/kg-d, the corresponding plant tissue concentration is 65 mg/kg.

For otter, there were no UCL/NOAEL models with HQ values above 1 for any COPC. These results indicate that risks from dietary exposure of river otter to concentrations of COPCs in the study area are below levels expected to cause ecological effects.

For mallard, the COPCs exceeding the lower threshold effects level (UCL/NOAEL HQs > 1), were chromium and zinc. HQs for lead were 2 for all of the study area scenarios as well as the reference, indicating negligible risk. Neither chromium or zinc of these exceeded the average/LOAEL thresholds using study area-wide average concentrations of invertebrate and plant doses or for models for HBHA Pond and HBHA wetland separately. Study area-wide risks of waterfowl to metals is low.

For shrew, the COPCs exceeding the lower threshold effects level (UCL/NOAEL HQs > 1), were antimony, arsenic, lead, mercury, and thallium. Other than arsenic, the majority of the exceedances of the lower effects threshold was observed for exposures to soils and soil invertebrates at station A6. Among these five inorganics, only arsenic concentrations in soil corresponded to HQs greater than 1 in the average/LOAEL models at stations A6, BE-2, HB02-2, and HB03-3. Using the food-chain model for shrew, an upper TEL was calculated to be 160 mg/kg arsenic in sediment, based on the LOAEL TRV (mouse, reproductive, 3 generations) of 1.5 mg/kgBW-d. Sediment concentrations at stations A6, BE-2, HB02-2, and HB03 in shrew habitat exceeded this upper TEL value (range 219 to 671 mg/kg in sediment/soil). LOAEL TRVs with growth endpoints ranged from 4 to 66 mg/kg-day for rats, although the few reported LOAEL levels for mice were lower (0.08 to 1.5 mg/kg). This indicates possible impacts to shrew or other small mammals due to exposure to arsenic in diet at these stations, although the uncertainty associated with this risk is high.

Fish tissue analysis results show higher exposure to inorganics resulting in higher body burdens of COPCs in the study area than fish collected at reference stations. The concentrations in fish tissue are high, particularly for arsenic, indicating significantly higher exposure to arsenic in the study area among small foraging fish and bottom feeding fish. Highest concentrations of arsenic in fish tissue were observed among bottom feeders (brown bullhead and white sucker). Evaluation of fish tissue residues suggests that with the exception of arsenic, there is no direct and widespread impacts of inorganic COPCs on individual fish, or to fish populations. Based on a comparison of arsenic tissue concentrations to benchmark values, the difference between benchmark and actual detected tissue concentrations is approximately a factor of 2 for bottom-feeding fish. The risk from exposure to arsenic appears to be negligible for predatory fish (largemouth bass) and low for small foraging fish (pumpkinseed). Based on tissue residue levels, those fish with feeding strategies providing highest exposure to sediments, show tissue residue values that are slightly above tissue residue effects level for ecological effects. There was no evidence of gross abnormalities observed on fish collected in any pond. The injuries observed were not unusual, and did not provide evidence of ecological effects potentially attributable to contamination. Ecological effects of contaminant toxicity to reduction in prey base or direct reduction in recruitment may be present, but can not be distinguished from other biotic or abiotic factors impairing fish populations in the study area. Exposure of fish, particularly bottom feeding fish, to sediment arsenic is high and the tissue data provide evidence of potential ecological effects, with the magnitude of the benchmarks exceedances fairly low (average HQs between 1.0 and 1.9, maximum HQs between 1.5 and 3.1). Fish population data are inconclusive about the role of toxicity in impairing fish populations in the study area ponds, since population impairment due to poor habitat may have masked effects of toxicity causing population effects.

Comparisons to available effects-based sediment benchmarks identified arsenic, cadmium, chromium, copper, lead, mercury, and zinc as exceeding threshold concentrations for the benthic invertebrates. In nature, the adverse effects at any given contaminant level likely vary from station to station depending on the composition, form, and magnitude of inorganic contamination, the physical characteristics of the sediments, suitability of the immediate habitats (*e.g.*, sedimentation, high flows during storms events, and other habitat characteristics). All or some of these factors may act to reduce (*e.g.*, ameliorate) or increase adverse effects observed on organisms exposed to a given contaminant concentration in sediment at a given station. Even with this inherent variation among stations, the distribution of the individual contaminants across the study area in comparison to these benchmarks aids in evaluating the extent and location of sediments posing the highest risk to biological communities.

Comparison to benchmarks does not necessarily indicate ecological effects. With inorganic COPCs in sediments, many factors may alter the study area-specific toxicity of each compound. These factors include, organic carbon content, particle size, AVS/SEM ratios, and concentration of other constituents in the sediments, such as iron or nutrients (Chapman *et al.*, 1998).

Consequently, the biological effects data (acute and chronic toxicity test, and benthic community structure) were used in combination with habitat information to evaluate biological effects for the benthic invertebrate community endpoint.

The weight of evidence analysis for the invertebrate endpoint resulted in the conclusion that there were adverse effects on the benthic community structure associated with the distribution of inorganic contaminants in the study area. The concentration of arsenic was most closely correlated to the benthic invertebrate endpoints. To a lesser degree, concentrations of copper, cadmium, chromium, lead, mercury, and zinc were elevated among the stations either showing sediment toxicity or impaired community composition, but the relationship of adverse effects to higher concentrations of these metals was less consistent. The synergistic or additive effects of the combinations of metals at these stations is not known.

The highest toxicity and impairment of invertebrate communities was observed in HBHA Pond, at stations MC-06, MC-05, and MC-07. Low dissolved oxygen concentrations may contribute to poor invertebrate community health in the deep stations in HBHA Pond (MC-05 and MC-07). However, toxicity observed at MC-05 and MC-07 indicated acute toxicity to invertebrates, in the laboratory where the effects of low in-situ dissolved oxygen is not a factor. The two stations in the study area with the highest sediment arsenic concentrations (MC-05 and MC-07) corresponded to the two stations with highest invertebrate toxicity. The concentration of arsenic at MC-05 and MC-07 was 2,390 mg/kg and 1,103 mg/kg, respectively, which are 33 and 72 times the SEL for arsenic, respectively.

Downstream of the HBHA Pond in the HBHA wetland, there was evidence of toxicity at MC-10 and MC-11 for some toxicity endpoints. Growth of *H. azteca* in 42-day tests was low at MC-10 and MC-11 (Table 41). Twenty-day survival at MC-10 and MC-11 for *C. tentans* was very low (19% and 17% respectively), however confidence in these results are reduced due to poor performance of laboratory controls. Both MC-10 and MC-11 also showed benthic community impairment in having a low taxa richness and diversity compared to reference communities.

Within the HBHA wetlands, the highest sediment arsenic for stations used in the invertebrate toxicity testing was measured at 1,200 mg/kg at MC-11, which is similar to that measured in HBHA pond, but with a very high corresponding iron concentration.

Fate and transport studies have documented the discharge of groundwater with dissolved arsenic to HBHA Pond. In addition, surface water data confirm that the dissolved arsenic fraction is higher at the outlet of HBHA pond than downstream in the wetlands. Samples collected downstream at the outlet of the HBHA wetlands show higher proportions of arsenic in fractions associated with particulate iron. The fate and transport studies indicate that arsenic in a dissolved form is discharged to HBHA pond and as the water travels downgradient, arsenic/iron complexes form and are deposited in study area sediments.

Only two samples of invertebrate tissue were collected from HBHA pond. The pattern of distribution of these relative to areas of greater or lesser surface water discharge cannot be established with these few samples. However, the chemistry of the pond suggests that arsenic is more available for uptake by biota exposed to the sediments, and the notably high concentration of invertebrate tissue arsenic at MC-06 may also be related to the arsenic availability in the sediment in the pond. Metals concentrations in invertebrate tissue at MC-06 indicate very high exposure to arsenic, lead, copper, chromium, and zinc. This station also exhibited acute toxicity and impaired invertebrate community structure. These results indicate that the toxicity and impairment to benthic invertebrates in HBHA Pond is likely related to the forms of metals in the sediment having higher toxicity and bioavailability than the same metals present in sediments downstream.

The weight of evidence for the benthic invertebrates indicates that benthic invertebrate community impairment is associated with high levels of arsenic and divalent metals in HBHA Pond. AVS/SEM data indicated that the likelihood for toxicity due to the six SEM metals (cadmium, copper, lead, mercury, nickel, and zinc), is low. This provides further support that other contaminants such as arsenic are producing the adverse effects on invertebrates, as opposed to the

SEM metals. Since the relationship of total metals concentrations in sediment to invertebrate impairment is influenced by many factors such as TOC, AVS, other nutrients and metal concentrations, the dose/response relationship of the sediment concentration of a single COPC such as arsenic can be masked by other variables. In the case of arsenic in sediments, when the amount of iron in the sediment increases and it is available to form less toxic, insoluble iron-arsenic complexes, the toxicity of the arsenic measured in bulk sediments may decrease. Representing this ameliorating effect of iron on arsenic toxicity by calculating a simple ratio of arsenic:iron concentration in sediment dramatically improved the relationship of arsenic to toxicity and community endpoints.

Surface water data were utilized in the BERA mainly in calculating dose to wildlife receptors. In Section 4.1.1 (Table 23), twelve COPCs were identified as possibly posing a risk to aquatic receptors including aquatic invertebrates or fish. Table 45 presents a comparison of surface water data to benchmarks. No screening criteria were available for vinyl chloride, cyclohexanone, and pyrene. Pyrene and vinyl chloride were detected only in the deep water of HBHA Pond. The risk from exposure to these organics is likely to be low, but is uncertain due to lack of reference values. Benzene exceeded the Tier II benchmark only in the deep water of HBHA pond. There is possible risk from exposure to benzene in the pond, but the magnitude of the risk is low.

Among the inorganics, barium and manganese posed a low risk to aquatic receptors throughout the study area. Dissolved silver concentrations were also above benchmarks in the HBHA wetland. Zinc was the only COPC which exceeded a NAWQC. The exceedance of the hardness-adjusted NAWQC by zinc represents a low risk to receptors in the shallow water of HBHA Pond.

6.3 Uncertainty

There is uncertainty associated with estimates of risk in any BERA, as the risk estimates are based on a number of assumptions regarding exposure and toxicity. There is uncertainty associated with the site conceptual model, with natural variation and parameter error, and with model error (USEPA, 1997). A thorough understanding of the uncertainties associated with risk estimates is critical to understanding predicted risks and placing them in proper perspective. In addition to those already discussed in Sections 5.1, 5.2, and 6.2, important sources of uncertainty associated with the BERA are addressed below.

Uncertainty associated with the conceptual model (Figure 7) includes assumptions about the sources of contaminants and the fate and transport of the contaminants in the study area. The prediction of risk in this assessment does not distinguish the sources of contaminants that are identified as COPCs, but is addressed in detail in the MSGRP RI. In an urban watershed, the source of contaminants such as inorganics in sediments can not always be distinguished from site-related compounds or other local run-off or up-stream sources. However, the number of sediment samples collected within the study area provides high confidence in the estimation of exposure from sediment, and utilization of reference areas provides a mechanism for evaluating the potential source of common contaminants.

There is uncertainty associated with assumptions used in screening and eliminating COPCs in the refinement of COPCs step. Although barium, beryllium, selenium, and thallium were detected in sediment samples above concentrations observed at reference locations, no effects-based sediment benchmarks were available to address risk to benthic organisms. This is a source of uncertainty. However, the concentrations of each were less than a factor of ten below reference locations and none of these metals are known to bioaccumulate. The lack of specific effects-based benchmarks for these compounds is not a major source of uncertainty. In addition, contribution to toxicity from these metals would have been addressed in the invertebrate toxicity testing.

There is some uncertainty in the selection of the receptors as representative of communities utilizing the habitat in the study area. Habitat quality for some of the species varies across the study area and will influence actual presence or exposure of species or communities in different portions of the study area. For example, the short-tailed shrew was selected as a small mammal that is likely to inhabit the drier areas of the study area habitat. Using wetland soil samples to estimate dietary exposures for shrews likely overestimated the exposure to sediment COPCs to small mammals. Therefore the calculated risk to shrew populations was associated with higher uncertainty.

6.3.1 Exposure Estimation. Exposure estimates for indicator species are a source of uncertainty in the BERA. Values for exposure parameters (*e.g.*, body weight, food intake rate, and sediment ingestion rate) were based on literature values, not study area-specific data. For instance, it was assumed, based on other studies, that approximately 30% of the shrew diet is comprised of earthworms for the average and UCL models. It was also assumed that contaminant body burdens in earthworms are far greater than would be found in any of the other prey items shrews typically consume. The accuracy of each of these assumptions may be debated. However, the approach maintained in the BERA was to utilize conservative exposure parameters while maintaining a realistic evaluation of the potential for risk.

The collection of study area-specific tissue data (plants, invertebrates, and fish) increased the confidence in the exposure estimates of dietary doses of receptors. There is uncertainty in using data collected in one portion of the study area to represent concentrations to which an indicator species may be exposed to in another portion of the study area. The assumption inherent in the modeling is that the conditions at specific sampling locations, representing the exposure point concentrations, adequately represents the exposure throughout the habitat.

Plant tissue concentration estimates were used for calculating exposure for muskrat and mallards. The plant material was assumed to compose 33% of the mallard diet and 90% of the muskrat diet. The bioaccumulative potential of plants varies among species, and even within different parts of the plant. Therefore, there are additional uncertainties in assuming the tissue concentrations from whole plants are representative of the exposure of a consumer, particularly for a species that might selectively graze on a specific species or part of a plant. Analysis of plant data presented in Section 5.1 confirms that the concentration of arsenic was significantly higher in roots of plants, than in the foliage (stem/leaf). The highest arsenic concentration in plant tissue (240 mg/kg) was observed for cattail roots at station MC-08 in the HBHA wetland (sediment arsenic concentration was 594 mg/kg) which is 10 times greater than the study area-wide average. Two samples of cattail were collected at MC-08 and the average arsenic concentration was 142 mg/kg. The study area-wide average concentration of arsenic in cattail roots was 58.5 mg/kg. If the diet of muskrat had been assumed to be 100% cattail roots with a concentration of 58.5 mg/kg, the muskrat HQs would have increased to 10 (using a TRV of 3.2 mg/kg-d) in the HBHA wetlands.

The tissue concentrations of arsenic at MC-06 was 39 mg/kg in roots and the stem tissue was 4.5 mg/kg (in non-cattail species) with a co-collected sediment sample having an arsenic concentration of 273 mg/kg. If the diet of muskrat had been assumed to be 100% roots, the muskrat HQ for HB01 would have increased from 4 to 6.

Otter models were based on a dietary composition of 100% fish. The whole body fish tissue data used for otter in the models was from pumpkinseed sunfish. The size of the fish represented by the pumkinseeds (1 to 44 grams and 48 to 127 mm) may have been small relative to actual fish available to otter in the study area. However, this did not contribute a large over- or under-estimation of risk to otter. The tissue concentrations of arsenic for reconstructed whole body concentration in larger fish species (largemouth bass, brown bullhead, and white sucker) from the same areas were similar to those observed in the small fish (pumpkinseed) tissue (Table 36).

Selection of stations to be utilized in the exposure calculations for each indicator species also carries a degree of uncertainty. The intensity and frequency of use of different stations, although not heavily factored into the assessment of risks, would also influence the potential for COPCs to impact individuals. However, since the emphasis in the risk analysis was placed on the average case scenarios for each indicator, the influence of any one station being included or excluded from a given model would have had little effect on the calculated HQs and the associated evaluation of risk. The average case exposure calculations are fairly robust with respect to selection of stations for each receptor species.

It is commonly assumed that the data used to characterize exposure (sediment, surface water, or tissue concentrations) are normally distributed. Ecological data, however, often do not fit a normal distribution, since they tend to have many low values and fewer high values. Since the mean is actually used in exposure estimated to represent a time-average, the arithmetic mean in some cases may over estimate exposure. Statistical analysis of the data used in the BERA revealed that some of the COPC concentrations are not normally distributed, however, arithmetic means were still used to evaluate exposure. This was a conservative assumption and a source of uncertainty, since the arithmetic means are usually higher than geometric means, which are appropriate for log-normally distributed data.

In general, there is high confidence that data collected for the BERA represent the types and distributions of contaminants within the study area. However, exposure estimates are always uncertain in that they are driven by available data and by the methods used to collect those data. For example, exposure uncertainty is associated with the removal, prior to sampling, of coarse organic material (leaf litter or detritus) overlaying sediment or soil. Analytical data reflect the concentration of COPCs in sediment and finer organic matter underlying the coarse organic matter at the surface. Therefore, analytical data may under- or overestimate exposures for invertebrates that inhabit or contact only coarse particulate organic matter at the substrate surface. Similarly, the majority of the sediment was removed from the plant tissue samples prior to processing. Residual sediment would be likely to overestimate COPC concentration in plant tissue. The potential exposure of herbivores through sediment ingestion was estimated by including a proportion of the diet as ingested sediment, based on literature values.

Conservative assumptions were also made about exposure duration and site use factors. Assumptions were made that exposure remains constant over the seasonal exposure duration of an individual animal. In fact, the home range of many species varies from one life stage to another. Migration of individuals in and out of the study area would also affect exposure duration. In particular, maximum exposure scenarios are very conservative, as they assume the highest station concentrations for a contaminant was spread evenly over the entire range of an organism's residence or foraging range. With the exception of some benthic invertebrates, this assumption is very conservative, because none of the vertebrate species would likely be confined to an area representative of a single station for a period of time approximating the exposure duration. Consequently, maximum exposure estimates are worst-case scenarios that tend to overestimate exposure, and were used only in screening models.

6.3.2 Toxicological Data. Toxicity values for indicator species and communities were based on literature values. As is the case for literature-based exposure parameter values, this is a major source of uncertainty in the BERA. The sensitivity of receptors in the study area may be different than the sensitivity of species used in tests reported in the literature.

Assumptions about the equality of contaminant form between laboratory tests and field conditions must also be made in the absence of speciation analyses. This is a source of uncertainty, since toxicity may vary with the form of the toxicant in the environment. Thus, the actual toxicities of COPCs evaluated in this BERA could be higher or lower than indicated by the TRVs used in the development of HQs.

Another source of uncertainty is the extrapolation of LOAELs to NOAELs using an uncertainty factor of ten. This approach is likely conservative. Dourson and Stara (1983 *cited in* USEPA, 1997) determined that 96% of the chemicals included in a data review had LOAEL/NOAEL ratios of five or less. The use of an uncertainty factor of 10, although potentially conservative, also serves to counter some of the uncertainty associated with interspecies extrapolations, for which a specific uncertainty factor was not used.

TRVs were available for only a few organic COPCs for avian species. There were no TRVs for screening VOCs. Avian TRVs were not available for all of the SVOCs, hoever, a reference value for low molecular weight PAHs was used for screening avian species. For COPCs having no available avian reference values, risk was estimated based on mammalian TRVs. The use of mammalian TRVs is uncertain, as the similar effects to birds may not be seen at doses affecting mammals. However, these estimates of doses for non-similar species were used rather than dropping the COPC with no evaluation due to a lack of a TRV. For COPCs carried forward to BERA evaluation, only antimony had no avian reference values. It is possible that the lack of avian TRVs in screening models may have resulted underestimating risk for some COPCs. This is not seen as a major data gap in the BERA, as all of the COPCs identified as having potential risk for other receptors were mainly inorganics, for which avian TRVs were available.

Based on the review of available studies for which possible LOAEL TRV values were given (Appendix 7C.13), a large source of uncertainty is the selection of a TRV for estimation of HQs. The results of different studies often varied several orders of magnitude, based on using various forms of the COPC, different species, and different endpoints. For muskrat, the risk evaluation was based on average case scenarios at 10 stations study area-wide and one reference scenario. All of the 10 stations, and the reference scenario, had HQs above 1 for arsenic for the UCL scenario. Using the LOAEL TRV for arsenic (3.22 mg/kg-d) eight stations exceeeded LOAEL TRV, but if a TRV of 24 mg/kg-d was used, all HQs drop below 1. The analysis dietary exposure is more sensitive to the selection of TRVs than the calculations of the exposure concentrations in each medium.

EPA suggests the use of TRVs derived in the Eco-SSL guidance for evaluating the exposure of receptors to soils. The TRVs utilized in this BERA were selected to be consistent with those used in the BERA for the Aberjona River Study Area (Wells G&H OU3). If, instead, the values from Eco-SSLs had been selected for use for evaluating the risk to avian and mammalian receptors in the current document, results would not have changed for any of the avian or mammalian risk estimates. In general, TRVs used in the BERA were similar to the TRVs recommended in the Eco-SSLs. Exceptions included a lower screening TRV used for barium, a lower TRV for cobalt and a higher TRV used for lead, as compared to the Eco-SSL values. A much lower mammalian NOAEL TRV was used for barium (5.1 mg.kg BW-day) in the BERA (one order of magnitude lower than the Eco-SSL of 51.8 mg/kg BW-day). However, results would not have changed, as the BERA did not identify risk from exposure to barium. Similarly, a lower TRV was used for cobalt (5 mg/kgBW-d versus an Eco-SSL value of 7.3 mg/kgBW-d) in the BERA, but results would not have changed, as the BERA did not identify risk from exposure to cobalt. For lead, the BERA used a TRV that was almost two times higher than the Eco-SSL (8 mg/KgBW-d versus an Eco-SSL value of 4.7 mg/kgBW-d), which could have potentially underestimated risk. However, lead was identified as a COPC using the higher value, and the results of the screening to select TRVs would not have changed using the lower Eco-SSL value.

One of the largest sources of uncertainty in all of these TRV values is the form of the chemical used to determine the laboratory exposure. The HQ approach uses the assumption that the absorption of the chemical from the diet will be the same as the absorption of the chemical in the form used in the laboratory. Often this assumption is very conservative, because absorption of metals ingested with sediment or plant material is greatly reduced from forms given in laboratory studies.

For the purposes of the maximum exposure models, an oral bioavailability factor of 1 was assumed for each chemical evaluated in the ingestion pathway. The use of a factor of 1 assumes that 100% of the chemical ingested in the diet is bioavailable and is utilized to be conservative. This bioavailability factor likely provides an overestimate of the dose to the receptors for many COPCs.

As seen from the swine study conducted in conjunction with the risk assessment for Wells G&H OU-3 Aberjona River Study, only approximately 50% of the arsenic in sediment fed to young swine was bioavailable. In the study, data were collected to calculate the relative bioavailability (RBA) of arsenic from these sediments. RBA is an estimate of the oral bioavailability to humans of arsenic from study area sediments compared to that of a reference arsenic compound administered in drinking water. "Best Estimate" RBA values determined in this study ranged from 37% to 51%, indicating that arsenic from sediments is absorbed less extensively than arsenic from drinking water.

If this RBA value of 51% was used to evaluate the oral toxicity of arsenic in sediments for mammals (muskrat and shrew), it would only have been applied to adjust the incidental sediment ingestion dose for each of the mammal models for arsenic. As such, the difference in the dose estimate would have been minimal, as sediment ingestion composes only 3% of the muskrat diet. Th majority of the dose of arsenic in muskrat is from the ingestion of plant tissue. The dose from plant material was not adjusted by this RBA, since no specific RBA for plants was derived, and this is a source of uncertainty for the muskrat model. If the RBA of 51% had been applied to plant ingestion, the upper TEL for muskrat would have been increased from 152 mg/kg arsenic in sediment to 304 mg/kg at a TRV of 3.2 mg/kg-d and from 1200 mg/kg to 2400 mg/kg at a TRV of 24 mg/kg-d. No RBA factors were used for any other COPC or receptor species, and this contributes a large proportion of the uncertainty in the dietary models.

In comparison to dietary exposure models, results made with comparison of study area-specific tissue data to tissue residue effects levels have a higher degree of certainty. Although there is variation in TRVs, in contrast to wildlife receptors, the reference values for invertebrate and fish tissue comparisons were generally based on similar species. The direct measurements of tissue have much higher certainty than dietary estimates of dose used for wildlife receptors. The main limitation in the analyses is in the availability and selection of reference values.

In addition there is uncertainty associated with the selection of tissue fractions used for the comparisons. In fish, arsenic accumulates mainly in the liver (ATSDR, 2000a) and comparisons of the carcass data for large fish species made to whole body literature values is likely to over-estimate risk. However, based on these comparisons, only tissue arsenic concentrations were detected at levels above these benchmarks. Since arsenic tissue concentrations did exceed benchmarks, calculations were made to estimate reconstructed whole body (RWB) tissue concentrations for arsenic. As shown in Table 36, the reconstructed whole body tissue values are consistently lower in arsenic concentration than the corresponding carcass values. The RWB tissue values were used in the evaluation of arsenic risk for large fish species as a more accurate estimate of risk.

Another source of uncertainty in the fish tissue comparisons was that of using TRVs for liver from different species. Liver tissue benchmarks are species-specific and, as the only values were available in the literature for pumpkinseed, fish species comparisons were made to this value. Tissue concentrations from liver of largemouth bass, brown bullhead, and white sucker were all below the liver benchmark for pumpkinseed, with the exception of the maximum white sucker liver concentration in HBHA wetlands. This comparison to liver concentrations of non-similar species has associated uncertainty.

In the evaluation of the tissue residue data for fish, aluminum and zinc were detected in some fractions of fish at concentrations above those observed at reference locations. However, no tissue residue effects benchmark was available for aluminum or zinc. The lack of these benchmarks leaves an uncertainty about ecological effects to fish associated with these metals.

6.4 Summary of Ecological Risks

Based on data collected between 1999 and 2004, the effects-based screening resulted in the selection of twelve COPCs in surface water including two VOCs in the deep water of HBHA Pond, four SVOCs, and six inorganics. Fifty-two COPCs were identified in sediment and soil, including eight VOCs, 20 SVOCs, five pesticides, and nineteen inorganics. Nine indicator species or indicator communities were selected to evaluate risks associated with exposure to the COPCs in the surface water, sediment, and biota of the Northern Study Area. Endpoints in the BERA were selected to represent ecological attributes that are to be protected (assessment endpoints) and a measurable characteristic of those attributes (measurement endpoints) that can be used to gauge the degree of impact that has or may occur. Endpoints and the corresponding risk summary are shown in Table 46.

Each endpoint has associated with it a magnitude of risk and a degree of uncertainty. The magnitude of risk incorporates both the degree to which the endpoint was exceeded and also the proportion of the habitat affected. Since the endpoints were based on effects on populations, a reasonable probability of risk was determined to be present only when a risk was present through the majority of the organism's habitat. If the NOAEL TRV (lower effects threshold) was exceeded across most of the study area, the contaminant was concluded to pose a low risk to populations. The highest risk was associated with contaminants that exceeded upper threshold effects levels based on LOAEL TRVs, and was present throughout a majority of the indicator species' habitat within the study area. If high HQs were present in only a small proportion of the habitat for the selected indicator species, the magnitude of the overall risk to the population from exposure to the COPC was considered low.

The uncertainty associated with the estimation of risk, summarized in section 6.3, was qualitatively assessed, and based on many factors. A major source of uncertainty for mammalian and avian indicators was the relevance of the available TRVs and the relative bioavailability of COPCs ingested as plant or animal tissue. High uncertainty was also associated with COPCs that had corresponding high concentrations at reference locations. In cases where the magnitude of risk was low, and was associated with high degree of uncertainty, the overall risk for that endpoint was considered negligible.

Based on the analysis of the nine selected indicators/endpoints in the BERA, there were no indications of significant ecological risk associated with VOC, SVOC, and pesticide/PCB contamination within the study area. The weight of evidence also indicates there are impacts from inorganic contaminants on invertebrate communities within the study area. However, the strength of evidence suggests that there is high exposure to arsenic for semi-aquatic mammals, terrestrial mammals, bottom feeding fish, and small forage fish in the study area. The magnitude of the risk and uncertainty associated with the ecological effects for each receptor are discussed below.

For muskrat, arsenic had HQs in the average/LOAEL model above 1 at all stations except BE-1 and BE-3. Using the food-chain model for muskrat, the upper TEL for arsenic was calculated to be 152 mg/kg arsenic in sediment with a TRV of 3.22 mg/kg-d and over 1200 mg/kg in sediment with a TRV of 24 mg/kg-d. The majority of the dietary exposure results from the ingestion of plant tissue (88% of dose). The LOAEL TRVs for muskrat ranged from 0.55 to 24 mg/kg-d.

Using a TRV of 0.55 mg/kg-d, the plant tissue concentration corresponding to an HQ of 1 is 1 mg/kg and using a TRV of 24 mg/kg-d, the corresponding plant tissue concentration is 65 mg/kg. One sample of cattail roots at MC-08 exceeded this concentration. Applying the TRV of 0.55 mg/kg-d may be over-protective, as it leads to a reference HQ of 2 and a TEL plant tissue concentration of 1.0 mg/kg which is lower than that observed at reference locations (1.7 mg/kg). The areas of highest risk correspond to locations in HBHA Pond and upper portions of the HBHA wetland with the highest plant tissue concentrations of arsenic. The dietary analysis indicates high exposure of muskrat to arsenic in HBHA Pond and upper portions of the HBHA wetland with uncertain risk due to low confidence in selection of a reference toxicity value representing harm.

Risk to sustainability of river otter populations or other piscivorus mammal populations was determined to be negligible since none of the HQs for COPCs were greater than the target HQ of 1 using the NOAEL TRV. The confidence in this evaluation is high due to the use of study area-specific fish tissue data, and use of a conservative NOAEL TRV. Based on the dietary exposure models, there is no evidence of significant ecological effects associated with contaminants on piscivorus mammals feeding in the study area.

Risk to sustainability of green heron populations or other piscivorus bird populations was determined to be negligible since none of the HQs for COPCs were greater than the target HQ of 1 using the NOAEL TRV. The confidence in this evaluation is high due to the use of study area-specific tissue data, and use of a conservative NOAEL TRV. Based on the dietary exposure models, there is no evidence of significant ecological effects associated with contaminants on piscivorus birds feeding in the study area. Similarly, for waterfowl, represented by mallard, there was no evidence of significant risk from exposure to COPCs in the study area.

For shrew, other than arsenic, the majority of the exceedances of the lower effects threshold was observed for exposures to soils and soil invertebrates at station A6. Among the five inorganics with potential risk, only arsenic concentrations in soil corresponded to HQs greater than one in the average/LOAEL models at stations A6, BE-2, HB02-2, and HB03-3. Soil concentrations at stations A6, BE-2, HB02-2, and HB03 in shrew habitat exceeded this upper TEL value (160 mg/kg). This indicates possible impacts to shrew or other small mammals due to exposure to arsenic in diet at stations (A6, BE-2, HB02-2, and HB03-3) bordering the HBHA pond and wetlands, although the uncertainty associated with this risk is high.

Surface water data were used in the dietary models to estimate dose, but were also used separately to evaluate potential effects on aquatic organisms (invertebrates and fish) in the water column. Among the COPCs in surface water, benzene was detected in deep samples from HBHA pond. The average concentration in deep samples indicated potential risk based on Tier II benchmarks. SVOCs were infrequently detected and represent a low risk to receptors in surface water. Average inorganic COPC concentrations, including barium, manganese, silver, and zinc exceeded surface water benchmarks (Table 45). Based on these comparisons, there is low risk to aquatic receptors. However, evaluation of fish tissue residues suggests that with the exception of arsenic, there is no direct and widespread impacts of inorganic COPCs on fish populations.

The USEPA chronic NAWQC for total arsenic is 340 μg/L and for dissolved arsenic is 150 μg/L (USEPA, 2002). The maximum dissolved arsenic concentration measured in surface water in the study area (120 μg/L) was below the NAWQC. The surface water concentrations are well below those expected to cause effects on fish. However, as described below, fish exposure to arsenic is high, and tissue level concentrations are slightly above values associated with adverse effects for species with feeding strategies that result in high exposure to sediment and ingestion of sediment-dwelling organisms. These data indicate that the main exposure of aquatic organisms is not through surface water exposure, but through sediment exposure or dietary exposure.

Risks to fish were evaluated from both tissue residue data and from population studies. There is a relatively high degree of confidence in the tissue residue comparisons. The magnitude of risk based on tissue concentrations of arsenic are generally low. There is a higher degree of uncertainty in the fish populations study data associated with the effects of poor habitat quality on fish population health. Fish tissue analysis indicates exposure to inorganics results in high body burdens of COPCs, particularly for arsenic, indicating high exposure to arsenic in the study area among small foraging fish and bottom feeding fish. The highest concentrations of arsenic in fish tissue were observed among bottom feeders (brown bullhead and white sucker). Evaluation of fish tissue residues suggests that with the exception of arsenic, there is no risk from exposure of inorganic COPCs to fish populations. The risk from exposure to arsenic appears to be negligible for predatory fish (largemouth bass) and low for small foraging fish (pumpkinseed). Fish tissue data indicate potential effects on brown bullhead, white sucker, and a low risk to pumpkinseed in the study area due to elevated concentrations of arsenic in tissue at levels potentially associated with harm. Ecological effects of contaminant toxicity resulting in a reduction in prey base for fish may be present, and a reduction in reproductive success or survival of young due to exposure to contaminants may be occurring, but these effects can not be distinguished from other biotic or abiotic factors impairing fish populations in the study area. Risks to fish are possibly underestimated based on the inability to discern any population impacts from the exposure to toxic substances from impacts associated with the poor habitat. The tissue data provide evidence of potential ecological effects, although population data are inconclusive about the role of toxicity in impairing fish populations in the study area ponds.

The weight of evidence for benthic invertebrate measurement endpoints indicates that there are impacts from inorganic contaminants on invertebrate communities within the study area. The comparison of sediment concentrations to effects-based benchmarks indicate that there are potential effects on benthic communities from inorganics, especially arsenic, cadmium, chromium, copper, lead, mercury, and zinc. The toxicity testing supports the conclusion that there are adverse ecological effects on the composition of the benthic community associated with high concentrations of metals in the sediment. The toxicity and community impairment is highly correlated to sediment arsenic concentrations, particularly when the effect of high iron concentrations is taken into account. Lack of toxicity of divalent metals predicted by AVS/SEM data is consistent with the lower correlation of benthic invertebrate impairment with divalent (SEM) metals than with sediment arsenic concentrations.

There is evidence of acute toxicity to benthic organisms at deep stations in HBHA Pond. The data also present strong evidence of toxicity to invertebrates at station MC-06 in the shallow area of HBHA Pond. Community composition data indicate highly impaired benthic community with low abundance and diversity in the sediments of HBHA Pond in deep water. In the shallow area of HBHA Pond, the community indices show evidence of impairment with limited number of taxa, low diversity, and high dominance of pollution-tolerant oligochaetes.

Downstream in the HBHA wetlands, evidence of invertebrate toxicity was also observed for stations SD-MC-10 and SD-MC-11, however the number of endpoints statistically significant from controls were limited. Toxicity data and benthic community data both indicate potential effects from exposure to sediments at these stations. Lower observed concentrations of metals in invertebrate tissue, indicate lower bioavailability, and are consistent with lower toxicity from sediments in these downstream wetlands.

The benthic invertebrate tissue data also add to the weight of evidence for the effects of arsenic, as the concentration of arsenic in invertebrate tissue exceed ecological effects levels and are greatly elevated at station MC-06. In general, elevated concentrations of metals in invertebrate tissue correspond to locations with high toxicity, but show less association with concentrations of the same metals in downstream sediments. These results indicate that the toxicity and impairment to benthic invertebrates in HBHA Pond are likely related to the forms of metals in the sediment having higher toxicity and/or bioavailability than the same metals present in sediments downstream.

Arsenic concentrations in sediment and biota contribute the majority of the potential ecological risk to ecological receptors in the study area. Arsenic bioaccumulates in plants and animals, but does not appear to biomagnify between trophic levels (Eisler, 1994; Farag *et al.*, 1998 *cited in* ATSDR, 2000a). In aquatic systems, the most significant transfer occurs between water and primary producers and the base of the food chain (herbivores). An extensive study of the factors affecting bioaccumulation of arsenic in Maryland streams found no evidence of biomagnification since arsenic concentrations decreased with increasing trophic level (Mason *et al.*, 2000 *cited in* ATSDR, 2000). Another study showed no difference in arsenic levels among different species of exposed fish, which included herbivorous, insectivorous, and carnivorous species (Kidwell *et al.*,1995 *cited in* ATSDR, 2000a). The study area results are consistent with these findings. Organisms with direct exposure to sediment arsenic (plants and benthic invertebrates) have the highest relative body burdens of arsenic. Herbivores (muskrat) and bottom-feeding fish have higher risk than predators (green heron, otter, and largemouth bass) from exposure to arsenic through the food chain.

7.0 REFERENCES

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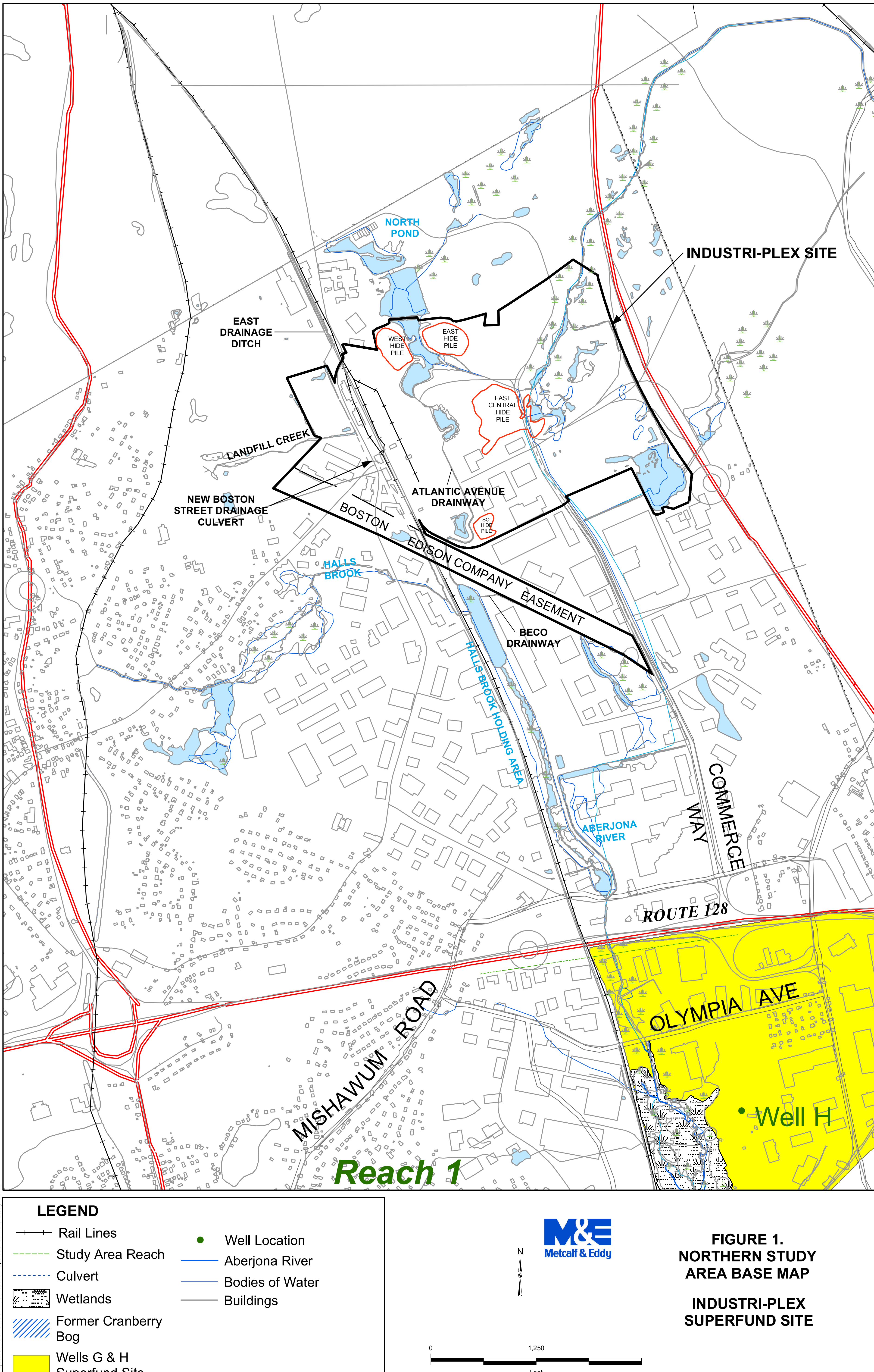

**FIGURE 1.
NORTHERN STUDY
AREA BASE MAP**

**INDUSTRI-PLEX
SUPERFUND SITE**