BASELINE HUMAN HEALTH AND ECOLOGICAL RISK ASSESSMENT REPORT

Wells G&H Superfund Site
Aberjona River Study
Operable Unit 3
Woburn, Massachusetts

September 2004

NOTE: This Report includes revisions to EPA's previous June 2003 Baseline Human Health Risk Assessment Report.

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BASELINE HUMAN HEALTH AND ECOLOGICAL RISK ASSESSMENT REPORT

Wells G&H Superfund Site
Aberjona River Study
Operable Unit 3
Woburn, Massachusetts

Volume I
Text and Figures

September 2004

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EXECUTIVE SUMMARY

A Baseline Risk Assessment of the Aberjona River Study area was conducted by Metcalf & Eddy, Inc. for USEPA Region I at the Wells G&H Superfund Site in Woburn, Massachusetts. The Aberjona River Study, or Operable Unit 3, is one of three operable units delineated for the Wells G&H Superfund Site. For the purposes of the risk assessments, the objective of the Aberjona River Study is to determine whether contaminated media (surface water, sediment, floodplain surface soil, and biota) within the study area pose risks to human health and the environment.

This document presents the revised baseline human health and ecological risk assessments for the Aberjona River Study area from Route 128 to the Mystic Lakes. The U.S. Environmental Protection Agency (USEPA) released the draft Aberjona River Study Baseline Human Health and Ecological Risk Assessment Report (M&E, 2003) for public comment in May 2003. USEPA initially released the baseline human health risk assessment in March 2003, followed by the combined human health and ecological risk assessment in May 2003. Based on USEPA responses to comments received during the comment period, the May 2003 draft risk assessment has been revised.

The Aberjona River Study area extends from Route 128 in Woburn to the Upper and Lower Mystic Lakes in Arlington (see Figure 1-1). The study area includes a six-mile reach of the nine-mile-long Aberjona, the Wells G&H 38-acre wetland south of Olympia Avenue, the 17 acre former cranberry bog south of Salem Street, the upper and lower forebays (the two northern basins) of Upper Mystic Lake, the southern basin of Upper Mystic Lake, and Lower Mystic Lake. For the purposes of the risk assessments, the Aberjona River Study area was divided into six reaches and 52 sampling stations (see Figures 1-3 and 2-2).

The Aberjona River Valley, particularly in Woburn, was historically a base for the leather and tanning industries. Woburn also supported a large chemical manufacturing industry in the upper reaches of the Aberjona River. Several manufacturing plants and a number of light industries are still active today.

The field program for the Aberjona River Study area was conducted in 1995, 1997, 1999, 2000/2001 and 2002, and included the collection and analysis of surface water, sediment, floodplain surface soil, and biota (plants, crayfish, and fish) samples, and bioassays with study area sediment. Chemical classes of concern included volatile organics (VOCs), semi-volatile organics (SVOCs), pesticides, polychlorinated biphenyls (PCBs), and inorganics. Within the six reaches of the study area, a total of 21 surface water and 53 sediment and/or soil sampling stations and an additional 12 reference stations were identified. Eight crayfish, six plant, and over 80 fish samples were also collected within the study area. In 2001, 15 stations and five reference locations were sampled using the sediment quality triad, which included laboratory sediment toxicity testing, sediment chemistry, and benthic invertebrate community analysis.

In response to comments provided on the May 2003 draft risk assessment report, the following supplemental data were collected:

- Sediment and floodplain surface soil data collected in 2004 from the Aberjona River south of Bacon Street in Winchester (station AJRW);
- Baseflow and storm event surface water data from six gauging stations (SW-05 through SW-10) along the length of the study area, collected between July 2001 and October 2002; and
- Sediment core data collected in February 2003 from nine locations (SC05 through SC13) along the length of the study area.

A few volatile organics, some pesticides, and various inorganics were detected in surface water. Some volatile and semi-volatile organics, pesticides, PCBs, and numerous inorganics were detected in sediments. In fish and crayfish samples, the frequency of detection was high for many pesticides, PCBs, and inorganics. Contaminants detected in plant tissue included poly-aromatic hydrocarbons (PAHs), pesticides, one PCB, and numerous inorganics. Generally, the highest concentrations of contaminants in all media were measured in the vicinity of the Wells G&H 38-acre wetland in Woburn.

Field investigations also included the collection of 2002 groundwater metals data from within the Wells G&H 38-acre wetland. These data were included in a screening-level evaluation to provide preliminary information on the potential impact of sediments contaminated with metals on groundwater quality. Because the June 21, 2004 Massachusetts Department of Environmental Protection (MADEP) *Groundwater Use and Value Determination* (Appendix A.8) supports a medium use and value for groundwater at the site, which includes drinking water ingestion, federal Maximum Contaminant Levels (MCLs), as standards protective of domestic water use, were selected for the screening-level evaluation. Groundwater is not currently used as a source of potable water. Shallow overburden arsenic concentrations in eight of the thirteen monitoring wells exceeded the arsenic MCL (10 Wg/L). No other exceedances of primary MCLs were noted in shallow, medium, or deep overburden wells. Secondary MCLs for aluminum, iron, and manganese were exceeded in shallow, medium, and deep overburden groundwater. Overall, there appears to be a decreasing trend of groundwater metals concentrations with increasing depth of groundwater. This trend is most noticeable with arsenic, suggesting a possible impact of sediment arsenic contamination on groundwater quality.

Human Health Risk Assessment Summary. Potential noncarcinogenic and carcinogenic human health risks were quantitatively estimated for the central tendency (CT) and reasonable maximum exposure (RME) cases for surface water, sediment, and/or floodplain surface soil at each station determined to be accessible to recreational human receptors currently or in the future (NR, 14, 22/TT-22, 13/TT-27, WH, NT-1, NT-2, NT-3, WG, WW, JY, WS/WSS, TT-30, TT-31, CB-01 through CB-07, 16/TT-33, 09, DA, AM, KF, 08, 07/DP, LP, AS, AJRW, and 06 through 01). Each station was evaluated as a separate exposure point. Risk estimation was also performed for the ingestion of fish fillet tissue from river reaches 3, 4, 5, and 6. In addition, potential risk to workers during future flood mitigation activities involving the dredging of contaminated river sediments were evaluated along with residential exposures to contaminants that may have migrated into residential yards near the river as a result of periodic flooding events.

Prior to completion of the risk assessment, an arsenic bioavailability study was performed to assist in the quantification of sediment risks. This site-specific study determined that arsenic is absorbed less efficiently from sediment than from a water medium. The relative bioavailability estimate determined in this recent study was used to quantify sediment ingestion risks at the study area. In addition, site-specific chromium VI (hexavalent chromium) data for sediments were collected and used in the risk assessment to more accurately characterize sediment risks at the study area.

Whenever possible, Upper Confidence Limits (UCLs), calculated using USEPA's software ProUCL version 3.0, were used as exposure point concentrations. Surface water, sediment, and surface soil risks were estimated for the CT and RME adult and young child recreational receptors exposed during recreational activities (i.e., swimming or wading) under current and future land-use conditions. Only those sediment samples collected from beneath two feet or less of standing water were used in the quantitative evaluation.

Under current land use, stations WS/WSS and CB-01 through CB-06 were evaluated as having the highest exposure potential (based on their proximity to nearby residences), stations 16/TT-33, DA, 09, 08, 07/DP, AJRW, and 06 through 01 as having "typical" exposure potential for recreational areas, and stations NR, 14, 22/TT-22, WH, WG, CB-07, TT-30, AM, and AS as having the lowest exposure potential (based on being partially isolated or in industrial locations). It was assumed that future land use would be the same as current land use, except for stations NR, 22/TT-22, WG, WH, and AS, the exposure potential of which were assumed to increase in the future as the areas near these stations becomes more developed. In addition, stations 13/TT-27, JY, WW, and TT-31 were evaluated under a future land use scenario assuming that the physical barriers limiting current access are removed. Stations NT-1, NT-2, and NT-3 were also evaluated under a future land use scenario due to potential redevelopment plans of the City of Woburn within the Wells G&H wetland that may include the construction of a nature trail (station

NT-3) with a possible boardwalk (station NT-1) or pier (station NT-2) extending out into the wetland. Other stations of the study area have not been quantitatively evaluated in the human health risk assessment because they are difficult to access due to the presence of dense vegetation, deep surface water, and/or their distance from the shoreline.

Potential future exposures to COPCs in sediment core samples (designated SC05 through SC13) collected from nine locations throughout the study area were evaluated to determine risk to workers involved dredging of contaminated sediments. Exposures to residents in the vicinity of stations WS/WSS, CB-05, KF, 07/DP, and AJRW were also evaluated, under the assumption that periodic flooding in these areas has resulting in the migration of contamination to residential yards. Residential exposures to COPCs were conservatively evaluated using storm event surface water data and flood plain surface soil data.

Estimated risks were compared to the USEPA target cancer risk range of 10^{-6} to 10^{-4} and a target hazard index (HI) of 1 for noncarcinogenic effects.

An overall summary of cancer and noncancer risk estimates for the recreational and dredging scenarios is presented in Table ES-1 and Figure ES-1. In Table ES-1, risks are summarized for both the RME and CT receptors only at stations and sediment core locations where site-specific risks exceed risk management guidelines. When risks were estimated for the recreational child and adult receptor, the child HIs are presented on Table ES-1 as the most conservative, while incremental lifetime cancer risks (ILCRs) presented on Table ES-1 are the sum of the child and adult risks (i.e., a total receptor cancer risk). Recreational surface water, sediment, and surface soil risks, presented by station, have been summed together under the assumption that each receptor is exposed to all three media during recreational activities. Fish fillet ingestion risks were not summed with other media. In cases where the total HI exceeded 1, COPCs having similar systemic effects were summed for each pathway and medium. Table ES-1 also summarizes the primary risk contributors for those receptors with estimated ILCRs greater than the target range of 10^{-6} to 10^{-4} and target organ-specific HIs greater than 1.

ILCRs and target organ-specific HIs, estimated for the surface water and surface soil exposure scenarios, were less than or within the target cancer risk range of 10^{-6} to 10^{-4} and less than the target HI of 1, respectively, for each of the exposure areas. When HIs were summed only for COPCs with similar target organs, segregated HIs for fish fillet ingestion were less than or equal to the target HI of 1 for all study area reaches. Total receptor ILCRs were below or within the target risk range for the fish ingestion pathway. In addition, potential residential risks based on exposures to floodplain surface soils and storm event surface water data collected from within the study area were within or below the target cancer risk range of 10^{-6} to 10^{-4} and less than the target HI of 1.

Estimated risks are within or below the target risk levels for exposure to sediment at stations NR, 14, 22/TT-22, WG, WW, JY, WS/WSS, TT-30, TT-31, CB-01, CB-02, CB-04, CB-05, CB-06, CB-07, 16/TT-33, 09, DA, AM, KF, 08, 07/DP, LP, AS, 06, 05, AJRW, 04, 03, 02, and 01. Estimated risks are also within or below the target risk level for worker exposures to sediment cores during possible future dredging activities at locations SC07, and SC09 through SC13.

For sediment, HIs exceeded 1 and/or ILCRs exceeded 10^{-4} at stations 13/TT-27 (future), WH (current/future), NT-1 (future), NT-2 (future), NT-3 (future), CB-03 (current/future), SC05 (future), SC06 (future) and SC08 (future). All stations noted are located within the Wells G&H 38-acre wetland, north of Salem Street in Woburn, with the exception of CB-03 which is located in the former cranberry bog immediately south of Salem Street. The exceedances were due primarily to the presence of arsenic in sediment. It should further be noted that the sediment RME EPC for arsenic at some of these stations is uncertain due to one or a small number of arsenic detects compared to the remainder of the data set. This uncertainty is specifically applicable to stations 13/TT-27 (samples SD-13-01-FW and SD-13-02-FW; 4210 mg/kg and 2480 mg/kg, respectively), WH (sample SD-12-01-ME; 3230 mg/kg), and CB-03 (sample CB-03-11; 1410 mg/kg). Benzo(a)pyrene was also a minor risk contributor at stations 13/TT-27, WH, NT-1, and NT-2. The risk associated with benzo(a)pyrene at these stations was between 2×10^{-6}

and 3×10^{-6}. The benzo(a)pyrene sediment concentrations at these study area stations fall within the range of concentrations detected at the reference stations.

An evaluation of lead in surface water, floodplain surface soil, sediment cores, and sediment at the study area indicated that exposures to lead did not result in estimated childhood or adult blood lead levels in excess of blood lead level goals. Average lead concentrations in fish fillet tissue are less than the published background lead level. Therefore, lead was determined not to be of concern for human receptors at the study area.

Human Health Risk Conclusions. Within the Aberjona River Study area, arsenic and/or benzo(a)pyrene in sediments at stations 13/TT-27, WH, NT-1, NT-2, NT-3, and CB-03 and at sediment core locations SC05, SC06, and SC08 exceeded risk management guidelines established for human exposures. Figure ES-1 visually summarizes the stations and sediment core locations within the study area where human health risks exceed risk management guidelines.

Ecological Risk Assessment Summary. The baseline ecological risk assessment was performed to evaluate the potential for contaminants in surface water, sediment, and biota to impact ecological receptor populations present within the Aberjona River Study area. Study area COPCs, 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. The screening process identified 58 COPCs for sediment.

Nine inorganic COPCs were identified in the initial surface water screening, and among these, seven were detected at concentrations above screening values in a more comprehensive surface water study conducted to support the Remedial Investigation. The surface water COPCs included: barium, cadmium, copper, iron, lead, mercury, and silver. Based on the low magnitude and frequency of the exceedences 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.

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, green heron, mallard, and short-tailed shrew. Additional indicators species/communities selected included fish 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. Fish populations were evaluated by reach, and heron and mallard, which have the largest foraging areas, were assessed on a site-wide scale.

Based on comparison of concentrations of COPCs in fish tissue to reference samples and tissue residue benchmarks, there was no evidence for impacts of COPCs on fish. Based on dietary exposure models, the risk to survival or reproduction of green heron or other semi-aquatic predatory birds, was also negligible.

The stations exhibiting ecological risk are shown in Figure ES-2. A summary of ecological receptor risks is presented in Table ES-2 and Figure ES-3. Risks to ecological receptors were located in depostional areas in the Wells G&H 38-acre wetland (reach 1) and the upper section of reach 2. Two stations in the Mystic Lakes had arsenic concentrations indicating risk to benthic invertebrates. Risks to ecological receptors were due to inorganic contamination, primarily from dietary exposure to sediments and/or ingestion of biota. There were no indications of significant ecological risk associated with VOC, SVOC, and pesticide/PCB, contamination to any ecological receptor within the Aberjona River Study area. The contribution of risk due to exposure to surface water was minor for the mammalian and avian species, indicating no strong evidence for risk from surface water COPCs.

The highest risks to muskrats were for arsenic exposure in reaches 1 and 2. Based on the BERA endpoint, arsenic was determined to pose a risk to muskrat populations based on the large number of stations in reach 1 exceeding levels of potential harm. The calculated threshold effects level was applied to stations in Reach 2. These results indicated a likelihood of potential impacts on survival or reproduction of mammal populations such as muskrat exposed to arsenic in the diet while foraging in the Aberjona River Study area. In addition to arsenic risks, the potential risks to muskrat from exposure to chromium, copper, lead, and mercury was high only at a few stations, and based on the endpoint criterion, exposure to chromium, copper, lead, and mercury presents a low risk to muskrat and other aquatic mammals, mainly in reach 1.

For mallard, chromium, lead, and mercury posed low risk site-wide and within the Wells G&H 38-acre wetland, primarily resulting from high sediment concentrations of these metals in reaches 1 and 2. Although chromium may present a risk at individual stations, the risks to waterfowl on a site-wide basis is low.

The evaluation of risk to shrew indicated low risk from exposure to arsenic in reaches 1 and 2. However, since the number of stations with high risk is small, site-wide the risk to small terrestrial mammals due to exposure to arsenic is low.

For the benthic invertebrates, comparison of sediment metals concentration to effects-based sediment benchmarks indicated potential effects from arsenic, chromium, copper, lead, mercury, and zinc, with highest sediment metals concentrations observed in reach 1. Toxicity testing results showed reductions in growth of laboratory test organisms that 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 risk to invertebrates from high arsenic concentrations in sediments was located mainly in reaches 1 and 2, with limited areas of reach 6 (Mystic Lakes) having arsenic concentrations minimally above levels associated with risk. However, no risk to other receptors, including fish, were identified from exposure to arsenic in the Mystic Lakes.

Additional sampling was conducted after the completion of the draft BERA. These data were evaluated separately and compared to risk conclusions. Based on the baseflow and storm event surface water data and sediment data collected at station AJRW, there is no significant ecological risk associated with exposure of receptors exceeding thresholds established in the BERA. If these data had been incorporated into the BERA calculations, there would have been no changes in the conclusions of sitewide risks calculated for heron or mallard, nor would there have been risk associated with exposure to receptors at this station for semi-aquatic mammals (muskrat) or benthic invertebrates.

Sediment core data collected throughout the study area indicated that sediment metals concentrations generally decreased with depth. The metals concentrations in the sediment core data were consistent with sediment data previously collected in the Aberjona River Study Area in support of the BERA. The sediment core sample results for the 0 - 1 foot interval were qualitatively assessed. Threshold effects levels were exceeded for SC05 through SC08, located in Reach 1, primarily due to arsenic, chromium, and mercury. These results are consistent with results presented for nearby Reach 1 sediment stations assessed as part of the BERA.

Ecological Risk Assessment Conclusions. Figure ES-3 visually summarizes the stations within the study area where there are risks to one or more ecological receptors. Risks were identified for muskrat, mallard, shrew, and the benthic invertebrate community. The highest risk to ecological receptors was found in reaches 1 and 2, associated with arsenic in sediment. Chromium, copper, lead, and mercury in sediment also contributed to risk to a lesser extent for one or more stations and/or receptors.

USEPA intends on expanding the study area by completing a second risk assessment for the Aberjona River north of Route 128. This second risk assessment will include environmental data collected from the Industri-Plex Superfund Site, the Halls Brook Holding Area, and the Aberjona River upstream of Route 128. Collectively, the two risk assessments will evaluate the environmental data collected along the entire river from the Industri-Plex Superfund Site in North Woburn to the Mystic Lakes. USEPA will incorporate this final comprehensive risk assessment into a comprehensive Remedial Investigation (RI) Report documenting all the data collected along the Aberjona River and Halls Brook Holding Area from North Woburn to the Mystic Lakes, and explain further the nature and extent of contaminants and their fate and transport mechanisms. The groundwater-sediment interaction for metals will also be discussed more fully in the comprehensive RI Report. The comprehensive RI Report is expected to be completed in the Fall of 2004.

SECTION 1.0
INTRODUCTION

1.1 PURPOSE, SCOPE AND ORGANIZATION OF REPORT

The Aberjona River Study is one of three operable units delineated for the Wells G&H Superfund Site. The objective of the Aberjona River Study is to determine potential risks to human health and the environment due to exposure to contaminated environmental media within the Aberjona River Study area (the study area). This objective has been addressed by collecting surface water, sediment, floodplain surface soil, and biota samples from the study area, targeting areas where human and environmental receptors have the greatest potential of exposure to contaminants.

This document presents the revised baseline human health and ecological risk assessments for the Aberjona River Study area from Route 128 to the Mystic Lakes. The U.S. Environmental Protection Agency (USEPA) released the draft Aberjona River Study Baseline Human Health and Ecological Risk Assessment Report (M&E, 2003) for public comment in May 2003. USEPA initially released the baseline human health risk assessment in March 2003, followed by the combined human health and ecological risk assessment in May 2003. Based on USEPA responses to comments received during the comment period, the May 2003 draft risk assessment has been revised. Notable aspects of this revision include:

- recalculation of exposure point concentrations based on the use of USEPA's updated software program ProUCL (version 3.0);
- evaluation of recent sediment and floodplain surface soil samples collected along the Aberjona River in Winchester, south of Bacon Street (station AJRW);
- evaluation of recent sediment core data collected from nine locations along the

Aberjona River between Route 128 and the Mystic Lakes (SC05 through SC13); and

- evaluation of recent surface water baseflow and storm event data collected from six surface water gauging stations along the Aberjona River between Route 128 and the Mystic Lakes (SW-05 through SW-10).

Comments received on the draft report along with USEPA responses to the comments are contained within Appendix F.

USEPA intends on expanding the study area by completing a second risk assessment for the Aberjona River north of Route 128. This second risk assessment will include environmental data collected from the Industri-Plex Superfund Site, the Halls Brook Holding Area, and Aberjona River upstream of Route 128. Collectively, the two risk assessments will evaluate the environmental data collected along the entire river from the Industri-Plex Superfund Site in North Woburn to the Mystic Lakes. USEPA will incorporate this final comprehensive risk assessment into a comprehensive Remedial Investigation (RI) documenting all the data collected along the Aberjona River and Halls Brook Holding Area from North Woburn to the Mystic Lakes, and explain further the nature and extent of contaminants and their fate and transport mechanisms. The comprehensive RI Report is expected to be completed in the Fall of 2004.

This revised risk assessment report also provides a screening-level evaluation of groundwater metals data collected within the Wells G&H 38-acre wetland. The groundwater screening will provide preliminary information on the potential impact of sediments contaminated with metals on groundwater quality. Only groundwater metals data have been included in the screening because metals are the primary contaminants within the 38-acre wetland sediments. Other less significant contaminants present within the wetland sediments (e.g., volatile organic compounds) are likely the result of impacts from the Wells G&H source area properties (OU-1). The groundwater-sediment interaction will be discussed more fully in the comprehensive RI Report.

The text of the report is presented in the following six sections of Volume I:

- Section 1.0, Introduction, presents a description of the study area, including its environmental setting, geology, hydrogeology and surface hydrology, background information on study area history and relevant previous investigations, and the study objectives;

- Section 2.0, Site Investigation, describes the scope and methods of field studies, laboratory investigations and data validation, and discusses the nature and extent of contamination;

- Section 3.0, Human Health Risk Assessment, evaluates the baseline human health risks associated with the study area;

- Section 4.0, Ecological Risk Assessment, evaluates the baseline ecological risks associated with the study area;

- Section 5.0, Summary and Conclusions, summarizes the report findings and describes the conclusions of the field investigation and human health and ecological risk assessments; and

- Section 6.0, References, contains the reference citations for the previous five sections of the report.

Volume I also contains the figures. Volumes II and III contain the tables referred to in Volume I, and Volumes IV, V, and VI contain the appendices which provide additional supporting materials for the field investigation (Appendix A), data presentation (Appendix B), the human health risk assessment (Appendix C), the ecological risk assessment (Appendices D and E), and USEPA responses to comments received on the May 2003 draft risk assessment report (Appendix F).

The remainder of this section of the report contains historical information relative to the study area (Section 1.2), a physical description of the study area (Section 1.3), and a discussion of the objectives of the Aberjona River Study (Section 1.4).

1.2 STUDY AREA HISTORY AND PREVIOUS INVESTIGATIONS

The Aberjona River Study area extends from the Aberjona River and associated wetlands just south of Route 128 (northern boundary of the study area), to the Upper and Lower Mystic Lakes in Arlington, including the public beach and swimming area (i.e., Sandy Beach) in Winchester (Figure 1-1). The study area includes a six-mile reach of the nine-mile-long Aberjona River, an associated Wells G&H 38-acre wetland south of Olympia Avenue, a 17-acre former cranberry bog south of Salem Street, and a channelized urban river system. The study area also includes the upper and lower forebays (the two northern basins) of Upper Mystic Lake, Sandy Beach on the northern shore of the southern basin of Upper Mystic Lake, and Lower Mystic Lake. Tributaries to the Aberjona River and uplands within their respective sub-watersheds are not within the study area boundaries.

The Aberjona River Valley, particularly in Woburn, was historically a base for the leather and tanning industries. Associated industries, such as shoe manufacturing, were also important to the local economy (Tarr, 1987). The peak years of the tanning and leather finishing industries were from the late 1870s to the 1920s (Durant *et al.*, 1990). A period of gradual decline followed, with the last plant closing in January 1988 (Durant *et al.*, 1990). Woburn also supported a large chemical manufacturing industry in the upper reaches of the Aberjona River. Several manufacturing plants are still active today.

Historical analysis of aerial photos of the Wells G&H Superfund Site from 1938 to 1988 show a land use shift in the Woburn area from predominantly agriculture to light industry (USEPA, 1988). A number of light industries such as machine shops and small manufacturing operations are found in the area. Some of these light industries and an automobile salvage yard located near the river have been or currently are under investigation by the Wells G&H Superfund Site Operable Unit 1 Settling Defendants as potential contaminant release sites.

There are two National Priority List (NPL) Superfund Sites in the valley (see Figure 1-2). The

Wells G&H Superfund Site is within the study area boundaries. The Industri-Plex Superfund Site is located just upstream of the study area's northern boundary and, therefore, potentially impacts the study area.

1.2.1 Wells G&H Superfund Site

The Wells G&H Superfund Site is a 330-acre site situated in east Woburn, Massachusetts. The site is bounded by Route128/Interstate 95 to the north, Interstate 93 (I-93) to the east, Massachusetts Bay transit Authority (MBTA) railroad tracks to the west, and Salem and Cedar Streets to the south. Wells G&H are two municipal water supply wells located in the Aberjona River Valley that supplemented the City of Woburn's water supply in the 1960s and 1970s (USEPA, 1989). In 1979, the Massachusetts Department of Environmental Quality Engineering (DEQE; now known as the Massachusetts Department of Environmental Protection (MADEP)) tested the water supply near Wells G&H in response to a local disposal problem. Several chlorinated volatile organic compounds (VOCs), including 1,1,1-trichloroethane (1,1,1-TCA), trans-1,2-dichloroethene (t-1,2-DCE), tetrachloroethene (PCE), and trichloroethene (TCE), were detected at concentrations ranging from 1 to 400 parts per billion (ppb). The wells were immediately closed based on the results of this sampling. Woburn then revived an existing agreement with the Metropolitan District Commission to receive supplemental water as a replacement. The City continues to receive this supplemental water.

In 1981, USEPA conducted a hydrogeologic investigation and groundwater quality evaluation of a ten-square-mile portion of east and north Woburn in response to the contamination found at Wells G&H and other disposal problems discovered in the area. The purpose of the investigation was to determine the extent and degree of contamination in the aquifer, and to potentially identify the sources of contamination. Based on the direction of groundwater flow, the areal extent of groundwater contamination and property inspections, USEPA identified the source areas for contamination at Wells G&H to be within a one-square-mile area surrounding the wells on either side of the river. Five facilities were identified as potential sources of contamination: W. R. Grace Company, Unifirst Corporation, New England Plastics, Wildwood Conservation Corporation (also referred to as the Beatrice property), and Olympia Nominee Trust. Based on these findings, Wells G&H was listed as a Superfund Site on the NPL on December 21, 1982.

Additional studies have been performed since the NPL listing. In 1982, the DEQE published the *Upper Mystic Lake Watershed Urban Runoff Project Main Report* characterizing contaminant sources, water quality, pollution impacts and control options associated with urban runoff. The dynamics of wet weather/dry weather effects were also measured and analyzed.

In 1985, the U.S. Geological Survey (USGS), under an agreement with USEPA, conducted a 30-day aquifer test to determine the zone of contribution to the wells (USGS, 1987). The study concluded that a hydraulic connection between the aquifer and the river exists under pumping conditions. Also in 1985, USEPA conducted an evaluation of the wetlands near the wells to determine the extent and type of wetlands that exist at the study area (PRC, 1986). In 1986, USEPA completed an RI that included installation of groundwater monitoring wells, analyses of groundwater, and collection and analysis of Aberjona River surface water and sediment samples over the area between Route 128 and Salem Street.

In 1988, USEPA completed a supplemental RI to gather additional soil and groundwater data as well as to collect additional surface water and sediment samples from the Aberjona River (Ebasco, 1988a). USEPA completed a risk assessment in 1988 (Ebasco, 1988b) that examined the current and future potential risks from exposure to contamination at the study area if no remediation were performed.

Over the last ten years, researchers from MIT have sampled sediments and investigated the spatial distribution and geochemical behavior of inorganics over the length of the Aberjona River from the headwaters in Reading to Upper Mystic Lake (Durant, 1991; Aurillio *et al.*, 1994; Hemond, 1995; Spliethoff and Hemond, 1996). The MIT group analyzed for arsenic and other inorganics, as indicator contaminants of past industrial discharges to the Aberjona River from chemical and leather industries located in the upstream portion of the watershed.

A Record of Decision (ROD) for the Wells G&H Superfund Site was signed in September of 1989 (USEPA, 1989), followed by a Consent Decree in 1991 between USEPA and several Potentially Responsible Parties (PRPs). The ROD required the cleanup of groundwater contaminated principally with volatile organic compounds (VOCs) and/or soil remediation at five identified source area properties owned by the W.R. Grace Co. - Conn., Unifirst Corporation, the Olympia Nominee Trust, Wildwood Conservation Corporation, and New England Plastics. The Consent Decree was entered with four of the five properties to clean-up their soil and groundwater contamination. The fifth property, Olympia Nominee Trust, did not participate in the Consent Decree and has been in negotiations with USEPA for the cleanup of their property. USEPA has identified the groundwater cleanup of these five source areas as the first Operable Unit (OU-1) for the study area. The ROD and Consent Decree also required further investigations be conducted for the Central Area of the study area and the Aberjona River. The Central Area of the study area is defined as the area within the study area boundaries not including the source area properties covered by the ROD or Aberjona River. The investigation of the Aberjona River is being conducted by the USEPA, while the Central Area investigation is being conducted by a group of PRPs under the 1991 Consent Decree. USEPA has identified the Central Area as operable unit 2 (OU-2) and the Aberjona River Study as operable unit 3 (OU-3). The majority of the field investigations conducted in support of the Aberjona River Study were completed by USEPA, U.S. Fish and Wildlife Service (USFWS) and Foster Wheeler in the summer and fall of 1995 (Foster Wheeler, 1996). Surface water, sediment, fish, crayfish and plant samples were collected from various locations within the study area in 1995. After transition of the study to Metcalf & Eddy (M&E) in 1996, a second round of sediment sampling was conducted in 1997 by M&E at selected locations to clarify some uncertainty associated with the 1995 sediment sampling data. This report includes the findings of the 1995 sampling effort conducted by Foster Wheeler, as well as, the supplemental field activities conducted in 1997 by M&E, 2000 through 2002 by Tetra Tech NUS (TtNUS), and 2002 through 2004 by USEPA. In addition, analytical data from reference locations and one study area location sampled by the settling defendants at the Industri-Plex Superfund Site in 1999 have been included to supplement the study area data set.

1.2.2 Industri-Plex Superfund Site

The Industri-Plex Superfund Site is a 245-acre industrial park located in Woburn. From the mid-1800s until 1969, the study area was used to manufacture chemicals (eg. sulfuric acid, lead arsenate, pesticides, and glue. Animal hides and residues used to manufacture glue were buried on-site. Since 1980, the study area has been under investigation by state and federal environmental agencies. It is now being remediated by the Industri-Plex Site Remedial Trust (ISRT), a group of settling defendants including Solutia, Inc. (formerly Monsanto), Stauffer Management Company (formerly Stauffer Chemical) and 22 land owners, in accordance with the 1989 Consent Decree. USEPA is the lead agency overseeing the Consent Decree. The 1986 Record of Decision and 1989 Consent Decree required the ISRT to cap soils contaminated with heavy metals (e.g. arsenic, chromium, lead), collect and treat hydrogen sulfide gas emissions from an on-site animal hide pile, treat hot spot groundwater contamination through an interim groundwater treatment system, and investigate the migration of site-related contaminants. In 1998, ISRT completed construction of the caps and gas collection treatment system. Since 1999, ISRT has conducted additional site-related groundwater, soil gas, sediment, and surface water investigations between Industri-Plex and Route 128, which are expected to be completed in 2004.

USEPA is responsible for incorporating ISRT's site-related investigation data and any other additional investigation data collected by USEPA into a Remedial Investigation/ Feasibility Study

(RI/FS), and determine if any other additional remedial activities are necessary. The Industri-Plex ROD identifies this RI/FS as the Multiple Source Groundwater Response Plan (MSGRP), which focuses on groundwater, surface water and sediments. Over the past few years, USEPA has expanded its investigations to include additional sampling around Industri-Plex and along the entire Aberjona River. All sediment and soil samples collected within the Aberjona River Study area have been incorporated into this baseline risk assessment. All the data collected along the Halls Brook Holding Area (HBHA) and Aberjona River, from the Industri-Plex Superfund Site to the Mystic Lakes, including the data in this baseline risk assessment, will be incorporated into a comprehensive RI. In addition, this baseline risk assessment will be expanded, amended, and/or supplemented to include all the MSGRP data and be finalized in the comprehensive RI for the entire river.

Over the past decade, the Industri-Plex Superfund Site has been extensively studied to determine levels of contamination and remedial approaches. The Phase 1 (1991) and Phase 2 (1992) Groundwater and Surface Water Investigation Plan (GSIP) by ISRT included surface water and sediment sampling from the Industri-Plex Superfund Site in North Woburn to Route 128, as well as reference locations. As indicated above, ISRT is expected to complete it's final investigation of site related contaminants from Industri-Plex to Route 128 by mid-2004. In addition, Massachusetts Institute of Technology (MIT) studies (Aurillio *et al.*, 1994; Hemond, 1995) have analyzed sediments sampled from Halls Brook Storage Area, an artificial nine-hectare impoundment created in the 1970s for flood control purposes, which is located immediately down stream of the Industri-Plex Site. The marshy southern portion of this impoundment discharges into the Aberjona River. MIT has also conducted several studies along the Aberjona River and Mystic Lakes.

1.3 STUDY AREA DESCRIPTION

In this subsection, the environmental setting, geology, surface hydrology, and hydrgeology are described.

1.3.1 Environmental Setting

The Aberjona River has its headwaters in west-central Reading, Massachusetts (near Forest Street and Main Street) and flows southwest from Reading through the City of Woburn and the Town of Winchester before entering the Mystic Lakes system (Figure 1-2). The Aberjona River passes through a mix of park land, residential, urban, and light industrial areas, with the industrial areas concentrated in the City of Woburn. Land use is highly developed along the entire length of the river. As such, the river and associated water bodies and wetlands are affected by a number of potential factors including neglect, indiscriminate disposal of debris, and local and upstream runoff, including non-point and point source discharges, as well as development, loss of flood storage, and culverting/channelizing.

The Aberjona River is classified as a Massachusetts Class B surface water and supports fish populations. Fish sampling confirmed the presence of warm water species throughout the study area. Class B waters are defined by the MADEP as “a habitat for fish, other aquatic life and wildlife, and for primary and secondary recreation. Where designated they shall be suitable as a source of public water supply with appropriate treatment. They shall be suitable for irrigation and other agricultural uses and for compatible industrial cooling and process uses. These waters shall have consistently good aesthetic value.” “Primary contact recreation” represents “any recreation or other water use in which there is prolonged and intimate contact with water and a significant risk of ingestion. These include, but are not limited to, wading, swimming, diving, surfing and water skiing.” “ Secondary contact recreation” represents “any recreation or other water use in which contact with water is either incidental or accidental. These include but are not limited to fishing, boating, and limited contact incidental to shoreline activities.” (MADEP, 1996 and 1997)

In slow-moving sections, the river meanders through vegetated wetlands (Hall Brook Holding Area and the Wells G&H 38-acre wetland in Woburn), while in some of the more urban areas (Route 128 area and in downtown Winchester), the river is culverted or artificially channeled.

The study area is classified as urban. The population has grown in recent years in the City of Woburn from 35,835 in 1990 to 37,528 in 2000 (City of Woburn, 2001) and from 21,221 in 1990 to 21,344 in 2000 in the Town of Winchester (Town of Winchester, 2001) with the expansion of light and technology-related industries along the major interstate highways (Routes 93 and 95/128). The overall increase in population and development has placed an additional demand on the use of open space areas, particularly on the park lands bordering the river in the area of the Woburn-Winchester town line and along the lower reaches of the Aberjona River as it enters the Upper Mystic Lake. Future land use is not expected to change significantly.

Wildlife habitat associated with the river and water bodies within the study area is generally restricted to a relatively narrow corridor. The width of this corridor varies from approximately 20 feet to 0.3 miles. At several locations, development encroaches to the waters edge. Habitats along the river include emergent, scrub/shrub and forested wetlands, fragmented upland forests, sub-mature woodlots, grassy meadows, and maintained park land. The study area is in an urban watershed. In areas not directly impacted by human activities through alterations of the river bank or channelization, the river habitat is indirectly influenced by stormwater run-off or proximity to human activity which can affect habitat quality for some species.

For the baseline human health and ecological risk assessments, the study area has been divided into six reaches (see Figure 1-3), based on information obtained during field reconnaissance activities conducted by the USEPA and USFWS, and information from previous studies, including the *Upper Mystic Lake Watershed Urban Runoff Project Main Report* (DEQE, 1982) and the *Wells G&H Wetlands Assessment Final Report* (PRC, 1986). For the purposes of the ecological risk assessment, the reaches have been defined based on similarity of habitat, species presence or absence, and accessibility. For continuity purposes, the same reaches were also applied to the human health risk assessment. The six defined reaches are as follows:

- Reach 1 — Just south of Rt 128 south to Salem Street including the Wells G&H 38-acre wetland;
- Reach 2 — Salem Street south into Winchester to the area where the river crosses Washington Street (including Cranberry Bog Conservation Area);
- Reach 3 — Washington Street south to Swanton Street (including Davidson Park);
- Reach 4 — Swanton Street south to Mill Pond located in downtown Winchester;
- Reach 5 — Mill Pond outlet south to Upper Mystic Lake Inlet; and
- Reach 6 — Upper Mystic Lake's upper and lower forebays and the Upper and Lower Mystic Lakes.

The Aberjona River in reach 1 and particularly the Wells G&H 38-acre wetland is generally shallow, slow-moving, and turbid as it meanders through the emergent wetland for most of this reach. Floating and emergent aquatic plants are common in this reach and invertebrates observed represent a relatively diverse assemblage of organisms. This wetland offers habitat to a variety of wildlife species, including muskrat, raccoon, red-winged blackbird, common yellowthroat, song sparrow, common grackle, barn swallow, Canada goose, mallard, eastern painted turtle, snapping turtle, bullfrog and green frog. Fish species observed during field investigations included brown bullhead, pumpkinseed, shiner and white sucker. The reach includes commercial, undeveloped, and residential property. Undeveloped property may include recreational use.

Reach 2 is characterized by relatively faster-flowing water and benthic substrates consisting of imbedded gravel and cobbles. However, the waters remain generally shallow. In sections, vegetation along the banks provides greater stream cover in this reach. Fish species observed during site visits were limited to redfin pickerel and white sucker. The Cranberry Bog Conservation Area, a location where local residents may engage in recreational activities including wading and fishing, is located within this reach. This reach also includes an area immediately south of Washington Circle, adjacent to a small park named Danielson Park. The reach includes commercial, undeveloped, and residential property.

Reach 3 is similar in character to reach 2, but includes a small, 1-acre pond at Davidson Park, a frequently used area where local residents may possibly wade and fish. Extensive portions of the stream banks have been stabilized with rip-rap in this reach. Fish species inhabiting reach 3 included largemouth bass, brown bullhead, pumpkinseed, yellow perch and white sucker. The reach includes commercial, undeveloped, and residential property.

Near the start of reach 4, the river is culverted south of Swanton Street, and follows an underground course for approximately 0.3 mile after which it discharges into Judkins Pond located in downtown Winchester. Largemouth bass, pumpkinseed and white sucker were collected in reach 4. The reach includes commercial, undeveloped, and residential property.

The river habitat in reach 5 is similar to that in Reaches 2 and 3, however, park land is the dominant land use along this reach. As in Reach 4, fish species included largemouth bass, pumpkinseed and white sucker. The reach includes commercial, undeveloped, and residential property.

Reach 6 constitutes Upper Mystic Lake's upper and lower forebays and the Upper and Lower Mystic Lakes. The upper and lower forebays of the lake are sediment deposition areas characterized by soft, fine-grained sediments. The lake and surrounding park are intensively used for recreation including swimming, hiking, wading, fishing and boating. Sandy Beach, located on the northern shore of the southern basin, is heavily used during the summer for public swimming and wading. Fish species inhabiting the lake include bluegill, common carp, largemouth bass, pumpkinseed, brown bullhead, yellow perch, black crappie, golden shiner, American eel, alewives, white sucker and chain pickerel. The reach includes commercial, undeveloped, and residential property.

1.3.2 Geology

The Aberjona River flows through a bedrock valley filled with glacial sediments. The principal bedrock type underlying the valley is the Salem gabbro-diorite which is flanked on either side of the valley by the Dedham granite (USGS, 1987). The depth to bedrock from land surface ranges from zero, where bedrock crops out at several locations along the east and west side of the valley, to approximately 140 feet in the center of the valley beneath the Aberjona River. The bedrock valley is filled with glacial outwash deposits and recent alluvial sediments.

These stratified sand and gravel deposits form the most important aquifer (Central Aquifer) in the study area. The unconsolidated valley fill deposits consist of interbedded sands, silts, clays and gravels. In general, these can be divided into three units. The uppermost unit is the sand, silt, clay and peat layer. This layer is underlain by a 10 to 50 foot layer of coarse sands, which in turn is underlain by a unit consisting of coarse sands and gravels. This lower unit ranges in thickness from 20 to 50 feet and directly overlies bedrock. Glacial till is found primarily in the uplands on either side of the Aberjona River Valley. A thin, discontinuous layer of highly compacted basal till (or lodgement till) directly overlying the bedrock has been observed in several of the well borings near Wells G&H.

1.3.3 Surface Hydrology

The Aberjona River, with a drainage area of approximately 25 square miles (USGS, 1990), flows north to south. The river originates north of the study area east of Route 93, in wetlands near the intersection of Forest and Main Streets in Reading, Massachusetts. The river flows away from the wetland in two directions, forming a north and a south branch. Both branches have been diverted into a series of man made channels and culverts as they flow to the southwest. The north and south branches merge west of Route 93, south of Atlantic Avenue, in a open channel and culvert which runs south along Commerce Way in Woburn, Massachusetts. The combined north and south branches discharge to an open channel north of Mishawam Road and merge with the Hall Brook Holding Area, a long narrow wetland north of Mishawam Road. The Hall's Brook Holding Area is a wetland providing wildlife habitat within a urban environment and serving as important flood storage capacity.

From the Hall Brook Holding Area, the Aberjona River flows south through an open channel and culvert under Route 128. Immediately south of Route128, the Aberjona River is channelized to Olympia Avenue. The river continues to flow south and enters the Wells G&H 38-acre wetland characterized by shallow, slow moving, meandering surface water. Approximately ¼ mile south of Olympia Avenue, and north of Well H, the river appears to split into an east and west channel. Based on aerial photographs, the eastern channel appears to be the main flow channel, and continues to have well-developed meanders. The western channel appears to have a less defined channel and more stagnant conditions. The connection between the main (east) channel and the west channel may be seasonally enhanced, potentially influenced by periodic beaver activity, and more well developed during periods of peak flow. Approximately ¼ mile south of where they split, the east and west channels reconverge west of Well G and north of Salem Street.

Downstream of the confluence, the combined channel is faster moving, and more well defined. South of Salem Street, the river flows through a former cranberry bog containing lateral irrigation channels, and continues south as a well defined channel. Between Montvale Avenue and Winchester Center, the river remains well channelized flowing through a series of small urban bermed impoundments, and approximately 1,125 feet of culvert under the grounds of Winchester

High School, and then discharges into Judkins Pond and Mill Pond. The Aberjona River then flows south through mostly open channels before discharging into Upper Mystic Lake.

Surface water elevations (above mean sea level) in the river range from approximately 100 feet in the headwaters area to approximately 13 feet at the entry to the Upper Mystic Lake; elevation near the Wells G&H Superfund Site is approximately 45 feet. Average discharge, as recorded by the USGS station at Winchester, one-half mile upstream of Upper Mystic Lake, is 28.6 cubic feet per second (cfs) over a 50 year period (USGS, 1990). Review of the 1989 flow data for this gauging station indicates low flows occur between July and October and during January. Higher flows occur from March through June (USGS, 1990).

Upper Mystic Lake is a 166-acre glacial kettle lake and comprises two shallow northern basins and one deep southern basin (DEQE, 1982). The Aberjona River discharges into the northernmost basin, referred to as the upper forebay, which is 25.5 acres in size with a maximum depth of 6 feet. The second northern basin, the lower forebay, is 14.5 acres in size with a maximum depth of 8 feet. The southern basin is 126 acres in size and has a maximum depth of 90 feet. Upper Mystic Lake has a total volume of 273 million cubic feet, of which 97 percent is contained in the southern basin. The lake discharges at the outlet dam on the southern shoreline to Lower Mystic Lake, which was reportedly constructed in 1864 (DEQE, 1975) to isolate the Upper Mystic Lake from salt water incursions from the Mystic River and provide a fresh water drinking water supply. The Lower Mystic Lake is 111 acres in size and has a maximum depth of 79 feet.

1.3.4 Hydrogeology

The majority of data on hydrogeology comes from previous Wells G&H studies. Over 200 observation wells have been installed at the Wells G&H Superfund Site to characterize the local hydrogeology and to determine the extent of groundwater contamination. In addition, a 30-day aquifer test was conducted by the USGS in December 1985 (USGS, 1987).

Under normal conditions, groundwater recharge is largely from precipitation which infiltrates through the stratified drift and underlying bedrock in the upland areas. Piezometric head data obtained from nested wells indicate that a downward vertical component of flow exists in the uplands of the Aberjona River Valley, and an upward vertical component of flow exists in the vicinity of the wetlands near the river. As a result, groundwater being recharged in the uplands migrates through the bedrock and stratified drift, ultimately discharging into the wetlands and Aberjona River through the shallow peat layer. Some groundwater migrates laterally down the river valley in a southerly direction.

At the Wells G&H Superfund Site, the elevation of the water table ranges from approximately 40 feet above mean sea level at the Aberjona River to about 95 feet above mean sea level north of the W. R. Grace portion of the study area. The depth to the water table ranges from zero feet in the wetland areas to 10 to 15 feet below ground surface (bgs) in the hilly areas along the east and west boundaries of the study area. The water table gradients are relatively steep in the eastern part of the site near Washington Street and relatively low in the wetlands near the central portion of the site.

The USGS aquifer test (USGS, 1987) concluded that Wells G&H obtained water from a zone of contribution that may be divided into two parts. Most water pumped by the wells was obtained from that part of the aquifer immediately surrounding both wells and through induced infiltration of surface water from the overlying river and wetland. The remaining parts of the zone of contribution were those areas of the Aberjona River drainage basin upgradient from and outside of the area of influence of Wells G&H and downgradient to a stagnation point caused by the groundwater southward flow towards the river and the Riley well (the Riley well was situated in the southwestern portion of the Wells G&H Site). A small amount of Aberjona River surface water entering the northern end of the study area, which is derived from groundwater discharge and surface water runoff in the upgradient drainage area, was induced from the river to the wells under pumping conditions.

1.4 STUDY OBJECTIVES

The overall purpose of the Aberjona River Study is to determine if contamination of the Aberjona River poses potential risks to human health and the environment. The general objectives of the Aberjona River Study are:

- to identify the contaminated environmental media and contaminants of potential concern (COPCs) in the Aberjona River Study area;
- to identify potential pathways of exposure and potential toxicological responses of human and ecological receptors to COPC concentrations occurring in contaminated environmental media at the study area;
- to identify sampling locations within the study area where COPC concentrations in environmental media do and do not appear to pose potential risk to human and ecological receptors, based on conservative exposure and toxicity assumptions;

A field sampling program was designed to collect the data needed to meet the objectives described above. Sampling objectives and design are described in Section 2.0 of this report, including specific details on study area reconnaissance, sampling effort allocation, sampling station definition, and sampling pattern.

This document addresses the study objectives related to human health and ecological receptors.

SECTION 2.0
SITE INVESTIGATION

Site investigation activities were initiated in 1995, and completed in 2004, to meet the objectives of the Aberjona River Study, introduced in Section 1.0 of this report. Sections 2.1 through 2.3 provide details pertinent to these investigational activities. Section 2.1, Field Investigations, describes the field sampling programs designed to collect, analyze, and validate the surface water, sediment, floodplain surface soils, sediment core, and biota data required for use in the baseline human health and ecological risk assessments. This information is also provided for the groundwater sampling conducted within the Wells G&H 38-acre wetland and included for the groundwater screening-level evaluation. The sampling stations are described in Section 2.1.1, sampling methodologies are described in Section 2.1.2, analytical methods in Section 2.1.3, data validation in Section 2.1.4, and data treatment in Section 2.1.5. Appendix A contains additional supporting information pertinent to the investigational activities. The analytical results for the collected environmental samples are presented and discussed in Section 2.2, Nature and Extent of Contamination, with data tables containing all analytical parameters for all collected samples presented in Appendix B. This section also provides a screening-level evaluation of groundwater data collected within the Wells G&H 38-acre wetland. The fate and transport of detected contaminants within the environmental setting of the study area are discussed in Section 2.3.

2.1 FIELD INVESTIGATIONS

Field investigations were conducted in 1995 for the collection of study area and reference surface water, sediment, fish, crayfish, and plant samples. Supplemental study area and reference sediment sampling was conducted in 1997. In 1999, additional reference locations and one study area location were sampled for surface water and sediment in support of the Industri-Plex Superfund Site Investigation. These 1999 data have been used to supplement the reference and study area surface water and sediment data collected for the study area in 1995 and 1997. In 2000 and 2001(2000/2001), additional sediment and surface soil sampling was conducted throughout the study area to fill data gaps for both the human health and ecological risk assessments based on the data collected in 1995 and 1997 and expected future use of the study area. In 2002, further sediment and surface soil samples were collected along the Aberjona River to further characterize potentially impacted areas along the river such as the former Cranberry Bog, flood plain surface soil conditions, and areas proposed by the City of Woburn for future use (e.g., City property containing Well G and Well H, Aberjona Auto Parts), and assist in the preparation of the risk assessments. In response to comments provided on the May 2003 draft risk assessment report, the following supplemental data were collected and incorporated into the risk assessment:

- Sediment and floodplain surface soil data collected in 2004 from the Aberjona River south of Bacon Street in Winchester (station AJRW);
- Baseflow and storm event surface water data from six gauging stations (SW-05 through SW-10) along the length of the study area, collected between July 2001 and October 2002; and
- Sediment core data collected in February 2003 from nine locations (SC05 through SC13) along the length of the study area.

Therefore, for the Wells G&H Aberjona River Study human health and ecological risk assessments, analytical data from study area samples collected between 1995 and 2004, and reference location samples collected in 1995, 1997, and 1999 were combined by media and station as appropriate. In addition, Triad sampling was conducted in June 2001 to further support the preparation of the ecological risk assessment.

Groundwater metals data collected in 2002 within the Wells G&H 38-acre wetland, to be included in the screening-level evaluation, are also discussed. The groundwater screening will provide preliminary information on the potential impact of sediments contaminated with metals on groundwater quality. Only groundwater metals data have been included because metals are the primary contaminants within the 38-acre wetland sediments. The groundwater-sediment interaction will be discussed more fully in the comprehensive RI Report. Other less significant contaminants present within the wetland sediments (e.g., volatile organic compounds) are likely the result of impacts from the Wells G&H source area properties (OU-1) and will be addressed, if necessary, by the remediation of the source areas (OU-1) and the central area (OU-2) of the Wells G&H Superfund Site.

2.1.1 Sampling Stations

Three distinct aquatic habitat types were sampled during the field effort. These included the main channel of the Aberjona River, impoundments along the river course, including Upper Mystic and Lower Mystic Lakes, and associated emergent and forested wetlands. Study area surface water, sediment, floodplain surface soils, and biota samples were collected from each of these three habitat types. In addition, local and regional reference areas, representative of various study area habitats, were selected from reference locations 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. The reference locations were also selected in areas unlikely to be significantly impacted by other sources of contamination.

The sampling stations for the five sampling rounds (i.e., 1995, 1997, 1999, 2000/2001, and 2002 sampling), the Triad sampling, and the supplemental sampling, added in response to comments on the draft report, are discussed in the following subsections. Figure 2-1 shows the locations of the sampling stations. Since 1995, various contractors have collected data within the study area for USEPA. Consequently, several sample identifier nomenclatures were used.

A total of 21 surface water and 53 sediment and/or soil stations were sampled within the aquatic habitats in the six-mile study area (stations 01 through 16 and stations 18 through 22, NR, BW, WG, WH, WW, WS/WSS, JY, TT-27 through TT-33, CB-01 through CB-07, DA, KF, LP, AM, AS, MP, AJRW, UF, UM, and LM). Additional samples collected at stations 04, 06, 07, and 22 in 2000/2001 were designated A0 (station 04), JP (station 06), DP (station 07), and TT-22 (station 22). Locations of the study area sampling stations, presented by river reach, are shown on Figure 2-2 and Figures 2-3 through 2-10. In addition, twelve reference stations (stations 23 through 27, 01-IP through 04-IP, 12-IP, SA-01-TR, and HB-00-TR) were sampled for the purpose of providing a reference for the study area with regard to contaminant distribution and potential sources (excluding the Wells G&H and Industri-Plex Superfund Sites) within the Aberjona River watershed (Figure 2-1 and Figures 2-11 through 2-20).

Since sediment samples were collected at each station, the designation of SD was generally used for sampling stations on the report figures. For example, SD-02-01 is the designation for sediment sample 1 at station 02. However, in the 2000/2001 sampling round, some additional designations were used. For example, the Cranberry Bog samples were designated CB-01-02 for CB station 01 sample 2 and TT- was used to designate stations 27 through 33 (e.g., TT-28-01), as well as the additional samples collected at station 22 (TT-22). Samples collected near the following areas were designated: Wildwood area (WW-), Well G (WG-),Well H (WH-), Salem Street (WS-), Montvale Avenue (AM-), Swanton Street (AS-), Judkins Pond (JP-), the Winchester Center Mill Pond (MP-), station 04 (A0-), Upper Forebay (UF-), Upper Mystic Lake (UM-), and Lower Mystic Lake (LM-). In addition, surface soil samples collected in Davidson Park (station 07) were designated (DP-). Triad sampling stations were designated as -TR. In 2002 and 2004, some additional designations were used: Aberjona River in Winchester south of Bacon Street (AJRW), Board walk (BW), Cranberry Bog (CB-04, -05, -06, -07), south of Danielson Park area (DA), Kraft Foods (KF), Leonard Pool (LP), Normac Road (NR), Junkyard (JY), and Salem Street (WSS). Sediment samples were collected from all these stations, and surface soil samples were collected from the flood plain (i.e., top of bank) at AJRW, CB-05, DA, KF, NR, and WSS. The designation of SO was generally used to represent floodplain surface soil samples at these stations.

Surface water samples collected from the six surface water gauging stations in 2001/2002 were designated SW, followed by a number between 5 and 10 to represent the locations (e.g., SW05). A surface water grab sample collected in 1999 from the Aberjona River between Normac Road and station 14 was designated SW-MC-13, corresponding to location 13 sampled by the Industri-Plex Site Remedial Trust with USEPA oversight. For the sediment core data, the designation SC was used, followed by a number between 5 and 13 to represent the locations (e.g., SC05).

All sediment locations sampled in 1997, 2000/2001, 2002, and 2004, the sediment core locations sampled in 2003, the surface water gauging stations sampled in 2001/2002, and all surface water/sediment locations sampled in 1999 were recorded using a GPS unit. In contrast, only one sampling location (-01) at each station was recorded using GPS for the 1995 sampling round. All other 1995 sampling locations were located using field notes and have been designated as "approximate" locations on the figures in this section. The 1995 field notes are provided in Appendix A. Figures 2-3 through 2-10 present study area sampling locations, by reach. Figures 2-11 through 2-20 present the reference area GPS-recorded locations for the five sampling rounds, as well as other sampling points designated as "approximate" for the 1995 sampling round. Figures 2-31 and 2-34 show the GPS-recorded locations of the surface water gauging stations and the sediment core sampling locations, respectively. Table 2-1 summarizes the distribution and functional role of the study area, reference, and Triad stations.

2.1.1.1 1995 Sampling Round. Samples for surface water, sediment and biota (fish, crayfish and plant tissue) were collected in 1995 from 21 study area stations (Appendix A.1). The field sampling approach included identification of significant habitats associated with the study area for ecological receptors, and areas where potential recreational use may allow for significant human exposure to contaminants in the surface water, sediment, and fish fillet tissue from the Aberjona River Study area. In addition, five stations (stations 23 through 27) were sampled for the same environmental media to provide a reference data set for the study area.

Stations 01 through 04 represent Upper Mystic Lake, with station 01 representing the main swimming beach, Sandy Beach, and station 03 representing a boat launching area. Ten stations (05 through 12, 14, and 16) were selected to represent the main channel of the Aberjona River from Olympia Avenue (station 14) downstream to near the inlet of Upper Mystic Lake (station 05). Seven stations (13, 15, and 18 through 22) were selected to represent the Aberjona River wetlands (Table 2-1 and Table 2-2).

Stations 23, 24, 26 and 27 were selected as regional reference locations. Station 23 was located at the confluence of Lubbers Brook. Station 24 was located in the riverine wetlands of Maple Meadow Brook. Station 26 was located at Wrights Pond, to the east of the study area, and station 27 was located in the Shawsheen River, to the northwest of the study area (Table 2-1). The reference station designated station 25 represents Horn Pond, an impoundment of Fowle Brook (a tributary of the lower Aberjona River).

Surface Water. With four exceptions, at least one surface water sample was collected from each of the 21 study area stations and five reference stations. No surface water samples were collected from stations 19 through 22 since standing water was not present during the 1995 field investigation. Sampling stations are shown by reach on Figure 2-2. Sampling stations are shown on Figures 2-3 through 2-10 for the six reaches of the study area. Figures 2-11 through 2-20 present the reference locations.

Analyses included volatile organic compounds (VOCs), semi-volatile organic compounds (SVOCs), pesticides, polychlorinated biphenyls (PCBs), inorganics (total metals), a suite of water quality parameters (including total dissolved solids [TDS] and total suspended solids [TSS]), and total dissolved carbon. Selected metals of potential interest are presented in Figure 2-21.

Sediment. Sediment sampling was conducted at all 21 study area stations and 5 reference area stations (stations 01 through 16 and 18 through 27; there is no station 17). Multiple sediment samples were collected from each sampling station because historical data identified this medium as containing the largest number of contaminants and because of its tendency to act as a sink for both organic and inorganic contaminants in aquatic environments. This type of sampling design allowed for the evaluation of localized contaminant distributions at each station.

Station	**Location**
01(-01 through -10)	Upper Mystic Lake, Sandy Beach
02 (-01through -03)	Upper Mystic Lake, as it narrows
03 (-01 through -03)	Upper Mystic Lake, near boat launch
04 (-01 through -03)	Upper Mystic Lake, Aberjona River Inlet
05 (-01 through -03)	Gauging Station, Mystic Valley Parkway
06 (-01 through -03)	Judkins Pond
07 (-01 through -10)	Davidson Park
08 (-01 through -03)	Aberjona River, upstream of Davidson Park
09 (-01 through -10)	Cranberry Bog Conservation Area/Danielson Park
10 (-01 through -03)	Salem Street Bridge
11 (-01 through -03)	Aberjona River wetlands, near Well G
12 (-01 through -03)	Aberjona River wetlands, near Well H
13 (-01 through -03)	Aberjona River wetlands, near railroad sidetrack of Wildwood Property
14 (-01 through -03)	Aberjona River wetlands, near Olympia Avenue
15 (-01 through -03)	Aberjona River wetlands, west of Well G
16 (-01 through -03)	Cranberry Bog Conservation Area
18 (-01 through -03)	Aberjona River wetlands, near rifle range road
19 (-01 through -03)	Aberjona River wetlands, near Well G (sediment only)
20 (-01through -03)	Aberjona River wetlands, near dirt road (sediment only)
21 (-01through -03)	Aberjona River wetlands, floating bog near wetlands (sediment only)
22 (-01 through -03)	Aberjona River wetlands, rifle range, west of Dewey St. (sediment only)

23 (-01 through -03)	Confluence of Lubbers Brook (reference)
24 (-01 through -03)	Maple Meadow Brook and Route 129 (reference)
25 (-01 through -03)	North end of Horn Pond, Fowle Brook (reference)
26 (-01 through -03)	Wrights Pond (reference)
27 (-01 through -03)	Shawsheen River (reference)

At stations with three samples, a central location was selected (e.g., the SD-01-01-FW sample) and two additional samples were collected within 50 feet of the central point (e.g., the SD-01-02-FW and SD-01-03-FW samples). This sampling design was used during the 1995 sampling round and allowed for the evaluation of local contaminant distributions at each station. Only the -01 sampling location at each station (e.g., SD-02-01-FW) was recorded with a GPS unit. All other 1995 sampling locations have been designated as "approximate" (Figures 2-3 through 2-10).

Expanded sampling was conducted at station 01 (Upper Mystic Lake - Sandy Beach), station 07 (Aberjona River at Davidson Park) and station 09 (Cranberry Bog Conservation Area on the Aberjona River) because these stations were identified as significant for the baseline human health risk assessment. Although these data were collected specifically for human risk assessment, data from stations 01, 07 and 09 were also collected in habitats relevant to the ecological risk assessment.

Analyses included VOCs, SVOCs, pesticides, PCBs, inorganics (metals and cyanide), total organic carbon (TOC), grain size, moisture, and AVS/SEM. In addition, 10 day benthic bioassays (e.g. toxicity tests) were conducted on sediments from seven study area and two reference sediment stations (see Appendix A.1).

Fish and Crayfish Tissue. Over 80 fish samples were collected throughout the study area between July and September 1995. In addition, thirty-two fish samples were collected from reference station 25 (Horn Pond) and 26 (Wrights Pond). Four categories of samples were collected and analyzed: small fish whole body and large fish fillet, offal, and whole body. Small fish were primarily collected to support the evaluation of the heron in the baseline ecological risk assessment. Contaminant concentrations in large fish fillet, offal, and whole body were used to evaluate the health of fish communities at the study area. However, fillet samples were analyzed primarily to support the baseline human health risk assessment. Small fish were used to provide tissue residue data to support the ecological risk assessment.

Crayfish sampling occurred in July, August and September 1995. Eight crayfish samples, each consisting of several individuals per sample, were collected from the study area: two from reach 1 (CF-RV-06 and CF-RV-07); two from reach 2 (CF-RV-03 and CF-RV-04); three from reach 3 (CF-RV-01, CF-RV-02, and CF-LK-01); and one from reach 5 (CF-RV-05 and CF-RV-08 [duplicate]). One reference sample (CF-RB-02 and CF-RB-03 [duplicate]) was collected in the Shawsheen River. Crayfish were collected to support the characterization of risks to indicator species which consume aquatic invertebrates (heron, muskrat and mallard). Contaminant concentrations in crayfish were also used to evaluate the health of the benthic invertebrate community at the study area.

Whole body, fillet, and offal fish samples and whole body crayfish samples were analyzed for SVOCs, pesticides, PCBs, inorganics, percent lipid, and percent moisture. Species, category, reach in which the collection occurred, lengths, weights, analyte list, and sample labels are provided in Appendix A.1.

Plant Tissue. Plant sampling was conducted during late August and early September 1995. Herbaceous plant samples were collected from three stations (18, 20 and 21) within the wells G&H 38-acre wetland at the northern end of the study area (Figure 2-2). Two composited plant tissue samples were obtained at each station (from the -01 and -02 sediment sampling locations) for a total of six samples. One composite sample (and a duplicate) was also collected from reference station 23. The methods for collection and compositing the plant samples are described in the *Final Field Operations Plan* (Ebasco, 1995). Samples included roots, stems and leaves, but only the basal portions of stems and leaves on tall emergents such as cattails and reeds. The samples were analyzed for SVOCs, pesticides, PCBs, metals, and percent moisture.

Samples PL-18-01 and -02 were collected adjacent to a ponded area characterized by cattails and common reed, with an understory of arrowhead, bulrush and spikerush. Other species included purple loosestrife and touch-me-not. The composite plant tissue sampled consisted of five species: common reed, cattail, arrowhead, spikerush and bulrush. Samples PL-20-01 and -02 were collected from a wetland area, adjacent to a nearly stagnant pool. Dominant species at this location included cattail, purple loosestrife and touch-me-not. The composite samples consisted of four species: cattail, burreed, arrowhead and spikerush. Samples PL-21-01 and -02 were collected from a floating bog in the northwest portion of the Wells G&H 38-acre wetland. The plant community in the sampling area was dominated by cattail, spikerush, arrowhead, grasses, sedges and rush. The composite samples consisted of four species: cattail, common reed, arrowhead and spikerush. References sample PL-23-01 and its duplicate, PL-23-02, were collected from the area adjacent to Lubbers Brook at Concord Street, just west of Route 93, two exits north of Route 128. The composite samples consisted of four species: cattail, pondweed, pickerelweed and burreed.

2.1.1.2 1997 Sampling Round. Additional sampling was conducted in the fall of 1997 to clarify some uncertainty associated with the 1995 sediment sampling results. Only sediment samples were collected during this round, which occurred between November 12, 1997 and November 20, 1997 (Appendix A.2). The 1997 study area sampling locations, recorded with a GPS unit, are shown on Figures 2-3 through 2-10. A total of 26 samples were collected from 17 of the 21 study area stations (stations 08, 09, 14 and 16 excluded). In addition, one sample was collected from two of the five reference stations (stations 24 and 25; see Figures 2-12 and 2-14). All 1997 samples were located as near to the original 1995 locations as possible.

Station	**Location**
01(-01 and -07)	Upper Mystic Lake, Sandy Beach

02 (-01 through -02)	Upper Mystic Lake, as it narrows
03 (-02)	Upper Mystic Lake, near boat launch
04 (-02 through -03)	Upper Mystic Lake, Aberjona River Inlet
05 (-03)	Gauging Station, Mystic Valley Parkway
06 (-03)	Judkins Pond
07 (-02, -05 and -10)	Davidson Park
10 (-01 through -02)	Salem Street Bridge
11 (-01)	Aberjona River wetlands, near Well G
12 (-01 and -03)	Aberjona River wetlands, near Well H
13 (-01 and -03)	Aberjona River wetlands, near railroad sidetrack
15 (-01)	Aberjona River wetlands, west of Well G
18 (-02 through -03)	Aberjona River wetlands, near rifle range road
19 (-01)	Aberjona River wetlands, near Well G (sediment only)
20 (-01)	Aberjona River wetlands, near dirt road (sediment only)
21 (-01)	Aberjona River wetlands, floating bog near wetlands (sediment only)
22 (-02)	Aberjona River wetlands, rifle range, west of Dewey St. (sediment only)
24 (-03)	Maple Meadow Brook and Route 129 (reference)
25 (-02)	North end of Horn Pond, Fowle Brook (reference)

In addition, 10 day benthic bioassays were conducted on six study area and two reference area sediment samples (see Appendix A.2)

For clarity, sediment samples collected in 1995 are referred to with the designation “FW”, and samples collected in 1997 are referred to with the designation “ME”. For example, sample SD-10-01-FW was collected in 1995 from station 10, location 01. Station 10, location 01, was resampled in 1997, and is designated as SD-10-01-ME. However, on Section 2.0 figures, different colors have been used to distinguish between the sampling rounds and GPS located and approximate locations based on field notes.

Analytes for the 1997 sampling round included: VOCs, polynuclear aromatic hydrocarbons (PAHs), pesticides, PCBs, low concentration metals, AVS/SEM, total combustible organics (TCO), grain size, pH, and percent moisture. Modified analytical procedures, based on standard USEPA methods, were used to minimize the influence of high sediment moisture content on quantitation limits.

2.1.1.3 1999 Sampling Round. Surface water and sediment samples were collected from five reference stations (01-IP through 04-IP and 12-IP; see Figures 2-17 through 2-20 and Appendix A.3) by the Industri-Plex Site Remedial Trust with USEPA oversight. Surface water data from one additional location sampled during this round (SW-MC-13) have also been included due to the location of SW-MC-13 within the study area boundary. Figure 2-1 shows the location of these reference station relative to the study area. Figure 2-31 shows the location of SW-MC-13. The 1999 reference and study area stations were located as follows:

Station	Location
01-IP	the Aberjona River east of Acadia Street
02-IP	South Pond
03-IP	Phillips Pond
04-IP	upstream of where Halls Brook runs through residential and commercial areas
12-IP	Halls Brook just upstream of 04-IP
SW-MC-13	Aberjona River, upstream of Wells G&H 38-acre wetland

These samples have been used to supplement the surface water and sediment study area and reference data collected in 1995 and 1997 (Tables 2-1 and 2-2).

Analytes for the 1999 sampling round included: VOCs, SVOCs, pesticides, PCBs, metals, total organic carbon (TOC), grain size, and AVS/SEM analyses for sediment, and VOCs, SVOCs, pesticides, PCBs, dissolved metals, and total metals for surface water.

2.1.1.4 2000/2001 Sampling Round. Sediment samples were collected at 21 new stations and three previously sampled stations (04, 06, and 22) and surface soils were collected at Davidson Park (station 07) within the site study area as listed below (Figures 2-3 through 2-10). Figure 2-1 illustrates the location of these new stations relative to the other study area and reference locations. Appendix A.4 contains the field sampling log sheets for this sampling round.

Station	Location
WG-01 through -20	eastern portion of Wells G&H Wetland
WH-01 through -10	eastern portion of Wells G&H Wetland
WW-01 through -12	western portion of Wells G&H Wetland
WS-01 through -10	Salem Street
TT-22-01 through -03 (station 22)	northeast of Wells G&H wetland
TT-27-01 through 04	western portion - Wells G&H Wetland (except -04)
TT-28-01 through -03	eastern portion of the Wells G&H Wetland
TT-29-01 through -03	eastern portion of the Wells G&H Wetland
TT-30-01 through -03	south of Salem St.
TT-31-01 through -03	south of Salem St.
TT-32-01 through -03	Cranberry Bog (north of stations 9 & 16)
TT-33-01 through -03	Cranberry Bog (north of stations 9 & 16)
CB-01-01 through -10	Cranberry Bog (north of stations 9 & 16)
CB-02-01 through -10	Cranberry Bog (north of stations 9 & 16)
CB-03-01 through -12	Cranberry Bog (north of stations 9 & 16)
AM-01	Montvale Ave
DP-01 through -26 (station 7)	Davidson Park (soils)
AS-01 through -02	Swanton Street

JP-01 (station 6)	Judkins Pond
MP-01 through -02	The Winchester Center Mill Pond
AO-01 through -05 (station 4)	Upper Forebay of the Mystic Lakes
UF-01 through -03	Upper Forebay of the Mystic Lakes
LF-01 through -02	Lower Forebay of the Mystic Lakes
UM-01 through -03	Upper Mystic Lake
LM- 01 through-03	Lower Mystic Lake

Since results of the 1995 and 1997 sampling rounds indicated that the contaminants of concern were predominately metals, the sediment and surface soil samples collected in 2000/2001 were analyzed for inorganics only.

2.1.1.5 2001 Triad Sampling. In June 2001, sediment samples were collected at 20 locations for sediment chemistry (volatiles, semi-volatiles, pesticides, PCBs, metals, TOC, grain size, AVS/SEM), benthic invertebrate community analysis and laboratory sediment toxicity testing (10 day and chronic testing). The stations sampled for the sediment quality triad analysis are shown in Figures 2-1, and 2-3 through 2-10. These stations included 15 stations in the Aberjona River Study area and three reference locations sampled previously for sediment chemistry and two new reference locations. HB-00 (Hall's Brook) and SA-01 (South Branch Aberjona River) represent new reference locations used for the Triad sampling only. The Triad locations are further designated by the suffix -TR. Appendix A.5 presents field sampling information for the Triad sampling.

Station	Location
01-IP-TR	east of Acadia Street
SA-01-TR	South Branch of Aberjona - Acadia Rd
03-IP-TR	Phillips Pond
04-IP-TR	Halls Brook north of residential/commercial areas
HB-00-TR	Hall's Brook, Danforth street

22-01-TR	northeast of Wells G&H wetland
13-01-TR	Aberjona River Wetlands
WH-07-TR	eastern portion of Wells G&H wetland
12-03-TR	Aberjona River
TT-29-03-TR	eastern portion of Wells G&H wetland
19-01-TR	Aberjona River wetland
18-02-TR	Aberjona River wetland
WW-06-TR	western portion of Wells G&H wetland
10-02-TR	Aberjona River
TT-30-01-TR	south of Salem Street
TT-32-02-TR	Cranberry Bog (north of stations 09 & 16)
TT-33-02-TR	Cranberry Bog (north of stations 09 & 16)
06-03-TR	Aberjona River (Judkins Pond)
UF-02-TR	Upper Forebay of Mystics Lakes
AO-03-TR	Upper Forebay of Mystics Lakes

A summary of the volatile organic compounds detected by organic group within the study area for the 1995, 1997, and 2000/2001, and 2001 Triad sediment sampling rounds are presented on Figure 2-22.

2.1.1.6 2002 Sampling Round. Sediment samples were collected at eight new stations and surface soils only were collected at five stations (two new stations and three of the eight new sediment stations; Figures 2-2 through 2-10) for metals, PCBs (selected sediments) and TOC analyses. Figure 2-1 and Figures 2-3 through 2-10 illustrates the location of these new stations relative to the other study area and reference locations. Appendix A.6 presents field logs for this sampling round.

NR-01 through -05	Normac Road north of Olympia Ave.(sediments)
NR-16 through -20	Normac Road north of Olympia Ave. (soils)

BW-01 through -05	future boardwalk in Wells G&H Wetland
WSS-01 through -05	north of Salem Street (soils)
JY-06 through -15	Junk Yard (north of Salem Street)
CB-04-01 through –09	Cranberry Bog (north of stations 09 & 16)
CB-05-01 through -10-02	Cranberry Bog (north of stations 09 & 16) (soils)
CB-06-01 through -10-02	Cranberry Bog (north of stations 09 & 16)
CB-07-01 through -06	Cranberry Bog (north of stations 09 & 16)
DA-01 through -05	Danielson Park (soils)
KF-01 through -10	Kraft Foods south of Montvale Ave (sediments)
KF-01 through -10	Kraft Foods south of Montvale Ave (soils)
LP-01 through -15	Leonard Pool (south of Davidson Park)

PAHs, pesticides, and PCBs detected in 1995,1997, 2000/2001, 2001 Triad, and 2002 sediment samples are summarized on Figure 2-23. Selected metals (arsenic, chromium, lead, and mercury) detected in sediments in the 1995, 1997, 2000/2001, 2001 Triad, and 2002 sampling rounds are presented on Figure 2-24 and Figures 2-25 through 2-29.

2.1.1.7 Supplemental Sampling. The supplemental sampling includes the collection of: (1) sediment and floodplain soil samples at station AJRW; (2) baseflow and storm event surface water samples collected at six surface water gauging stations within the study area; and (3) sediment core samples at nine locations within the study area. The following subsections describe the supplemental sampling, by media. Appendix A.7 presents field log sheets for the supplemental sampling.

Sediment. Sediment samples were collected at station AJRW (Figures 2-2 and 2-10) for metals and TOC. Six deep sediment samples (SD-AJRW-01 through AJRW-06) were collected in March 2004, and nine shallow sediment samples (SD-AJRW-07, -09, -11, -13, -15, -17, -19, -21, and -23) were collected in April of 2004. Figure 2-1 illustrates the location of this new station relative to the other study area and reference locations. Figure 2-10 demonstrates the locations of the individual sediment samples.

Floodplain Surface Soil. Floodplain surface soil samples were collected at station AJRW (Figures 2-2 and 2-10) for metals and TOC. Nine floodplain surface soils (SO-AJRW-08, -10, -12, -14, -16, -18, -20, -22, and -24) were collected in April of 2004. Figure 2-1 illustrates the location of this new station relative to the other study area and reference locations. Figure 2-10 demonstrates the locations of the individual soil samples. Selected metals (arsenic, chromium, lead, and mercury) detected in soils in the 2000/2001, 2002, and 2004 sampling rounds are presented on Figure 2-30.

Surface Water. Baseflow surface water grab samples were collected from six surface water gauging stations within the study area between July 2001 and October 2002 for total metals, dissolved metals, and TSS. Storm event data were also collected from the surface water gauging stations in 2002 during April, May, July, August, September, and October storm events. The surface water gauging stations, designated SW-05 through SW-10, are shown on Figure 2-31. Gauging stations SW-05, SW-06, and SW-07 were located at the Salem Street Bridge (south of the Wells G&H 38-acre wetland), downstream of the Montvale Avenue bridge (Woburn), and at the Swanton Street bridge (Winchester), respectively. Gauging station SW-08 was located at the USGS gauging stations on the Mystic Valley Parkway (Winchester). Gauging stations SW-09 and SW-10 were located at the outlets to the Upper (Arlington) and Lower (Medford) Mystic Lakes, respectively. Selected metals (arsenic, chromium, lead, and mercury) detected in the 2001/2002 baseflow and storm event surface water are presented on Figures 2-32 and 2-33, respectively.

Sediment Cores. Sediment core samples were collected at nine locations (SC05 through SC13) within the study area for metals, hexavalent chromium, and TOC. Figure 2-34 demonstrates the locations of the individual sediment core samples. At each sediment core location, samples were collected from four discrete intervals: 0-1 foot, 1-2 foot, 2-3 foot, and 3-4 foot. All samples were collected in February of 2003. SC05 through SC08 were collected from the Wells G&H 38-acre wetland, SC09 and SC10 from the former Cranberry Bog, SC11 from Davidson Park, SC12 from Judkins Pond, and SC13 from the outlet of the river as it flows into the upper forebay of Upper Mystic Lake (Figure 2-34). Selected metals (arsenic, chromium, lead, and mercury) detected in the 2003 sediment cores are presented on Figure 2-35.

2.1.1.8 Groundwater Sampling. The groundwater sampling data were collected by TRC Environmental (TRC) in March, April, and October 2002. TRC installed and sampled nine wells within the Wells G&H 38-acre wetland and sampled four previously installed wells, also located within the wetland area of interest. The monitoring well locations are shown on Figure 2-36 and include MW-003 through MW-011 (installed by TRC) and S88, S89, S92, and S93. Groundwater sampling was conducted within the shallow overburden, medium overburden, and deep overburden flow zones. Wells were sampled under ambient, non-pumping conditions. The report entitled *"Data Trend Evaluation, Wells G&H Superfund Site, Operable Unit 1 - Olympia Property"* (TRC, 2002b) presents field log sheets for the wells installed in 2002.

2.1.2 Sampling Methodologies

In the following subsections, the specific sampling methodologies used for the 1995, 1997, 1999, 2000/2001, and 2002 sampling rounds, the 2001 Triad sampling, and the supplemental sampling are presented. Sampling methods for groundwater are also briefly discussed.

2.1.2.1 1995 Sampling Round. Surface water was collected by approaching the sampling location from a downstream location and filling the sampling container directly from the body of water. Where the surface water was too shallow to dip the sampling container or while sampling for VOCs, a small transfer container was used. Sample aliquots for VOC analyses were collected in pre-preserved bottles, while other analyses requiring preservatives were preserved post-collection.

Sediment samples were collected as near to the surface water sampling location as possible where undisturbed sediment existed from a depth of 0 - 6 inches. A stainless-steel sampling device was used to collect the sediments. Sediment aliquots for VOC analyses were placed directly into the sample bottle. Additional sediment was then placed into a stainless-steel bowl and thoroughly mixed. The required number of sample bottles were then filled for the required analyses.

Aliquots of sediment were also collected for bioassays with benthic invertebrates. The purpose of the toxicity testing was to determine whether the survival or growth of organisms exposed to study area sediment would be significantly different from the survival or growth observed in organisms exposed to reference or control sediment.

In the 1995 sampling round, sediments for bioassays were collected from eight sampling stations (SD-07-04-FW, SD-12-03-FW, SD-16-01-FW, SD-18-01-FW, SD-19-01-FW, SD-19-02-FW and two reference stations, SD-23-01-FW and SD-24-01-FW) (USEPA, 1996b). A field duplicate was collected at SD-19-01. Stations 07, 12, 16 and 23 were located in riverine habitat while stations 18, 19 and 24 were located in wetland habitat. Ten-day growth and survival tests were conducted on eight (8) sediment samples at the USEPA New England Office of Ecosystem Assessment with the amphipod, *Hyalella azteca*, and the chironomid, *Chironomus tentans*, in accordance with USEPA (1994) and USEPA Region I ESD Biology Section SOP #13.1.2. Unless otherwise indicated, statistical evaluations of toxicity test data were performed by USEPA.

Fish, crayfish, and wetlands vegetation were also sampled during the 1995 sampling round, between July 31 and September 12. Crayfish and fish sampling were conducted jointly by Foster Wheeler and United States Fish and Wildlife Services (USFWS). Although the primary method of fish collection was electroshocking, seines and trot lines were also used. Crayfish were collected by seining. Plant samples were collected by hand. Fish, crayfish and plants were collected and analyzed for contaminants to support the evaluation of risks to ecological receptors. Fish fillet data were utilized to evaluate risks to human receptors.

Samples for VOCs, SVOCs, pesticide/PCB and inorganic analyses were chilled to 4°C and shipped to environmental laboratories for Routine Analytical Services (RAS) analyses via the USEPA CLP system and to Delivery of Analytical Services (DAS) laboratories for TOC, grain size and AVS/SEM analyses. Tissue samples and sediment samples for bioassay testing were sent to laboratories for analyses in coordination with USFWS and USEPA.

Additional information regarding the field program, as well as the toxicity testing report are presented in *Preliminary Data Compendium, Wells G&H RI/FS, Operable Unit III, Aberjona River Study, sampling period July 31, 1995 to September 12, 1995,* (Foster Wheeler, 1996) and in Appendix A.1.

2.1.2.2 1997 Sampling Round. Sediment was the only medium sampled during the 1997 sampling round, and samples were collected from a depth of 0 - 6 inches. At most locations, sediment samples were collected using an Eckman dredge equipped with a six-foot handle. Sediment was collected by inserting the dredge into the river or lake bottom deep enough to allow the loose plant material and decaying organic matter to float out the top and then closing the bottom “jaws” of the dredge. The dredge collected a semisolid “cube” of sediment from the location which was then subsampled when brought into the boat. At shallow locations, sediment was sampled directly with tubular modified syringes and coring devices (e.g., bulb planters).

Aliquots for VOC analyses were placed directly into pre-weighed septum jars containing sufficient methanol to immerse the sample. Aliquots for AVS/SEM analyses were placed directly into wide-mouth jars with teflon-lined lids. Jars were filled to minimize the headspace and prevent the loss of hydrogen sulfide. Sediment for the remaining analyses was placed into a stainless-steel colander lined with large pieces of filter paper to promote dewatering. Extra filter paper was placed onto the top of the sediment to speed the dewatering process. The dewatered sediment was then homogenized, placed into appropriate sample jars and chilled to 4°C ±2°C. The samples were shipped to environmental laboratories that were specifically chosen to analyze high moisture samples and had successfully completed an analytical pre-test analysis.

As in the 1995 sampling round, sediments were also collected for bioassays with benthic invertebrates. In the 1997 sampling round, 10 day toxicity test sediments were collected from eight stations (SD-06-03-ME, SD-07-10-ME, SD-10-02-ME, SD-12-03-ME, SD-18-02-ME, SD-19-01-ME and two reference stations, SD-25-02-ME and Fowle Brook) (USEPA, 1998b). Fowle Brook (near Horn Pond) was sampled for toxicity testing only. Stations 10, 12, 18 and Fowle Brook (SD-FB) were located in riverine habitat while stations 06, 07, 19 and 25 were located in pond/wetland habitat.

Ten-day survival tests were conducted at the USEPA New England Office of Ecosystem Assessment with *Hyalella azteca*, and *Chironomus tentans,* in accordance with USEPA (1994) and USEPA OEME Biology Section Standard Operating Procedure Number 2.7. Growth was also an endpoint evaluated for *C. tentans*. Unless otherwise indicated, statistical analyses of toxicity test data were performed by USEPA.

Additional details regarding 1997 sampling methodologies and the toxicity testing report are presented in the *Supplemental Data Compendium for Wells G&H Superfund Site, Aberjona River Study (Operable Unit 3), Woburn, Massachusetts* (M&E, 1998) and in Appendix A.2.

2.1.2.3 1999 Sampling Round. Surface water samples were collected by lowering decontaminated sample bottles oriented upstream to the desired depth. In water > 2 ft deep, decontaminated Niskin or Kemmerer water sample devices were used. In water < 2 ft deep, samples were collected using a decontaminated wide-mouth sampling jar.

Sediment samples were collected using a Tall Eckman dredge sampler. It was estimated that 6 grabs would be needed to collect sufficient volume for all parameters and the toxicity tests. Samples for VOCs and AVS/SEM were obtained from the first grab. All other samples were drawn from a composite of the upper two inches of sediment from the six grabs. For VOCs, the sample was collected using a syringe and placed in a “low-level” prepared VOC sampling container.

Additional details regarding the Industri-Plex sampling methodologies are presented in *Toxicological Surface Water Sediments Sampling and Fish Sampling Work Plan and Quality Assurance Project Plan for the Industri-Plex Site, Woburn Massachusetts,* (Menzie Cura, 1999) and Appendix A.3.

2.1.2.4 2000/2001 Sampling Round. Sediment samples were collected from the 0 - 6 inch interval in the Wells G&H wetland/floodplain using a hand tool. The sediment was homogenized in a stainless-steel bowl and any free water present was decanted. The sediment was then placed in the appropriate containers for analysis. An Eckman grab sampler was used to collect the sediments from the Aberjona River and Mystic Lakes. These sediment samples were collected from downstream to upstream locations. Any free water present was decanted from the top of the Eckman sampler. The depth of the sample in the Eckman sampler was measured before placing the sediment into a stainless-steel bowl for homogenization. The sediment was then placed in the appropriate containers for analysis.

Surface soil samples were collected from Davidson Park and were obtained from a depth of 0 - 6 inches bgs using a hand auger. Equal volumes of soil were collected from four different points close to and surrounding the sample location to ensure that the sample was a representative composite of the location. The soil was placed into a stainless-steel bowl and thoroughly mixed to homogenize the soil from the four locations. The soil was then placed in the appropriate containers for analysis.

2.1.2.5 2001 Triad Sampling. During the June 2001 triad sampling, an Eckman grab sampler was used to collect sediments from the 0-6 inch interval for chemistry analyses, benthic invertebrate community composition and toxicity testing. Sediment aliquots for AVS/SEM and VOC analyses were collected first and were placed directly into the sample bottles. Three separate Eckman grabs were collected for macroinvertebrate identification. These samples were sieved in the field through a 600 µm mesh (#30) sieve and the material remaining on the sieve was placed in a sample container and preserved with formalin. Additional dredge samples were collected from undisturbed locations, placed in a large stainless steel container, and thoroughly mixed. The required number of sample bottles were then filled for the chemistry and laboratory toxicity testing analyses.

In the 2001 Triad sampling round, sediments for toxicity testing were collected from 20 sampling stations. Stations 12, TT-29, 18, 10, TT-30, 01-IP, and 04-IP were located in riverine habitat while stations WH, 22, 19, TT-33, TT-32, WW, 13, SA, and HB were located in wetland habitat and stations 06, UF, AO, and 03-IP were located in lake/pond habitat (Tables 2-1 and 2-2). Laboratory toxicity tests were conducted on twenty sediment samples with the amphipod, *Hyalella azteca*, and the chironomid, *Chironomus tentans*, in accordance with *Methods for Measuring the Toxicity and Bioaccumulation of Sediment-associated Contaminants with Freshwater Invertebrates* (USEPA, 2000). Ten-day acute toxicity tests were performed using both *Hyalella azteca* and *Chironomus tentans*. In addition, 42-day chronic toxicity tests were conducted with *Hyalella azteca* and life cycle chronic tests were conducted with *Chironomus tentans*. Unless otherwise indicated, statistical evaluations of toxicity test data were performed in accordance with USEPA (2000) methodology.

2.1.2.6 2002 Sampling Round. Sediment and surface soil samples were collected from the 0-6 inch interval for chemical analyses. To maintain consistency, sample collection procedures used in the 2000/2001 sampling round (described in Section 2.1.2.4) were followed for the 2002 sampling round.

2.1.2.7 Supplemental Sampling. To maintain consistency, sample collection procedures used in the 2000/2001 sampling round (described in Section 2.1.2.4) were followed for the collection of the 2003 sediment cores samples and the 2004 sediment and floodplain surface soil samples. Sediment and floodplain surface soil samples were collected from the 0-6 inch interval. For the 2003 sediment core samples, four collection intervals were sampled: 0-1 foot, 1-2 foot, 2-3 foot, and 3-4 foot.

Monthly baseflow surface water grab samples were collected using the "direct-dip" method: submerging a sample bottle to the required depth (60 percent of the total water column) and retrieving a sample. Samples to be analyzed for dissolved metals were filtered immediately after collection using a peristaltic pump. Storm event sampling was conducted using automated samplers, equipped with 24 1-liter plastic sampling bottles to collect samples for metals analysis. The samplers would engage at the same time when triggered by pre-set conditions. The auto-samplers also performed field measurements of velocity, pH, conductivity, specific conductance, oxidation-reduction potential, dissolved oxygen, temperature, turbidity, and rainfall. Storm event samples were collected using a slotted PVC pipe sampler which collected water from the entire water column near the center of the river channel.

2.1.2.8 Groundwater Sampling. Newly installed wells were developed prior to sampling, as described in the report entitled *"Data Trend Evaluation, Wells G&H Superfund Site, Operable Unit 1 - Olympia Property"* (TRC, 2002b). Groundwater sampling was performed using low-flow procedures in accordance with USEPA's July 30, 1996 Revision 2 Region I low-flow purging and sampling guidelines. Additional sampling details are provided in TRC's February 2002 *Quality Assurance Project Plan for Pre-Design Investigation and Site Characterization* (TRC, 2002a).

2.1.3 Analytical Methods

In the following subsections, the specific analytical methods used for the 1995, 1997 and 1999, and 2000/2001 sampling rounds, the 2001 Triad sampling, the 2002 sampling round, and the supplemental sampling are presented. The method references by analytical parameter for each of the rounds of sampling are presented following the text.

2.1.3.1 1995 Sampling Round. Analysis of surface water and sediment samples for the 1995 sampling round were performed by USEPA Contract Laboratory Program (CLP) laboratories using USEPA Routine Analytical Services (RAS) methods. Other selected laboratories performed Delivery of Analytical Services (DAS) methods. The RAS methods used for the analyses of the target analyte list (TAL) and target compound list (TCL) analytes did not take into account the low percent solids in the sediments, which may be associated with high moisture content and/or high organic content, resulting in the rejection of some non-detect data during validation because of high moisture content, in accordance with USEPA Region I data validation guidelines (USEPA, 1996a). Data for TOC, grain size, and AVS/SEM were unaffected. Water quality was monitored at each sampling location by measuring pH, temperature, specific conductivity, dissolved oxygen, and hardness immediately prior to surface water sampling.

1995 Sampling Round - Foster Wheeler
Sediment, Surface Water, and Biota (Fish Tissue, Crayfish, Plants)

Parameter	Method Reference
Volatile Organics	DAS and RAS CLP SOW OLM03.1
Semivolatile Organics	DAS and RAS CLP SOW OLM01.9
Pesticides/PCBs	RAS CLP SOW OLM01.9
Metals-total	DAS (ILM03.0)
Cyanide (sediment only)	DAS (ILM03.0)
Dissolved Organic Carbon (surface water)	DAS (SW846-9060-mod)
Water Quality Parameters (surface water)	DAS methods
AVS/SEM (sediment)	DAS (USEPA 1992 draft method)
Total Organic Carbon (sediment)	DAS (Lloyd Kahn)
Grain Size (sediment)	DAS (sieve)
Moisture	DAS (CLP gravimetric-mod)
Bioassays (sediment)	USEPA 100.1, 100.2
Lipid Content (fish tissue, crayfish only)	DAS (USEPA-OB)

2.1.3.2 1997 Sampling Round. For sediments collected during the 1997 sampling round, DAS analytical methods for analysis of all parameters were used (M&E, 1997). Sediments for VOC analysis were field preserved/extracted in purge and trap grade methanol and analyzed via a modified version of USEPA SW846 Method 8260 incorporated in the DAS specification. PAH analysis was conducted using a modified version of the CLP Statement of Work (SOW) OLM03.1 incorporated in the DAS specification. Provisions for selected ion monitoring (SIM) were included in the method to provide the required quantitation limits. Pesticides and PCBs were also analyzed using a modified version of the CLP SOW OLM03.1 incorporated in the DAS specification. An additional compound, 2,4'-DDT, was added to the target compound list and a freeze drying technique was used to increase the percent solids in the sediment matrix.

The DAS method for the analysis of low concentration metals used a modified version of the CLP SOW ILM04.0 incorporated in the DAS specification requiring lower detection limits for antimony, arsenic, beryllium, cadmium, mercury and silver as needed for the risk assessments (M&E, 1997). An oven-drying process at 60°C was used to increase the percent solids in the sediment matrix. The AVS/SEM USEPA method, the "Narragansett Method", was used as written because it was designed for sediment determinations. Specific quantitation limits were required of the laboratory for risk assessment use. The TCO content of the sediments was greater than one percent, therefore, the low concentration TOC procedure was not performed for any samples. Grain size (both sieve and hydrometer), moisture content, and pH in the sediments were determined by routine methods.

1997 Sampling Round - Metcalf & Eddy
Sediment

Parameter	Method Reference
Volatile Organics	DAS (8260A- mod)
Polyaromatic Hydrocarbons	DAS (OLM03.1 mod)
Pesticides/PCBs	DAS (OLM03.1 mod)
Metals	DAS (ILM04.0 mod)
Total Combustible Organics	DAS (ASTM D2974-87)
Grain Size	DAS (ASTM D422-63)
AVS/SEM	DAS (AVS by USEPA1993 method, SEM by ILM04.1-mod)
pH	DAS (SW846-9045C)
moisture content	DAS (ASTM D2974-87)
10 day Toxicity	USEPA 100.1, 100.2

2.1.3.3 1999 Sampling Round. Woods Hole Group Environmental Laboratory (Woods Hole) in Raynham, Massachusetts performed the analyses of surface water and sediment samples for the 1999 sampling round. USEPA SW-846 methods were used for analysis of all parameters with the exception of AVS/SEM and grain size (see below). Woods Hole used a method based on USEPA's 1992 draft method for the analysis of AVS/SEM. Method references by parameter type are included in the table below. The laboratory slightly modified these methods to meet the data quality objectives (e.g., detection limits) presented in the quality assurance project plan (Menzie-Cura, 1999). Woods Hole performed freeze-drying of the sediment samples prior to sample preparation and extraction to increase the solids content of the sample. This technique was employed to minimize the rejection of data in samples with low solids content, as required by USEPA Region I data validation guidelines (USEPA, 1996a). Technical staff at USEPA Region I support the use of freeze-drying sediment samples to decrease the moisture content. VOC and AVS/SEM samples did not undergo freeze-drying prior to analysis since volatile compounds may be lost upon freeze-drying.

1999 Sampling Round- Industri-Plex
Sediment, Surface Water, and Biota (Benthic Invertebrates)

Parameter	Method Reference
Volatile Organics	SW-846 8260B
Semivolatile Organics	SW-846 8270C
Polyaromatic Hydrocarbons (biota)	SW-846 8270C
Pesticides	SW-846 8081A
Polychlorinated Biphenyls	SW-846 8082
Metals-dissolved (surface water)	SW-846 6010B and/or 7000 series
Metals-total and Cyanide	SW-846 6010B and/or 7000 series
Total Organic Carbon	SW-846 9060 (modified for sediments)
AVS/SEM (sediment)	USEPA 1992 draft method
moisture content	ASTM D2974-87
Grain Size (sediment)	ASTM D422

2.1.3.4 2000/2001 Sampling Round. Southwest Research Institute, Ceimic Corporation, Mitkem Corporation, and Chemtech Consulting Group performed the sediment and soil analyses for metals, cyanide, and/or sulfides. USEPA CLP SOW ILM04.1 and DAS were used for the analysis of metals and cyanide and SW-846 was used for hexavalent chromium on 20% of the samples collected. TtNUS DAS methods, which are based on USEPA methods, were also used for the analysis of metals, cyanide, and sulfides (see below).

For samples with low solids content submitted for metals and cyanide analyses, the samples were either oven-dried or the sample aliquot was increased prior to digestion/distillation in order to achieve the required quantitation limits and to avoid rejection of data as required by USEPA Region I data validation guidelines (USEPA, 1996a). Samples submitted for sulfide analyses did not undergo sample preparation modifications to account for low solids content.

2000/2001 Sampling Round - TtNUS
Sediment and Surface Soil

Parameter	Method Reference
Metals and Cyanide	RAS CLP ILM04.1 and DAS (ILM04.1-mod)
Hexavalent Chromium	DAS (SW-846 7196A)
Sulfides (sediment)	DAS (SW-846 9030B, 9034)

2.1.3.5 2001 Triad Sampling. Chem Tech performed the sediment analyses for organics and inorganics. USEPA CLP SOW OLM04.2 was used for the analyses organics (volatiles, semi-volatiles, and pesticide/ PCBs) and ILM04.1 for metals, inorganics, AVS/SEM, and TOC. Auqatec Biological Sciences conducted the biological analyses: Biotoxicity Method 100.1, 100.2 100.4 100.5 and Benthic Macroinvertebrates Assessments Method 10500D. TtNUS DAS methods, which are based on USEPA methods, were also used for the analysis.

For samples with low solids content, the samples were either oven-dried or the sample aliquot was increased prior to digestion/distillation in order to achieve the required quantitation limits and to avoid rejection of data as required by USEPA Region I data validation guidelines (USEPA, 1996a). Samples submitted for sulfide analyses did not undergo sample preparation modifications to account for low solids content.

2001 Triad Sampling Round - TtNUS
Sediment and Biota (Benthic Macroinvertebrates)

Parameter	Method Reference
Volatile Organics	SW- 5035/OLMO4.2 modified for sediments
Semi-volatile organics	CLP OLM04.2 modified version
Pesticides/PCBs	CLP OLM04.2 modified version
Metals	RAS CLP SOW ILM04.1 and DAS (ILM04.1-mod)
Total Organic Carbon	DAS (Lloyd Kahn Method)
Grain Size	DAS (ASTM D422-63)
AVS/SEM	DAS (AVS by Allen and Fu Method (December 1991), SEM by ILM04.1)
Toxicity	USEPA 100.1, 100.2, 100.4, 100.5
Benthic Macroinvertebrates	Standard Method 10500D

2.1.3.6 2002 Sampling Round. American Analytical and Technical Services conducted the metal analysis. Southwest Research Institute conducted the hexavalent chromium and total sulfide analyses and USEPA Region I Office of Environmental Measurement and Evaluation (OEME) conducted the PCB and TOC analyses.

For samples with low solids content, the samples were either oven-dried or the sample aliquot was increased prior to digestion/distillation in order to achieve the required quantitation limits and to avoid rejection of data as required by USEPA Region I data validation guidelines (USEPA, 1996a). Samples submitted for sulfide analyses did not undergo sample preparation modifications to account for low solids content.

2002 Sampling Round - USEPA Region I & TtNUS
Sediment and Surface Soil

Parameter	Method Reference
PCBs (selected sediments only)	SW-846 8082-modified
Hexavalent Chromium	SW-846 Method 3060A/7199
Metals	RAS CLP SOW ILM04.1
Total Organic Carbon	SW-846 9060 (modified for sediments)

2.1.3.7 Supplemental Sampling. For sediment (including sediment cores) and floodplain surface soil, USEPA Region I Office of Environmental Measurement and Evaluation (OEME) conducted the metals, hexavalent chromium, and TOC analyses. CompuChem in Cary, North Carolina performed the analyses for surface water. USEPA SW-846 methods were used for analysis of TOC and TSS. USEPA CLP method ILM04.1 was used for metals analysis. Method references by parameter type are included in the table below.

For sediment samples with low solids content, the samples were either oven-dried or the sample aliquot was increased prior to digestion/distillation in order to achieve the required quantitation limits and to avoid rejection of data as required by USEPA Region I data validation guidelines (USEPA, 1996a).

Supplemental Sampling - USEPA Region I & TtNUS
Surface Water, Sediment, Sediment Cores, and Surface Soil

Parameter	Method Reference
Metals-dissolved (surface water)	DAS CLP SOW ILM04.1
Metals-total (surface water)	DAS CLP SOW ILM04.1
Total Suspended Solids	SW846 160.2
Hexavalent Chromium	SW-846 Method 3060A/7199
Metals	RAS CLP SOW ILM04.1
Total Organic Carbon	SW-846 9060 (modified for sediments)

2.1.3.8 Groundwater Sampling. The groundwater samples were analyzed for VOCs, SVOCs, pesticides/PCBs, total metals, cyanide, perchlorate, and wet chemistry parameters (e.g., pH, dissolved oxygen, and conductivity). Only total metals data have been included because metals are the primary contaminants within the 38-acre wetland sediments. Samples were analyzed in accordance with TRC's February 2002 *Quality Assurance Project Plan for Pre-Design Investigation and Site Characterization* (TRC, 2002a).

2.1.4 Data Validation

The following subsections describe the data validation procedures used for the 1995, 1997, 1999, 2000/2001, and 2002 analytical data sets, the Triad sampling, and the supplemental data sets are summarized in data validation memoranda. The data validation process was consistent with USEPA Region 1 data validation guidance by following the most current (at the time of validation) USEPA Region I data validation guidance. Quality control (QC) samples were taken and submitted for laboratory analysis to monitor precision, accuracy and potential contamination throughout the sampling episode. These samples included trip blanks, equipment blanks, field duplicates, laboratory duplicates and matrix spike/matrix spike duplicate samples.

The data validation reports summarize the samples reviewed, quality control elements reviewed, any nonconformances with the established criteria and any validation actions (including data qualifiers). The data qualifiers used consist of the following:

- J - The associated numerical value is the approximate concentration for the analyte in the sample.
- UJ - The analyte was not detected above the sample reporting limit; however, the reporting limit is approximate.
- U - The sample was analyzed for, but was not detected above the reporting limit.
- R - The sample result was rejected due to serious quality control deficiencies. The presence or absence of the analyte cannot be verified.

Approximated and nonqualified results were used in further evaluations, but the qualified data were first reviewed to establish their usability.

2.1.4.1 1995 Sampling Round. Surface water, sediment and biota samples collected during the 1995 sampling round were validated in accordance with then current USEPA Region I data validation guidelines (USEPA, 1988 and USEPA, 1989c) by Foster Wheeler. These validations included both RAS analyses for sediment VOCs, SVOCs, pesticides, PCBs, and DAS analyses. The DAS analyses usually included stricter QC criteria such as lower detection limits, smaller recovery windows and more frequent QC sample analysis. All of the surface water and biota samples were analyzed under the DAS program, as were the sediment inorganic and classical chemistry analyses. The stricter DAS QC requirements are carried over into the data validation requirements and often result in more qualifications.

The low percent solids resulted in the rejection of some nondetect data in a number of sediment samples in accordance with USEPA Region I data validation guidelines (USEPA, 1996a). These sediment samples were analyzed under the RAS program where no modifications for correcting for low percent solids can be made.

2.1.4.2 1997 Sampling Round. All sediment analytical data generated from the supplemental field investigation were validated by M&E to USEPA Region I Tier III in accordance with USEPA Region I data validation guidelines (USEPA, 1996a), which were modified to include the QC criteria set forth in the DAS analytical specifications. Typical actions implemented during validation of analytical data are described in the *Supplemental Data Compendium for Wells G&H Superfund Site, Aberjona River Study (Operable Unit 3), Woburn, Massachusetts* (M&E, 1998; Appendix A.2).

The percent solids in each sediment sample was increased (the moisture content decreased) using field techniques and laboratory procedures specifically designed for this project. Only sediment samples collected for VOC and AVS/SEM analyses were analyzed at the "true" or unmanipulated moisture content found in the field.

2.1.4.3 1999 Sampling Round. Surface water and sediment samples collected during the 1999 sampling round were validated by New Environmental Horizons (NEH) according to data validation procedures similar to Tier II or Tier III. NEH's data validation (DV) report format does not adhere to USEPA Region I requirements, however, the technical content and approach is consistent with USEPA Region I DV guidelines. NEH used quality control criteria from three sources to assess the quality and useability of these data. The sources included: 1) the analytical method; 2) the site-specific QAPP; and 3) USEPA Region I DV guidelines. AVS/SEM data were validated according to NEH's DV procedures similar to Tier II, and the remaining data were validated according to NEH's DV procedures similar to Tier III. Freeze-drying of the sediment samples was successful and no data were rejected during validation due to low solid content. NEH did not reject VOC or AVS/SEM data for sediments due to low solid content since the results represent the best available methodology for generating these data.

NEH applied "B" qualifiers to data that were potentially affected by laboratory or field contamination. Prior to data evaluation, M&E replaced the "B" qualifiers with "J" qualifiers since these data are estimated and may be biased high due to potential laboratory and field contamination. Otherwise, the qualifiers used by NEH are consistent with the standard data validation qualifiers.

2.1.4.4 2000/2001 Sampling Round. Data for the sediment and soil samples collected during the 2000/2001 sampling round were validated by TtNUS to Tier III in accordance with USEPA Region I data validation guidelines (USEPA, 1989c, 1996a). Provisions were included in TtNUS's DAS methods for metals and cyanide in sediments to account for samples with low solids content, therefore, sediment sample results in this data set did not require rejection due to low solids content.

Hexavalent chromium data from sediments collected in February 2001 was questioned due to limitations of the colorimetric method (i.e. SW-846 Method 3060A/7196A) when analyzing matrices exhibiting reducing conditions as seen in the wetlands sediments. Consequently, all hexavalent chromium results, both positive and non-detects, were rejected with the exception of two samples. In June 2001, TtNUS re-sampled three locations in areas where elevated concentrations of hexavalent chromium were previously reported in the February 2001 sample round. The results of these samples indicated low concentrations of hexavalent chromium ranging from 5.98 mg/Kg to 16.8 mg/Kg (see January 9, 2002 Data Validation Memorandum prepared by TtNUS in Appendix C.4).

During 2001, USEPA and TtNUS evaluated alternative analytical methods for hexavalent chromium to overcome technical difficulties associated with matrix interferences and reducing conditions present in the wetland sediments. This evaluation included collecting duplicate samples and conducting parallel analysis using both the colorimetric and ion chromatography methods. USEPA and TtNUS determined that for these wetland sediments, the ion chromatography method (SW-846 Method 3060A/7199) was the best method to account for the potential matrix interferences and reducing conditions.

2.1.4.5 2001 Triad Sampling. Data for the sediment during the 2001 Triad sampling round were validated by TtNUS to Tier III for most analyses and Tier II for TOC in accordance with USEPA Region I data validation guidelines (USEPA, 1989c, 1996a). Provisions were included in TtNUS's DAS methods to account for samples with low solids content, therefore, sediment sample results in this data set did not require rejection due to low solids content. M&E performed a review of the data validation memoranda generated by TtNUS for this sampling round to verify, to the extent possible, that the data were qualified as specified in the data validation memoranda and in accordance with USEPA Region I data validation guidelines (USEPA, 1989a). This review did not reveal any discrepancies with Region I data validation procedures. However, in a few instances, M&E would have used a more conservative approach in flagging data.

2.1.4.6 2002 Sampling Round. Data for the sediment and soil sampling in the 2002 sampling rounds collected by the USEPA were validated by USEPA to Tier II in accordance with USEPA Region I data validation guidelines (USEPA, 1989c, 1996a, 2002). Since USEPA had conducted the data validation, an independent review of the data validation by Metcalf & Eddy was assumed to be unnecessary.

To support the risk assessment and further evaluate the presence of hexavalent chromium, in October 2002, TtNUS re-sampled sediments from six previously sampled areas that exhibited the highest total chromium concentrations and analyzed them for total chromium and hexavalent chromium using the ion chromatography method. This data was validated to Tier III criteria in accordance with USEPA Region I data validation guidelines (USEPA, 1989c, 1996a, 2002). The results for hexavalent chromium were from non-detect in five of the samples to 17.3 mg/Kg in the sample with the highest total chromium concentration (13,400 mg/Kg).

The data support TtNUS' opinion that it is unlikely that hexavalent chromium exists in the wetland sediments where elevated sulfide concentrations and reducing conditions are present or if the total chromium was present and concentrations were elevated, hexavalent chromium may exist, but at very low concentrations. Although this data was not included in the data tables in section 2, it was used to develop a ratio of hexavalent chromium to total chromium for the Human Health Risk Assessment presented in Section 3. See December 4, 2002 letter from TtNUS in Appendix C.4.

2.1.4.7 Supplemental Sampling. For the supplemental sampling, data for the sediment, sediment cores, and floodplain surface soil sampling collected by the USEPA and data for surface water sampling collected by TtNUS were validated by USEPA and TtNUS, respectively, to Tier II in accordance with USEPA Region I data validation guidelines (USEPA, 1989c, 1996a, 2002). An independent review of the data validation by Metcalf & Eddy was assumed to be unnecessary.

2.1.4.8 Groundwater Sampling. Groundwater data were validated as described in TRC's February 2002 *Quality Assurance Project Plan for Pre-Design Investigation and Site Characterization* (TRC, 2002a).

2.1.5 Data Treatment

This subsection discussed the use and treatment of the analytical data prior to use in the baseline human health and ecological risk assessments.

The following criteria were applied to the analytical data:

- If a value is not flagged, the value was used as reported (a detected value);
- If a value is flagged with "J", the value was used as reported (a detected value);
- If a value is flagged with "R" or "UR", the value was considered not to exist and was not used (a rejected value); and
- If the value is flagged with "U" or "UJ", the result was considered a nondetected (an undetected) value.

Prior to using analytical data for a primary sample with an associated field duplicate, the analytical values for the primary sample and the field duplicate were averaged together (USEPA, 1989a and 1989b) to provide a single set of values for the field duplicate pair. The following conventions were used for averaging field duplicate samples together:

- If both samples have detected values (flagged with "J" or unflagged), both values were averaged together. If one value or both values are flagged with "J" prior to averaging, the resulting averaged value was flagged with "J".
- If both samples have nondetected values (flagged with "U" or "UJ"), the lower value and its flag were used.

- If one sample has a nondetected value (flagged with "U" or "UJ") and the other sample has a detected value (flagged with "J" or unflagged) the following is done:

  - If the detected value is less than or equal to the nondetected value, the detected value and its flag were used; or

  - If the detected value is greater than the nondetected value, the detected value and 1/2 the nondetected value were averaged together. The resulting averaged value was flagged with "J".

- If one sample has a nonrejected value (flagged with "J", "U", "UJ" or unflagged) and one sample has a rejected value (flagged with "R" or "UR"), the nonrejected value and its flag were used.

The range of detection limits was determined based on the individual sample-specific detection limit (or sample quantitation limit; SQL) for each analyte. Because of sample dilution and/or sample weights, laboratory detection limits for individual samples can be higher than the method-specified detection limits. Minimum and maximum SQLs were determined for each analyte using each sample's SQL for all samples analyzed, regardless of whether the analyte was detected in any particular sample.

The frequency of detection is the number of samples with detected values per the number of samples analyzed. The number of samples with detected values was determined by totaling all samples with detected values (flagged with "J" or unflagged). The number of samples analyzed was determined by totaling all samples with detected or nondetected values (flagged with "U", "UJ", "J" or unflagged). Rejected values (flagged with "R" or "UR") were not included in the total number of samples analyzed. The mean of the field duplicate sample and corresponding sample was included when determining the number of samples analyzed and the number of detected values.

Arithmetic mean concentrations and Upper Confidence Limits (UCLs) were calculated using USEPA's ProUCL version 3.0 and included all detected values (flagged with "J" or unflagged) and ½ of the SQL for non-detected values (flagged with "U" or "UJ"). In some cases, the mean or UCL was greater than the maximum value because of high or widely varying detection limits, because a detected value is below the SQL (flagged with "J" on the laboratory report), or because a small data set was used. Detected values below the SQL are considered to be estimated concentrations, but are used in the risk assessments.

2.2 NATURE AND EXTENT OF CONTAMINATION

Previous studies of surface water, sediment and biota have been conducted for the study area (Section 1.2). Historical surface water data have indicated VOCs (e.g., trichloroethene (TCE), 1,2 dichloroethene (1,2-DCE), 1,1-dichloroethane and toluene) and inorganic (e.g., aluminum, arsenic, chromium, copper, iron and manganese) contamination. In addition, VOCs (e.g., acetone, 2-butanone, toluene and various chlorinated solvents), SVOCs (e.g., PAHs and phthalate esters), and inorganics (e.g., arsenic, chromium, copper, lead and zinc) were detected in historical sediment samples. Biota samples have contained aluminum, arsenic, cadmium, lead and mercury.

As previously described, field investigations of the Aberjona River Study area and reference areas were conducted most recently in 1995, 1997, 1999, 2000/2001, 2001 Triad, 2002, and the supplemental sampling rounds. Samples of surface water, sediment, surface soil, sediment cores, and biota were collected from the Aberjona River, wetlands associated with the Aberjona River, and Upper Mystic Lake as well as from local and regional reference stations. The nature and extent of contamination, by medium, for these sampling rounds are briefly discussed in this section. The discussion focuses on VOCs, SVOCs, pesticides, PCBs, and heavy metals. Results for essential nutrients (i.e., calcium, magnesium, potassium and sodium) have not been discussed. Results from the 1995 and 1997 investigations have been presented in Appendix A.1 and A.2 (Foster Wheeler, 1996; M&E, 1998) and analytical data from the rounds of sampling have been compiled and are presented in Tables B-1 through B-10 in Appendix B. Summarized analytical data tables for study area and reference samples, by medium, and data tables showing all detected compounds, by sample, are presented in Volume II (Section 2 Tables).

Section 2.2.7 provides a screening-level evaluation of groundwater metals data collected within the Wells G&H 38-acre wetland. As previously described, the groundwater screening will provide preliminary information on the potential impact of sediments contaminated with metals on groundwater quality. The groundwater-sediment interaction will be discussed more fully in the comprehensive RI Report.

2.2.1 Surface Water

Reference Stations. Surface water samples from ten reference stations were collected as part of investigational activities conducted for the study area. The reference data for surface water are presented in summarized form by station in Tables 2-3 through 2-12, in summarized form for all reference stations combined in Table 2-13 and by individual sample showing all detected compounds in Table 2-14. Eighteen inorganics, one SVOC, and one pesticide were detected in surface water. The SVOC, bis(2-ethylhexyl)phthalate, was detected at station 03-IP (sample SW-MC-03-01) at 3 Wg/L. The pesticide, gamma-chlordane, was detected in the reference wetland sample SW-24-01 at 0.0017 Wg/L. No VOCs were detected in reference surface water samples.

Relative to other reference locations, the maximum concentration of several inorganics (aluminum, arsenic, barium, cobalt, copper, iron, lead, manganese, nickel, vanadium and zinc) were measured at river reference sample SW-MC-01, and mercury was detected in the highest concentration at wetland reference sample (0.13 Wg/L, SW-24-01; Table 2-13).

Study Area Stations. The study area data for the twenty one surface water stations are presented in summarized form by station in Tables 2-15 through 2-31, in summarized form for all study area stations combined in Table 2-32 and by individual sample showing all detected compounds in Table 2-33. Four VOCs were detected at relatively low concentrations (i.e., below 5 µg/L) from study area surface water samples. cis-1,2-Dichloroethene (cis-1,2-DCE) and TCE were detected at the Cranberry Bog Conservation Area (sample SW-09-01) and in the Cranberry Bog wetlands (sample SW-16-01). In addition, chloroform and tetrachloroethene (PCE) were present in a sample from the lower portion of the Aberjona River (SW-05-01), and cis-1,2-DCE was identified in an upper Aberjona River sample (SW-11-01).

No SVOCs were detected in the surface water samples, and five pesticides (i.e., heptachlor epoxide, endosulfan II, methoxychlor, and alpha- and gamma-chlordane) were present in at least one of three Aberjona River locations (see table below). These locations were SW-11-01 (Aberjona River near Well G), SW-12-01 (Aberjona River near Well H), and SW-13-01 (Aberjona River wetland near Well H).

Elevated concentrations in comparison to reference samples were detected for various inorganics in river and lake samples. Sample SW-10-01 (Aberjona River near Salem Street Bridge) contained elevated levels of nine inorganics in comparison to river reference sample SW-MC-01. These analytes were antimony (1.5 µg/L), arsenic (77.1 µg/L), beryllium (0.18 µg/L), cadmium (2.5 µg/L), chromium (146 µg/L), copper (111 µg/L), lead (75.9 µg/L), mercury (0.88 µg/L), and zinc (626 µg/L). Sample SW-10-01 also had the highest TSS concentration, which may account for the high concentrations of inorganics detected in the sample. In addition, elevated levels of arsenic, chromium, copper, and zinc were detected in some of the other surface water samples from the Aberjona River, associated wetlands, and Upper Mystic Lake in comparison to river, wetland and lake reference samples (see table below). Selected metals of potential interest are presented in Figure 2-21.

Surface Water

Analyte	Study Area Locations			Reference Locations		
	Range of Detects	**Detection Frequency**	**Average Concentration***	**Range of Detects**	**Detection Frequency**	**Average Concentration***
<u>VOCs</u> (µg/L)						
cis-1,2-DCE	1 - 4	3 / 16	0.91	ND	0 / 11	0.77
TCE	2 - 2	2 / 16	0.69	ND	0 / 11	0.77

Surface Water

Analyte	Study Area Locations			Reference Locations		
	Range of Detects	Detection Frequency	Average Concentration*	Range of Detects	Detection Frequency	Average Concentration*
PCE	2	1 / 16	0.59	ND	0 / 11	0.77
Chloroform	1	1 / 16	0.53	ND	0 / 11	0.77
<u>Pesticides</u> (μg/L)						
Heptachlor Epoxide	0.0011 - 0.0019	3 / 17	0.021	ND	0 / 11	0.014
Endosulfan II	0.0015	1 / 17	0.047	ND	0 / 11	0.025
Methoxychlor	0.015	1 / 17	0.24	ND	0 / 11	0.13
alpha-Chlordane	0.0021	1 / 17	0.024	ND	0 / 11	0.014
gamma-Chlordane	0.0014	1 / 17	0.024	0.0017	1 / 11	0.012
<u>Metals</u> (μg/L)						
Antimony	1.3 - 1.5	2 / 17	0.68	ND	0 / 11	5.2
Arsenic	2.8 - 77.1	13 / 17	14	1.1 - 15.7	8 / 11	3.1
Beryllium	0.043 - 0.18	3 / 17	0.051	ND	0 / 11	0.083
Cadmium	0.16 - 2.5	3 / 17	0.30	ND	0 / 11	0.25
Chromium	0.72 - 146	17 / 17	13	0.34 - 9	4 / 11	3.0
Copper	1.7 - 111	17 / 17	12	0.45 - 13.8	8 / 11	2.6
Lead	0.94 - 75.9	11 / 17	7.4	0.665 - 51.4	8 / 11	6.9
Mercury	0.04 - 0.88	8 / 17	0.15	0.097 - 0.13	4 / 11	0.058

Surface Water

Analyte	Study Area Locations			Reference Locations		
	Range of Detects	Detection Frequency	Average Concentration*	Range of Detects	Detection Frequency	Average Concentration*
Zinc	10.8 - 626	16 / 17	72	1.375 - 71.7	7 / 11	13

ND - not detected

* - Average concentrations include values of half the detection limit for all non-detects.

2.2.2 Sediment

Reference Stations. Reference sediment samples were also collected as part of investigational activities conducted for the study area. The reference data for sediment are presented in summarized form by station in Tables 2-34 through 2-45, in summarized form for all reference stations combined in Table 2-46, by station showing individual samples and all detected compounds in Tables 2-47 through 2-58 and by individual samples (all detected compounds) for all stations combined in Table 2-59. Most of the analytes detected within the study area were also detected in sediment collected from the reference stations. Twenty-three inorganics, eight VOCs, twenty-five SVOCs, eighteen pesticides, and two PCBs were detected in sediment. The maximum concentration of four volatile compounds were found at SD-MC-03-TR. SVOCs detected were primarily PAHs and phthalates. Reference station 24 had the maximum levels of eighteeen of the twenty-five SVOCs detected in study area or reference sediments. In addition, seven of the eighteen pesticides detected in the reference samples were present at maximum levels exceeding those detected in study area samples. The two PCBs, Aroclor-1248 and -1260, were detected at levels of 290 µg/kg and 200 µg/kg, respectively, at station 24.

Study Area Stations. The study area data for sediment are presented in summarized form by station in Tables 2-60 through 2-109, in summarized form for all study area stations combined in Table 2-

110, by station showing individual sample and all detected compounds in Tables 2-111 through 2-160 and by individual sample (all detected compounds) for all stations combined in Table 2-161. Chlorinated VOCs (including vinyl chloride, methylene chloride, 1,1-dichloroethane (1,1 DCA), 1,1,2-trichoro-1,2,2-trifluoroethane, 1,1,1-trichloroethane, cis-1,2-DCE, trans-1,2-DCE, TCE, and PCE) were detected in study area sediments during the field investigations. Detected levels for these compounds are presented in the table below. The more elevated levels were present at station 20 (wetland associated with western channel of the upper Aberjona River), with TCE at 2,025 µg/kg, and station 22 (wooded wetland associated with the upper Aberjona River), with PCE at 3,164 µg/kg. In addition, benzene, ethylbenzene, naphthalene, m- and p-xylene, methyl acetate, acetone, carbon disulfide, 2-butanone, and toluene were detected. A summary of the volatile organic compounds detected by organic group within the study area for the 1995, 1997, and 2001 Triad sampling rounds are presented on Figure 2-22.

SVOCs were detected in numerous locations, with most of the compounds identified being PAHs and phthalate esters. Sediment samples from station 07, located in Davidson Park, contained the maximum levels for seventeen of the twenty-four SVOCs detected in the sediments. Generally, the SVOC levels detected in river, wetland, and lake sediments were within an order of magnitude of those present at reference stations. PAHs were the most frequently detected organics and were measured at higher levels. Also, the highest detected levels tended to occur with the higher molecular weight PAHs.

Pesticides were frequently detected and distributed throughout the sampling stations, including reference stations. A total of 20 pesticides were detected, with gamma-chlordane having the highest level (650 µg/kg at station 13, wetland associated with side channel of the Aberjona River). DDT, DDD, DDE, and alpha- and gamma-chlordane were the predominant pesticides measured in terms of frequency of detection and concentrations. In addition, Aroclor-1260 was detected in 38 of the 105 sediment samples. Levels of Aroclor-1260 ranged from 11 µg/kg (SD-19-01-ME, Aberjona River near Well G) to 2,400 µg/kg (SD-JY-07), (just north of Salem Street). Aroclor-1248 was detected in sediment samples with concentrations ranging from 8.1 µg/kg at station SD-22-02-ME to 560 µg/kg at SD-10-01-ME. PAHs, pesticides, and PCBs detected in 1995, 1997, and 2001 Triad samples are summarized on Figure 2-23.

As in the previous investigations, numerous metals were detected in sediments, with many of these detections at levels greater than reference sample levels. Arsenic, a historical contaminant in the study area, was present in 99 percent of the samples with a maximum level of 4,550 mg/kg (SD-12-03-ME, Aberjona River near Well H). Other inorganics detected include: antimony, cadmium, chromium, copper, mercury, and zinc (see table below). Generally, higher concentration of metals were detected in the Wells G&H 38-acre wetland. The average chromium concentration was 1,016 mg/kg, however in the Wildwood area, concentrations ranged from 3,670 mg/kg to 24,600 mg/kg. Lead was generally found in similar concentrations throughout the study area with an average of 512 mg/kg, however a maximum concentration of 41,000 mg/kg was measured at TT-22-01 near the rifle range. Although the average mercury concentration was <3 mg/kg, a maximum level of 89.2 mg/kg was measured at TT-30-03. The next highest concentration of mercury was 44.8 mg/kg at Station 12. Selected metals detected in the 1995, 1997, 2000/2001, 2001 Triad, and 2002 sampling rounds are presented on Figure 2-24. Average concentrations of arsenic, chromium, lead, and mercury by station are presented in Figures 2-25 through 2-29.

Sediment

Analyte	Study Area Locations			Reference Locations		
	Range of Detects	**Detection Frequency**	**Average Concentration ***	**Range of Detects**	**Detection Frequency**	**Average Concentration ***
<u>VOCs</u> (µg/kg)						
Vinyl Chloride	2 - 255	2 / 87	34	ND	0 / 24	9.7
Methylene Chloride	28 - 100	2 / 101	38	ND	0 / 26	16
1,1-DCA	3	1 / 87	32	ND	0 / 24	9.7

Sediment

Analyte	Study Area Locations			Reference Locations		
	Range of Detects	Detection Frequency	Average Concentration *	Range of Detects	Detection Frequency	Average Concentration *
1,1,2-trichloro-1,2,2-trifluoroethane	37.5	1 / 5	21	52	1 / 5	24
cis-1,2-DCE	7 - 562	18 / 28	78	ND	0 / 10	9.3
Acetone	58 - 7300	22 / 81	317	23 - 2300	12 / 24	185
Benzene	6 - 22	8 / 87	10	4	1 / 24	7.9
Carbon Disulfide	18 - 29	3 / 63	9.2	3 - 3	2 / 22	7.9
Ethylbenzene	5 - 9	4 / 87	10	ND	0 / 24	7.8
PCE	41-3164	5 / 90	82	ND	0 / 24	9.7
Toluene	3 - 22	3 / 63	9.0	7.9 - 73	3 / 24	12
TCE	6 - 2025	18 / 91	51	ND	0 / 24	8.2
Xylenes m,p	10 - 25	2 / 28	28	ND	0 / 10	9.3
SVOCs (µg/kg)						
Benzo(a)anthracene	40 - 9600	85 / 109	1033	110 - 5900	18 / 26	1151
Benzo(a)pyrene	33 - 10000	84 / 106	1093	130 - 5500	18 / 26	1117
Benzo(g,h,i)perylene	44 - 5300	62 / 103	636	190 - 2200	11 / 26	511
Benzo(k)fluoranthene	45 - 14000	84 / 109	1200	400 - 9600	16 / 26	1420
Chrysene	40 - 10000	88 / 110	1307	140 - 7300	18 / 26	1389

Sediment

Analyte	Study Area Locations			Reference Locations		
	Range of Detects	Detection Frequency	Average Concentration *	Range of Detects	Detection Frequency	Average Concentration *
Fluoranthene	42 - 23000	102 / 116	2258	71 - 15000	24 / 26	2547
<u>Pesticides/PCBs</u> (µg/kg)						
gamma-Chlordane	0.12 - 650	66 /106	14	0.31 - 4.6	6 / 26	2.9
DDT	0.29 - 47	62 / 104	6.7	2.1 - 180	16 / 26	22
DDE	0.089 - 160	91 / 115	20	1.6 - 470	19 / 26	30
DDD	0.46 - 310	74 / 111	27	1.1 - 390	20 / 26	48
Aroclor 1248	8.1 - 560	26 / 103	68	56 - 290	2 / 26	65
Aroclor 1260	11 - 2400	38 / 105	155	47 - 200	2 / 26	61
<u>Metals</u> (mg/kg)						
Antimony	0.325 - 329	190 / 289	6.6	0.5 - 5.6	20 / 27	1.0
Arsenic	2.4 - 4550	324 / 325	245	2.5 - 44.5	27 / 27	17
Cadmium	0.045 - 37.7	293 / 326	6.3	0.087 - 6.1	20 / 27	1.3
Chromium	3.3 - 24600	327 / 327	1016	8.9 - 512	27 / 27	89

Sediment

Analyte	Study Area Locations			Reference Locations		
	Range of Detects	Detection Frequency	Average Concentration *	Range of Detects	Detection Frequency	Average Concentration *
Copper	4.2 - 3760	327 / 327	365	1.9 - 344	27 / 27	63
Lead	1.9 - 41000	327 / 327	512	5.6 - 755	27 / 27	194
Mercury	0.05 - 89.2	285 / 316	2.1	0.021 - 0.71	19 / 26	0.19
Zinc	15.2 - 8750	322 / 327	1250	10.4 - 645	27 / 27	195

ND - not detected

* - Average concentrations include values of half the detection limit for all non-detects.

2.2.3 Surface Soil

Study Area. Sixty-one surface soil samples were collected including ten samples at station CB-05, five samples at Station DA (Danielson Park), twenty six samples collected at Station DP (Davidson Park), ten samples at station KF (Kraft Foods), five samples at station NR (Normac Road), and five samples at station WSS (Salem Street) are presented in summarized form in Table 2-168, summarized by individual station in Tables 2-162 through 2-167, all detected compounds by station and individual sample in Tables 2-169 through 2-174, and detected compounds at all locations in Table 2-175. Samples were analyzed for metals only. Numerous metals were detected at concentrations similar to sediment samples collected at stations CB, 09 and 16, 07, KFSE, NRSE, and WS (Tables 2-84 to 2-89, 2-68 and 2-75, 2-66, 2-91, 2-96, and 2-108). Fifteen metals were detected at more than 80% of the soil locations with the highest concentrations of aluminum, arsenic, beryllium, cadmium, chromium, cobalt, copper, lead, manganese, mercury, and nickel detected at SO-DA-03 south of

Danielson Park (Figure 2-4). Also, the highest concentrations of iron, selenium, thallium, and zinc were detected at SO-DP-16 located near the entrance of Davidson Park (see Figure 2-7). Although locations such as SO-DA-03, SO-DP-16, SO-DP-26, and SO-NR-20 exhibited higher concentrations of most metals, similar concentrations of metals were generally found in soil within stations. Selected metals detected in the 2000/2001, and 2002 sampling rounds are presented on Figure 2-30. Concentrations of arsenic, chromium, lead, and mercury for soil and sediment samples collected at Normac Road (NRSO/NRSE), Salem Street (WSS/WS), soil CB-05 and nearby sediment CB-03, Davidson Park (DP/07), and Kraft Foods (KFSO/KFSE) are presented in the table below. This comparison of surface soil to sediment concentrations for these metals indicated generally higher concentrations of these metals in sediments relative to upland soils collected at the same station.

Surface Soil

Analyte mg/kg	Soil Stations			Sediment Stations		
	Range of Detects	Detection Frequency	Average Concentration*	Range of Detects	Detection Frequency	Average Concentration*
	NRSO			**NRSE**		
Arsenic	39.6 - 266	5 / 5	111	106 -221	5 / 5	166
Chromium	40 - 228	5 / 5	98	104 - 258	5 / 5	186
Lead	35.6 - 187	5 / 5	85	80.8 - 249	5 / 5	161
Mercury	0.38 - 2	5 / 5	0.8	0.77 - 5.9	5 / 5	2.63
	WSS			**WS**		
Arsenic	8.1 - 10.9	5 / 5	9	17.7 - 339	10 / 10	168
Chromium	9.3 - 19.7	5 / 5	13	67.7 - 1320	10 / 10	504
Lead	55.5 - 93.3	5 / 5	68	165 - 490	10 / 10	295
Mercury	0.09 - 0.13	5 / 5	0.10	0.22 - 1.8	8 / 8	0.98
	CB-05			**CB-03**		
Arsenic	16.8 - 86.3	10 / 10	30	9.1 - 1410	12 / 12	272
Chromium	13.4 - 211	10 / 10	56	38.7 - 768	12 / 12	457
Lead	28.9 - 90.9	10 / 10	50	24.4 - 443	12 / 12	196
Mercury	0.07 - 0.6	10 / 10	0.2	0.058 - 3.6	12 / 12	1.2

Surface Soil

Analyte mg/kg	Soil Stations			Sediment Stations		
	Range of Detects	Detection Frequency	Average Concentration*	Range of Detects	Detection Frequency	Average Concentration*
		DP			**07**	
Arsenic	6.05 - 219	26 / 26	33	10.3 - 129	13 / 13	62
Chromium	24.4 - 316	26 / 26	110	31.9 - 442	13 / 13	184
Lead	25 - 261	26 / 26	119	41.2 - 480	13 / 13	267
Mercury	0.22 - 2.35	21 / 26	0.77	0.28 - 5.7	12 / 13	1.4
		KFSO			**KFSE**	
Arsenic	19.9 - 54.55	10 / 10	40	11.7 - 90.4	10 / 10	38
Chromium	48.4 - 141	10 / 10	102	20.8 - 113	10 / 10	68
Lead	86.5 - 265.5	10 / 10	160	31.6 - 188	10 / 10	98
Mercury	0.27 - 0.855	10 / 10	0.6	0.1 - 1.6	10 / 10	0.51

units - mg/kg

* - Average concentrations include values of half the detection limit for all non-detects.

2.2.4 Crayfish and Fish Tissue

Reference Stations. Reference samples for crayfish and fish tissue were also collected as part of investigational activities conducted at the study area. The reference samples were collected from the same water bodies as the surface water and sediment reference samples. The data for crayfish tissue are presented in summarized form for reference stations in Table 2-176 and by detected compounds in Table 2-177. Small fish tissue are presented in summarized form for reference stations in Table 2-184 and by detected compounds in Table 2-185. Large fish fillet are presented in summarized form for reference stations in Table 2-194 and by detected compounds in Table 2-195. Large fish offal are presented in summarized form for reference stations in Table 2-202 and by detected compounds in

Table 2-203. Large fish whole body are presented in summarized form for reference stations in Table 2-210 and by detected compounds in Table 2-211. All contaminants detected in fish were either pesticides, PCBs or inorganics, with one exception. The PAH, benzo(g,h,i)perylene, was detected at 400 µg/kg in one fillet sample (LF-LB-08-F). The greatest number of contaminants were detected in the whole large fish.

Detected compounds in reference crayfish and fish tissue included: 4,4'-DDD, 4,4'-DDE, some 4,4'-DDT, and Aroclor-1260 for organic compounds and aluminum, arsenic, barium, cadmium, chromium, cobalt, copper, iron, manganese, selenium, silver, and zinc for inorganics.

Study Area Stations. The study area data for crayfish tissue samples are presented in summarized form by reach in Tables 2-178 through 2-181 and in summarized form for all study area samples combined in Table 2-182 and by individual sample (all detected compounds) for all reaches combined in Table 2-183.

The study area data for small fish tissue samples are presented in summarized form by reach in Tables 2-186 through 2-191 and in summarized form for all study area samples combined in Table 2-192 and by individual sample (all detected compounds) for all reaches combined in Table 2-193.

The study area data for large fish fillet samples are presented in summarized form by reach in Tables 2-196 through 2-199 and in summarized form for all study area samples combined in Table 2-200 and by individual sample (all detected compounds) for all reaches combined in Table 2-201.

The study area data for large fish offal tissue samples are presented in summarized form by reach in Tables 2-204 through 2-207 and in summarized form for all study area samples combined in Table 2-208 and by individual sample (all detected compounds) for all reaches combined in Table 2-209.

The study area data for large fish whole body tissue samples are presented in summarized form by reach in Tables 2-212 through 2-215 and in summarized form for all study area samples combined in

Table 2-216 and by individual sample (all detected compounds) for all reaches combined in Table 2-217.

Fish tissues samples were not analyzed for VOCs. No SVOCs were detected with the exception of diethylphthalate, which was detected in one large fish sample. Many pesticides, PCBs, and inorganics were frequently detected in large fish samples.

A number of contaminants were detected in crayfish tissue collected within the study area. Fourteen pesticides and two PCBs were detected in crayfish tissue collected within the study area. 4,4'-DDT, endrin aldehyde, and Aroclor-1260 were detected in all crayfish samples. The range of detected concentrations for 4,4'-DDD, 4,4'-DDE, dieldrin, alpha-chlordane, gamma-chlordane, Aroclor-1254, and Aroclor-1260 are presented in the table below.

A number of contaminants were detected in small fish tissue collected within the study area. Sixteen pesticides and three PCBs were detected in small fish tissue collected within the study area. 4,4'-DDD, 4,4'-DDE, alpha-chlordane, dieldrin, gamma-chlordane, Aroclor-1254, and Aroclor-1260 were detected in at least 90% of the small fish samples. The range of detected concentrations for 4,4'-DDD, 4,4'-DDE, dieldrin, alpha-chlordane, gamma-chlordane, Aroclor-1254, and Aroclor-1260 are presented in the table below.

A number of contaminants were detected in large fish fillet tissue collected within the study area. Seventeen pesticides and three PCBs were detected in large fish tissue collected within the study area. 4,4'-DDD, 4,4'-DDE, gamma-chlordane, Aroclor-1254, and Aroclor-1260 were detected in over 88% of the large fish fillet tissue samples. The range of detected concentrations for 4,4'-DDD, 4,4'-DDE, dieldrin, alpha-chlordane, gamma-chlordane, Aroclor-1254, and Aroclor-1260 are presented in the table below.

A number of contaminants were detected in large fish offal tissue collected within the study area. Sixteen pesticides and three PCBs were detected in large fish offal tissue collected within the study area. 4,4'-DDD, 4,4'-DDE, alpha-chlordane, gamma chlordane, and Aroclor-1254, and Aroclor-1260 were detected in at least 95% of the large fish offal samples. The range of detected concentrations for 4,4'-DDD, 4,4'-DDE, dieldrin, alpha-chlordane, gamma-chlordane, Aroclor-1254, and Aroclor-1260 are presented in the table below.

A number of contaminants were detected in large fish whole body tissue collected within the study area. Sixteen pesticides and two PCBs were detected in large fish whole body tissue collected within the study area. 4,4'-DDD, 4,4'-DDE, 4,4'-DDT, alpha-chlordane, dieldrin, gamma chlordane, and Aroclor-1254, and Aroclor-1260 were detected in over 90% of the large fish whole body samples. The range of detected concentrations for 4,4'-DDD, 4,4'-DDE, dieldrin, alpha-chlordane, gamma-chlordane, Aroclor-1254, and Aroclor-1260 are presented in the table below.

Crayfish & Fish Tissue

Analyte	**Study Area Locations**			**Reference Locations**		
	Range of Detects	**Detection Frequency**	**Average Concentration***	**Range of Detects**	**Detection Frequency**	**Average Concentration***
Crayfish:						
<u>Pesticides/PCBs</u> (µg/kg)						
alpha-Chlordane	0.52 - 1.8	6 / 8	0.93	ND	0 / 2	0.25
gamma-Chlordane	0.25 - 0.83	5 / 8	0.40	ND	0 / 2	0.25
DDD	1.6 - 4.1	6 / 8	2.4	0.63 - 0.94	2 / 2	0.79
DDE	4.6 - 8.2	6 / 8	5.0	4 - 5.3	2 / 2	4.7
Dieldrin	0.81 - 2.4	7 / 8	1.4	ND	0 / 2	0.49
Aroclor 1254	12 - 24	7 / 8	16	ND	0 / 2	2.5
Aroclor 1260	10.55 - 41	8 / 8	26	5.1 - 5.2	2 / 2	5.2
<u>Metals</u> (mg/kg)						

Crayfish & Fish Tissue

Analyte	Study Area Locations			Reference Locations		
	Range of Detects	Detection Frequency	Average Concentration*	Range of Detects	Detection Frequency	Average Concentration*
Arsenic	1.1 - 4.4	7 / 8	2.2	ND	0 / 2	0.19
Cadmium	0.043 - 0.16	7 / 7	0.070	0.07 - 0.09	2 / 2	0.080
Chromium	0.2 - 2.5	8 / 8	1.1	0.18	1 / 1	0.18
Lead	0.212 - 1.5	8 / 8	0.78	ND	0 / 2	0.033
Mercury	0.02	1 / 8	0.0092	ND	0 / 2	0.006
Zinc	18.35 - 29.1	8 / 8	25	18 - 18.3	2 / 2	18
Small Fish:						
<u>Pesticides/PCBs</u> (µg/kg)						
alpha-Chlordane	3.9 - 42	28 / 30	18	0.33 - 1.3	4 / 6	0.80
gamma-Chlordane	2 - 20	29 / 30	8.2	ND	0 / 6	0.65
DDD	4.2 - 45	29 / 30	24	0.82 - 3.2	5 / 6	2.0
DDE	9 - 59	27/30	30	5.5 - 24	6 / 6	13
Dieldrin	2.5 - 39	28 / 30	11	ND	0 / 6	1.3
Aroclor 1254	41 - 160	28 / 30	89	ND	0 / 6	6.5
Aroclor 1260	30 - 110	30 / 30	68	9.2 - 33	6 / 6	19
<u>Metals</u> (mg/kg)						
Arsenic	0.59 - 1.4	3 / 33	0.22	ND	0 / 6	0.049
Cadmium	0.013 - 0.084	31 / 32	0.028	0.017 - 0.018	3 / 6	0.012
Chromium	0.21 - 0.66	32 / 33	0.34	0.3 - 0.35	4 / 6	0.26

Crayfish & Fish Tissue

Analyte	Study Area Locations			Reference Locations		
	Range of Detects	Detection Frequency	Average Concentration*	Range of Detects	Detection Frequency	Average Concentration*
Lead	0.071 - 0.68	10 / 33	0.13	ND	0 / 6	0.035
Mercury	0.015 - 0.054	17 / 33	0.021	0.062 - 0.098	4 / 6	0.059
Zinc	21 - 42	33 / 33	26	13.9 - 38.3	6 / 6	28
Large Fish Fillet:						
<u>Pesticides/PCBs</u> (µg/kg)						
alpha-Chlordane	1.3 - 11	21 / 26	3.5	0.29 - 0.33	3 / 13	0.47
gamma-Chlordane	0.56 - 5.1	25 / 26	1.8	ND	0 / 13	0.46
DDD	2.1 - 27	23 / 26	7.1	6.4	1 / 13	1.2
DDE	3.8 - 120	24 / 26	24	2.8 - 32	12 / 12	9.1
Dieldrin	0.7 - 2.9	19 / 26	1.7	ND	0 / 13	0.88
Aroclor 1254	9.6 - 180	24 / 26	44	ND	0 / 13	4.5
Aroclor 1260	14 - 315	26 / 26	52	5.1 - 130	13 / 13	22
<u>Metals</u> (mg/kg)						
Arsenic	0.17	1 / 26	0.070	0.0219 - 0.0806	16 / 29	0.044
Cadmium	0.012 - 0.023	12 / 26	0.011	ND	0 / 29	0.13
Chromium	0.053 - 0.16	18 / 26	0.069	0.057 - 0.81	11 / 29	0.32
Lead	0.062 - 2.3	4 / 26	0.13	0.059 - 0.061	3 / 29	0.28

Crayfish & Fish Tissue

Analyte	Study Area Locations			Reference Locations		
	Range of Detects	Detection Frequency	Average Concentration*	Range of Detects	Detection Frequency	Average Concentration*
Mercury	0.018 - 0.58	22 / 26	0.16	0.1 - 1	25 / 29	0.31
Zinc	4.8 - 7.9	24 / 26	6.0	3.3 - 8.3	25 / 29	5.1

Crayfish & Fish Tissue

Analyte	Study Area Locations			Reference Locations		
	Range of Detects	Detection Frequency	Average Concentration*	Range of Detects	Detection Frequency	Average Concentration*
Large Fish Offal:						
<u>Pesticides/PCBs</u> (µg/kg)						
alpha-Chlordane	3.9 - 81	25 / 26	27	1.4 - 4.9	4 / 13	2.5
gamma-Chlordane	2.6 - 36	25 / 26	11	ND	0 / 13	2.1
DDD	6.2 - 150	25 / 26	55	3 - 11	7 / 13	5.6
DDE	8.9 - 540	26 / 26	173	22 - 110	12 / 13	51
Dieldrin	2 - 25	22 / 26	9.9	ND	0 / 13	4.1
Aroclor 1254	29 - 770	25 / 26	274	ND	0 / 13	20
Aroclor 1260	37 - 720	26 / 26	297	33 - 230	12 / 13	100
<u>Metals</u> (mg/kg)						
Arsenic	0.096 - 2.5	5 / 26	0.28	ND	0 / 13	0.045
Cadmium	0.014 - 0.12	23 / 25	0.035	0.014 - 0.04	10 / 13	0.019
Chromium	0.24 - 2.6	26 / 26	0.64	0.2 - 0.38	10 / 13	0.29
Lead	0.066 - 3.2	9 / 26	0.37	1.3	1 / 13	0.14
Mercury	0.0092 - 0.33	22 / 26	0.085	0.091 - 0.6	10 / 13	0.26
Zinc	17.3 - 37.6	26 / 26	27	16.1 - 23.9	13 / 13	20
Large Fish Whole Body:						
<u>Pesticides/PCBs</u> (µg/kg)						
alpha-Chlordane	16 - 110	15 / 15	35	0.63 - 29	12 / 13	7.5

Crayfish & Fish Tissue

Analyte	Study Area Locations			Reference Locations		
	Range of Detects	Detection Frequency	Average Concentration*	Range of Detects	Detection Frequency	Average Concentration*
gamma-Chlordane	7.9 - 26	15 / 15	16	0.62 - 16	8 / 13	4.8
DDD	19 - 360	15 / 15	75	4.3 - 150	13 / 13	32
DDE	20 - 820	15 / 15	148	16 - 280	13 / 13	86
Dieldrin	5.9 - 59	14 / 15	15	2.8	1 / 13	3.0
Aroclor 1254	69 - 260	14 / 15	140	72 - 250	5 / 13	55
Aroclor 1260	33 - 1100	15 / 15	256	25 - 520	13 / 13	140
<u>Metals</u> (mg/kg)						
Arsenic	ND	0 / 15	0.059	ND	0 / 13	0.048
Cadmium	0.011 - 0.054	12 / 15	0.021	0.014 - 0.56	9 / 13	0.065
Chromium	0.1 - 0.39	11 / 15	0.20	0.086 - 0.3	5 / 13	0.12
Lead	0.1 - 1	8 / 15	0.18	0.19 - 0.42	4 / 13	0.15
Mercury	0.023 - 0.13	10 / 15	0.042	0.11 - 0.59	5 / 13	0.15
Zinc	18.6 - 43.6	15 / 15	26	17.5 - 36.3	13 / 13	24

ND - not detected

* - Average concentrations include values of half the detection limit for all non-detects.

2.2.5 Plant Tissue

Reference Stations. The reference data for plant tissue are presented in summarized form for all reference stations combined in Table 2-218, and by individual sample (all detected compounds) for all stations combined in Table 2-219. SVOCs and PCBs were not detected in the reference plant sample. Pesticides detected included heptachlor, 4,4'-DDE, 4,4'-DDD, 4,4'-DDT and alpha-chlordane. Detected levels were, in general, low (i.e., only slightly above the detection limits). Inorganics detected in the reference plant sample included aluminum, arsenic, barium, cadmium, chromium, cobalt, copper, iron, lead, manganese, nickel, vanadium, and zinc.

Study Area Stations. The study area data for plant tissue are presented in summarized form for all study area stations combined in Table 2-220, and by individual sample (all detected compounds) for all stations combined in Table 2-221. Over 30 contaminants were detected in plant tissue collected from the study area. Higher concentrations of contaminants were detected in study area samples relative to reference samples (see table below). Among the eight PAHs detected, maximum detected concentrations for individual compounds ranged from 38 µg/kg for phenanthrene to 120 µg/kg for benzo(k)fluoranthene and fluoranthene. Eleven pesticides and one PCB were detected in plant tissue collected within the study area. Maximum detected concentrations were below 2.0 µg/kg for all compounds except alpha-chlordane, endrin aldehyde, and Aroclor-1260 (see table below).

Nineteen inorganics were detected with 100% detection frequency for all except selenium, which was detected in one out of six samples. Antimony, beryllium, cyanide, silver, and thallium were not detected in any of the samples. The range of detected and average concentrations for aluminum, arsenic, lead, mercury, and zinc compared to reference locations is presented in the table below.

Plant

Analyte	Study Area Locations			Reference Locations		
	Range of Detects	Detection Frequency	Average Concentration*	Range of Detects	Detection Frequency	Average Concentration*
<u>SVOCs</u> (µg/kg)						
Benzo(b)fluoranthene	55 - 100	4 / 6	103	ND	0 / 2	165
Benzo(k)fluoranthene	52 - 120	3 / 6	123	ND	0 / 2	165

Plant

Analyte	Study Area Locations			Reference Locations		
	Range of Detects	Detection Frequency	Average Concentration*	Range of Detects	Detection Frequency	Average Concentration*
Chrysene	54 - 98	3 / 6	119	ND	0 / 2	165
Fluoranthene	31 - 120	6 / 6	54	ND	0 / 2	165
Phenanthrene	25 - 38	2 / 6	121	ND	0 / 2	165
Pyrene	33 - 110	5 / 6	87	ND	0 / 2	165
<u>Pesticides/PCBs</u> (µg/kg)						
alpha-Chlordane	0.84 - 2.3	6 / 6	1.4	0.43 - 0.45	2 / 2	0.44
gamma-Chlordane	0.32 - 0.56	6 / 6	0.47	ND	0 / 2	0.25
DDD	0.58 - 1.1	3 / 6	0.63	4.7 - 5.2	2 / 2	5.0
DDE	0.36 - 1.2	6 / 6	0.70	0.85 - 0.87	2 / 2	0.86
Endrin Aldehyde	0.63 - 2.9	4 / 6	1.0	ND	0 / 2	0.49
Aroclor 1260	5.6 - 8.9	6 / 6	7.0	ND	0 / 2	2.5
<u>Metals</u> (mg/kg)						
Aluminum	154 - 348	6 / 6	256	69.3 - 83.9	2 / 2	77
Arsenic	2.9 - 15.9	6 / 6	10	1.4 - 1.5	2 / 2	1.5
Lead	9.4 - 13.9	6 / 6	12	3 - 3.3	2 / 2	3.2
Mercury	0.04 - 0.29	6 / 6	0.11	ND	0 / 2	0.0050
Zinc	76 - 149	6 / 6	107	19.7 - 22.3	2 / 2	21

ND - not detected

* - Average concentrations include values of half the detection limit for all non-detects.

2.2.6 Supplemental Sampling

Nature and extent of contamination for the supplemental sampling are summarized, by medium, in the following subsections.

Surface Water. Baseflow and storm event surface water data were collected from six surface water gauging stations throughout the study area. Gauging stations SW-05, SW-06, and SW-07 were located at the Salem Street Bridge (south of the Wells G&H 38-acre wetland in Woburn), downstream of the Montvale Avenue bridge (Woburn), and at the Swanton Street bridge (Winchester), respectively. Gauging station SW-08 was located at the USGS gauging stations on the Mystic Valley Parkway (Winchester). Gauging stations SW-09 and SW-10 were located at the outlets to the Upper (Arlington) and Lower (Medford) Mystic Lakes, respectively (Figure 2-31). Data for gauging station SW-05 have been combined with data for SW-MC-13. The baseflow and storm event data for the six surface water gauging stations are presented in summarized form by gauging station in Tables 2-222 through 2-227 and Tables 2-234 through 2-239, respectively. Summaries by individual sample for each gauging station, showing all detected compounds, are presented in Tables 2-228 through 2-233 for baseflow data, and in Tables 2-240 through 2-245 for storm event data.

The 1995 and 2001/2002 data sets display overall consistent results for total metals concentrations. The storm event data indicated slightly elevated average concentrations in comparison to baseflow samples for aluminum, antimony, arsenic, cadmium, chromium, cobalt, copper, iron, lead, mercury, nickel, vanadium, and zinc. Baseflow samples contained the highest average concentrations of barium, beryllium, manganese, selenium, silver, and thallium. For these listed metals, baseflow average concentrations exceeded storm event average concentrations due to slightly elevated non-detect levels in the baseflow data set. The maximum concentration of aluminum, arsenic, barium, beryllium, cadmium, chromium, cobalt, iron, lead, manganese, mercury, nickel, selenium, silver, thallium, vanadium, and zinc were noted in storm event samples collected from gauging station SW-08. Selected metals of potential interest are presented in Figures 2-32 and 2-33. A comparison of overall metals concentrations in the baseflow and storm event samples is presented in the table below.

Surface Water

Analyte Wg/L	Baseflow			Storm Event		
	Range of Detects	Detection Frequency	Average Concentration*	Range of Detects	Detection Frequency	Average Concentration*
Aluminum	60.6 - 363	12 / 97	55.4	109.4 -3397	12 / 36	259.6
Antimony	ND	0 / 97	1.29	0.82 - 3.9	10 / 36	1.33
Arsenic	1.5 - 28	64 / 98	7.71	1.95 - 29.9	28 / 36	9.91
Barium	29.9 - 62.9	97 / 97	40.1	22.6 - 95.2	36 / 36	34.8
Beryllium	ND	0 / 97	0.17	0.094 - 0.27	6 / 36	0.116
Cadmium	ND	0 / 97	0.19	0.1 - 1.26	7 / 36	0.22
Chromium	0.5 - 17.3	63 / 97	3.08	0.61 - 59.1	25 / 36	7.2
Cobalt	0.55 - 2.2	22 / 97	0.69	0.3 - 4.1	10 / 36	0.74
Copper	1.7 - 21.1	48 / 97	3.94	2.1 - 112	21 / 36	13.9
Iron	75.5 - 3460	87 / 97	1073	77.8 - 8858	29 / 36	1421
Lead	1.4 - 17.4	33 / 97	2.33	1.04 - 83	19 / 36	7.4
Manganese	12.9 - 764	97 / 97	301.9	10.9 - 807	36 / 36	189.3
Mercury	0.1 - 0.36	2 / 97	0.057	0.054 - 0.24	11 / 36	0.068
Nickel	0.59 - 2.7	55 / 97	1.21	0.55 - 6.5	19 / 36	1.48
Selenium	ND	0 / 97	1.62	1.05 - 1.69	6 / 36	1.28
Silver	ND	0 / 97	0.354	0.25 - 0.4	6 / 36	0.32
Thallium	5.2	1 / 97	1.86	1.16 - 2.15	6 / 36	1.52
Vanadium	0.52 - 5.4	21 / 97	0.633	0.56 - 10.15	16 / 36	1.3
Zinc	0.73 - 172	88 / 97	39.6	2.4 - 307.2	27 / 36	61.8

units - Wg/L

* - Average concentrations include values of half the detection limit for all non-detects.

Sediment/Floodplain Surface Soil. Fifteen sediment samples and nine floodplain surface soil samples were collected from station AJRW, in Winchester south of Bacon Street. Sediment and surface soil data are presented in summarized form in Tables 2-246 and 2-248, respectively, and summarized for detected compounds by individual sample in Tables 2-247 and 2-249, respectively.

Samples were analyzed for metals only. Numerous metals in surface soil were detected at concentrations similar to sediment samples. The highest maximum detected concentrations of aluminum, arsenic, barium, cobalt, copper, iron, manganese, mercury, nickel, and vanadium were detected in soil, while the highest maximum detected concentrations of beryllium, cadmium, chromium, lead, and zinc were detected in sediment. Selected metals detected in the AJRW sediment and surface soils samples are presented on Figure 2-30. Concentrations of metals in soil and sediment samples collected at station AJRW are presented in the table below. This comparison of surface soil to sediment concentrations for metals indicated consistent concentrations of metals in sediments relative to upland soils collected at station AJRW.

Station AJRW Surface Soil and Sediment

Analyte mg/kg	Surface Soil			Sediment		
	Range of Detects	Detection Frequency	Average Concentration*	Range of Detects	Detection Frequency	Average Concentration*
Aluminum	6400 - 17000	9 / 9	11311	3600 - 14000	15 / 15	8407
Arsenic	24 - 98	7 / 9	46	20 - 57.5	7 / 14	22
Barium	20 - 220	9 / 9	62	20 - 89.5	15 / 15	52
Beryllium	ND	0 / 9	0.49	0.98	1 / 15	0.53
Cadmium	ND	0 / 9	1.5	3.6 - 3.8	3 / 15	1.9
Chromium	19 - 90	9 / 9	40	11 - 235	15 / 15	89
Cobalt	3.5 - 21	9 / 9	9.1	4.1 - 17	15 / 15	8.7
Copper	26 - 200	9 / 9	64	11 - 190	15 / 15	92
Iron	11000 - 34000	9 / 9	17667	8800 - 25000	15 / 15	14553
Lead	88 - 930	9 / 9	298	35 - 1000	15 / 15	259
Manganese	63 - 600	9 / 9	229	91 - 445	15 / 15	221
Mercury	0.22 - 1.2	9 / 9	0.43	0.051 - 1.045	14 / 15	0.51
Nickel	7 - 33	9 / 9	14	6.9 - 24	15 / 15	14
Vanadium	29 - 70	9 / 9	44	17 - 53	15 / 15	30

Station AJRW Surface Soil and Sediment

Analyte mg/kg	Surface Soil			Sediment		
	Range of Detects	Detection Frequency	Average Concentration*	Range of Detects	Detection Frequency	Average Concentration*
Zinc	25 - 250	9 / 9	112	51 - 820	15 / 15	334

units - mg/kg

* - Average concentrations include values of half the detection limit for all non-detects.

Sediment Cores. Thirty-six sediment core samples were collected including four samples at each of locations SC05 through SC13. SC05 through SC08 were collected from the Wells G&H 38-acre wetland, SC09 and SC10 from the former Cranberry Bog, SC11 from Davidson Park, SC12 from Judkins Pond, and SC13 from the outlet of the river as it flows into the upper forebay of Upper Mystic Lake (Figure 2-34). The four samples collected at each location included one sample from each of the following depth intervals: 0 - 1 foot, 1 - 2 foot, 2 - 3 foot, and 3 - 4 foot. The samples are summarized by individual location in Tables 2-250 through 2-258 and all detected compounds at each location by individual sample in Tables 2-259 through 2-267. Samples were analyzed for metals only.

Concentrations of metals generally decreased with depth in the sediment cores except for sediment cores SC05, SC11, and SC12. SC05 (reach 1 by station TT-28) arsenic concentrations were higher in the 2 - 3 foot interval and 3 - 4 foot interval (900 mg/kg and 770 mg/kg, respectively). SC11 (reach 3 in Davidson Park) and SC12 (reach 4 in Judkins Pond) lead concentrations increased with depth with the 3 - 4 foot interval concentrations at 1200 mg/kg and 1300 mg/kg, respectively.

Overall, average arsenic concentrations decreased from 411 mg/kg in the 0 - 1 foot interval to 136 mg/kg in the 3 - 4 foot interval. Average total chromium concentrations decreased from 857 mg/kg in the 0 - 1 foot interval to 270 mg/kg in the 3 - 4 foot interval. Despite overall decreasing concentrations with depth, a small number of metals showed variable trends between specific depth intervals. For example, copper, lead, and mercury average concentrations decreased over the 0 - 1 foot and 1 - 2 foot intervals, but showed slightly increased concentrations in the 2 - 3 foot interval. Cadmium, selenium, and silver average concentrations decreased over the 0 - 1 foot, 1 - 2 foot, and 2 -3 foot intervals, but increased slightly in the 3 - 4 foot interval. Samples collected from the Wells G&H 38-acre wetland (SC05 through SC08) displayed the highest concentrations of arsenic, antimony, barium, beryllium, cadmium, cobalt, copper, iron, manganese, mercury, nickel, and zinc. The highest concentrations of aluminum and vanadium were detected at SC13, collected from the outlet of the river as it flows into the upper forebay of Upper Mystic Lake. The single detection of selenium was noted at location SC12, collected from Judkins Pond. The highest concentration of lead was also detected at SC12. Silver was detected in its highest concentration at SC11, located within the pond at Davidson Park. Selected metals detected in the sediment core samples, by depth interval, are presented on Figure 2-35. A comparison of metals concentrations in the four depth intervals is presented in the table below.

Sediment Cores

Analyte mg/kg	Range of Detects	Detection Frequency	Average Concentration*	Range of Detects	Detection Frequency	Average Concentration*
	0 - 1 Foot Interval			1 - 2 Foot Interval		
Aluminum	5300 - 22000	8 / 8	14675	4600 - 19000	9 / 9	11667
Antimony	ND	0 / 9	10.5	ND	0 / 9	6.34
Arsenic	67 - 1300	9 / 9	411.4	21 - 1700	9 / 9	292.3
Barium	19 - 150	9 / 9	88.8	17 - 130	9 / 9	59.2
Beryllium	1.1 - 1.5	2 / 9	1.26	1 - 1.3	4 / 9	0.94
Cadmium	4.2 - 25	7 / 9	10.9	3.6 - 48	7 / 9	10.3
Chromium	130 - 2700	9 / 9	856.7	28 - 1200	9 / 9	441.4
Chromium VI	1.39 - 29	9 / 9	9.2	0.3 - 12.9	9 / 9	4.74
Cobalt	3.6 - 68	9 / 9	26.9	3.3 - 29	9 / 9	15.4
Copper	47 - 1410	9 / 9	539.1	16 - 950	9 / 9	331.6

Sediment Cores

Analyte mg/kg	Range of Detects	Detection Frequency	Average Concentration*	Range of Detects	Detection Frequency	Average Concentration*
Iron	11000 - 120000	9 / 9	42578	6500 - 48000	9 / 9	22633
Lead	28 - 810	9 / 9	433.7	19 - 800	8 / 9	296.7
Managanese	110 - 2600	9 / 9	831.8	74 - 840	9 / 9	407.1
Mercury	0.18 - 14	9 / 9	4.13	0.033 - 8	9 / 9	2.76
Nickel	7.1 - 49	7 / 9	26.5	7.8 - 40	6 / 9	17.7
Selenium	ND	0 / 9	16.2	ND	0 / 9	9.96
Silver	1.8	1 / 9	1.3	3.1	1 / 9	1.2
Vanadium	11 - 87	9 / 9	52.7	9.2 - 67	9 / 9	36.9
Zinc	150 - 4800	9 / 9	1768	96 - 4600	9 / 9	1602
	2 - 3 Foot Interval			**3 - 4 Foot Interval**		
Aluminum	1800 - 19000	9 / 9	10100	1700 - 20000	9 / 9	9133
Antimony	13	1 / 9	5.8	ND	0 / 9	4.64
Arsenic	40 - 900	7 / 9	191.3	32 - 770	7 / 9	136.3
Barium	7.5 - 130	9 / 9	58.3	6.3 - 140	9 / 9	55.7
Beryllium	1 - 1.6	4 / 9	0.89	1.1 - 1.5	3 / 9	0.76
Cadmium	3.95 - 14	6 / 9	5.5	2.7 - 24	5 / 9	5.8
Chromium	5.6 - 1500	9 / 9	408.4	5.5 - 740	9 / 9	269.6
Chromium VI	6 - 16.1	9 / 9	4.38	0.059 - 7.9	9 / 9	2.9
Cobalt	3.4 - 22	8 / 9	11.8	4.5 - 22	6 / 9	9.2
Copper	15 - 1600	8 / 9	337.7	20 - 1400	8 / 9	294.6
Iron	1800 - 50000	9 / 9	19122	2000 - 34000	9 / 9	15089
Lead	25 - 1200	7 / 9	306.1	20 - 1300	7 / 9	345.3
Manganese	35 - 640	9 / 9	307.7	25 - 470	9 / 9	239.2
Mercury	0.37 - 26	7 / 9	3.87	0.28 - 8.3	7 / 9	1.7
Nickel	6.7 - 40	8 / 9	16.3	9 - 45	6 / 9	16.2

Sediment Cores

Analyte mg/kg	Range of Detects	Detection Frequency	Average Concentration*	Range of Detects	Detection Frequency	Average Concentration*
Selenium	ND	0 / 9	6.88	10	1 / 9	6.96
Silver	2 - 3.5	2 / 9	1.19	1.5 - 4.6	3 / 9	1.58
Vanadium	11 - 68	8 / 9	33.5	5.3 - 86	8 / 9	31.9
Zinc	64 - 1700	9 / 9	892	43 - 1900	9 / 9	800

units - mg/kg

* - Average concentrations include values of half the detection limit for all non-detects.

2.2.7 Groundwater Sampling

Groundwater sampling data were collected from thirteen monitoring wells within the Wells G&H 38-acre wetland. Shallow overburden data were available from all thirteen monitoring well locations. Medium and deep overburden data were available from six of the thirteen monitoring well locations. Figure 2-36 shows the location of each monitoring well. Metals data are presented, by well location and flow zone (shallow, medium, and deep overburden), in Table 2-268.

In order to evaluate the potential impact of sediments contaminated with metals on groundwater quality, a screening-level evaluation has been conducted which consists of a comparison of groundwater metals data to USEPA primary and secondary Maximum Contaminant Levels (MCLs; USEPA, 2003). The Massachusetts Department of Environmental Protection (MADEP) has recently re-evaluated the use and value of the groundwater aquifer within the Wells G&H Superfund Site. The determination supports a medium use and value for groundwater at the site, and states that domestic water use, including drinking water ingestion, should be considered (see *MADEP Groundwater Use and Value Determination* included as Appendix A.8 of this report). Therefore, MCLs, as standards protective of domestic water use, have been selected for the screening-level evaluation, even though groundwater is not currently used as a source of potable water. Only groundwater metals data have been included in the screening because metals are the primary contaminants within the 38-acre wetland sediments. Other less significant contaminants present within the wetland sediments (e.g., volatile organic compounds) are likely the result of impacts from the Wells G&H source area properties (OU-1).

As shown in Table 2-268, shallow overburden arsenic concentrations in monitoring wells MW-003, MW-004, MW-006, MW-007, MW-010, MW-011, S88, and S92 exceed the arsenic MCL (10 Wg/L). Arsenic exceedances ranged from 10.6 Wg/L (at monitoring well S92) to 142 Wg/L (at monitoring well MW-010). No other exceedances of primary MCLs were noted in shallow overburden wells. No exceedances of primary MCLs were noted in the medium and deep overburden wells included in the sampling program. Locations of monitoring wells displaying exceedances of primary MCLs are noted on Figure 2-37.

Secondary MCLs for aluminum, iron, and manganese were exceeded in shallow, medium, and deep overburden groundwater. Manganese secondary MCL exceedances occurred most frequently. All shallow overburden well locations, and all but one of the medium and deep overburden well sampling locations (monitoring well S92) demonstrated levels above the secondary MCL of 50 Wg/L. Manganese exceedances ranged from 5,930 Wg/L (in shallow overburden groundwater at monitoring well MW-S89) to 197 Wg/L (in deep overburden groundwater at monitoring well MW-009). Iron concentrations at 12 of 13 shallow overburden well locations, 5 of 6 medium overburden well locations, and 2 of 6 deep overburden well locations exceeded the secondary MCL for iron (300 Wg/L). Iron exceedances ranged from 42,200 Wg/L (in shallow overburden groundwater at monitoring well S88) to 492 Wg/L (in medium overburden groundwater at monitoring well MW-011). Aluminum concentrations at 8 of 13 shallow overburden well locations, 3 of 6 medium overburden well locations, and 4 of 6 deep overburden well locations exceeded the lower end of the secondary MCL range for aluminum (50 Wg/L). Only shallow overburden concentrations at monitoring well locations S89 (324 Wg/L) exceeded the upper end of the aluminum MCL range (200 Wg/L).

Overall, there appears to be a trend of groundwater metals concentrations decreasing with increasing depth of groundwater. This trend is most noticeable with arsenic, suggesting a possible impact of sediment arsenic contamination on groundwater quality. Groundwater is not currently being used as a source of potable water. The groundwater-sediment interaction for metals will be discussed more fully in the comprehensive RI Report.

2.3 CONTAMINANT FATE AND TRANSPORT

An understanding of the environmental fate and the potential transport mechanisms of the contaminants present in the Aberjona River study area samples is necessary to determine the potential for continued contaminant migration and to assess the potential for exposure to the contaminants. While a site-specific analysis of fate and transport mechanisms was beyond the scope of this study, a general discussion of fate and transport mechanisms is presented in this section. Two major characteristics affecting the fate and transport of a chemical are the mobility and the persistence of the chemical in environmental media.

Mobility is the tendency of a chemical to migrate through the environment. Mobility is controlled by both the physicochemical environment at a site and the behavioral characteristics of individual chemicals. Important factors relating to the physicochemical environment of a site include the local climate, the configuration of surface water and groundwater bodies and the nature of underlying soils and bedrock. Factors that control the behavior of individual compounds include aqueous solubility, the susceptibility of a chemical to sorption and volatility.

Persistence is the tendency of a chemical to remain in the environment. Persistence is influenced by many of the factors affecting chemical mobility (including solubility, sorption and volatility), but is also a function of oxidation rates, hydrolytic and photolytic reactions, and biochemical processes (such as biodegradation and bioaccumulation).

2.3.1 Factors Affecting Fate and Transport

Major factors affecting environmental fate and transport of chemicals are briefly defined below:

Solubility is the measure of a chemical's ability to dissolve in a solvent and is expressed in units of chemical mass per unit volume of solvent (e.g., Wg/L or mg/L). Aqueous solubility, which is used for environmental conditions, is an important determinant of chemical concentration and residence time in water. In addition, solubility often predicts the ease with which chemicals are leached from wastes and soils.

Volatilization describes the movement of a chemical from the surface of a liquid or solid matrix to a gas or vapor phase. Volatilization from the liquid phase is measured by the Henry's Law constant, which can be expressed as the quotient of the chemical's vapor pressure to its solubility at a specific temperature. Lyman *et al.* (1982) described compounds as readily, significantly or limitedly volatilized based on the values of their Henry's Law constants. These values in atm-m^3/mol are $\geq 10^{-3}$, 10^{-3} to 10^{-5} and $< 10^{-5}$, respectively.

Sorption (adsorption/desorption) is the reversible binding of a chemical to a solid matrix. Both soluble nonpolar and insoluble chemicals usually adsorb strongly to sediments, suspended solids and soils. Sorption of these compounds to a solid phase limits the fraction available for other fate processes such as volatilization and hydrolysis.

Partition coefficients are expressed as concentration ratios; higher values indicate a greater tendency for a chemical to associate with a non-aqueous phase (i.e., organic phase) than with an aqueous phase. Partition coefficients useful in describing the environmental behavior of a compound include K_{ow}, K_d and K_{oc} and are defined as follows:

K_{ow}: The octanol-water partition coefficient is the ratio of chemical concentration in octanol (an organic solvent) to that in water at steady-state conditions.

K_d: The soil-water partition coefficient is the ratio of chemical concentration in aqueous and solid phases at steady-state conditions (usually applied to inorganic species).

K_{oc}: The organic carbon partition coefficient is the K_d normalized to the concentration of organic carbon in the soil or sediment, since in many soils or sediments, the organic carbon is the dominant sorbent for hydrophobic organic compounds.

Bioaccumulation is the net accumulation of a chemical by an organism as a result of uptake from all routes of exposure. Plants or animals may bioaccumulate chemicals from environmental media via ingestion, inhalation, and/or direct absorption through the organism's exposed surfaces. After entering the organism, contaminants may become concentrated within specific tissues. Compound-specific information on bioconcentration (uptake from water), bioaccumulation and biomagnification is provided in Section 3.0.

Biotransformation/biodegradation is the metabolic transformation of complex molecules into other compounds by microorganisms. Products of biotransformation/biodegradation may or may not be toxic to other organisms, and these products may then undergo further biotransformation/ biodegradation.

Hydrolysis is the reaction of a chemical with water or with hydrogen (H^+) or hydroxyl (OH^-) ions. The extent of chemical hydrolytic reactivity depends on both the pH (acidity/alkalinity) and the molecular structure of the specific chemical.

Photolysis is a chemical decomposition process induced by sunlight. The rate of loss of a chemical from photochemical reactions depends on its molecular structure, the proximity and wavelength of the light source, and the presence of other reactive compounds.

Oxidation is a chemical reaction which involves the removal of electrons from an element or compound. Conversely, electrons are added to chemical substrates in **reduction** reactions. Both oxidation and reduction reactions are environmentally significant in that they influence the mobility and fate of chemicals in environmental matrices. Oxidized and reduced forms of the same element or compound may have significantly different chemical, environmental and/or toxicological properties.

For the purposes of this discussion, the compounds detected during the investigations were grouped into four generalized classes sharing similar physicochemical and behavioral characteristics. These classes are: (1) VOCs, (2) PAHs, (3) pesticides/PCBs and (4) inorganics.

This section of the report is focused on the fate and transport processes that may affect contaminants in the Aberjona River Study area. Environmental characteristics that influence contaminant fate and transport, environmental behavioral characteristics of the contaminant classes, and a summary of the anticipated fate for the chemical classes are discussed in the following subsections. Specific transport and migration pathways that may affect the contaminants and the contaminant fate and transport analysis findings are also discussed.

The fate and transport of contaminants in the Aberjona River Study area are affected by the geology and soil type, geochemistry, hydrology, and climate of the area. These characteristics are discussed in detail in Section 1.3.

2.3.2 Fate and Transport Data

This section summarizes the chemical characteristics and available fate and transport data for organic and inorganic contaminants. Each generalized contaminant class is discussed along with a summary of the anticipated environmental fate characteristics. Much of the information presented in this section is from USEPA (1979, 1985), in addition to other sources, to which the reader is referred for more detailed discussions of the chemical characteristics affecting the fate and transport of the study area contaminants.

Volatile Organic Compounds. VOCs are commonly used as solvents and degreasers in a variety of industrial processes, and they are also widely found in petroleum products. VOCs are generally segregated into halogenated and non-halogenated VOCs. Since primarily halogenated VOCs have been detected in the study area, only halogenated VOCs are discussed further.

Halogenated VOCs are generally mobile and not very persistent in the environment, principally due to their high volatility, low sorption to soils, inability to substantially bioaccumulate and high aqueous solubility. Because of these characteristics, the primary fate and transport mechanisms affecting these VOCs are volatilization into air and migration in the groundwater. However, under certain conditions which restrict or eliminate these compounds' contact with the surface atmosphere (e.g., presence in deep soil/sediment strata or deep groundwater), volatilization into the air may be of minor importance. The halogenated VOCs also tend to undergo degradation reactions in anaerobic and aerobic systems. These degradation reactions involve the progressive loss of halogens and/or the opening of the double-bond structure. Generally, these reactions result in a sequential increase in the mobility of the resulting compounds within environmental media.

Polycyclic Aromatic Hydrocarbons. PAHs are components of asphalt, fuels, oils and greases. Anthropogenic sources of PAHs in the environment include combustion processes used in industry, heating and power generation, and petroleum refining.

PAHs are persistent and generally immobile in soil/sediment matrices under normal environmental conditions. This is primarily due to their low aqueous solubility, their resistance to photolytic, oxidative and hydrolytic degradation, and their high affinity for sorption to organic matter and soil/sediment particles. However, in the presence of highly mobile organic compounds (e.g., VOCs) which can act as co-solvents, the mobility of PAHs in solid (soils/sediments) and/or aqueous matrices (groundwater/surface water) can be greatly enhanced. PAHs can be degraded by microbial populations; however, this is generally a slow process in the environment.

Pesticides/PCBs. Although pesticides and PCBs have quite different chemistries, uses and sources, these compounds are considered together as they have relatively similar environmental behavioral characteristics. Pesticides are commonly used in agriculture, as well as in commercial and residential areas to control insect and other nuisance populations. The principal use of PCBs was as dielectric fluids in electrical transformers and capacitors.

Pesticides and PCBs are generally persistent in the environment, primarily due to their resistance to degradation, low aqueous solubility and volatility, and very high adsorptive affinity for soils/sediments and organic matter. Within these classes of compounds, the degree of persistence in the environment is related to the degree of chlorination as well as chemical structure. In general, the greater the number of chlorine atoms and the more double bonding in the structure, the more persistent the pesticide or PCB. They are typically highly resistant to biodegradation and, when it does occur, it is a very slow process. Degradation products of 4,4'-DDT are 4,4'-DDD and 4,4'-DDE. Primarily as a result of their very high sorption to soils/sediments, pesticides and PCBs are essentially immobile in soil/sediment matrices under normal environmental conditions. However, in the presence of highly mobile organic co-solvents (e.g., VOCs), the mobility of pesticides and PCBs can be greatly enhanced.

Inorganics. This subsection summarizes environmental fate data for those inorganics exhibiting atypical environmental concentrations. These constituents are antimony, arsenic, beryllium, cadmium, chromium, copper, lead, manganese, mercury, selenium, silver, vanadium, and zinc.

Many of the fate and transport mechanisms that may be important for organic compounds have little impact on the inorganics. Volatilization, for example, only applies to a select few inorganics (e.g., mercury, some organo-metallic compounds), and then only under special conditions. Photolysis is also of negligible importance to the environmental behavior of inorganics. Inorganics are difficult to discuss in terms of behaviorally similar groups. The characteristics of individual inorganics are generally better understood.

The most important factors controlling inorganic fate and transport are solubility, redox behavior, aqueous speciation and sorption behavior, all of which are functions of the ambient geochemical environment. Biotransformation processes can be important for some inorganics (e.g., arsenic, copper, lead) under certain environmental conditions. All inorganics are subject to cation-exchange reactions with minerals present in the environment. The extent to which cation-exchange occurs is dependent on the mineral species present and on pH, as well as on the characteristics of the individual inorganics.

The mobility of inorganics within environmental matrices depends upon numerous factors such as the relative stabilities of individual valence states, oxygen content, pH and Eh conditions, and the presence of available complexing agents. Chemical speciation determines the relative degree of sorption among different species of a particular inorganic. Based on the data available for the sediments, sorption is most probably a significant fate process for inorganics. The aerobic conditions in the surface water are likely to promote the precipitation of ferromanganese oxides and oxyhydroxides to which other inorganics will readily adsorb.

Antimony. The hydrogeochemical behavior of antimony is analogous to that of arsenic. Like arsenic, antimony most commonly occurs in the Sb(+3) and Sb(+5) valence states, and in existing field and laboratory investigations, antimony appears to be largely controlled by redox conditions. In aerobic waters, insoluble Sb(+5) is adsorbed to ferromanganese oxides and oxyhydroxides, however the sorption of Sb(+5) may not be as strong as As(+5), and the solubility of Sb(+5) is generally higher than that of As(+5). In anaerobic waters, antimony is reduced to Sb(+3) which is soluble.

Arsenic**.** In the environment, arsenic occurs predominantly in the As(+3) and As(+5) valence states and, although certain conditions may promote the formation of arsenious (H_3As0_3) or arsenic (H_3As0_4) acid, the oxidation state of arsenic is the factor that seems to control arsenic solubilization. The inorganic state is dominant even though arsenic is involved in biological cycling that can form soluble organic complexes.

The redox chemistry of arsenic is highly analogous to that of iron and manganese, and arsenic tends to be closely associated with these two elements in aqueous systems. Under aerobic conditions, As(+5) is the predominant species. Arsenic (+5) is highly insoluble and tends to be strongly adsorbed on ferromanganese precipitates; i.e., As (+5) follows the oxidized species of iron (Fe(+3)) and manganese (Mn(+4)). Thus, in oxidated water, arsenic is primarily associated with particulate phases. Under reducing conditions, arsenic is reduced to As(+3), which is soluble in anoxic waters. Arsenic may also form complexes with anthropogenically introduced organic compounds that may affect the geochemical behavior of arsenic.

Arsenic is adsorbed principally onto clays, aluminum hydroxides, ferromanganese oxides, and organic compounds. In general, pentavalent arsenic has a greater adsorptive affinity than trivalent arsenic. For arsenic, sorption is most important in aerobic, acidic fresh water with sorption decreasing above pH 9 for As(+3) and above pH 7 for As(+5).

Beryllium. Beryllium is always found in the +2 valence state in aqueous matrices and may form stable compounds with anions if they are present (e.g., fluoride). At low pH (< 4), Be+2 ions are the predominant species, whereas at very high pH (>12), $HBeO_2$ is the more prevalent form in water. Within normal pH ranges in the environment, the slightly soluble $Be(OH)_2$ is the dominant species. Very little data exist for beryllium sorption behavior because of its low solubility; however, the available data suggest that beryllium sorbs to clay at low pH. At high pH, complexation into insoluble compounds appears to be favored over sorption mechanisms.

Cadmium. Cadmium may exist in soluble organic complexes or as ionic species in water. Cadmium ions in solution are always present in the (+2) valence state in aqueous environmental matrices. Cadmium may also be associated with the particulate phase. Cadmium is principally adsorbed by clays, organics, carbonates, and aluminum and iron oxides, with sorption generally increasing as the pH increases.

Chromium. In aqueous systems, chromium can theoretically occur in two oxidation states: Cr(+3) and Cr(+6). In many ways, the hydrogeochemical behavior of chromium is the opposite of iron, manganese, arsenic and antimony. The oxidized state of chromium, Cr(+6), is relatively soluble, forming complex anions in aqueous solution. The most important of these are chromate (CrO_4^{-2}) and hydrochromate ($HCrO_4^-$). However, Cr(+6) species are not stable aqueous complexes under virtually all naturally occurring redox conditions. In natural waters, trivalent chromium is the stable and predominant aqueous form of chromium. In its trivalent form, chromium rapidly precipitates as insoluble oxides or hydroxides, or adsorbs onto clays or oxides of other inorganics.

Copper**.** Copper(+2) is the most prevalent form of copper in aqueous systems as most of the stable cuprous (+1) forms in waters are highly insoluble. Copper may also exist in water as the hydrated divalent cupric ion. However, in general, most copper in aqueous solution is in a complex form with organic or inorganic ligands and these are expected to be the predominant dissolved aqueous species of copper at the study area. Copper is sorbed by clays, mineral surfaces, organics, carbonate, and iron and manganese oxide precipitates. Copper sorption is highly pH dependent and the presence of other anionic species can increase copper sorption.

Lead. Lead(+2) is the most common stable ionic aqueous species with hydroxyl, carbonate, sulfide and sulfate anions acting as solubility controls. Under aerobic conditions, $PbSO_4$ and to a lesser extent $PbCO_3$, control lead solubility; whereas, under anaerobic conditions, PbS concentrations mediate aqueous lead solubility. Lead may also exist in soluble organic complexes (i.e., humic and fulvic acids) in aqueous matrices. Lead adsorbs principally to clays, hydrous iron and manganese oxides, mineral surfaces and organic compounds. Lead sorption is very pH-dependent, with low pH conditions favoring desorption.

Manganese. Manganese occurs in the (+2) and (+4) oxidation states in aqueous systems. In oxidated waters, Mn(+4) is the stable form. Mn(+4) is insoluble and precipitates, along with Fe(+3), to form ferromanganese oxides and oxyhydroxides. In anaerobic waters, manganese is reduced to Mn(+2) which is soluble under continuing reducing conditions. Studies of natural systems have shown that

Mn(+4) is the first (i.e., the least soluble) inorganic to precipitate of the behaviorally analogous group manganese, iron and arsenic. Similarly, the reduction of Mn(+4) to Mn(+2) and the accompanying resolution occurs before the reduction of Fe(+2) or As(+5). As long as aerobic conditions persist in the groundwater, transport of manganese in aqueous solutions will be of minor significance. Manganese readily forms insoluble oxides in aerobic waters. The formation of manganese oxides often requires nucleation on a particle resulting in "manganese coatings". The formation and continued growth of manganese coatings is an sorption process. Sorption is an important process under aerobic conditions, but is readily reversed if conditions become anaerobic.

Mercury. Mercury may exist in the (0), (+1), or (+2) valence states in natural waters, depending on conditions. Above pH 5 and under moderately oxidizing conditions, dissolved elemental mercury [Hg(0)] is expected to be the predominant elemental aqueous species. Mercury readily complexes with organic matter via biologically and non-biologically mediated processes. As a result, dissolved methyl mercury ion and undissociated dimethyl mercury may be present in aqueous matrices if mercury is present. Some studies have found mercury concentrations in surface waters vary with the biological cycle (i.e., vary seasonally with biological activity). Mercury is strongly adsorbed to many inorganic surfaces and organic matter. Desorption may occur under low pH conditions.

Selenium**.** The geochemical behavior of selenium is similar to that of sulfur, and selenium occurs in both cationic (+4) and anionic (-2) states. More rarely, selenium can occur in the native (0) state. However, this occurs only under anoxic conditions which may be present in deep soil strata, deep groundwater and/or marshland soil/sediment exhibiting high BOD/COD values.

Silver. Silver (+1) is the predominant species in natural waters, although it may also occur in the (+2) and (+3) valence states as a complexed ion. Silver tends to be closely associated with iron-manganese oxide and oxyhydroxide precipitates under aerobic conditions. Strong sorption to inorganic precipitates, clay minerals, and organic matter limits the mobility of silver, especially at higher (>7.0) pH values. Silver also forms insoluble silver salts with many of the common inorganic ligands (e.g., chloride and carbonate).

Vanadium. Vanadium can occur in the (+3), (+4), and (+5) valence states in the normal range of environmental conditions. In addition to the complexity introduced by the multiple oxidation states, the aqueous geochemistry of vanadium is further complicated by the variety of complex ions vanadium may form. In simplified form, however, the chemical behavior of vanadium somewhat resembles that of chromium. In reducing environments, vanadium is insoluble, and its solubility increases as conditions becoming increasingly oxidizing. Vanadium is readily adsorbed by clays and organic matter. Sorption by organic matter is probably more correctly a reductive, and therefore, an immobilizing, reaction.

Zinc. In most natural waters, zinc occurs as the hydrated divalent (+2) cation. In organically polluted waters, complexation with organic compounds may be an important process. The solubility of zinc is strongly dependent on pH, with low pH favoring increased solubility. Zinc has a strong affinity for sorption to hydrous inorganic oxides, clays and organic matter. Sorption of zinc is strongly favored at higher (>7) pH values.

2.3.3 Transport and Mechanisms of Migration

Contamination of matrices within the Aberjona River Study has occurred most likely from past industrial disposal practices. The importance of a given mechanism is controlled by the specific physical, geochemical, climatic and hydrologic conditions at a given site, as well as by the physicochemical characteristics of the contaminated media. In this section of the report, the following potential pathways for the fate and transport of contaminant classes within the various matrices in the Aberjona River Study area will be considered:

- Migration of contaminants from potential source areas to environmental media;
- Migration of contaminants into and within surface waters;
- Re-suspension and migration of sediments;
- Migration of contaminants into air.

The importance of the pathways mentioned above to the classes of contaminants found within the study area will be discussed in the sections that follow.

Migration of Contaminants from Potential Source Areas to Environmental Media. Contamination of matrices within the Aberjona River Study area has occurred in the past as a result of disposal practices, most likely from upgradient NPL and/or industrial sites. Chemicals from these potential sources of contamination may migrate with and/or into the surrounding environment in several ways. Constituents present in potential solid sources (e.g., soils) may be transported into the underlying groundwater by the infiltration of rain and/or gravity. Contaminants may migrate with the groundwater flow, and may enter surface waters/wetland areas by groundwater discharge. In addition, localized storm events can generate surface runoff and ponding, and the more water-soluble contaminants may migrate into and/or within the surface water runoff. The runoff may also be able to transport fine particulates with associated contaminants to other areas. Dry, windy weather may result in the entrainment of contaminated particulates, with subsequent deposition over adjacent areas. Volatile contaminants may volatilize from environmental sources and be emitted into the atmosphere.

Migration of Contaminants into and within Surface Waters. As stated above, contaminants may migrate into the local surface waters through groundwater recharge and surface runoff. Upon entering the groundwater through percolation, contaminants may migrate within the groundwater, and enter the nearby surface water (i.e., the Aberjona River) and associated wetland areas by groundwater discharge. In addition, localized storm events can generate surface runoff and ponding. The more water-soluble contaminants may migrate into and/or within the surface water runoff. The runoff may also be able to transport fine particulates with associated contaminants. Finally, limited deposition of airborne particulates may occur.

Surface water transport can occur in two ways: through transport of contaminated material in the sediment load and through transport of dissolved components. Sediment transport is controlled by physical processes and is dependent on the rate of flow, which determines the sediment load capacity of the water body. In more stagnant bodies of water (e.g., lakes, wetlands), contaminated sediments may accumulate without further significant transport. In rivers and streams, increased flow rates increase the capacity to transport sediment away from sources (i.e., downstream). Points of deposition where sediments may accumulate are dependent on the river/stream system. In contrast to sediment transport, transport of dissolved components is both a physical and a chemical process. Chemistry in the transport process is controlled by the rate of release of the contaminant from the source, by the solubility of the contaminant, and by the rate of influx of a contaminated media (i.e., rate of groundwater recharge into surface water). The amount of contaminant transported is a function of the equilibrium dissolution/precipitation conditions of both the constituent and the water system. The transport of dissolved components is also physically dependent on the rate of flow of the surface body (i.e., higher flow levels increase rate of transport and distance from source).

VOCs are not expected to persist in surface waters due to their rapid volatilization. In addition, they have low affinities for sorption and high aqueous solubilities, indicating that VOCs are not likely to occur within the sediments. This is substantiated by the VOC data, which indicated only a small number of VOCs in the surface water and sediment samples. In contrast, PAHs, PCBs, and pesticides have moderate to low aqueous solubilities and volatilization rates, and high affinities for sorption. Therefore, these compounds are more likely to be present in the sediments and/or to be transported in the sediment load. These behavioral characteristics are supported by the sampling data. No SVOCs or PCBs, and only low levels of pesticides, were detected in the surface water samples. Sediment samples, in comparison, contained numerous occurrences of PAHs, pesticides, and PCBs. Transport of inorganics in surface waters is controlled by the solubility of the individual inorganics, which is in turn controlled by the physicochemical characteristics of the water body.

Re-suspension and Migration of Sediments within the Study Area. Contaminants in sediments may also migrate during periods of high river flow and/or storm events. Thus, sediment with high concentrations of contaminants may act as source areas for contaminant transport.

Migration of Contaminants into Air. Contaminants may migrate into air via two distinct emission mechanisms: entrainment of particles by the wind (i.e., dust emissions) and volatilization, primarily of organic compounds. Particulate entrainment is a relatively unfeasible transport mechanism from surface waters and sediments. Volatilization from surface matrices is essentially unrestricted, and as such, is governed only by the physicochemical characteristics of a given compound under ambient conditions. Volatile organic compounds, due to their high volatility, would be likely to migrate from the surface of water bodies (i.e., river and lakes) and wetlands. The concentrations of VOCs detected in the surface waters, though, indicate that volatile emissions would be relatively insignificant for the study area.

2.3.4 Contaminant Fate and Transport Analysis Summary

The migration of contaminants from their source(s) into the surface waters that comprise the Aberjona River Study area will most likely occur through stormwater surface runoff and groundwater discharge. Upon entering the Aberjona River, contaminants will migrate with the local flow until dilution and removal mechanisms such as sorption, degradation, precipitation and limited volatilization result in their eventual non-detection, or until the contaminated water discharges to wetlands, other local streams or Upper Mystic Lake.

Storm runoff and to some extent groundwater discharge are expected to play a role in the environmental fate and transport mechanisms at the study area. An upward vertical flow component for groundwater exists in the vicinity of the wetland areas near the river. This flow results in groundwater discharge into the wetlands and Aberjona River. The migration pathway is especially important for highly soluble constituents including VOCs and certain inorganic species. In addition, since most of the area's stormwater runoff discharges to the Aberjona River and low lying areas, dissolved constituents/ions and/or particulates with associated contaminants can become incorporated into the water body system. This migration pathway is especially important for VOCs, PAHs, pesticides, PCBs, and inorganics.

The migration of contaminants within the surface waters will be a principal environmental fate and transport mechanism for the study area. Surface water transport can occur in two ways: through transport of dissolved components and through transport of contaminated material in the sediment load. The sediment load migration pathway is especially important for constituents with moderate to low aqueous solubility and high affinity for sorption to fine particles (e.g., PAHs, pesticides, PCBs, and inorganics). Soluble inorganic species, and to a lesser extent VOCs, can also be transported as dissolved components. In addition, contaminants in sediments may also be re-suspended during high flow and/or storm conditions, and contribute towards contaminant migration.

The emission of VOCs into the atmosphere from contaminated surface runoff and perennial and transitory water bodies, and their further migration via the prevailing wind, could be a viable transport mechanism. The concentrations of VOCs detected in the surface waters, though, indicate that this migration pathway would be relatively insignificant for the study area.

SECTION 3.0
BASELINE HUMAN HEALTH RISK ASSESSMENT

3.1 INTRODUCTION

This section of the report contains the baseline human health risk assessment for the Wells G&H Superfund Site, Operable unit 3 (the Aberjona River Study). Local residents living along or near the Aberjona River Study area could potentially contact contaminants in surface water, sediments, floodplain surface soil, and fish tissue while engaging in recreational activities including wading, swimming, and fishing.

The U.S. Environmental Protection Agency (USEPA) released the draft baseline human health risk assessment in March 2003, followed by a combined human health and ecological risk assessment in May 2003. Based on USEPA responses to comments received during the comment period, the draft risk assessment has been revised. Notable aspects of this revision include:

- recalculation of exposure point concentrations based on the use of USEPA's updated software program ProUCL (version 3.0);
- evaluation of recent sediment and floodplain surface soil samples collected along the Aberjona River in Winchester, south of Bacon Street (station AJRW);
- evaluation of recent sediment core data collected from nine locations along the Aberjona River between Route 128 and the Mystic Lakes (SC05 through SC13) for a dredging worker scenario;
- evaluation of recent surface water baseflow and storm event data collected from six surface water gauging stations along the Aberjona River between Route 128 and the Mystic Lakes (SW-05 through SW-10); and

- the extrapolation of recreational risks to a residential scenario to estimate the potential risks to residents whose property may have been impacted by flooding of the river.

The focus of this human health risk assessment is the quantitative evaluation of potential risks to local residents at stations[1] where human receptors have the potential for current and/or future exposure to contaminants. Local residents are potentially utilizing portions of the Aberjona River for current recreational activities. However, some areas of the river are currently difficult to access due to the presence of dense vegetation, steep banks and/or deep water, or have physical barriers to access such as fencing. Some of these difficult-to-access stations may become accessible in the future should the Wells G&H wetland area be developed by the City of Woburn or should future development or land use changes result in the removal of the physical barriers limiting current access.

Board of Health officials from the City of Woburn and Town of Winchester were contacted to determine the likelihood that individuals, engaging in recreational fishing, are also harvesting other aquatic species from the Aberjona River. Since it is the belief of these public health officers that individuals are not harvesting and consuming crayfish and eel from the river, only the human consumption of fish fillet tissue has been included in the quantitative human health evaluation (M&E, 2000).

3.1.1 Purpose and Scope

[1] The term station is used to describe a distinct area along the river where human exposures are reasonably assumed to occur currently or in the future. Specific surface water, sediment and soil samples are assigned to a station in order to estimate risk associated with exposures occurring at that station. See Appendix C.1 for sediment and soil samples that comprise each station.

The purposes of the baseline human health risk assessment are: 1) to evaluate the potential human health risks that may be posed by chemical contamination of the surface water, sediment, floodplain surface soil, and fish tissue within currently accessible and future potentially accessible portions of the study area; and 2) to provide a basis for decisions as to whether remedial action is necessary. This baseline risk assessment may also be used qualitatively to identify site conditions (chemicals, exposure pathways, locations) of greatest potential concern.

According to USEPA guidelines (USEPA, 1989), the baseline risk assessment generally consists of four basic steps summarized below:

> **Hazard Identification.** Determination of the nature and amount of chemicals that could potentially be encountered at a site, and selection of those chemicals that are of potential concern for the assessment of the impact on human health.
>
> **Exposure Assessment.** Quantification of the extent, frequency, and duration of actual or potential exposure to chemicals by pathways relevant to a site and the activities of potential receptors.
>
> **Toxicity Assessment.** Identification of the types of health effects that could be associated with exposure to these chemicals, determination of the relationship between exposure (dose) and the probability of occurrence of the health impact (response).
>
> **Risk Characterization.** Estimation of the probability that an adverse health impact may occur as a result of exposure to chemicals in the amount and by the pathways identified and the uncertainty in those estimates.

The baseline human health risk assessment for the study area was conducted using methodologies required by USEPA guidelines (USEPA, 1989; 1992; 1993; 1994b; 1995; 1996; 1997a; 2001; 2002b; 2002c; 2003a; 2004c; and 2004d). A baseline risk assessment is intended to be site-specific; therefore, site-specific information was incorporated into the evaluation whenever available. In the absence of site-specific information, default assumptions, as specified by USEPA guidance, or professional judgement were used.

The baseline human health risk assessment provides estimates of risk, under both current use and hypothetical future use scenarios, to both the central tendency (CT) receptor and the reasonable maximum exposure (RME) receptor. The CT receptor is used to represent average exposures occurring at a station while the RME receptor is used to represent the maximum (upper-bound) exposure that is reasonably expected to occur at a station. Exposure pathways and exposure routes are selected based on current and future land use. Exposure assessments model human exposure by these pathways according to algorithms in relevant guidelines. Variables contributing most to estimates of risk or to the uncertainty in the risk assessment have been identified. Each of these steps is discussed in more detail in the appropriate sections of the report.

This baseline human health risk assessment consists of several sections. Section 3.2, Hazard Identification, describes the environmental samples used for the risk assessment, the selection of chemicals of potential concern (COPCs) from among the chemicals identified at the study area, and the determination of exposure point concentrations (EPCs). Section 3.3, Exposure Assessment, describes the selection of receptors and exposure pathways to be evaluated and the calculation of dose to the receptors selected. Section 3.4, Toxicity Assessment, summarizes the toxicity of the COPCs including both potential carcinogens and noncarcinogens. Section 3.5, Risk Characterization, includes a summary of study area risks and an uncertainty analysis. Table 3-1 (Selection of Exposure Pathways) provides a conceptual model for the study area, identifying the exposure media, exposure points, receptors, and routes of exposure quantitatively evaluated as part of the baseline human health risk assessment.

3.1.2 Identification of Current/Future Exposure Stations

Sampling stations have been identified and described in Section 2.1.1. For the baseline human health risk assessment, all stations previously identified have been considered for evaluation.

Because both surface soil and sediment have been collected from some sampling stations, a small number of sampling stations identified in Section 2.1.1 have been combined to evaluate potential cumulative risk to human receptors. These stations include:

- stations 07 (sediment) and DP (surface soil) (station 07/DP); and
- station WS (sediment) and WSS (surface soil) (station WS/WSS).

Due to the proximity of some stations to one another or the availability of specific future use information, other stations identified in Section 2.1.1 have been combined into a single exposure area to evaluate potential risk to human receptors. These stations include:

- stations 12 (sample SD-12-01-ME only) and WH (station WH);
- stations 13 and TT-27 (station 13/TT-27);
- stations 16 and TT-33 (station 16/TT-33);
- stations 19 (sample SD-19-01-ME only) and WG (station WG);
- stations 22 and TT-22 (station 22/TT-22);
- stations BW, WG, TT-29, 19, and sample SD-12-01-ME (station NT-1);
- stations WG, 12, and sample SD-19-01-ME (station NT-2);
- station WG and samples SD-12-01-ME and SD-19-01-ME (station NT-3); and
- station 06 and sample JP-01 (station 06).

The naming designations that will be used for the remainder of the human health risk assessment for these combined stations appear in parentheses above. Table C.1-1 in Appendix C.1 identifies the stations of the study area considered for evaluation in the baseline human health risk assessment under current and potential future land use conditions. No station is defined as station 17.

Current Exposure Stations. Local residents are potentially utilizing the Aberjona River for current recreational activities at stations:

- NR, 14, 22/TT-22, WH, WG, WS/WSS (reach 1);
- TT-30, CB-01 through CB-07, 16/TT-33, 09, DA, and AM (upper reach 2);
- KF and 08 (lower reach 2);
- 07/DP, LP, and AS (reach 3);
- 06 (reach 4);
- 05 and AJRW (reach 5);
- 04, 03, and 02 (upper reach 6); and
- 01 (lower reach 6; Sandy Beach).

Each of these stations is currently accessible to human receptors since no barriers are in place to prevent access to the river or the area surrounding the river. These stations are in areas of mild to moderate vegetation, generally shallow (i.e., less than two feet) and slow moving surface water, and gradual river banks with few, if any, physical barriers present (e.g., fencing or other access obstacles). Table 2-2 provides a listing of the sediment samples collected at each station and the depth of surface water above each sediment collection point. This information is important since only sediment data from surficial near-shore locations should be used to evaluate direct contact exposures in a human health risk assessment (USEPA, 1989). Therefore, sediment sampling locations below greater than two feet of surface water have not been quantitatively evaluated in this risk assessment.

For some of these current stations, individual sediment sampling locations have been excluded from use in the baseline human health risk assessment due to the presence of more than two feet of surface water above the sediment sampling location, the presence of physical barriers and/or the distance of the sampling point from the shoreline. At three of these stations, all sediment sampling locations are considered difficult to access due to the presence of surface water above sediments in excess of two feet. The three stations where current surface water contact is possible, but sediment contact is not likely are stations 06 (Judkins Pond), 04 (the upper forebay of Mystic Lake), and 02 (the lower forebay of Mystic Lake). For these stations, surface water in excess of two feet is present a short distance from a rocky and steep river bank, preventing sediment contact by humans. The rationale for exclusion of individual sediment sampling points is presented in Appendix C.1 on Table C.1-1.

Other stations of the river are currently difficult to access due to the presence of dense vegetation, steep banks, deep water, soft organic deposits or physical barriers to access such as fencing. Due to their current low human exposure potential, these stations have not been evaluated as current sediment exposure areas in the quantitative human health risk assessment. The stations where sediment is considered currently inaccessible to humans are presented on Table C.1-1. Surface water, if collected at these stations, is considered accessible due to its mobile nature, rendering exposures possible at the waters edge after the surface water migrates to the banks.

A number of these currently difficult-to-access stations where sediment contact has not been quantitatively evaluated are within the Wells G&H 38-acre wetland, including stations 10, 11, 12, 13/TT-27, 15, 18, 19, 20, 21, WW, JY, BW, TT-28, and TT-29. Station 10 is located in an area 20 to 30 feet from shore in ponded water approximately 2 feet in depth. There is an additional 2 to 3 feet of soft organic deposits above the depositional sediments. Hip waders, which would prevent contact with sediments, would be required to access this station. Stations 11, 12 (except sample SD-12-01-ME), 15, 18, 19 (except sample SD-19-01-ME), 20, 21, BW, TT-28, and TT-29 are all located in areas at least 30 feet from shore in dense phragmites and underbrush. Many sampling locations at these stations also contain soft organic deposits above the depositional sediments. Humans would access these stations with significantly less frequency, if at all, than the stations located along the edges of the wetland, which are considered currently accessible. In addition, for some of the sampling locations, standing water above sediments exceeds 2 feet. Stations JY, WW, and 13/TT-27 are classified as currently difficult to access due to the presence of an intact fence and concrete wall preventing access to the junkyard (station JY), and locked gates controlling access to the soil-vapor extraction system operating as part of Wells G&H OU1 (controlling current access to stations WW and 13/TT-27). These conditions cause these stations to be classified as currently difficult to access.

In the Cranberry Bog, sediment sampling stations TT-31 and TT-32 are classified as currently difficult to access. Station TT-31 is located behind commercial buildings in an area where access is restricted by a fence at the nearby trucking company. Station TT-32 is located within an area more than 75 feet from shore, along the stream channel in the center of the bog, in dense phragmites and briars. Again, humans would access these stations with significantly less frequency than the stations located along the edges of the Cranberry Bog, which are considered currently accessible. These conditions cause these stations to be classified as currently difficult to access.

Within reaches 4, 5, and 6, sediment sampling stations MP, UF, AO, LF, UM, and LM are considered currently difficult to access. Station MP is located in Winchester at the Town Hall Pond. Sediment samples at this station were collected approximately 20' from shore in water in excess of 2 feet. A dam located at this pond maintains the water level such that human contact with sediments is unlikely. Stations UF, AO, LF, UM, and LM are all located within Mystic Lake. Surface water depth at these stations ranges from 4 to 7 feet (at stations AO, UF, and LF) to 50 to 85 feet (at stations UM and LM). Therefore, due to the presence of deep surface water above sediments, these stations were classified as difficult to access.

Future Exposure Stations. Those stations considered currently accessible to humans are considered accessible under future land use conditions as well. However, some of the stations classified as currently difficult to access may become accessible in the future should the Wells G&H wetland area be developed by the City of Woburn or should the physical barriers preventing current access be removed. Therefore, stations NT-1, NT-2, NT-3, 13/TT-27, JY, WW, and TT-31 have been quantitatively evaluated as potential future exposure areas.

Stations NT-1, NT-2, and NT-3 are being evaluated based on potential redevelopment plans proposed by the City of Woburn for the Wells G&H wetland area. Exposures at these stations assume future development of the wetland area for passive recreational use including the construction of a nature trail in the 38-acre wetland that might also include an elevated walkway into the wetland or a pier extending into the wetland near Well H (stations NT-1 and NT-2, respectively). Construction of a nature trail only (adjacent to the wetland) is the exposure scenario assumed for station NT-3. The "Wells G&H Superfund Redevelopment Initiative Advisory Committee Information Package", dated September 4, 2002, contains draft plans for the development of the wetland and the construction of the nature trail, elevated walkway, and/or pier. The information package is included in Appendix C.2. Figures 1 (station NT-1) and 2 (station NT-2) of the information package show the proposed locations of the nature trail with boardwalk or pier, respectively. Figure 3 (station NT-3) of the information package shows the location of the proposed nature trail without access to the wetland. These figures in Appendix C.2 serve as the basis for the sample groupings identified for these three human health exposure stations.

Stations JY, WW, and 13/TT-27 are considered accessible in the future, because future development may result in the removal of the fencing, locked gates, and concrete wall currently preventing access. Likewise, TT-31 may become accessible should the fencing and buildings currently preventing access be removed. Future development of the surrounding areas near stations JY, WW, 13/TT-27, and TT-31 would also increase the exposure potential of these stations.

Because the remaining stations are considered difficult to access due to distance from the shoreline and/or the presence of dense vegetation, steep banks, and/or deep water, stations BW, 10, 11, 12, 15, 18, 19, 20, 21, AO, LM, UF, LF, UM, MP, TT-28, TT-29, and TT-32 are considered to remain inaccessible in the future. The conditions limiting access currently are unlikely to change in the future. No quantitative human health evaluation has been conducted for these stations.

3.2 HAZARD IDENTIFICATION

The purpose of this section is the determination of the type and amount of chemicals present at the study area and the selection of the COPCs with regard to human health. In addition, this section summarizes the methodology used to determine EPCs for COPCs in each medium.

3.2.1 Reference Stations

Reference samples for surface water, sediment, and fish fillet tissue were collected as part of investigational activities conducted for the study area. In addition, surface water, sediment, and fish fillet reference samples collected in support of the Industri-Plex Superfund Site Investigation have been used to augment the study area reference investigation. Reference stations are identified as stations 23 through 27, 01-IP through 04-IP, 12-IP, HB, and SA. Their locations are shown relative to the study area in Section 2.0, Figure 2-1, and by station in Figures 2-11 through 2-20. For both efforts, samples were collected from areas not considered to be affected by site activities and not displaying visual evidence of contamination. Table C.1-1 in Appendix C.1 identifies the reference stations, individual sediment samples collected at each station, and the rationale for any individual sediment samples excluded from consideration in the human health risk assessment.

The reference data for the media evaluated for human exposures are presented in data summary tables in Appendix C.3; Table C.3-2.1 for surface water, Table C.3-2.5 for sediment, and Table

C.3-2.9 for fish fillet tissue. Analytical data for individual reference samples are presented in Section 2.0, Tables 2-14, 2-59 and 2-195 for surface water, sediment, and fish fillet tissue, respectively. Reference analyte concentrations do not impact the selection of COPCs (subsection 3.2.3) or EPCs (subsection 3.2.4).

3.2.2 Data Used in Risk Assessment

Detailed discussions of sampling methodologies and the quality assurance and control activities implemented during the collection of the data are provided in Sections 2.1.2 through 2.1.4. The sampling data were validated according to USEPA Region 1's Contract Laboratory Program (CLP) procedures and guidelines, as described in Section 2.1.4. The analytical results are discussed in Section 2.2. The following process used to summarize the analytical data is in accordance with *Risk Assessment Guidance for Superfund (RAGS)* (USEPA, 1989) and supplemental guidance (USEPA, 1992).

The analytical data were summarized by environmental medium and grouped into exposure areas (i.e., stations or reaches). For the baseline human health risk assessment, thirty-seven current and/or future exposure areas were selected for quantitative evaluation: Stations NR, 14, 22/TT-22, 13/TT-27, WH, NT-1, NT-2, NT-3, WG, WW, JY, WS/WSS, TT-30, TT-31, CB-01 through CB-07, 16/TT-33, 09, DA, AM, KF, 08, 07/DP, LP, AS, 06, 05, AJRW, 04, 03, 02, and 01. Exposures at these stations are assumed to vary over time, depending on changes in land use, and to vary in intensity and frequency based on accessibility, types of recreational activities reported or anticipated to occur, and proximity to residences. It is unlikely that human exposures will occur at other locations within the study area, as previously discussed.

3.2.2.1 Surface Water. Surface water samples were collected and analyzed in 1995 within a sampling period extending from July 31 to September 12. A total of 17 stations were sampled during this period. For the baseline human health risk assessment, surface water samples have been grouped into river reaches or reach segments. This approach acknowledges that surface water is a mobile medium. Therefore, humans accessing a specific station may encounter surface water from all sampling locations within the vicinity of that station (i.e., a river reach or reach segment). Surface water samples available for evaluation at stations within each river reach or reach segment have been grouped as follows:

Reach 1	(SW-10-01 through SW-15-01 and SW-18-01);
Upper Reach 2	(SW-09-01 and SW-16-01);
Lower Reach 2	(SW-08-01);
Reach 3	(SW-07-01);
Reach 4	(SW-06-01);
Reach 5	(SW-05-01);
Upper Reach 6	(SW-02-01 through SW-04-01); and
Lower Reach 6	(SW-01-01).

No surface water samples were collected from stations:

- NR, 22/TT-22, WH, NT-1, NT-2, NT-3, WG, WW, JY, and WS/WSS (reach 1)
- TT-30, TT-31, CB-01 through CB-07, DA, and AM (upper reach 2)
- KF (lower reach 2)
- LP and AS (reach 3).
- AJRW (reach 5).

However, surface water exposures at these stations were evaluated using surface water samples collected within the same river reach or reach segment. This approach recognizes the mobile nature of surface water and its exposure potential at nearby areas.

Analytical results of detected compounds for each of these samples are shown in Table 2-33. The surface water sample results are summarized in Table 3-2.1, and station locations are shown in Section 2.0, Figures 2-3 through 2-10. A comprehensive reporting of these data may be found in Appendix B, Table B-1, and in the *Preliminary Data Compendium, Wells G&H RI/FS, Operable Unit III, Aberjona River Study* (Foster Wheeler, 1996; Appendix A). Surface water sample results are summarized by reach or reach segment on Tables 3-2.1.1 through 3-2.1.8.

Supplemental baseflow surface water grab samples were collected from six surface water gauging stations within the study area between July 2001 and October 2002. Storm event data were also collected from the surface water gauging stations in 2002 during April, May, July, August, September, and October storm events. The surface water gauging stations, designated SW-05 through SW-10, are shown on Figure 2-31. Gauging stations SW-05, SW-06, and SW-07 were located at the Salem Street Bridge (at the juncture of reaches 1 and 2), downstream of the Montvale Avenue bridge (reach 3), and at the Swanton Street bridge (reach 4), respectively. Gauging station SW-08 was located at the USGS gauging stations on the Mystic Valley Parkway (reach 5). Gauging stations SW-09 and SW-10 were located at the outlets to the Upper (upper reach 6) and Lower (lower reach 6) Mystic Lakes, respectively. Surface water data from the gauging stations have also been grouped into river reaches as follows:

Reach 1	SW-05;
Upper Reach 2	SW-05;
Lower Reach 2	SW-06;
Reach 3	SW-06;
Reach 4	SW-07;
Reach 5	SW-08;
Upper Reach 6	SW-09; and
Lower Reach 6	SW-10.

Analytical results of detected compounds for the baseflow and storm event samples are shown in Tables 2-228 through 2-233 and Tables 2-240 through 2-245, respectively. The baseflow and storm event surface water sample results for each gauging station are summarized in Table 3-2.7. A comprehensive reporting of these data may be found in Appendix B.6.

3.2.2.2 Sediment. Between one and 22 sediment samples were collected from each of the human health exposure areas during five rounds of sampling. However, because human exposures are likely to occur only to sediments located below two feet or less of standing water, a subset of the collected sediment samples were used in the quantitative human health risk assessment. The sediment samples collected from appropriate sampling depths (below two feet or less of standing water) and applied to the human health risk assessment are collectively summarized in Table 3-2.2. Station AJRW sediment results are summarized separately in Table 3-2.5. All sediment samples collected from within the study area, including those applicable to the human health risk assessment, are presented in Appendix C.1, Table C.1-1. This table also provides the rationale for the exclusion of specific samples from the human health risk assessment. Sediment sampling locations are shown in Section 2.0, Figures 2-3 through 2-10. On Figures 2-3 through 2-10, samples highlighted in yellow represent the 1995 sampling round, samples highlighted in blue represent the 1997 sampling round, samples highlighted in pink represent the 2001 triad sampling event, and samples highlighted in white represent the 2000-2002 and supplemental sampling rounds.

In 1995, the first round of sediment samples was collected from an appropriate sampling depth (below two feet or less of standing water) at stations 01, 03, 05, 07/DP, 08, 09, 13/TT-27, 14, 16/TT-33, and 22/TT-22. At each of these stations, between two and ten sediment samples were collected. The second round of sediment sampling was conducted in November 1997 and included the collection of one additional sediment sample collected from an appropriate sampling depth at stations 03, 05, 12, 19, and 22/TT-22 and two additional samples of appropriate depth at stations 07/DP and 13/TT-27. The third round of sediment sampling was conducted between July

2000 and August 2001. During this third round, between ten and 22 sediment samples of appropriate depth were collected from stations WS/WSS, WG, WH, WW, CB-01, CB-02, and CB-03; three additional samples were collected from station 22/TT-22; three samples were collected from station 13/TT-27; and one sample was collected from each of stations AS and AM. The fourth round of sediment sampling was conducted between July 2002 and September 2002. The fourth sampling round included the collection of: (1) five samples from stations NR and BW; (2) ten samples from stations JY, CB-04, CB-06 and KF; (3) six samples from station CB-07; and (4) 15 samples from station LP. In June 2001, triad sediment sampling was conducted for the ecological risk assessment. This effort included the collection and analysis of one sample of appropriate depth from stations 19, 22/TT-22, WH, 13/TT-27, TT-29, WW, and 16/TT-33. Finally, in response to comments received on the draft risk assessment report, supplemental sampling was conducted in 2004 in Winchester, south of Bacon Street, that included the collection of nine shallow sediments from station AJRW.

As previously noted, no sediment samples were collected from an appropriate depth during any sampling round for stations 02, 04, or 06. All sediment samples at these stations were collected from below a depth of greater than two feet of standing water. In addition, sediment sample SD-TT-27-04 has not been included because it was collected from a tributary outside the 38-acre wetland. This tributary does not contain elevated levels of site-related contaminants and, therefore, is not representative of the other station 13/TT-27 sediment samples. Sediment sample results are summarized by station on Tables 3-2.2.1 through 3-2.2.32. Sediment sample results are summarized for station AJRW on Table 3-2.5.

Analytical results of detected compounds for each of the sediment samples are presented in Section 2.0, Table 2-161. Table 2-247 provides the analytical results of detected compounds for station AJRW. A comprehensive reporting of the 1997 supplemental data ("-ME" samples) is provided in the *Supplemental Data Compendium for Wells G&H Superfund Site, Aberjona River Study (Operable Unit 3)* (M&E, 1998; Appendix A). A comprehensive reporting of the 1995 data (“-FW” samples) may be found in the *Preliminary Data Compendium, Wells G&H RI/FS, Operable Unit III, Aberjona River Study* (Foster Wheeler, 1996; Appendix A). Appendix A of this report also contains field notes associated with the triad sampling effort (“-TR” samples) and the more recent 2000-2002 and the 2004 supplemental sampling rounds. A comprehensive reporting of the sediment data is provided in Appendix B, Table B-2. Appendix B.6 presents the 2004 supplemental sediment data for station AJRW.

3.2.2.3 Floodplain Surface Soil. Floodplain surface soil samples were collected from stations NR, WS/WSS, CB-05, DA, KF, 07/DP, and AJRW. All samples were collected from the 0-6" interval to characterize potential contamination within the river floodplain. The first round of floodplain surface soil samples were collected and analyzed in 2000 within a sampling period extending from July 31 to September 12. During this first round, 20 floodplain samples were collected from station 07/DP. A second round of surface soil sampling was conducted during the summer of 2002 to further characterize the floodplain along the river. The second round included the collection of: (1) five top-of bank samples from station NR; (2) five top-of-bank samples from station WS/WSS (in the backyard of the Salem Street residence); (3) ten samples along a walking trail within the floodplain area of station CB-05; (4) five top-of-bank samples from station DA; and (5) ten top-of-bank samples from station KF. In response to comments received on the draft risk assessment report, supplemental sampling was conducted in 2004 in Winchester, south of Bacon Street, that included the collection of nine top-of-bank samples from station AJRW.

Surface soil samples collected during the three sampling rounds include:

Station NR	(SO-NR-16 through SO-NR-20 with duplicate for sample -18);
Station WS/WSS	(SO-SS-01 through SO-SS-05 with duplicate for sample -02);
Station CB-05	(SO-CB-05-01 through SO-CB-05-10 with duplicate for sample -10);

Station DA	(SO-DA-01 through SO-DA-05 with duplicate for sample -01);
Station KF	(SO-KF-01 through SO-KF-10 with duplicate for sample -05);
Station 07/DP	(SO-DP-01 through SO-DP-26 with duplicates for samples -06, -20 and -26); and
Station AJRW	(SO-AJRW-08, -10, -12, -14, -16, -18, -20, -22, and -24 with duplicate for sample -10).

Analytical results of detected compounds for each of these samples are shown in Table 2-175. Analytical results of detected compounds at station AJRW are shown in Table 2-249. The surface soil sample results are summarized in Table 3-2.3, except for station AJRW data which are summarized in Table 3-2.6. Sample locations are shown in Section 2.0, Figures 2-3, 2-4, 2-6, 2-7, and 2-10. A comprehensive reporting of these data may be found in Appendices B.3 and B.6. Surface soil sample results are summarized by station on Tables 3-2.3.1 through 3-2.3.6 and Table 3-2.6 (station AJRW).

3.2.2.4 Fish Fillet Tissue. The risks associated with fish fillet ingestion were evaluated for reaches 3 through 6 of the study area. No fish fillet tissue was available for reaches 1 and 2 due to the lack of large fish present in these reaches. Consequently, no fish fillet tissue was collected from those reaches. Reaches 4 and 5 were combined since these reaches support similar species and sizes of fish, and both reaches are located in close proximity to populated areas of Winchester. Reaches 3 and 6 were evaluated as distinct exposure areas where anglers might limit their sport to either pond (Davidson Park/Leonard Pool) or lake (Upper Mystic Lake) fishing. Fillet samples obtained within these three exposure areas (reaches or combination of reaches) were not separated by species under the assumption that individual anglers do not target one particular type of fish. Therefore, three separate species/locations were identified and are listed below:

Reach 3	fillet from brown bullheads, largemouth bass, and yellow perch;

Reaches 4 and 5	fillet from largemouth bass; and
Reach 6	fillet from brown bullheads and largemouth bass.

Fish tissue sampling was conducted in 1995 within the time period extending from July 31 to September 12. Even though whole body, offal, and crayfish samples were collected from the study area, it was assumed that human receptors reasonably consume only fish fillet tissue. Fish fillet samples were collected from four of the six river reaches encompassing the study area (reaches 3, 4, 5 and 6). These river reaches are described in detail in Section 1.0, subsection 1.3.1. Fillet samples evaluated quantitatively in the human health risk assessment include:

Reach 3	LF-LK-06-F through LF-LK-10-F and LF-RV-01-F through LF-RV-04-F;
Reaches 4 and 5	LF-LK-11-F through LF-LK-14-F, LF-RV-05-F, LF-RV-06-F, LF-RV-10-F, and LF-RV-13-F through LF-RV-15-F; and
Reach 6	LF-LK-20-F through LF-LK-25-F.

Analytical results of detected compounds for each of the fish fillet samples are shown in Section 2.0, Table 2-201. Fillet tissue sampling results are collectively summarized in Table 3-2.4 and fillet sample results are summarized by reach in Tables 3-2.4.1 (reach 3), 3-2.4.2 (reaches 4 and 5), and 3-2.4.3 (reach 6). A comprehensive reporting of these data may be found in Appendix B, Table B-4, and in the *Preliminary Data Compendium, Wells G&H RI/FS, Operable Unit III, Aberjona River Study* (Foster Wheeler, 1996; Appendix A). Table A.1-2 in Appendix A.1 of this report identifies the fish fillet samples collected and the corresponding river reach.

3.2.2.5 Sediment Cores. Sediment core samples were collected at nine locations (SC05 through SC13) within the study area. Figure 2-34 demonstrates the location of the individual sediment core samples. At each sediment core location, samples were collected from four discrete intervals: 0-1 feet, 1-2 feet, 2-3 feet, and 3-4 feet. All samples were collected in February of

2003. SC05 through SC08 were collected from the Wells G&H 38-acre wetland, SC09 and SC10 from the former Cranberry Bog, SC11 from Davidson Park, SC12 from Judkins Pond, and SC13 from the outlet of the river as it flows into the upper forebay of Upper Mystic Lake.

Analytical results of detected compounds for each of the sediment core samples are presented in Section 2.0, Tables 2-259 through 2-267. The sediment core sample results are summarized in Table 3-2.8. Appendix A.7 of this report contains field notes associated with the 2003 sediment core sampling effort. A comprehensive reporting of the sediment core data is provided in Appendix B.6.

3.2.2.6 Data Evaluation. The data were qualified by the analytical laboratory and validated as described in Section 2.0, subsection 2.1.4. The qualification and validation of the analytical data included a comparison of the study area data to corresponding blank (laboratory, field, equipment, and trip) concentration data. Data rejected by the validation ("R" qualified) were not used. Estimated values (e.g., "J" qualified) were used in the risk assessment without modification. Analytical data from duplicate samples were combined as described in Section 2.0, subsection 2.1.5. Frequency of detection was calculated as the number of samples in which the chemical was detected over the total number of samples analyzed after the exclusion of rejected ("R" qualified) data.

Since sediment samples were collected during multiple sampling rounds, more than one set of analytical results were available for some sampling locations. For these sampling locations, the multiple results were treated as unique samples rather than as duplicate samples (i.e., the multiple results were not averaged as described in subsection 2.1.5 for duplicates). Therefore, in determining the frequency of detection for sediment, the analytical results from the different sampling rounds were considered as separate values.

All surface water and 1995/1997 sediment samples were analyzed for TCL organics including VOCs, SVOCs, pesticides/PCBs, and TAL inorganics. After reviewing the 1995/1997 comprehensive analyses, metals were determined to be the primary contaminants detected within the study area. Therefore, subsequent analyses were narrowed to TAL inorganics. Sediment samples collected in 2000-2004 were analyzed for TAL inorganics. Fish fillet tissue samples were analyzed for SVOCs, pesticides/PCBs, and TAL inorganics. Surface soil samples were analyzed for TAL inorganics only.

In February and June 2001, a subset of the 2000/2001 sediment sampling locations were sampled for chromium VI analysis using a colorimetric method (Method 7196A). Due to limitations with the analytical method that resulted in some data rejection, additional sediment samples were collected in October 2002 for chromium VI analysis using ion chromatography (Method 7199). These data support the opinion that it is unlikely that chromium VI exists in sediments at appreciable levels. Chromium VI results and data validation information have been included in Appendix C.4. Chromium VI was only present at a very low concentration (17.3 mg/kg) in a single sample (SD-WW-06) that contained a significantly elevated total chromium value (13,400 mg/kg). Samples with total chromium results less than 930 mg/kg were all non-detect for chromium VI.

Using the information from the 2002 ion chromatography study, it can be conservatively inferred that approximately 0.13% (17.3 mg/kg of 13,400 mg/kg) of total chromium in sediments is present as chromium VI. Chromium VI levels can be assumed to be non-detect for stations with maximum total chromium results less than 930 mg/kg. Therefore, for sediment stations with maximum total chromium results in excess of 930 mg/kg, 0.13% of the total chromium value for each sample was assumed to be present as chromium VI unless station-specific chromium VI analysis had been performed.

Stations with maximum total chromium results in excess of 930 mg/kg include stations WH, NT-1, NT-2, NT-3, WG, WW, JY, WS/WSS, TT-30, CB-06, and 16/TT-33. Of these stations, NT-2, NT-3, and WG have station-specific results demonstrating that chromium VI was non-detect at the location of the maximum detected total chromium value. Therefore, total chromium at these stations is assumed to be present as chromium III. For stations WH, NT-1, WW, JY, WS/WSS, TT-30, CB-06, and 16/TT-33, chromium VI values were calculated by assuming that 0.13% of the total chromium result represents chromium VI. The calculated chromium VI results for these eight station are presented on Tables 3-2.2.5, 3-2.2.6, 3-2.2.10. 3-2.2.11, 3-2.2.12, 3-2.2.13, 3-2.2.19 and 3-2.2.21. Because chromium VI analysis was not performed for surface water, floodplain surface soil or fish fillet tissue, all chromium detected in these media are conservatively assumed to exist as chromium VI. For the 2003 sediment core samples, chromium VI analysis was performed using the ion chromatography method. Therefore, total chromium results for the sediment core samples have been quantitatively evaluated as chromium III.

Summary tables (Tables 3-2.1 through 3-2.8) for chemicals detected in each of the environmental media provide the frequency of detection, range of detection limits, range of detected concentrations and location of maximum detected result for all analytes. In addition, Tables 3-2.2.1 through 3-2.2.32 and Tables 3-2.3.1 through 3-2.3.6 provide a summary of chemicals detected in sediment and floodplain surface soil, respectively, for each of the stations quantitatively evaluated in the human health risk assessment. Station AJRW sediment and floodplain surface soil data are summarized separately in Tables 3-2.5 and 3-2.6, respectively. Tables 3-2.1.1 through 3-2.1.8 and Tables 3-2.4.1 through 3-2.4.3 provide a summary of chemicals detected in surface water and fish fillet tissue for each of the reaches/reach segments quantitatively evaluated in the human health risk assessment. Baseflow and storm event surface water data and sediment core data are summarized separately by sampling location in Tables 3-2.7 and 3-2.8, respectively.

3.2.3 Identification of COPCs

The scope of the baseline human health risk assessment includes identification of COPCs based on the chemical substances found at the study area. The list of COPCs was developed using the simple screening process described below. For surface water, sediment, and surface soil, all available and appropriate data for the human health exposure areas were combined for each media to select COPCs, except for the supplemental sampling data. All fish fillet tissue data from the four reaches of the river were combined to select fish fillet tissue COPCs. Combining all data for a given medium results in a conservative list of COPCs for that medium. COPCs were selected separately for the supplemental sampling data which includes 2004 station AJRW sediment and floodplain surface soil data, 2001/2002 baseflow and storm event surface water data, and 2003 sediment core data.

3.2.3.1 Selection Criteria. The maximum detected concentration of a chemical in surface water, sediment, or surface soil was compared to preliminary remedial goals (PRGs) published by USEPA Region 9 (USEPA, 2002a). Because Region 9 has not developed PRGs for the fish ingestion pathway, the maximum detected concentration of a chemical in fillet tissue was compared to USEPA Region III risk-based concentrations (RBCs; USEPA, 2004b). Both PRGs and RBCs are chemical concentrations back-calculated using toxicity criteria and either a 1×10^{-6} target risk level for potential carcinogens or a hazard quotient (HQ) of 1 for noncarcinogens. For purposes of this screening analysis, a HQ of 0.1 was used to add a ten-fold measure of safety to reduce the chance of omitting chemicals from the list of COPCs that could contribute to a total hazard index (HI) of 1. To accomplish this, PRGs and RBCs for noncarcinogenic chemicals were divided by 10 prior to comparison to maximum detected values. Tap water PRGs, residential soil PRGs, and fish RBCs were used for comparison to maximum detected surface water, sediment, and fish fillet tissue concentrations, respectively. Residential soil PRGs were also used for comparison to maximum detected surface soil concentrations. The comparison of surface water concentrations to tap water PRGs provides a conservative screening evaluation. Ambient Water Quality Criteria (AWQCs) (USEPA, 2002d) developed to be protective of human health following the ingestion of water and organisms from fishable surface water bodies, were also used as screening criteria for surface water.

A maximum detected study area chemical concentration less than its screening value indicated that the excess lifetime cancer risk associated with exposure to that chemical concentration would be less than one in one million and the HQ associated with exposure would be less than 0.1. Chemicals detected at concentrations below their screening criteria (and also below AWQCs for surface water) were, therefore, eliminated from further evaluation. All chemicals with maximum concentrations greater than the relevant screening criteria (or relevant AWQCs for surface water) were selected as COPCs. Comparisons of maximum concentrations to screening criteria are presented in the data summary tables for each medium (Tables 3-2.1, 3-2.2, 3-2.3, and 3-2.4 for surface water, sediment, surface soil, and fish fillet tissue, respectively). This comparison is presented for the supplemental sampling data in Table 3-2.5 (station AJRW sediment), Table 3-2.6 (station AJRW surface soil), Table 3-2.7 (baseflow and storm event surface water data), and Table 3-2.8 (sediment core data). For certain analytes lacking compound-specific screening criteria (e.g., endrin aldehyde), a surrogate compound was selected (e.g., endrin) and its screening criteria was used for COPC screening. Specific instances where surrogate assignments were made are identified in footnotes on Tables 3-2.1 through 3-2.8.

For four essential human nutrients that lacked screening criteria (i.e., calcium, magnesium, potassium, and sodium), the maximum detected concentrations were compared to concentrations in drinking water, soil, and fish fillet tissue that would not significantly increase the dietary Allowable Daily Intakes (ADIs), as follows: for calcium (400,000 Wg/l water; 1,000,000 mg/kg soil; 50,000 mg/kg fish); for magnesium (805,000 Wg/l water; 1,000,000 mg/kg soil; 100,630 mg/kg fish); for potassium (100,000 Wg/l water; 1,000,000 mg/kg soil; 12,500 mg/kg fish); and for sodium (100,000 Wg/l water; 1,000,000 mg/kg soil; 12,500 mg/kg for fish). Derivations of these ADIs are provided in Appendix C.5. Back-calculated soil concentrations for magnesium and calcium have been adjusted so as not to exceed 1,000,000 mg/kg (i.e., 100%). If no concentrations exceeded the ADIs, these chemicals were not further evaluated.

Since PRGs and RBCs were not available for lead, the maximum detected lead concentration in sediment and surface soil was evaluated relative to the residential soil screening level of 400 mg/kg (USEPA, 1994a). The maximum lead concentration in surface water was evaluated relative to a drinking water concentration of 15 Wg/l, a criterion protective of blood lead levels in children (USEPA, 2003b). No lead screening criterion is available for fish fillet tissue. Therefore, lead has been selected as a COPC for this medium. Detected lead levels in fish fillet tissue will be discussed relative to average background fish tissue levels for lead (0.34 mg/kg; USEPA, 1994c) in the risk characterization (subsection 3.5.2.3).

Three additional inorganic chemicals, aluminum, iron, and cobalt, were eliminated as COPCs at the direction of USEPA (1998) because Region 1 does not concur with the provisional toxicity criteria provided by the Superfund Technical Support Center for these analytes. These metals are abundant in the earth's crust and are unlikely to cause substantial toxicity at concentrations commonly encountered.

3.2.3.2 Chemicals Selected as COPCs. This subsection describes the chemicals selected as COPCs and refers to lists of the selected chemicals.

COPCs in Surface Water. Surface water analytical results from the study area stations with available data are summarized in Table 3-2.1. Supplemental baseflow and storm event surface water data are summarized in Table 3-2.7. Tables 3-2.1 and 3-2.7 list all chemicals detected in surface water from these stations as well as the chemicals selected as COPCs in surface water based on comparison to tap water PRGs and human health AWQCs. For the 1995 surface water data, the maximum detected results for chloroform, tetrachloroethene (PCE), trichloroethene (TCE), heptachlor epoxide, arsenic, cadmium, chromium, lead, manganese, and mercury exceed their respective PRGs and/or AWQCs and were selected as surface water COPCs. No essential nutrients were detected at maximum concentrations in excess of their respective ADIs for surface water. For the 2001/2002 surface water gauging stations, baseflow and storm event COPCs are summarized as follows:

SW-05 Baseflow: arsenic, chromium, manganese, mercury
Storm event: antimony, arsenic, chromium, manganese, mercury

SW-06 Baseflow: arsenic, chromium, manganese
Storm event: antimony, arsenic, chromium, lead, manganese, mercury

SW-07 Baseflow: arsenic, manganese
Strom event: arsenic, chromium, lead, manganese, mercury

SW-08 Baseflow: arsenic, manganese
Storm event: antimony, arsenic, chromium, lead, manganese, mercury, thallium

SW-09 Baseflow: arsenic, manganese
Storm event: antimony, arsenic

SW-10 Baseflow: arsenic, mercury
Storm event: arsenic, mercury

COPCs in Sediment. Sediment analytical results from the study area stations quantitatively evaluated in the human health risk assessment are summarized in Table 3-2.2, except for station AJRW which are summarized in Table 3-2.5. The specific samples summarized and evaluated are listed in Appendix C.1, Table C.1-1. Except for station AJRW, Table 3-2.2 lists all chemicals detected in sediment samples from these stations as well as the chemicals selected as COPCs in sediment based on comparison to residential soil PRGs. The maximum detected results for PCE, TCE, benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(k)fluoranthene, dibenz(a,h)anthracene, indeno(1,2,3-cd)pyrene, phenanthrene, Aroclor 1254, Aroclor 1260, antimony, arsenic, barium, cadmium, chromium, chromium VI, copper, lead, manganese, mercury, nickel, thallium, vanadium, and zinc exceed their respective PRGs and were selected as sediment

COPCs. For station AJRW, arsenic, lead, manganese, and mercury were selected as sediment COPCs. No essential nutrients were detected at maximum concentrations in excess of their respective ADIs for sediment.

COPCs in Surface Soil. Surface soil analytical results from the study area are summarized in Table 3-2.3, except for station AJRW which are summarized in Table 3-2.6. Except for station AJRW, Table 3-2.3 lists all chemicals detected in surface soil as well as the chemicals selected as COPCs in surface soil based on comparison to residential soil PRGs. The maximum detected results for arsenic, cadmium, chromium, copper, manganese, mercury, thallium, vanadium, and zinc exceed their respective PRGs and were selected as surface soil COPCs. For station AJRW, arsenic, chromium, lead, manganese, mercury, and vanadium were selected as surface soil COPCs. No essential nutrients were detected at maximum concentrations in excess of their respective ADIs for soil.

COPCs in Fish Tissue. Fish fillet tissue analytical results from 25 study area samples are summarized in Table 3-2.4. Table 3-2.4 lists all chemicals detected in fish fillet tissue samples from the river as well as the chemicals selected as COPCs in fish tissue based on comparison to fish RBCs. The maximum detected results for 4,4'-DDD, 4,4'-DDE, 4,4'-DDT, aldrin, alpha-chlordane, Aroclor 1248, Aroclor 1254, Aroclor 1260, delta-BHC, dieldrin, heptachlor epoxide, antimony, arsenic, lead, mercury, and selenium exceed their respective RBCs and were selected as fish fillet tissue COPCs. No essential nutrients were detected at maximum concentrations in excess of their respective ADIs for fish.

COPCs in Sediment Cores. Sediment core analytical results from the study area are summarized in Table 3-2.8. Table 3-2.8 lists all chemicals detected in sediment core samples as well as the chemicals selected as COPCs at each sediment core locations based on comparison to residential soil PRGs. At each location, all four depth intervals (0-1 feet, 1-2 feet, 2-3 feet, and 3-

4 feet) were combined to select COPCs. The maximum detected results for the following compounds exceed their respective PRGs and were selected as sediment core COPCs:

SC05	antimony, arsenic, cadmium, copper, lead, manganese, mercury, vanadium
SC06	arsenic, cadmium, copper, lead, manganese, mercury, vanadium, zinc
SC07	arsenic, cadmium, copper, lead, manganese, mercury, vanadium, zinc
SC08	arsenic, cadmium, copper, lead, manganese, mercury, vanadium, zinc
SC09	arsenic, mercury
SC10	arsenic
SC11	arsenic, cadmium, copper, lead, manganese, mercury, vanadium
SC12	arsenic, cadmium, copper, lead, manganese, mercury, vanadium
SC13	arsenic, cadmium, copper, lead, manganese, mercury, vanadium, zinc

No essential nutrients were detected at maximum concentrations in excess of their respective ADIs for soil.

3.2.4 Determination of Exposure Point Concentrations

To evaluate the magnitude of potential human exposures, the concentration of each COPC in each exposure medium must be estimated. An estimate of this concentration is referred to as an EPC. EPCs were determined for the COPCs in each medium.

USEPA requires the use of the UCL providing 95% coverage on the arithmetic mean concentration for the estimation of both the CT and RME risk (USEPA 1989; 1992; 1994b; 2002c; and 2004c). Therefore, whenever possible, the UCL has been calculated and used as the EPC for both the RME and CT exposure cases. UCLs were calculated using USEPA's program "ProUCL Statistical Software" (Version 3.0). Appendix C.6 contains documentation for the calculation and selection of the UCL values. UCL values could be calculated by this program if four or more samples were available for summarization from a station. When less than four samples were available, the program was unable to calculate a UCL value.

When one sample was available for an exposure area, the maximum detected COPC concentrations were used as the EPCs for both the CT and RME exposure cases. When the UCL value for a COPC exceeded the maximum detected concentration because of small sample sizes or high variability or if the UCL could not be calculated (< 4 samples), the maximum detected value was used as the EPC for the RME scenario, and the arithmetic mean value was used as the EPC for the CT exposure case (USEPA 1989 and 1994b). In cases where the arithmetic mean value exceeded the maximum detected value, the maximum detected value was used as the EPC for both the RME and CT cases.

Surface Water. For surface water, each of the human health exposure areas (i.e., reaches or reach segments) were quantitatively evaluated using COPCs which were selected using all 1995 data from the stations combined (see Table 3-2.1). For current and future scenarios, Tables 3-3.1.RME and 3-3.1.CT list, by reach or reach segment, the surface water COPCs detected and the EPCs selected for the RME and CT scenarios, respectively. Since only one surface water sample was collected from lower reach 2, reach 3, reach 4, reach 5 and lower reach 6 (Sandy Beach), the arithmetic mean and UCL values were not calculated for these areas. In addition, UCLs could not be calculated for upper reach 2 and upper reach 6 since less than four samples were available for summarization.

For the 2001/2002 baseflow and storm event surface water data sets, UCL values were calculated for each gauging station since adequate numbers of surface water grab samples were available from each gauging station (Table 3-3.8.RME and 3-3.8.CT).

Sediment. For sediment, each of the human health exposure stations was quantitatively evaluated as a separate exposure location using COPCs which were selected using all data from the stations combined (see Table 3-2.2), except for station AJRW. For the current exposure scenarios, Tables 3-3.2.RME and 3-3.2.CT list the sediment COPCs detected, by station, and the EPCs selected for the RME and CT scenarios, respectively. Tables 3-3.3.RME and 3-3.3.CT identify EPCs for the future scenarios. Tables 3-3.6.RME and 3-3.6.CT identify EPCs for station AJRW. UCL values have been calculated when four or more sediment samples of appropriate depth were collected from a station. Arithmetic mean and UCL values have not been calculated for stations TT-30 and AM since only one sample was available for quantitative evaluation.

Surface Soil. For surface soil, stations NR, WS/WSS, CB-05, DA, KF, and 07/DP were quantitatively evaluated as separate exposure locations using COPCs which were selected using all surface soil data combined (see Table 3-2.3). COPCs for station AJRW were selected separately (see Table 3-2.6) For current and future scenarios, Tables 3-3.4.RME and 3-3.4.CT list the surface soil COPCs detected along with the EPCs selected for the RME and CT scenarios, respectively. Tables 3-3.7.RME and 3-3.7.CT identify EPCs for station AJRW. Arithmetic mean and UCL values have been provided because more than four surface soil samples were collected from each of the stations.

Fish Fillet Tissue. For fish fillet tissue, four reaches (reaches 3 through 6) were quantitatively evaluated using COPCs which were selected using all 1995 data from the four reaches combined (see Table 3-2.4). For current and future scenarios, Tables 3-3.5.RME and 3-3.5.CT list the fish fillet tissue COPCs detected, by river reach, along with the EPCs selected for the RME and CT scenarios, respectively. No fish fillet tissues samples were collected from reaches 1 and 2. Fish fillet samples were grouped by river reaches that might be targeted by individuals engaging in recreational fishing. The four river reaches from which fillet samples were collected are described in detail in Section 1.0, subsection 1.3.1.

Sediment Cores. For sediment cores, sampling locations SC05 through SC13 were quantitatively evaluated as separate exposure locations using COPCs which were selected individually for each location (see Table 3-2.8). For the future scenario, Tables 3-3.9.RME and 3-3.9.CT list the sediment core COPCs at each sampling location along with the EPCs selected for the RME and CT scenarios, respectively. Arithmetic mean and UCL values have been provided because four sediment core samples were collected from each sampling location.

3.3 EXPOSURE ASSESSMENT

The purpose of the exposure assessment is the quantification of the extent, frequency and duration of actual or potential exposure to chemicals by pathways relevant to the study area and activities of the potential receptors.

3.3.1 Identification of Potentially Exposed Populations and Potential Exposure Pathways

As part of the exposure assessment, current and potential future exposure pathways were determined through which identified populations may be exposed to the COPCs at the study area.

An exposure pathway describes the course a chemical follows while moving through environmental media to a receptor. An exposure pathway may consist of a mechanism of release of contaminants to an environmental medium (e.g., surface water), an exposure route (e.g., ingestion), and a receptor (e.g., recreational child). An exposure pathway is considered complete when contact by a receptor with contaminated media may occur currently or in the future. For purposes of this risk assessment, only potentially complete exposure pathways were quantitatively evaluated.

USEPA (1989 and 1991) guidance requires that plausible exposures under both current and future land-use scenarios be evaluated in a baseline risk assessment. Accordingly, potential human exposure pathways were identified for both current and potential future land-use scenarios at the study area. The current land-use scenario examines the potential for human exposure under current study area conditions, while the future land-use scenario evaluates potential exposures following possible changes in study area use (assuming no remedial action occurs).

3.3.1.1 Potential Exposure Pathways and Receptors Under Current Land-Use Conditions. Local residents along or near the Aberjona River could potentially contact contaminants in surface water, sediment, floodplain surface soil and fish tissue at specific locations while engaging in recreational activities in the river such as wading, swimming, and fishing. Table 3-1 presents a summary of the current exposure routes quantitatively evaluated in the baseline human health risk assessment as well as the human health exposure points and receptors.

As previously discussed, local residents may currently engage in recreational activities in and along the river at:

- stations NR, 14, 22/TT-22, WH, WG, WS/WSS (reach 1);
- stations TT-30, CB-01 through CB-07, 16/TT-33, 09, DA, and AM (upper reach 2);
- stations KF and 08 (lower reach 2);
- stations 07/DP, LP, and AS (reach 3);
- station 06 (reach 4);
- stations 05 and AJRW (reach 5);
- stations 04, 03, and 02 (upper reach 6); and
- station 01 (lower reach 6; Sandy Beach).

Exposures of adult and young child local residents to surface water, sediment, and surface soil were evaluated at these stations. Due to the presence of shallow surface waters, wading is likely to be the primary activity at each of these locations, with the exception of station 01 (Sandy Beach) where surface waters are deeper and swimming is known to occur. As noted previously, stations 02, 04, and 06 are in areas where only surface water contact was assumed to occur due to the lack of a complete exposure pathway for sediments.

Evaluated exposure pathways associated with swimming include incidental ingestion of and dermal contact with both surface water and sediment. For wading, exposure pathways include incidental ingestion of and dermal contact with sediment (except at stations 02, 04, and 06), and dermal contact with surface water. Ingestion of surface water for wading-related exposures was not assessed since the water is shallow, making it unlikely that a wader would ingest more than a negligible amount of surface water. Inhalation of compounds volatilizing from surface water was not evaluated because only low levels of volatile compounds are present, and dilution and dispersion into ambient air would result in further attenuation of airborne concentrations of volatile compounds. For the recreational surface water and sediment exposure scenarios, adults and young children (i.e., 1 to 6 years old) were considered as receptor populations. The selection of these two receptor populations assumes that a recreational adult would accompany a young child to the study area, which is reasonable based on the recreational scenario assumed to occur.

Human receptors may also contact floodplain surface soils at stations NR, WS/WSS, CB-05, DA, KF, 07/DP, and AJRW while engaging in recreational activities along the river. Therefore, exposures to surface soil were evaluated at these stations. Surface soil exposure pathways evaluated include incidental ingestion of and dermal contact with soil. Inhalation of volatile or particulate materials from surface soil was not evaluated because dilution and dispersion of contaminants into ambient air would result in significant attenuation of airborne concentrations of compounds. As for the surface water and sediment exposure scenarios, adults and young children (i.e., 1 to 6 years old) were considered as receptor populations for exposure to surface soils.

Even though surface water, sediment, and surface soil exposures reasonably may occur at each of these stations, the frequency of receptor exposures is likely to vary based on the accessibility and attractiveness of the stations to human receptors, and on the proximity of the stations to residential areas. Therefore, different exposure frequencies (days per year) have been assumed for individual stations (see subsection 3.3.2.2). For current use, stations WS/WSS and CB-01 through CB-06 were assumed to have the greatest exposure potential due to their proximity to residences. For these stations, residences directly abut the wetland areas. Stations NR, 14, 22/TT-22, WH, WG, TT-30, CB-07, AM, and AS were assumed to have the lowest exposure potential due to their location either further from residential areas (i.e., in somewhat remote areas; stations 22/TT-22, WH, and WG), further from the shoreline (station CB-07) or in industrialized areas (station NR, 14, TT-30, AM and AS). The exposure potential of these latter nine stations may increase in the future due to changes in land use. The remaining stations (16/TT-33, 09, DA, KF, 08, 07/DP, LP, AJRW, and 01 through 06) were considered to be of "average" or "typical" exposure potential for the study area (i.e., in areas not immediately adjacent to residential areas, but easily accessed for recreational activities). At these stations, park land is adjacent to the river rather than residential or industrial areas.

The exposure pathway evaluated for the recreational angler was the ingestion of fish fillet tissue. Dermal contact with fish tissue was not assessed since exposure to contaminants in fish is unlikely through this pathway. For the local angler exposure scenario, adults and older children (i.e., 7 to 16 years old) were considered as receptor populations. Older children were selected as a more appropriate receptor population for a fishing scenario since it is unlikely that young children (ages 1 to 6) would engage in this activity with any degree of frequency.

Comments provided on the draft baseline human health risk assessment indicated concerns about the potential migration of study area contaminants during flooding events. To address these concerns, storm event surface water data were evaluated for young child and adult exposures that may occur within residential yards or basements during periodic flooding events. In addition, floodplain surface soil data in locations proximate to residential homes were evaluated for young child and adult residential exposures under the conservative assumption that contaminant concentrations present in floodplain surface soils are present in residential yards as a result of periodic flooding. Floodplain surface soil data for stations WS/WSS, CB-05, 07/DP, KF, and AJRW were selected for residential evaluation due to their proximity to residential homes that may be impacted by flooding events.

Since industrial operations near the study area would result in little, if any, contact by local workers with sediment, surface water, and surface soil associated with the study area, exposures of workers to these media were not quantitatively evaluated. However, the evaluation of the adult recreational receptor is likely to be a conservative representation of commercial exposures, should they be occurring.

3.3.1.2 Potential Receptors and Exposure Pathways Under Future Land-Use Conditions. Table 3-1 presents a summary of the future exposure routes quantitatively evaluated in the baseline human health risk assessment along with the human health exposure points and receptors. To evaluate potential future exposures, it was assumed that no remedial action was taken, and that the levels of contamination currently existing at the study area would remain the same in the future.

For the purposes of this baseline human health risk assessment, the exposures described under current land-use conditions for recreational users are likely to remain unchanged in the future, except for stations NR, 14, 22/TT-22, AS, WG, and WH. For these stations, it was assumed that future land-use conditions would result in increased accessibility to human receptors due to future development of the surrounding area. Therefore, stations 22/TT-22, AS, WG, and WH have been evaluated as “average” or “typical” exposure cases under future land-use conditions (see subsection 3.3.2.2). Stations NR and 14 are located in industrial areas where currently undeveloped land may be developed for commercial purposes in the future, resulting in a greater potential for human exposures. Therefore, future exposures at these two stations may be greater than current exposures, but not as high as those at the other stations located in more recreational or residentially populated areas.

Based on redevelopment plans (see Appendix C.2), it was assumed that portions of the interior Wells G&H wetland would become accessible in the future due to the construction of a boardwalk or pier extending into the wetland area. Therefore, stations NT-1, NT-2, and NT-3 have been evaluated as "average" or "typical" exposure cases under a future land-use scenario. Stations JY, WW, and 13/TT-27 are also assumed to become accessible in the future due to removal of the fencing, locked gates, and concrete wall currently preventing access. Likewise, TT-31 may become accessible should the fencing and buildings currently preventing access be removed. Therefore, these stations have also been evaluated as "average" or "typical" exposure cases under future land-use conditions.

In the future, excavation worker exposures may occur to sediment core samples collected from areas that may be targeted for dredging activities to control flooding. Therefore, worker-related dredging exposures to sediment core COPCs have been quantitatively evaluated.

3.3.1.3 Summary of Pathways and Receptors Selected for Consideration. The following items summarize the pathways evaluated for each exposure scenario.

` Recreational user scenario (wader), current/future
  Ingestion pathways: sediment, surface soil
  Dermal contact pathways: surface water, sediment, surface soil

` Recreational user scenario (swimmer; station 01), current/future
  Ingestion pathways: surface water, sediment
  Dermal contact pathways: surface water, sediment

` Recreational angler scenario, current/future
  Ingestion pathway: fish fillet tissue

` Residential scenario, current/future
  Ingestion pathway: surface soil
  Dermal contact pathways: surface water, surface soil

` Dredger scenario, future
    Ingestion pathway: sediment
    Dermal contact pathway: sediment

3.3.2 Calculation of Dose

The purpose of the exposure assessment is to identify exposure equations to be used in the risk assessment and to document assumptions made for each of the parameters used in these equations. USEPA Region 1 *Risk Updates, No. 2* (USEPA, 1994b) requires the calculation of CT exposure and RME estimates and provides default exposure parameters for each of these estimations. The risk assessment used the default CT exposure parameters to evaluate average exposures and high-end exposure parameters to calculate RME estimates. USEPA guidance or documents used in the exposure assessment include *RAGS, Part A* (USEPA, 1989); *Exposure Factors Handbook* (USEPA, 1997a); *Supplemental Guidance for Dermal Risk Assessment* (USEPA, 2004d); and *Risk Updates, No. 2* (USEPA, 1994b).

3.3.2.1 Selection of Exposure Equations. Equations are presented for the calculation of chronic daily intake (CDI) values for the ingestion and dermal pathways of exposure. The equations are used for calculating a lifetime average daily dose (LADD) relevant to cancer risk (i.e., cancer intake) or for calculating an average daily dose (ADD) relevant to noncancer risk (i.e., noncancer intake). The medium-specific equations used for the calculation of carcinogenic and noncarcinogenic intakes of the COPCs are presented in Tables 3-4.1 through 3-4.11. Additional equations used in calculating dose following dermal exposure to organic compounds in surface water are contained in Appendix C.7.

3.3.2.2 Exposure Parameters. The exposure parameters used for each of the receptors evaluated in the risk assessment are described below and are presented in Tables 3-4.1 through 3-4.11. Since exposure parameters vary depending on the exposure pathway and receptor being evaluated, the exposure parameters are presented by pathway in the tables and are discussed by receptor.

Adult Recreational User Exposure Parameters. The exposure parameters for the adult recreational user are shown in Tables 3-4.1 (surface water; current land use), 3-4.2 (surface water, future land use), 3-4.3 (sediment; current land use), 3-4.4 (sediment; future land use), 3-4.5 (surface soil; current land use), 3-4.6 (surface soil; future land use), 3-4.7 (fish fillet tissue; current/future land use), and 3-4.8 (baseflow surface water; current/future land use). These exposure parameters rely partially on default CT and RME exposure parameters presented in *Risk Updates, No. 2* (USEPA, 1994b) and *Supplemental Guidance for Dermal Risk Assessment* (USEPA, 2004d).

For surface water, incidental ingestion was assumed to occur during swimming, but not wading. A surface water ingestion rate of 50 ml/hour (USEPA, 1989) was used to evaluate both CT and RME exposures during swimming. Exposure time for swimming was set at 0.5 hours/day for the CT case, and 1 hour/day for the RME case (professional judgement). For the surface soil and sediment ingestion pathway, the default CT and RME soil ingestion rates (50 mg/kg and 100 mg/kg, respectively; USEPA 1994b) for adult residents were used. Use of these values provides a conservative evaluation of sediment exposure.

Since the weather at the study area is cold and not conducive to outdoor activities for about 6 months of the year, it was assumed that the adult resident may venture onto the study area and engage in activities resulting in surface water, sediment and surface soil exposure for 1 to 4 days per week for the warmest 6 months of the year (26 to 104 days/year), depending on the exposure potential of the station. Therefore, the exposure frequency parameter varies by station. The exposure frequency values used for the CT and RME exposure cases for sediment or surface soil ingestion were either 26 and 26 days/year, respectively, for stations with low exposure potential (current exposures at stations NR, 14, 22/TT-22, WG, WH, and AS; current/future exposures at stations TT-30, CB-07 and AM); 26 and 52 days/year for stations in industrial areas with open land with the potential for future development (future exposures at stations NR and 14); 26 and 78 days/year, respectively, for stations with typical exposure potential (current/future exposures at stations 01 through 06, 07/DP, 08, 09, 16/TT-33, DA, KF, LP, and AJRW; future exposures at stations 22/TT-22, NT-1, NT-2, NT-3, WG, WH, JY, WW, 13/TT-27, TT-31, and AS); or 78 and 104 days/year, respectively, for stations WS/WSS and CB-01 through CB-06, which have the highest exposure potential. Table C.1-2 in Appendix C.1 summarizes the exposure frequency assumptions for the study area by station for current and potential future land use scenarios.

Exposure frequency values for surface water were identical to those listed above for sediment and surface soil except at station 01 where exposure to surface water during swimming was assumed to occur 3 days per week for the warmest 3 months of the year (39 days/year) for the RME case. For the CT case, ingestion of surface water during swimming was assumed to occur 5 days/year. These values are based on professional judgement.

The fraction of sediment or surface soil ingested from the study area was assumed to be 50% for both the CT and RME cases. Use of a fraction ingested term assumes that a receptor ingests a portion of the daily sediment/soil intake from the study area and a portion from wetland and upland areas not impacted by the site (i.e., background areas). This assumption is reasonable for the study area since receptors are likely to spend a portion of the day in residential yards or other background areas and incur a portion of their daily sediment/soil ingestion from these background areas. Using a 50% fraction ingested term assumes that half of the daily sediment/soil ingested is from the study area.

For the dermal pathway, skin surface areas were selected for the body parts that could contact surface water, sediment, or surface soil, using statistical distributions of surface areas provided in the *Supplemental Guidance for Dermal Risk Assessment* (USEPA, 2004d). Adult recreational users were assumed to contact sediments, surface soils, and surface water during wading with

5,700 cm^2 of body surface area for both the CT and RME cases (50th percentile value; USEPA, 2004d). The surface area assumes exposure to the face, forearms, hands, and lower legs. A soil-to-skin adherence factor of 0.07 mg/cm^2-day was used for both the CT and RME cases (USEPA, 2004d). This value is a 50th percentile weighted adherence factor for gardeners, the activity selected to represent a reasonable high-end activity for the adult. The same surface area and soil-to-skin adherence factors selected for soil have also been used for sediment since USEPA (2004d) suggests using the same approach for sediment as that used for soil. For surface water exposures during swimming, a body surface area of 18,000 cm^2 was used for both the CT and RME cases (USEPA, 2004d).

Per USEPA guidance (2004d), cadmium, arsenic, PCBs, and carcinogenic polycyclic aromatic hydrocarbons (PAHs) were assessed for dermal exposures to sediment and surface soil through the use of chemical-specific dermal absorption factors. Dermal absorption factors of 1%, 3%, 14%, and 13% for cadmium, arsenic, PCBs, and benzo(a)pyrene (B(a)P), respectively, were used in both the CT and RME cases (USEPA, 2004d). In the absence of recommended dermal absorption factors, dermal exposures to the remaining sediment COPCs were not assessed as recommended in USEPA, 2004d.

For the surface water dermal exposure pathway, absorbed doses were calculated for each chemical using equations and chemical-specific factors provided by USEPA (2004d). The dermal absorbed dose was calculated using chemical-specific permeability coefficients and, for organic compounds, molecular weight and octanol-water partition coefficients, as detailed in Appendix C.7. For this exposure, event times of 0.5 hours and 1 hour were assumed for the CT and RME cases, respectively, based on professional judgement. The remaining exposure parameters used for the dermal exposure pathway (i.e., exposure frequency, exposure duration, body weight, and averaging time) were the same as the values described for the surface water and sediment ingestion pathways.

For the ingestion of fish tissue, time-weighted ingestion rates of 5 g/day and 13 g/day were used for the CT and RME cases, respectively (USEPA, 1997) along with an exposure frequency of 365 days/year. The ingestion rates are recommended mean and 95th percentile values for freshwater anglers from the New England region. The fraction of dietary fish ingested from the study area was assumed to be 50% for both the RME and CT cases. Using a 50% fraction ingested term assumes that half of the daily fish ingested is from the study area. This assumption is reasonable since there are a number of attractive surface water bodies in the vicinity of the study area where fishing is known to occur.

The default high-end exposure duration of 24 years was used for the RME case, while an average exposure duration of 7 years was used for the CT exposure case (USEPA, 1994b). The default value of 70 kg for an adult body weight was used for both CT and RME exposures (USEPA, 1994b). Finally, as recommended in *RAGS* (USEPA, 1989), the averaging time for noncarcinogens was set equal to the exposure duration, and the averaging time for carcinogens was the standard USEPA lifetime duration (70 years).

Young Child Recreational User Exposure Parameters. The exposure parameters for the young child recreational user (1 to 6 years of age) are shown in Tables 3-4.1 (surface water; current land use), 3-4.2 (surface water, future land use), 3-4.3 (sediment; current land use), 3-4.4 (sediment; future land use), 3-4.5 (surface soil; current land use), 3-4.6 (surface soil; future land use), and 3-4.8 (baseflow surface water; current/future land use). As with the adult resident receptor, these exposure parameters rely partially on default CT and RME parameters presented in *Risk Updates, No. 2* (USEPA, 1994b) and *Supplemental Guidance for Dermal Risk Assessment* (USEPA, 2004d).

For surface water, incidental ingestion was assumed to occur during swimming, but not wading. A surface water ingestion rate of 50 ml/hour (USEPA, 1989) was used to evaluate both CT and RME exposures during swimming. Exposure time for swimming was set at 0.5 hours/day for the

CT case, and 1 hour/day for the RME case (professional judgement). For the sediment and surface soil ingestion pathway, the default CT and RME soil ingestion rates (100 mg/kg and 200 mg/kg, respectively; USEPA, 1994b) for young child residents were used. Use of these values provides a conservative evaluation of sediment exposure.

As for the adult resident, it was assumed that the young child resident may venture onto the study area and engage in activities resulting in surface water, sediment, and surface soil exposure for 1 to 4 days per week for the warmest 6 months of the year (26 and 104 days/year), depending on the exposure potential of the station. The exposure frequency values used for the CT and RME exposure cases for sediment and surface soil ingestion were either 26 and 26 days/year, respectively, for stations with low exposure potential (current exposures at stations NR, 14, 22/TT-22, WG, WH, and AS; current/future exposures at stations TT-30, CB-07, and AM); 26 and 52 days/year for stations in industrial areas with open land with the potential for future development (future exposures at stations NR and 14); 26 and 78 days/year, respectively, for stations with typical exposure potential (current/future exposures at stations 01 through 06, 07/DP, 08, 09, 16/TT-33, DA, KF, LP, and AJRW; future exposures at stations 22/TT-22, NT-1, NT-2, NT-3, WG, WH, JY, WW, 13/TT-27, TT-31, and AS); or 78 and 104 days/year, respectively, for stations WS/WSS and CB-01 through CB-06, which have the highest exposure potential. As for the adult, the fraction of sediment and surface soil ingested from the study area was assumed to be 50% for both the CT and RME cases. Table C.1-2 in Appendix C.1 summarizes the exposure frequency assumptions for the study area.

Exposure frequency values for surface water were identical to those listed above for sediment except at station 01 where exposure to surface water during swimming was assumed to occur 3 days per week for the warmest 3 months of the year (39 days/year) for the RME case. For the CT case, ingestion of surface water during swimming was assumed to occur 5 days/year. As for the adult, these exposure frequencies were based on professional judgement.

For the dermal pathway, skin surface areas were selected for the body parts that could contact surface water, sediment, or surface soil, using statistical distributions of surface areas provided in the *Supplemental Guidance for Dermal Risk Assessment* (USEPA, 2004d). Young child recreational users were assumed to contact sediments, surface soils and surface water during wading with 2,800 cm^2 of body surface area for both the CT and RME cases (50^{th} percentile value; USEPA, 2004d). The surface area assumes exposure to the face, forearms, hands, lower legs, and feet. A soil-to-skin adherence factor of 0.2 mg/cm^2-day was used for both the CT and RME cases (USEPA, 2004d). This value is a 50^{th} percentile weighted adherence factor for children playing in wet soil, the activity selected to represent a reasonable high-end activity for the child. The same surface area and soil-to-skin adherence factors selected for soil have also been used for sediment since USEPA (2004d) suggests using the same approach for sediment as that used for soil. For surface water exposures during swimming, a body surface area of 6,600 cm^2 was used for both the CT and RME cases (USEPA, 2004d).

Cadmium, arsenic, PCBs, and carcinogenic PAHs were assessed for dermal exposures to sediment and surface soil as previously described for the adult recreational user. For the surface water dermal exposure pathway, absorbed doses were calculated for each chemical using equations and chemical-specific factors previously described for the adult recreational user. The remaining exposure parameters used for the dermal exposure pathway (i.e., exposure frequency, exposure duration, body weight, and averaging time) were the same as the values described for the surface water and sediment ingestion pathways.

The default high-end exposure duration of 6 years was used for the RME case, while an average exposure duration of 2 years was used for the CT exposure case (USEPA, 1994b). The value of 15 kg for a young child body weight was used for both CT and RME exposures (USEPA, 1994b). The averaging time for noncarcinogens was set equal to the exposure duration, and the averaging time for carcinogens was the standard USEPA lifetime duration (70 years; USEPA, 1989).

Older Child Angler Exposure Parameters. Since the older child angler age range of 7 to 16 years falls between the receptor groups (i.e., child and adult) presented in the *Risk Updates* (USEPA, 1994b) guidance, many of the exposure parameters for the older child group were selected based on professional judgement, while others were determined given site-specific information. Exposure parameters for this receptor are shown in Table 3-4.7 and are discussed for older child ingestion of fish tissue only. For the ingestion of fish fillet tissue, time-weighted ingestion rates of 2.5 g/day and 6.5 g/day were used for the CT and RME cases, respectively, under the assumption that a child ingests approximately half of the daily fish intake of an adult. These ingestion rates were used in conjunction with an exposure frequency of 365 days/year (USEPA, 1994b). The fraction of dietary fish ingested from the study area was assumed to be 50% for both the RME and CT cases. Using a 50% fraction ingested term assumes that half of the daily fish ingested is from the study area. This assumption is supported by data presented in the *Exposure Factors Handbook* (USEPA, 1997) indicating that among individuals who reside in households with recreational fish consumption, approximately half of the total fish consumed is recreational fish. The exposure duration of 6 years for the RME case assumed that the receptor would ingest fish from the study area for 6 of the 10 years within the given age range, while the receptor was assumed to ingest fish from the study area for 2 years for the CT exposure case (USEPA, 1994b). The body weight for the older child (31 kg) that was used for both the RME and CT cases was consistent with age-specific data presented by USEPA (2004d). Finally, the standard USEPA lifetime averaging time for carcinogens, 70 years, was used, and for noncarcinogens, the exposure duration was used as the averaging time (USEPA, 1989).

Dredger Exposure Parameters. For this scenario, it is assumed that workers may be exposed to contaminated sediments up to four feet in depth, should the river be dredged as a flood control measure in the future. The exposure parameters for the dredger scenario are shown in Table 3-4.10 (sediment cores; future land use). The exposure parameters rely partially on default CT and RME exposure parameters presented in *Exposure Factors Handbook* (USEPA, 1997a) and *RAGS, Part E* (USEPA, 2004d).

For the sediment ingestion pathway, the default contact intensive soil ingestion rate of 200 mg/day (USEPA, 1997a) is used for both the CT and RME cases to provide a conservative evaluation of exposure. Based on site-specific information provided by the Town of Winchester, it is assumed that dredgers may be exposed to soil on-site for 83 days/year for the CT scenario (a four-month project) or 167 days/year for the RME scenario (an eight-month project). The fraction of soil ingested from the site is assumed to be 100% for both the CT and RME cases.

Assumed exposure durations of 1 year and 2 years, respectively, are used for the CT and RME cases. The default value of 70 kg for an adult body weight is used for both CT and RME exposures (USEPA, 1997a). Finally, as recommended in *RAGS* (USEPA, 1989), the averaging time for noncarcinogens is set equal to the exposure duration, and the averaging time for carcinogens is the standard USEPA lifetime duration (70 years).

For the dermal pathway, skin surface areas are calculated for the body parts that could contact sediment, using statistical distributions of surface areas provided in *RAGS, Part E* (USEPA, 2004d). Dredgers are assumed to contact soils with 3,300 cm^2 of body surface area for both the CT and RME cases (50^{th} percentile value; USEPA, 2004d). A soil-to-skin adherence factor of 0.2 mg/cm^2-day was used for both the CT and RME cases (USEPA, 2004d). COPCs were assessed for dermal exposures as previously described. The remaining exposure parameters used for the dermal exposure pathway (i.e., exposure frequency, exposure duration, body weight, and averaging time) are the same as the values described for the sediment ingestion pathway.

Young Child Area Resident Exposure Parameters. The area resident is potentially exposed to contaminants that may have migrated from the river to residential yards and basements during flooding events. The exposure parameters for the young child area resident are shown in Tables 3-4.9 (storm event surface water; current/future land use) and 3-4.11 (surface soil; current/future land use). These exposure parameters rely partially on default CT and RME parameters presented in *Risk Updates, No. 2* (USEPA, 1994b), *Exposure Factors Handbook* (USEPA, 1997a), and *RAGS, Part E* (USEPA, 2004d).

Floodplain surface soil data from stations WS/WSS, CB-05, 07/DP, KF, and AJRW were evaluated for residential exposures under the conservative assumption that the contaminants present in the floodplain may also be present in residential yards, as a result of periodic flooding events. Since the weather in the area is cold and not conducive to outdoor activities for about 6 months of the year, it is assumed that the young child resident engages in activities resulting in soil exposures 150 days/year for both the RME and CT scenarios (USEPA, 1994b). The fraction of soil ingested from the site is conservatively assumed to be 100% for both the CT and RME cases. The young child ingestion rate for soil is set at 200 mg/day for the RME receptor and 100 mg/day for the CT receptor (USEPA, 1997a).

To evaluate storm event surface water exposures, site-specific data were gathered from a frequently flooded neighborhood along the river, the Forest Street/Brookside Drive neighborhood that abuts Davidson Park. Based on information provided by residents of this neighborhood, an exposure frequency of 42 days/year had been used for RME exposures. This exposure frequency accounts for the occurrence of one major flooding event when river waters are present in basements and yards for up to one week, and twelve minor flooding events when river waters remain on the residential properties for two to three days. For CT exposures, an exposure frequency of 21 days/year has been assumed.

The default high-end exposure duration of 6 years is used for the RME case, while an average exposure duration of 2 years is used for the CT exposure case (USEPA, 1994b). The default value of 15 kg for a young child body weight was used for both CT and RME exposures (USEPA, 1997a). Finally, as recommended in *RAGS* (USEPA, 1989), the averaging time for noncarcinogens is set equal to the exposure duration, and the averaging time for carcinogens is the standard USEPA lifetime duration (70 years).

For the soil and surface water dermal pathways, skin surface areas are calculated for the body parts that could contact these media, using statistical distributions of surface areas provided in the *RAGS, Part E* (USEPA, 2004d). Young child residents were assumed to contact surface water and soil during outdoor activities with 2,800 cm^2 of body surface area for both the CT and RME cases (50th percentile value; USEPA, 2004d). The surface area assumes exposures to the face, forearms, hands, lower legs, and feet. A soil-to-skin adherence factor of 0.2 mg/cm^2-day was used for both the CT and RME cases (USEPA, 2004d). This value is the 50th percentile weighted adherence factor for children playing in wet soil, the activity selected to represent a reasonable high-end activity for the child.

Cadmium, arsenic, PCBs, and PAHs were assessed for dermal exposures to soil as previously described for the recreational receptor. For the surface water dermal exposure pathway, absorbed doses were calculated for each chemical using equations and chemical-specific factors previously described for the recreational user. The remaining exposure parameters used for the dermal exposure pathway (i.e., exposure frequency, exposure duration, body weight, and averaging time) were the same as the values described for the soil ingestion pathway.

Adult Area Resident Exposure Parameters. The area resident is potentially exposed to contaminants that may have migrated from the river to residential yards and basements during flooding events. The exposure parameters for the adult area resident are shown in Tables 3-4.9 (storm event surface water; current/future land use) and 3-4.11 (surface soil; current/future land use). These exposure parameters rely partially on default CT and RME parameters presented in *Risk Updates, No. 2* (USEPA, 1994b), *Exposure Factors Handbook* (USEPA, 1997a), and *RAGS, Part E* (USEPA, 2004d).

Floodplain surface soil data from stations WS/WSS, CB-05, 07/DP, KF, and AJRW were evaluated for residential exposures under the conservative assumption that the contaminants present in the floodplain may also be present in residential yards, as a result of periodic flooding events. Since the weather in the area is cold and not conducive to outdoor activities for about 6 months of the year, it is assumed that the adult resident engages in activities resulting in soil exposures 150 days/year for both the RME and CT scenarios (USEPA, 1994b). The fraction of soil ingested from the site is conservatively assumed to be 100% for both the CT and RME cases. The adult ingestion rate for soil is set at 100 mg/day for the RME receptor and 50 mg/day for the CT receptor (USEPA, 1997a).

To evaluate storm event surface water exposures, site-specific data were gathered from a frequently flooded neighborhood along the river, the Forest Street/Brookside Drive neighborhood that abuts Davidson Park. Based on information provided by residents of this neighborhood, an exposure frequency of 42 days/year had been used for RME exposures. This exposure frequency accounts for the occurrence of one major flooding event when river waters are present in basements and yards for up to one week, and twelve minor flooding events when river waters remain on the residential properties for two to three days. For CT exposures, an exposure frequency of 21 days/year has been assumed.

The default high-end exposure duration of 24 years is used for the RME case, while an average exposure duration of 7 years is used for the CT exposure case (USEPA, 1994b). The default value of 70 kg for an adult body weight was used for both CT and RME exposures (USEPA, 1997a). Finally, as recommended in *RAGS* (USEPA, 1989), the averaging time for noncarcinogens is set equal to the exposure duration, and the averaging time for carcinogens is the standard USEPA lifetime duration (70 years).

For the soil and surface water dermal pathways, skin surface areas are calculated for the body parts that could contact these media, using statistical distributions of surface areas provided in the *RAGS, Part E* (USEPA, 2004d). Adult residents were assumed to contact surface water and soil during outdoor activities with 5,700 cm^2 of body surface area for both the CT and RME cases (50^{th} percentile value; USEPA, 2004d). The surface area assumes exposure to the face, forearms, hands, and lower legs. A soil-to-skin adherence factor of 0.07 mg/cm^2-day was used for both the CT and RME cases (USEPA, 2004d). This value is a 50^{th} percentile weighted adherence factor for gardeners, the activity selected to represent a reasonable high-end activity for the adult.

Cadmium, arsenic, PCBs, and PAHs were assessed for dermal exposures to soil as previously described for the recreational receptor. For the surface water dermal exposure pathway, absorbed doses were calculated for each chemical using equations and chemical-specific factors previously described for the recreational user. The remaining exposure parameters used for the dermal exposure pathway (i.e., exposure frequency, exposure duration, body weight, and averaging time) were the same as the values described for the soil ingestion pathway.

3.4 TOXICITY ASSESSMENT

The toxicity assessment presented here was conducted in accordance with USEPA guidance (1989). The methodology used for classifying health effects from exposure to chemicals is recommended by USEPA (1989). The toxicity assessment considers chronic (long-term) exposures. For potentially carcinogenic chemicals, less than chronic exposures (i.e., subchronic exposures) would result in less risk than chronic exposure. Therefore, if chronic risk is below a regulatory limit, risk from subchronic exposures will also be below the regulatory limit. In other words, as exposure decreases, so does the risk for potential carcinogens.

For noncarcinogenic chemicals, acute (i.e., shorter exposures than subchronic) and subchronic risks could be assessed; however, only irritating substances such as sulfur dioxide would likely present an acute risk. Chronic exposures would result in higher risks than subchronic exposures, assuming exposures of equal intensity. Therefore, again, if chronic risks are below a regulatory limit, subchronic risks are also below the regulatory limit.

The chronic toxicity criteria were obtained from USEPA's Integrated Risk Information System (IRIS) (USEPA, 2004a). This source lists the most recent toxicity values recommended by USEPA for use in human health risk assessments. In the event that toxicity values for a COPC were not available through IRIS, provisional toxicity values were obtained from the National Center for Environmental Assessment (NCEA), a division of USEPA. Values from IRIS are the preferred criteria, if available, followed by NCEA provisional values. Toxicity criteria from the Health Effects Assessment Summary Tables (HEAST; USEPA, 1997b) were used only if values were not available from either IRIS or NCEA.

3.4.1 Toxicity Information for Noncarcinogenic Effects

Systemic toxic effects other than cancer can be associated with exposures to chemicals. The reference doses (RfDs) are the toxicity values that are used to evaluate the potential of developing noncarcinogenic effects as a result of exposure to potentially toxic chemicals. The RfDs have been developed on the premise that there are protective mechanisms that must be overcome before an appreciable risk of adverse health effects is manifested during a defined exposure period. It is assumed that there is a threshold dose that must be exceeded before adverse effects can occur.

Chemicals classified as carcinogens may also produce other systemic effects. These chemicals were also evaluated for potential noncarcinogenic toxic effects and were included in the determination of chronic toxicity HQs, which characterize noncancer hazards. Carcinogenic effects, however, are usually manifested at levels that are lower than those associated with systemic toxic effects; thus, cancer is usually the predominant adverse effect for contaminants that may elicit carcinogenic as well as noncarcinogenic responses. Table 3-5.1 summarizes the noncarcinogenic toxicity values (i.e., RfDs) and the corresponding critical effects for the COPCs at the study area.

Table 3-5.1 contains both chronic and subchronic toxicity values. Subchronic toxicity values are applicable to the dredger scenario where exposures are expected to occur over a brief (i.e., less than 2 year) duration. Chronic toxicity values are applicable to all other receptors whose exposures are expected to occur over a longer interval of time. Subchronic toxicity values are not found in IRIS. Instead, subchronic toxicity values have been developed from chronic toxicity values. According to USEPA guidance (USEPA, 1989), if a chronic toxicity value has been developed based on subchronic data, a subchronic toxicity value may be developed by removal of the uncertainty factor used to extrapolate from subchronic to chronic exposures (typically a factor of 10). If subchronic data are not available and the chronic toxicity value is derived from chronic data, the chronic toxicity value is adopted as the subchronic toxicity value.

Oral RfDs for manganese were developed based on USEPA Region I guidance (USEPA, 1996) as recommended in IRIS (USEPA, 2004a). These RfDs were based on a total allowable manganese intake of 10 mg/day (USEPA, 2004a). After adjusting for background intake (the average dietary manganese intake in the U.S. population; 5 mg/day), the remaining intake (5 mg/day) was then normalized for body weight (70 kg) to arrive at the manganese RfD for sediment, surface soil, and fish tissue exposures (0.07 mg/kg-day).

For mercury, the RfD for inorganic mercury was used to evaluate surface water and soil exposures. However, since mercury in sediments and fish tissue is likely to exist as organic mercury compounds, the RfD for organic mercury was used to evaluate sediment and fish ingestion exposures.

All chromium in surface water, surface soil, and fish fillet tissue from the study area was conservatively assumed to be chromium VI since no speciation data were collected for these media. Based on chromium VI analysis performed on study area sediments and sediment cores, chromium detected in sediments is assumed to be chromium III except at stations WH, NT-1, WW, JY, WS/WSS, TT-30, CB-06, and 16/TT-33 where low levels of chromium VI were either detected or assumed to be present based on available data (see subsection 3.2.2.5 for discussion).

Additional information on the noncarcinogenic effects for each COPC is presented in the toxicity profiles in Appendix C.8. Chemical-specific permeability coefficients (K_ps), used to evaluate the surface water dermal pathway, are provided in Appendix C.7.

3.4.2 Toxicity Information for Carcinogenic Effects

The potential for human carcinogenic effects is evaluated based on the chemical-specific slope factors (SFs) and the weight-of-evidence classification of the USEPA. The SF is the toxicity value that quantitatively defines the dose-response relationship of a known or suspected carcinogen. The SF is an estimate of an upper-bound lifetime probability of an individual developing cancer following exposure to a potential cancer-causing agent over his or her lifetime. The SFs for chemicals are generally expressed as the 95% UCL of the slope of the dose-response curve and are derived by assuming low-dose linearity and applying a computer model to extrapolate from the relatively high doses administered to animals (or the exposures observed in epidemiological studies) to the lower environmental exposure levels that generally occur in humans. The USEPA has developed SFs for chemicals classified as carcinogens, based on the premise that there is no threshold, i.e., there is no level of exposure below which there is no risk of a carcinogenic effect.

Because the SF is generally the 95% UCL of the probability of a response per unit intake of a chemical over a lifetime exposure, the use of such SFs is expected to result in a conservative (i.e., upper-bound) estimate of potential cancer risk. The true risk to humans is not likely to exceed the upper-bound estimate, but could be lower and may even be zero. Further, because the dose-response curve is assumed to be linear in the low-dose region, the accuracy of the SF may be limited if this region should, in reality, exhibit nonlinearity.

Table 3-6.1 summarizes the carcinogenic toxicity values (i.e., SFs) and the corresponding weight-of-evidence classifications. For PAHs, the SF for benzo(a)pyrene, along with the appropriate relative potency factors (USEPA, 1993), have been used to evaluate the potency of the individual carcinogenic PAHs. Additional discussion on each COPC is provided in toxicity profiles presented in Appendix C.8.

3.4.3 Adjustment of Toxicity Factors

No RfDs or SFs are available for evaluating dermal exposure. Therefore, cancer risks and HIs associated with dermal exposure may be evaluated using an oral SF or RfD, adjusted such that the toxicity value is appropriate for the dermal pathway. As detailed by USEPA (1989), for purposes of evaluating dermal exposure, it is generally necessary to adjust an oral toxicity factor (i.e., RfD or SF) from an administered (i.e., applied) dose to an absorbed (i.e., internal) dose. Because the toxicity values for the COPCs at the study area are expressed as orally administered doses (i.e., applied or intake-based), it is necessary to adjust both the RfDs and SFs for these substances in estimating exposure on an absorbed-dose basis when assessing dermal exposure.

The oral RfDs and oral SFs for each COPC were modified according to the following equations (USEPA, 1989) for use in assessing dermal exposure:

$$ERfD_o = RfD_o \times BF_{o,a}$$
$$ESF_o = SF_o / BF_{o,a}$$

where:

$ERfD_o$	=	effective absorbed-dose oral RfD for each chemical (i.e., adjusted dermal RfD)
RfD_o	=	oral RfD for each chemical
$BF_{o,a}$	=	absolute oral bioavailability factor for each chemical (i.e., oral to dermal adjustment factor)
ESF_o	=	effective absorbed-dose oral SF for chemical (i.e., adjusted dermal SF)
SF_o	=	oral SF for each chemical

Tables 3-5.1 and 3-6.1 present the oral to dermal adjustment factors used to adjust the oral toxicity criteria for the COPCs evaluated in the dermal exposure pathways. Oral bioavailability values used were obtained from USEPA (2004d). No adjustment for oral absorption efficiency has been applied to any COPC with an absorption efficiency of greater than 50%. These COPCs include all VOCs, PAH compounds, pesticides, PCBs, arsenic (except for sediments; see subsection 3.4.4), copper, organic mercury, selenium, thallium, and zinc. Additional information on compound-specific oral to dermal adjustment factors is provided in Appendix C.8.

3.4.4 Toxicity Information for Arsenic in Sediment

To more accurately assess the oral toxicity of arsenic in sediments at the study area, a site-specific oral bioavailability study was conducted. This study was initiated because current default information on the oral bioavailability of arsenic from environmental media indicates that arsenic may be absorbed from the gastrointestinal tract with an efficiency approaching 100%. However, oral bioavailability studies at other sites have indicated that the actual oral bioavailability of arsenic from some soils is significantly less than 100%.

Appendix C.9 contains the report, *Relative Bioavailability of Arsenic in Sediments from the Aberjona River,* that details the methods and results of the study conducted for the Aberjona River Study Area. In this study, young swine were fed sediments from the study area that contained arsenic at various known levels. Data were collected to calculate the relative bioavailability (RBA) of arsenic from these sediments. RBA is an estimate of the oral bioavailability 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. These site-specific RBA estimates are also less than the default value of 100% for oral absorption efficiency of arsenic. The most conservative RBA value determined for study area sediments (51%) was selected as the most appropriate to evaluate the oral toxicity of arsenic in sediments at all stations within the study area.

The site-specific RBA value of 51% was used to adjust the oral RFD and SF for arsenic to derive a site-specific estimate of oral toxicity of arsenic in sediments. The oral RfD and oral SF for arsenic were modified according to the following equations (see Appendix C.9) for use in assessing oral sediment exposures for arsenic:

$$RfD_{adjusted} = RfD_{IRIS} / RBA$$

$$SF_{adjusted} = SF_{IRIS} \times RBA$$

where:

$RfD_{adjusted}$ = adjusted oral RfD for arsenic in sediment

RfD_{IRIS} = oral RfD for arsenic as listed in IRIS (USEPA, 2004a)

RBA = site-specific relative bioavailability factor for arsenic (i.e., 0.51)

$SF_{adjusted}$ = adjusted oral SF for arsenic in sediment

SF_{IRIS} = oral SF for arsenic as listed in IRIS (USEPA, 2004a)

Tables 3-5.3 and 3-6.3 present the adjusted oral RfD and adjusted oral SF for arsenic, respectively. These adjusted toxicity values were used to evaluate ingestion exposures to arsenic in sediment only. Arsenic toxicity values were not changed for the evaluation of arsenic in other media or by the dermal route of exposure. Since inhalation exposures are not quantitatively evaluated in this report, Tables 3-5.2 and 3-6.2 are not applicable to the study area.

3.4.5 Toxicity of Lead

Lead was selected as a COPC in surface water, sediment, and fish fillet tissue. No RfD or SF is available for lead. Therefore, USEPA has recommended some alternative approaches to evaluate lead exposures. For sediment and surface water, childhood lead exposures were evaluated through the use of the Integrated Exposure Uptake Biokinetic (IEUBK) Model (USEPA, 2002b). Appendix C.10 contains summary information showing the IEUBK model inputs. This model uses algorithms to determine whether exposure to a soil lead concentration will result in an exceedance of a childhood blood lead level goal of 10 Wg/dL. The average time-weighted sediment lead concentration at each station was used as the soil concentration in the model. The average surface water lead concentration at each station was used to calculate a site-specific contribution from sources other than soil (i.e., an alternative source; see Table C.10-3). Default values, as recommended in the model, were used for all other inputs (see Table C.10-4).

Adult lead sediment exposures were evaluated through the use of methodology provided in *Recommendations of the Technical Workgroup for Lead for an Approach to Assessing Risk Associated with Adult Exposures to Lead in Soil* (USEPA, 2003a). This methodology uses algorithims to relate soil lead intake to blood lead concentrations in women of childbearing age. The model determines whether exposure to a soil lead concentration will result in an exceedance of a site-specific maternal blood lead level that is protective of a 95th percentile fetal blood level goal of 10 Wg/dL. Appendix C.10, Table C.10-1, documents the calculation of a site-specific maternal blood lead level of 4.2 Wg/dL, using a geometric standard deviation (GSD) in intake and biokinetics of 1.8, which is typical of populations in small areas dominated by a single source of lead exposure. A typical blood lead concentration in women of child-bearing age in the absence of study area exposures was assumed to be 2.0 Wg/dL, which is a mid-range default assumption (USEPA, 2003a). All other model inputs are presented on Table C.10-5.

Average lead levels in fish fillet tissue from the study area were evaluated relative to average background fish tissue levels for lead (0.34 mg/kg; USEPA, 1994c) in the risk characterization (subsection 3.5.2.3).

3.5 RISK CHARACTERIZATION

Risk characterization combines estimates of exposure with toxicity data to develop estimates of the probability that an adverse effect will occur under the specified conditions of exposure. The risk characterization was divided into three phases: 1) risk estimation; 2) risk description; and 3) uncertainty analysis.

Risk estimation is undertaken by combining the toxicity factors and exposure assessment equations to calculate estimates of risks. Noncarcinogenic risks are reported as pathway-specific HIs, which are the sum of individual COPC HQs for that pathway. Only HQs from COPCs that affect the same target organ are summed to generate HIs. Estimates of carcinogenic risks are reported as incremental (above background) lifetime cancer risks (ILCRs). Current practice considers carcinogenic risks to be additive when assessing exposure to a mixture of hazardous substances. Risk description entails several discussions, including the relative contributions of individual exposure pathways to the total risk for each medium. The significance of the risk estimates are relative to risk management criteria set forth in USEPA policy. The uncertainty analysis describes and quantifies, where possible, the impact of data uncertainty and variability, exposure assumptions, and toxicity values on estimates of risk.

3.5.1 Risk Estimation

Noncancer risk is estimated by means of a HQ. To calculate noncarcinogenic HQs, the ADDs, calculated as described in subsection 3.3.2, were divided by the RfDs as follows:

$$HQ = ADD / RfD$$

The sum of this ratio for all chemicals within a station and pathway that have the same target organ or type of toxicity is termed the pathway HI. The HI is useful as a reference point for gauging potential effects of environmental exposures to complex mixtures. In general, HIs that are less than 1 are not of regulatory concern; however, a HI of greater than 1 does not automatically indicate that an adverse effect will occur and should not automatically be interpreted as posing an unacceptable risk to the exposed population.

The total pathway HI for each station was calculated by summing the HQs for the COPCs having similar systemic effects. Total HIs for each receptor, by medium, were calculated by summing the total pathway HIs across pathways within media (e.g., summing dermal and ingestion sediment risks). As a first approximation, all COPCs are assumed to have additive effects. Total pathway HIs, assuming additivity of effects, are presented on Tables 3-7.1 through 3-7.13. However, in cases where the total pathway HI for a receptor exceeded 1, only COPCs having similar systemic effects (i.e., target organs) were summed for each pathway and medium. Target organ HIs are presented on Tables 3-9.1 through 3-9.104.

The cancer risk of each receptor is estimated for each medium by means of an ILCR. USEPA (1991) states that where the cumulative incremental current or future carcinogenic risk to a receptor is less than 10^{-4}, and where the noncarcinogenic HI is less than 1, action generally is not warranted unless there are adverse environmental impacts.

To calculate ILCR, the chemical- and pathway-specific LADDs, calculated as described in subsection 3.3.2, were multiplied by SFs as follows:

$$ILCR = SF \times LADD$$

The resulting value represents the upper-bound probability that an individual could develop cancer over his or her lifetime due to exposure to potential carcinogens under the conditions specified in the exposure scenario. For example, carcinogenic risk levels of 10^{-6} and 10^{-4} represents a one-in-one-million chance and a one-in-ten-thousand chance, respectively, that an individual could develop cancer over a lifetime.

The cancer risk for each pathway (e.g., the sediment ingestion pathway) was calculated by summing the risks from each COPC at each station within the pathway, while receptor risks for each medium were calculated by summing ILCRs for each pathway within the medium (e.g., the sediment ingestion and dermal contact pathways). Receptor cancer risk from exposure to sediment, surface water, and surface soil within a station was determined by adding the risk from sediment and surface soil ingestion to the risk from dermal contact with sediment, surface soil, and surface water from this location. ILCRs were further summed for child and adult receptors to derive a total receptor cancer risk for the recreational receptor. The total receptor ILCRs are presented on Tables 3-9.1 through 3-9.104.

Risks associated with surface soil exposures were not added to sediment and surface water risks at station CB-05. This station represents a walking trail elevated above and located away from the wetland areas of the Cranberry Bog. It is likely that receptors using this trail do not contact surface water and sediment in the wetland. Therefore, only surface soil risks have been presented for this station.

Total receptor cancer risk from each medium is presented by station. Risk was not summed across stations within the study area since the parameter values used assume maximal exposure within each exposure area. This approach assumes that an individual would not be maximally exposed to sediment at more than one station (e.g., station 07/DP and station 01).

3.5.2 Risk Description

This subsection summarizes the human health risks potentially associated with exposures to environmental media (surface water, sediment, surface soil, and fish tissue). Individual chemical-specific carcinogenic risks are expressed as probabilities of developing cancer (i.e., ILCRs), while noncarcinogenic risks are expressed as HIs. All carcinogenic and noncarcinogenic risks were calculated using both CT and RME methods. The RME represents the reasonable maximum exposure and risk a receptor may receive from a station. The CT represents the average exposure and risk at a station.

The risk description for the study area is provided below in two parts. First, the relative contributions of the various exposure pathways within each medium are analyzed for each receptor. Second, the relative contributions of each contaminant are analyzed for each receptor. The noncarcinogenic and carcinogenic risks associated with each medium for the adult and child recreational scenarios are presented in Tables 3-7.1 through 3-7.13 for the RME and CT cases (e.g., 3-7.1.RME and 3-7.1.CT).

Tables 3-9.1 through 3-9.104 present target-organ specific HIs, which are discussed if a medium-specific HI exceeds 1. For the recreational receptor, child and adult ILCRs have been summed to present the total receptor cancer risk. However, because the child receptor is the most sensitive receptor for the estimation of noncarcinogenic risks, only the child receptor HIs have been presented on Tables 3-9.1 through 3-9.104.

3.5.2.1 Description of HI Estimates. HI estimates represent the risk of health effects other than cancer from exposure to contaminants within the study area, as described in subsection 3.5.1. Tables 3-7.1 through 3-7.13 present the noncarcinogenic risks by receptor and medium. When a receptor-specific HI for an exposure medium exceeded 1, HIs were segregated by target organ and discussed as to whether target organ-specific HIs exceed the risk management criterion. These target organ HIs are presented on Tables 3-9.1 through 3-9.104.

Adult Recreational Receptor. The estimated HIs for each pathway and medium, presented by station, are listed for the adult recreational receptor in Tables 3-7.1 and 3-7.2 for current and future land use, respectively. Surface water HIs in these tables are based on 1995 data. Surface water HIs based on the 2001/2002 baseflow surface water data are provided in Table 3-7.9. Sediment and surface soil HIs for station AJRW are presented on Table 3-7.7.

Surface Water. HIs for current and/or future surface water ingestion and/or dermal contact, based on 1995 data, were less than the target HI of 1 for all stations. The exposure area with the highest HI from contact with surface water was station WS/WSS, with a HI of 0.05 for the current and future RME individual (Tables 3-7.1.RME and 3-7.2.RME). For the 2001/2002 baseflow surface water data, current and/or future HIs were also less than the target HI of 1 for all stations.

Sediment/Surface Soil. For the CT and RME receptors, current and/or future sediment ingestion and dermal contact risks were less than or equal to the target HI of 1 for all stations. The exposure area with the highest HI for sediment ingestion and dermal contact was station 13/TT-27, with a HI of 1 for the future individual (Table 3-7.2.RME).

HIs for current/future surface soil ingestion and dermal contact were less than the target HI of 1. For surface soil, the station with the highest HI was station DA with a current and future RME HI of 0.2 (Tables 3-7.1.RME and 3-7.2.RME).

Fish Fillet Tissue. When HQs were summed only for COPCs with similar target organs, current and future CT and RME segregated HIs for fish fillet ingestion were less than or equal to the target HI of 1 for all study area reaches.

Young Child Recreational Receptor. The estimated HIs for each pathway and medium, presented by station, are listed for the young child recreational receptor in Tables 3-7.3 and 3-7.4 for current and future land use, respectively. Surface water HIs in these tables are based on 1995 data. Surface water HIs based on the 2001/2002 baseflow surface water data are provided in Table 3-7.10. Sediment and surface soil HIs for station AJRW are presented on Table 3-7.8.

Surface Water. HIs for current and/or future surface water ingestion and/or dermal contact, based on 1995 data, were less than the target HI of 1 all stations. The exposure area with the highest HI from contact with surface water was station WS/WSS, with a HI of 0.1 for the current and future RME individual (Tables 3-7.3.RME and 3-7.4.RME). For the 2001/2002 baseflow surface water data, current and/or future HIs were also less than the target HI of 1 for all stations.

Sediment/Surface Soil. For the CT receptor, current and/or future sediment ingestion and dermal contact target organ HIs were less than or equal to the target HI of 1 for all stations, except for stations 13/TT-27 and NT-1. For these stations, the future CT HIs for sediment exposure were 3 and 2, respectively (Table 3-7.4.CT). For the RME receptor, HIs for current sediment ingestion and dermal contact exceeded the target HI of 1 at stations WH and CB-03 (Table 3-7.3.RME). For future land-use conditions, sediment exposure HIs exceeded the target HI at stations 13/TT-27, WH, NT-1, NT-2, NT-3, and CB-03 (Table 3-7.4.RME). In all cases, the largest contributor to the HI exceedances at these stations was arsenic. For all other stations, RME target organ HIs for current and future exposures were at or below the target HI of 1.

Target organ HIs for current/future surface soil ingestion and dermal contact were less than or equal to the target HI of 1. For surface soil, the station with the highest total receptor HI was station DA with a future RME HI of 2. However, when HQs were summed only for COPCs with similar by target organs, the future RME segregated HIs were less than 1 (Table 3-9.56.RME).

Older Child Recreational Receptor. The estimated HIs for each pathway and medium, presented by reach, are listed for the older child recreational receptor in Tables 3-7.5 and 3-7.6 for current and future land use, respectively. When HQs were summed only for COPCs with similar target organs, current and future CT and RME segregated HIs for fish fillet ingestion were less than or equal to the target HI of 1 for all study area reaches.

Young Child/Adult Area Resident Receptor. The estimated HIs for current/future land use are listed for the area resident receptor in Tables 3-9.85 through 3-9.90 for surface water, presented by gauging station, and Tables 3-9.100 through 3-9.104 for floodplain surface soil, presented by station. Surface water HIs in these tables are based on 2001/2002 storm event data. Surface soil HIs are presented for stations WS/WSS, CB-05, 07/DP, KF, and AJRW, where periodic flooding of the river may have impacted residential yards.

<u>Surface Water</u>. HIs for current/future storm event surface water dermal contact, based on 2001/2002 data, were less than the target HI of 1 for all gauging stations. The exposure area with the highest HI from contact with surface water was gauging station SW-05 (located at the juncture of reaches 1 and 2), with a HI of 0.01 for the current/future RME individual (Tables 3-9.85.RME).

Surface Soil. For the CT and RME receptors, current/future soil ingestion and dermal contact total receptor HIs exceeded the target HI at stations CB-05 (CT amd RME), 07/DP (RME), and AJRW (RME) with HIs of 2 noted in all cases. However, target organ HQs were less than or equal to the target HI of 1 for all stations evaluated.

Dredger Receptor. The estimated HIs for sediment ingestion and dermal contact, presented by sediment core location, are listed for the dredger receptor in Table 3-7.13 for future land use. Target organ HIs are presented, by location, in Tables 3-9.91 through 3-9.99. For the CT receptor, future sediment ingestion and dermal contact target organ HIs were less than or equal to the target HI of 1 for all locations. For the RME receptor, HIs for future sediment ingestion and dermal contact exceeded the target HI of 1 at locations SC05, SC06, and SC08 (Tables 3-9.91.RME, 3-9.92. RME, and 3-9.94.RME, respectively). In all cases, the largest contributor to the HI exceedances at these locations was arsenic. For all other locations, RME HIs for future exposures were at or below the target HI of 1.

3.5.2.2 Description of ILCR Estimates. Estimates of ILCR represent the incremental risk of cancer from the study area, as described in subsection 3.5.1. Tables 3-7.1 through 3-7.13 present the cancer risks by receptor and medium. ILCRs were summed for the child and adult receptors to derive a total receptor risk for the recreational receptor. The total receptor cancer risks, summed for the adult and child receptors, are presented on Tables 3-9.1 through 3-9.104.

Current Recreational Receptor. The estimated ILCRs for each pathway and medium, presented by station and reach, are listed for the current recreational receptor in Tables 3-9.1 through 3-9.32. Surface water ILCRs in these tables are based on 1995 data. Surface water ILCRs based on the 2001/2002 baseflow surface water data are provided in Tables 3-9.73 through 3-9.84. Sediment and surface soil ILCRs for station AJRW are presented on Table 3-9.72.

Surface Water. ILCRs for surface water ingestion and dermal contact were estimated, based on 1995 data, to be below 10^{-6} for all stations. The exposure areas with the highest ILCR from contact with surface water were stations WS/WSS and 05, each with an ILCR of 9×10^{-7} for the RME individual (Tables 3-9.6.RME and 3-9.25.RME). For the 2001/2002 baseflow surface water data, current and/or future surface water ILCRs were also below 10^{-6} for all stations.

Sediment/Surface Soil. For the current CT and RME receptors, ILCRs for sediment ingestion and dermal contact were estimated to be below or within the target risk range of 10^{-6} to 10^{-4} for all stations quantitatively evaluated. The exposure areas with the highest ILCR from contact with sediment were stations WH and CB-03, each with an ILCR of 1×10^{-4} for the RME individual (Tables 3-9.4.RME and 3-9.10.RME).

ILCRs for current surface soil ingestion and dermal contact were below or within the target risk range of 10^{-6} to 10^{-4} for all stations. The exposure area with the highest ILCR from contact with surface soil was station DA, with an ILCR of 6×10^{-5} for the RME individual (Table 3-9.17.RME).

Fish Fillet Tissue. ILCRs for fish fillet tissue ingestion were estimated to be above 10^{-6}, but below 10^{-4}. The reaches with the highest ILCR from fillet ingestion were reaches 4, 5, and 6, with ILCRs of 3×10^{-5} (Tables 3-9.31.RME and 3-9.32.RME).

Future Recreational Receptor. The estimated ILCRs for each pathway and medium, presented by station and reach, are listed for the future recreational receptor in Tables 3-9.33 through 3-9.71. Surface water ILCRs in these tables are based on 1995 data. Surface water ILCRs based on the 2001/2002 baseflow surface water data are provided in Tables 3-9.73 through 3-9.84. Sediment and surface soil ILCRs for station AJRW are presented on Table 3-9.72.

Surface Water. ILCRs for surface water ingestion and dermal contact were estimated, based on 1995 data, to be below 10^{-6} for all stations. The exposure areas with the highest ILCR from contact with surface water were stations WS/WSS and 05, each with an ILCR of 9×10^{-7} for the RME individual (Tables 3-9.44.RME and 3-9.64.RME).

Sediment/Surface Soil. For the CT receptor, ILCRs for sediment ingestion and dermal contact were estimated to be below or within the target risk range of 10^{-6} to 10^{-4} for all stations quantitatively evaluated. For the RME receptor, ILCRs for sediment ingestion and dermal contact were estimated to be above 10^{-4} for stations 13/TT-27 (7×10^{-4}; Table 3-9.36.RME), WH (4×10^{-4}; Table 3-9.37.RME), NT-1 (5×10^{-4}; Table 3-9.38.RME), and NT-2 (3×10^{-4}; Table 3-9.39.RME). The largest contributor to the RME ILCRs in excess of 10^{-4} was arsenic. Benzo(a)pyrene was also a minor risk contributor at stations 13/TT-27, WH, NT-1, and NT-2. The risk associated with benzo(a)pyrene at these stations was between 2×10^{-6} and 3×10^{-6}.

ILCRs for future surface soil ingestion and dermal contact were below or within the target risk range of 10^{-6} to 10^{-4} for all stations. The exposure area with the highest ILCR from contact with surface soil was station DA, with a ILCR of 6×10^{-5} for the RME individual (Table 3-9.56.RME).

Fish Fillet Tissue. ILCRs for fish fillet tissue ingestion were estimated to be above 10^{-6}, but below 10^{-4}. The reaches with the highest ILCR from fillet ingestion were reaches 4, 5, and 6, with ILCRs of 3×10^{-5} (Tables 3-9.70.RME and 3-9.71.RME).

Current/Future Area Resident Receptor. The estimated ILCRs for current/future land use are listed for the area resident receptor in Tables 3-9.85 through 3-9.90 for surface water, presented by gauging station, and Tables 3-9.100 through 3-9.104 for floodplain surface soil, presented by station. Surface water ILCRs in these tables are based on 2001/2002 storm event data. Surface soil ILCRs are presented for stations WS/WSS, CB-05, 07/DP, KF, and AJRW, where periodic flooding of the river may have impacted residential yards.

Surface Water. ILCRs for current/future storm event surface water dermal contact, based on 2001/2002 data, were below the target risk range of 10^{-6} to 10^{-4} for all gauging stations. The exposure areas with the highest ILCR from contact with surface water were gauging station SW-06 and SW-08, each with an ILCR of 2×10^{-7} for the current/future RME individual (Tables 3-9.86.RME and 3-9.88.RME).

Surface Soil. For the CT and RME receptors, current/future soil ingestion and dermal contact total receptor ILCRs were within the target risk range of 10^{-6} to 10^{-4} for all station. The exposure areas with the highest ILCR from contact with surface soil were stations CB-05 and AJRW, each with an ILCR of 7×10^{-5} for the current/future RME individual (Tables 3-9.104.RME and 3-9.102.RME).

Dredger Receptor. The estimated ILCRs for sediment ingestion and dermal contact, presented by sediment core location, are listed for the dredger receptor in Table 3-9.91 through 3-9.99 for future land use. ILCRs for future sediment ingestion and dermal contact were below or within the target risk range of 10^{-6} to 10^{-4} for all locations. The location with the highest ILCR from contact with sediment was location SC06, with an ILCR of 6×10^{-5} for the RME individual (Table 3-9.92.RME).

3.5.2.3 Risks Associated with Exposure to Lead. Lead was selected as a COPC for sediment, surface water and fish fillet tissue. Childhood surface water and sediment lead exposures at the study area were evaluated through use of the IEUBK model (USEPA, 2002b). Adult sediment lead exposures were evaluated using the methodology provided by USEPA (2003a). The results of the lead evaluation for the study area are contained in Appendix C.10. For adult exposures, the calculated central estimate of the blood lead concentration in women of childbearing age did not exceed the goal of 4.2 Wg/dL for current and future land use. Likewise, assumed childhood lead exposures were not estimated to result in blood lead levels exceeding the goal of 10 Wg/dL. Appendix C.10 provides inputs and outputs for both of these models.

Lead in fish fillet tissue was not evaluated through the use of these models. Instead, lead is discussed relative to average background levels provided in USEPA, 1994c. The average fish fillet tissue lead level for the study area is 0.043 mg/kg for reaches 3 through 6 combined. Average fish fillet tissue background levels are 0.34 mg/kg. Study area fish fillet average lead levels are less than the published background level. Therefore, the risk associated with lead in fish fillet tissue is consistent with that associated with fish tissue not impacted by the study area.

3.5.3 Description of Uncertainties

Estimation of risks to human health that may result from exposure to chemicals in the environment is a complex process that often requires the combined efforts of multiple disciplines. Each assumption, whether regarding the toxicity value to use for a particular chemical or the value of a parameter in an exposure equation, has a degree of variability and uncertainty associated with it. In each step of the risk assessment process, beginning with the data collection and analysis and continuing through the toxicity assessment, exposure assessment, and risk characterization, conservative assumptions are made that are intended to be protective of human health and to ensure that risks are not underestimated. For the study area, there is a probability of overestimating health risks or hazards for a number of reasons. The following subsections provide a discussion of the key uncertainties that may affect the final estimates of human health risk in this risk assessment. Uncertainties are arranged by topic.

3.5.3.1 Environmental Sampling and Analysis. The process of environmental sampling and analysis results in uncertainties from several sources, including errors inherent in sampling procedures or analytical methods. One area of uncertainty is sampling procedures. Since it is not possible to sample the entire area of interest at a given site, several samples are taken from each medium within each area of a site, and the results are considered to be representative of the chemicals present throughout the area. For fish tissue, it was assumed that the samples collected were representative of the chemicals present in all fish within the study area. This assumption may overestimate or underestimate risk. The concentrations of COPCs in floodplain surface soils immediately adjacent to the river were assumed to be representative of the concentrations of COPCs in area yards potentially impacted by flooding events. This assumption likely results in an overestimation of risk since concentrations likely decrease with increasing distance from the river.

Analytical methods also involved uncertainties. Due to uncertainty of quantification, individual chemicals were sometimes listed as detected, but with the value qualified as estimated by laboratory qualification or validation procedures. The estimated value was used in the risk assessment. This uncertainty may either over- or underestimate risk depending on how close the estimated value is to the true value. In some cases, analytical errors or sampling errors resulted in the rejection of data, which decreased the amount of data available and increased uncertainty associated with the representativeness of the detected chemical concentrations. Again, this may result in either an overestimation or underestimation of risk.

To decrease the uncertainty associated with this risk assessment, two specific analytical studies were conducted. The site-specific relative bioavailability study was performed to decrease the uncertainty associated with the ingestion of arsenic-containing sediment. This study involved the feeding of arsenic to swine in a sediment matrix. The oral absorption of arsenic from the sediment matrix was quantified and determined to be less than the absorption of arsenic from a water medium. Two relative bioavailability estimates, representing the mean bioavailability values for two different sediment types, were determined from the study. The most conservative relative bioavailability estimate was then used in the human health risk assessment to more accurately characterize the risk associated with sediment ingestion at the study area. Use of the most conservative estimate of oral bioavailability may have resulted in an overestimate of risk.

The second analytical study performed involved the determination of the most reliable analytic method to measure chromium VI in sediments. This study resulted in site-specific chromium speciation data that were used to more accurately characterize risk associated with chromium exposures at the study area. The chromium speciation study indicated that approximately 0.13% of total chromium in sediments existed in the hexavalent state, the more toxic form. This 0.13% value is based on the result of a single detected concentration of chromium VI in those samples analyzed by ion chromatography, the method demonstrated to be the most reliable. This uncertainty may result in either and underestimate or overestimate of risk depending on the representativeness of that single sample to the entire data set.

With respect to determining exposure point concentrations for this evaluation, one assumption was that the concentrations of chemicals in the medium evaluated would remain constant over time. Depending on the properties of the chemical and the medium in which it was detected, this assumption may overestimate risks, depending on the degree of chemical degradation to less toxic species or transport to other media. Conversely, environmental bioactivation of chemicals to more toxic chemicals (e.g., the environmental conversion inorganic mercury to organic forms of mercury) was also not considered. Therefore, this assumption may underestimate risk if bioactivation mechanisms are significant.

3.5.3.2 Selection of Chemicals for Evaluation. A comparison of maximum detected chemical concentrations to USEPA Region 9 PRGs for surface water, sediment and soil, and Region III RBCs for fish was conducted. PRGs and RBCs are conservative risk-based values that are used when selecting COPCs so as not to omit a chemical that might contribute significantly to risk. Chemicals whose maximum concentrations were below their respective cancer screening value or 10% of their noncancer screening value were not carried through the assessment. It is unlikely that this risk-based screening excluded chemicals that would be of concern, based on the conservative exposure assumptions and conservatively derived toxicity criteria that are the basis of the screening criteria. Although following this methodology does not provide a quantitative risk estimate for all chemicals, it focuses the assessment on the chemicals accounting for the greatest risks (i.e., chemicals whose maximum concentrations exceeded their respective PRGs or

RBCs), and, although the overall risk estimates are uncertain, it is not expected that actual risks will be significantly greater than estimated risks.

AWQCs for human health were also used to select COPCs for surface water. It should be noted that the arsenic AWQC is currently under review and may be revised to a higher value in the future. Regardless, arsenic would still be selected as a surface water COPC since arsenic concentrations exceed Region 9 PRGs, additionally used to select surface water COPCs.

3.5.3.3 Toxicological Data. Uncertainty is associated with the toxicity values and toxicity information available to assess potential adverse effects.

One of the major contributors to uncertainty is the accuracy of the toxicity values used. The assumptions used by the USEPA in the dose-response extrapolation model for carcinogens were based on a 95% UCL of the maximum likelihood estimate. Other assumptions include the following: 1) the extrapolation of data from high-dose exposures in human and animal studies to the low-dose exposure region of the general population is linear and does not have a threshold; 2) there is an interspecies (i.e., animal to man) correlation, based on body surface area; and 3) there is a conditional probability that cancer incidence demonstrated in animal studies will be similar to the incidence in potentially exposed humans. To the extent that any of these assumptions are incorrect, the extrapolated risks may be over- or underestimates. However, it should be noted that in the derivation of toxicity values, conservative assumptions are employed. Therefore, toxicity values tend to be biased toward overestimating risk.

One chemical for which there is some evidence of a nonlinear dose-response is arsenic (Chen *et al.*, 1992; Tseng, 1977; Tseng *et al.*, 1968). Since arsenic is a primary contributor to potential cancer risks to recreational and dredging receptors from the ingestion of sediment, the interpretation of whether there is a non-toxic threshold for arsenic could affect whether arsenic levels in sediment result in risks in excess of risk management criteria. The quantitative estimates of risk presented in this risk assessment assumes no threshold for carcinogenicity from arsenic, which may overestimate risks. More recent epidemiological studies (Lewis *et al.*, 1999; Moore *et al.*, 2002), which were not available at the time of development of the cancer slope factor for arsenic, failed to demonstrate a significant correlation between arsenic exposure and cancer. However, exposure levels were much lower than those believed to have occurred in study that serves as the basis for the oral slope factor derivation.

Toxicity values were lacking for a small number of COPCs. For example, there are no non-carcinogenic toxicity values (i.e., RfDs) for the carcinogenic PAHs. However, the noncarcinogenic effects of these compounds are likely to be adequately protected against by the evaluation of carcinogenic risks (i.e., carcinogenic effects appear at a lower does than non-carcinogenic effects). However, the lack of toxicity values for some COPCs contributes to an underestimation of risk.

3.5.3.4 Exposure Assessment. The primary areas of uncertainty affecting exposure parameter estimation involve the assumptions regarding exposure pathways, the estimation of exposure point concentrations, and the parameters used to estimate chemical doses. The uncertainties associated with these various sources are discussed below.

For dermal exposure pathways, the absence of dermal toxicity criteria necessitated the use of oral toxicity data. To calculate risk estimates for the dermal pathway, absolute oral bioavailability factors that reflect the toxicity study conditions were used to modify the oral toxicity criteria. For the chemicals with oral absorption exceeding 50% (i.e., the PAHs), a default oral absorption factor of 100% was used. The risk estimates for the dermal pathways may be over- or underestimated depending on how closely these values reflect the difference between the oral and dermal routes. Dermal absorption fractions (USEPA, 2004d), which estimate the penetration of sediment- or soil-associated compounds through the skin, are additionally used to assess dermal exposures for sediments and soils. These estimates are uncertain, and may result in either an overestimation or underestimation of risk.

To better quantify exposure point concentrations, USEPA's software program, Pro UCL version 3.0, was used to determine UCLs. This software has been extensively reviewed and provides the best available science for the statistical determination of EPCs. The use of this program is believed to result in the more accurate estimation of EPCs than previously used methods. However, in cases where there is high degree of variability between the data points for a COPC, a UCL may be uncertain. For example, the sediment RME EPC for arsenic at some stations is uncertain due to one or a small number of arsenic detects compared to the remainder of the data set. This uncertainty is specifically applicable to stations 13/TT-27 (samples SD-13-01-FW and SD-13-02-FW; 4210 mg/kg and 2480 mg/kg, respectively), WH (sample SD-12-01-ME; 3230 mg/kg), and CB-03 (sample CB-03-11; 1410 mg/kg). This uncertainty may results in either an overestimate or underestimate of risk.

The exposure assumptions selected for this evaluation were based on CT and RME case exposures. RME risks are conservative since estimated risks are based on upper-bound exposure assumptions. The RME individual is also assumed to be exposed to the UCL concentration of every chemical in sediment each time they visit the area. Note that the maximum concentration of different COPCs often occurs in different locations within the exposure area. Additionally, recreational activities assumed in this analysis may occur less frequently than assumed. Each of these assumptions may result in an overestimate of risk.

The parameter values used to describe the extent, frequency, and duration of exposure are associated with some uncertainty. Actual risks for some individuals within an exposed population may vary from those predicted depending upon their actual intake rates (e.g., sediment ingestion rates) or body weights. The exposure assumptions were selected to produce an upper-bound estimate of exposure in accordance with USEPA guidelines regarding evaluation of potential exposures at Superfund sites (e.g., exposures were assumed to occur for 9 to 30 years for recreational adults). Therefore, exposures and estimated potential risks for the majority of the evaluated receptors are likely to be overestimated.

Only sediment samples collected from below two feet or less of standing water were used in the human health risk assessment. This approach limited the number of samples available to calculate sediment EPCs at some stations. For these stations, the maximum detected level of a COPC in sediment may have been used as the RME EPC. Use of the maximum detected result instead of the UCL value for the RME EPC most likely results in an overestimate of risk. Depending on the representativeness of the available samples to the study area as a whole, this approach may have resulted in an over- or underestimation of risk.

3.5.3.5 Risk Characterization. Cancer risks and HIs for each receptor were not summed across all media. For example, the risks to the recreational receptor from surface water, sediment and surface soil ingestion and dermal contact were not summed with those from fish fillet tissue ingestion. In addition, risks from a given medium were not summed across exposure areas (i.e., stations). That is, for the recreational receptor, risks from ingestion of and dermal contact with sediment were assumed to occur within a given station. This assumption is uncertain since a given recreational receptor may spend half his time in one exposure area and half in another. Risks to such an individual would be intermediate between the risks to individuals exposed solely within each exposure area.

The quantification of risk based on one sample, as with stations AS and AM, may result in risk estimates associated with a high degree of uncertainty.

3.5.3.6 Overall Uncertainty. This risk assessment contains many layers of conservative assumptions. For example, in the RME case, the value selected for each parameter in each equation used to calculate risks to the RME individual is a maximum or upper-bound assumption.

Therefore, the estimated risk is likely to be greater than the UCL of all potential risks. If the risk assessment was able to capture the uncertainty and variability associated with each parameter, it is likely that the actual potential risk to the RME individual would be less than the risks estimated in this assessment.

3.5.4 Summary of Human Health Risks

An overall summary of cancer and noncancer risk estimates for the adult, young child and older child recreational receptors, adult and young child area residents, and adult dredging workers are presented in Tables 3-9.1 through 3-9.104. In these tables, risks are summarized for both the RME and CT receptors. When noncarcinogenic risks were estimated for a child and adult receptor, the child HIs are presented as the most conservative since exposures to children factor in higher ingestion rates and lower body weights. ILCRs presented for these receptors are the sum of the child and adult risks (i.e., a total receptor cancer risk). Adult and child HIs are not summed since noncarcinogenic effects occur over a short timeframe, unlike carcinogenic effects that are cumulative over one's lifetime.

Surface water, sediment, and surface soil risks, presented by station, have been summed together under the assumption that each receptor is exposed to all three media during recreational activities. Fish fillet ingestion risks, presented by reach, are summarized separately. Based on comments received on the draft baseline human health risk assessment, residential risks associated with storm event surface water and floodplain surface soil exposures as well as workers risks associated with a future dredging scenario have been estimated. In addition, HIs, segregated by systemic effects, are presented. In cases where the total HI exceeded 1, COPCs having similar systemic effects were summed for each pathway and medium. Tables 3-10.1 through 3-10.11 summarize the primary risk contributors for those receptors with estimated ILCRs greater than the target range of 10^{-6} to 10^{-4} and target organ-specific HIs greater than 1.

ILCRs and HIs, estimated for the surface water and surface soil exposure scenarios, were less than or within the target cancer risk range of 10^{-6} to 10^{-4} and less than and HI of 1, respectively, for each of the exposure areas. When HIs were summed only for COPCs with similar target organs, segregated HIs for fish fillet ingestion were less than or equal to the target HI of 1 for all study area reaches. In addition, total receptor ILCRs were below or within the target risk range for the fish ingestion pathway.

For RME recreational exposure to sediment, HIs exceeded 1 and/or ILCRs exceeded 10^{-4} at stations 13/TT-27 (future), WH (current/future), NT-1 (future), NT-2 (future), NT-3 (future), and CB-03 (current/future). All stations noted are located within the Wells G&H 38-acre wetland, north of Salem Street in Woburn, with the exception of CB-03 which is located in the former cranberry bog immediately south of Salem Street. The exceedances were due primarily to the presence of arsenic in sediment. Benzo(a)pyrene was also a minor risk contributor at stations 13/TT-27, WH, NT-1, and NT-2. The risk associated with benzo(a)pyrene at these stations was between 2×10^{-6} and 3×10^{-6}.

For RME worker exposures to sediment cores during potential future dredging of the river, HIs exceeded 1 at locations SC05, SC06, and SC08, located within the Wells G&H 38-acre wetland. The exceedances were due primarily to the presence of arsenic in sediment.

It should be noted that arsenic was present in all sediment samples collected from the reference stations. However, detected levels at the reference stations ranged from 3.8 mg/kg to 44.5 mg/kg (see Appendix C.3; Table C.3–2.5) compared to a range at station 13/TT-27 of 15.9 mg/kg to 4,210 mg/kg (Table 3-2.2.4), station WH of 4.7 mg/kg to 3,230 mg/kg (Table 3-2.2.5), station NT-1 of 6.6 mg/kg to 4,250 mg/kg (Table 3-2.2.6), station NT-2 of 6.6 mg/kg to 3,230 mg/kg (Table 3-2.2.7), station NT-3 of 6.6 mg/kg to 3,230 mg/kg (Table 3-2.2.8), and station CB-03 of 9.1 mg/kg to 1,410 mg/kg (Table 3-2.2.17). It should further be noted that the sediment RME EPC for arsenic at some of these stations is uncertain due to one or a small number of arsenic detects compared to the remainder of the data set. This uncertainty is specifically applicable to stations 13/TT-27 (samples SD-13-01-FW and SD-13-02-FW; 4210 mg/kg and 2480 mg/kg, respectively), WH (sample SD-12-01-ME; 3230 mg/kg), and CB-03 (sample CB-03-11; 1410 mg/kg).

Benzo(a)pyrene was detected in 12 of 16 sediment samples collected from the reference stations. Detected levels at the reference stations ranged from 0.13 mg/kg to 5.5 mg/kg (see Appendix C.3; Table C.3–2.5) compared to a range at station 13/TT-27 of 0.15 mg/kg to 1.7 mg/kg (Table 3-2.2.4), station WH of 1 mg/kg (Table 3-2.2.5), station NT-1 of 0.16 mg/kg to 1 mg/kg (Table 3-2.2.6) and station NT-2 of 0.16 mg/kg to 1 mg/kg (Table 3-2.2.7). The benzo(a)pyrene sediment concentrations at these study area stations fall within the range of concentrations detected at the reference stations.

An evaluation of lead in surface water, sediment, and surface soil at the study area indicated that exposures to lead did not result in estimated childhood or adult blood lead levels in excess of blood lead level goals. Lead concentrations in fish fillet tissue were consistent with published background lead levels. Therefore, lead was determined not to be of concern for human receptors at the study area.

SECTION 4.0
BASELINE ECOLOGICAL RISK ASSESSMENT

4.1 SCREENING-LEVEL ECOLOGICAL EFFECTS EVALUATION

The objective of this Baseline Ecological Risk Assessment (BERA) is to determine whether contaminants present in surface water, sediment, and biota of Wells G&H Superfund Site Operable Unit 3 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 U.S. Environmental Protection Agency (USEPA) released the draft BERA in May 2003. Based on USEPA responses to comments received during the comment period, the draft risk assessment has been revised. Notable aspects of this revision include:

- recalculation of exposure point concentrations based on the use of USEPA's updated software program ProUCL (version 3.0);
- consideration of recent sediment samples collected along the Aberjona River in Winchester, south of Bacon Street (station AJRW);
- consideration of recent sediment core data collected from nine locations along the Aberjona River between Route 128 and the Mystic Lakes (SC05 through SC13); and
- consideration of recent surface water baseflow and storm event data collected from six surface water gauging stations along the Aberjona River between Route 128 and the Mystic Lakes (SW-05 through SW-10).

The first step of the BERA is the screening-level ecological effects evaluation. Between 1995 and 2001, surface water and sediment samples were collected along a 6-mile stretch of the Aberjona River system from Route 128 in Woburn extending south into the Upper and Lower Mystic Lakes in Arlington (see Figure 1-1). During this screening-level effort, chemicals of potential concern (COPCs) were selected and carried forward for quantitative 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. The recently collected sediment, surface water, and sediment core data have not been quantitatively used in the BERA. Instead, these data have been qualitatively evaluated. A discussion of these data and the results of the qualitative evaluation is included in Appendices E.4 and E.5. The results of the qualitative evaluation are also discussed relative to the results of the BERA in Section 4.5.5 (Summary of Ecological Risks) and Section 5.0 (Summary and Conclusions).

4.1.1 Surface Water and Sediment Sampling Locations

Site-related surface water and sediment samples were collected within the Aberjona River, Aberjona River wetlands, and Upper and Lower Mystic Lakes (see Figures 1-3 and 2-2). Local and regional reference areas, representative of various on-site 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. Reference stations are identified as stations 23 through 27, 01-IP through 04-IP, 12-IP, HA, and SB. Their locations are shown relative to the study area in Section 2.0, Figure 2-2. A detailed discussion of the site-related and reference sampling stations, sampling design, and analytical results are presented in Sections 2.1 and 2.2.

4.1.2 Surface Water Screening

Surface water data for Upper Mystic Lake (stations SW-01 through SW-04), the main channel of the Aberjona River (stations SW-05 through SW-12, SW-14, and SW-16), and the Aberjona River wetlands (stations SW-13, SW-15, and SW-18 through SW-22) were screened separately (station locations are shown in Figures 2-3 through 2-10). These stations were divided in this manner because water quality parameters (*e.g.*, hardness) and hydrology (*e.g.*, capacity to dilute contaminant inputs) of the these three habitats (lake, river, and wetlands) differ greatly.

Station SW-12 is a tributary channel to the Aberjona River, and was included in the river samples based on habitat quality. SW-15 was collected in a shallow blind side channel and was grouped with the wetland samples. In addition, samples were collected from 10 reference locations and grouped together to represent reference conditions. Water samples were not filtered prior to analysis. A comprehensive reporting of surface water data is provided in Section 2 text, tables, and figures, as well as Appendix B, Table B-1.

Maximum concentrations of analytes detected in the surface waters of Upper Mystic Lake, the main channel of the Aberjona River, and the Aberjona River wetlands were each compared to USEPA freshwater chronic National Ambient Water Quality Criteria (NAWQC), or, if unavailable, Lowest Observed Effect Levels (LOELs) (USEPA, 1986a,b; 1987; 1992a, 1998b). If no freshwater chronic criterion was available for a particular analyte, the freshwater acute criterion was selected as the screening value, if available. Tier II screening values presented in Suter and Tsao (1996) were used for analytes without NAWQC. Tier II values were calculated using the Great Lakes Water Quality Initiative Tier II methodology (USEPA, 1993a) and are analogous to NAWQCs.

NAWQCs for cadmium, chromium, copper, lead, nickel, and zinc were normalized for habitat-specific average hardness: 103 mg/L as $CaCO_3$ for Upper Mystic Lake; 192 mg/L for the main channel of the Aberjona River; and 244 mg/L for the Aberjona River wetlands. Potassium, sodium, calcium, and magnesium were excluded from the screening process because they are nutrients and occur naturally at high concentrations.

Frequency of detection was not utilized as a screening tool for surface water due to the limited number of samples collected within each habitat type (maximum of 10 samples in the main channel of the Aberjona River). Data from reference locations were also not used as a screening tool to avoid eliminating any potential COPCs based on reference data collected within an urban watershed, but were used as a comparison for local/regional conditions. Surface water screening data are presented in Table 4-1 through 4-3.

Additional baseflow and storm event surface water data were collected from surface water gauging stations located along the entire length of the river. These data are discussed separately in Appendix E.4.

4.1.2.1 Upper Mystic Lake. A total of thirteen inorganics (Table 4-1) were detected in the four surface water samples collected from Upper Mystic Lake (SW-01-01 through SW-04-01). Of these, aluminum, barium, iron, lead, and manganese had maximum detected concentrations in excess of screening values. With the exception of lead, detection percentage was 100% for these inorganics. Freshwater chronic NAWQCs for aluminum, iron, and lead were exceeded in the Upper Mystic Lake. Surface water aluminum and lead concentrations exceeded the NAWQC at both stations SW-01-01 and SW-03-01 and the surface water iron concentration exceeded the NAWQC at SW-03-01, only. The TIER II (Suter and Tsao, 1996) screening values were exceeded at all Upper Mystic Lake stations for barium and 2 of 4 stations for manganese.

4.1.2.2 Main Channel of the Aberjona River. Ten surface water samples (SW-05-01 through SW-12-01, SW-14-01, and SW-16-01) were collected within the main channel of the Aberjona River from Olympia Avenue (SW-14-01) southward near the inlet to Upper Mystic Lake (SW-05-01). VOCs, pesticides, and inorganics were detected in these samples (Table 4-2). VOCs included cis-1,2-DCE, chloroform, TCE, and PCE. All four compounds are associated with groundwater contamination identified at the Wells G&H Superfund Site (PRC, 1986). Four pesticides (heptachlor epoxide, methoxychlor, alpha-chlordane, and gamma-chlordane) and sixteen inorganics were present at one or more of the ten sampling stations. With the exception of manganese, the maximum detected concentration of every inorganic occurred at station 10, immediately upstream of Salem Street in Woburn. This water sample (SW-10-01) also had the highest total suspended solids (TSS) concentration.

The maximum detected concentrations of the detected VOCs and pesticides were all less than their respective screening criteria. Inorganics detected at concentrations in excess of screening criteria included aluminum, barium, cadmium, copper, iron, lead, manganese, mercury, and zinc. Frequency of detection of these inorganics ranged from 2 of 10 to10 of 10. Cadmium, copper, mercury, and zinc concentrations only exceeded surface water NAWQCs at SW-10-01. Lead exceeded NAWQCs at stations SW-10-01 and SW-16-01. Iron concentrations exceeded the NAWQC at 7 of the 10 stations and aluminum at 4 of 10 stations sampled in the main channel of the Aberjona River. The TIER II (Suter and Tsao, 1996) screening values were exceeded at all main channel stations for barium and manganese.

4.1.2.3 Aberjona River Wetlands. Three surface water samples were collected within the Aberjona River wetlands (SW-13-01, SW-15-01, and SW-18-01). At the time of sampling (during the 1995 field investigation), there was no standing water at the other wetland sampling stations scheduled to be sampled (19, 20, 21, and 22). Of the three surface water samples collected, VOC analysis was conducted only on samples collected at stations 15 and 18.

Pesticides and inorganics were detected within the surface water of the Aberjona River wetlands (Table 4-3). Pesticides included endosulfan II and heptachlor epoxide. Fourteen inorganics were detected. The maximum detected concentrations of the pesticides and nine of the inorganics were less than their respective screening values. Inorganics detected in excess of screening values were aluminum, barium, iron, and manganese, with frequency of detection either 2 of 3 or 3 of 3. This group of inorganics represents a subset of those inorganics selected as COPCs within the main channel of the Aberjona River. Surface water aluminum and iron concentrations exceeded the NAWQC at both stations SW-13-01 and SW-18-01. The TIER II (Suter and Tsao, 1996) screening values were exceeded at all Aberjona River wetland stations for barium and manganese.

4.1.2.4 Reference Stations. Eleven surface water samples were collected at ten reference locations (SW-23-01 through SW-27-01, SW-01-IP through SW-04-IP, SW-12-IP). Two samples were collected at station SW-03-IP. There were nine surface water COPCs selected in the screening of surface water from the Aberjona River Study Area (Table 4-4). All of the selected surface water COPCs were inorganics. Reference location concentrations of these nine surface water COPCs are provided in Tables 4-5 to 4-15 for comparison purposes. NAWQCs for chromium, copper, lead, and zinc were normalized for site-specific hardness values.

With the exception of cadmium, mercury, and zinc, the concentrations of each of the inorganic COPCs in surface water samples from reference locations exceeded their respective screening values at one or more location. Inorganics detected at concentrations in excess of screening criteria included aluminum, barium, copper, iron, lead, and manganese. The TIER II (Suter and Tsao, 1996) screening values were exceeded at all stations for barium and 8 of 10 stations for manganese. The NAWQCs for aluminum, iron, and lead were also exceeded in at least half of the 10 reference locations. Surface water copper concentrations exceeded the NAWQC at only one (SW-01-IP) of the 10 reference locations.

4.1.3 Sediment Screening

For the purpose of sediment screening, all data from the study area were combined since there are depositional areas at various locations along the entire length of the Aberjona River where contaminants from upstream sources may have accumulated, including Upper Mystic Lake. Data for organics were from the 1995, 1997, and 2001 (triad) sampling rounds. Inorganic data sets screened for sediments included samples collected in 2000 and 2001, in addition to the 1995 and 1997 sampling rounds. For the metals, the total number of sediments samples for each analyte was between 317 to 355, with 122 samples analyzed for cyanide.

Additional sediment data collected in 2003 and 2004 were not included in this screening or in the risk calculations. These data include sediment cores collected at nine locations throughout the study area, and additional samples collected at station AJRW in reach 5. These data are discussed separately in Appendix E.5.

Maximum detected levels of chemicals in the BERA data set were compared to sediment quality criteria (Table 4-16). The sets of criteria used in the screening, in the order of selection, included:

- 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, 1996);

- Ontario Ministry of Environment and Energy (OMEE) Lowest Effect Levels (LELs) (Persaud *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 Level (TEL) (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%. With the exception of two samples (SD-01-07-FW and SD-09-06-FW), the organic carbon content of sediment samples with maximum detections of one or more organic analytes was greater than 1%. However, to maintain a conservative screening process, no screening criterion was adjusted upward to account for an organic carbon content of greater than 1%. Screening criteria were adjusted downward in those cases where the organic carbon content at the location of maximum detection was less than 1%.

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

Since more sediment samples were collected than surface water samples, frequency of detection was utilized 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.*, a pond utilized by amphibians for reproduction, a plunge pool in a river, 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.

Over 90 analytes were detected in study area sediments, including a number of VOCs, SVOCs, pesticides/PCBs, and inorganics (Table 4-16). Although detected less frequently than other chemical classes, nineteen VOCs were detected in 1% to 69% of the samples. Locations of maximum detections were mostly north of Salem Street, within the Wells G&H 38-acre wetland complex at the northern end of the study area, or at station TT-30 in the Cranberry Bog just south of Salem Street. VOCs detected at concentrations exceeding screening criteria included 2-butanone, acetone, carbon disulfide, cis-1,2-DCE, PCE, and TCE. With the exception of carbon disulfide, these compounds were selected as COPCs. Carbon disulfide was excluded from further consideration since it was detected in only 4% of samples (3 of 63). Although no criterion was available for vinyl chloride, it was not selected as a COPC since it was only detected in 2% of samples (2 of 87). 1,1,2-trichloro-1,2,2-trifluoroethane (freon 113), and methyl acetate were detected in a limited number of samples collected in June 2001. These compounds are both know to be laboratory contaminants, and were not selected as COPCs. Among the 5 VOCs selected as COPCs, only acetone and 2-butanone were detected at reference locations. The maximum detected concentrations at reference stations of both VOCs were observed at station 02-IP (Table 4-18).

Twenty-four SVOCs were detected in study area sediments. The detection percentage for many of the PAHs was greater than 80%. Locations of maximum detections varied, but a number of maximum levels were found in samples collected within the pond at Davidson Park (station 07). SVOCs detected at levels exceeding screening criteria included 2-methylnaphthalene, acenaphthylene, anthracene, benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(g,h,i)perylene, benzo(k)fluoranthene, chrysene, dibenz(a,h)anthracene, dibenzofuran, fluoranthene, fluorene, indeno(1,2,3-cd)pyrene, naphthalene, phenanthrene, and pyrene. These compounds were selected as COPCs. No screening criteria were available for carbazole, N-nitrosodiphenylamine, or di-n-octylphthalate. Of these, only di-n-octylphthalate was eliminated from further consideration since it was detected in only 4% of all samples (3 of 73). Among the 19 SVOCs selected as COPCs, only N-nitrosodiphenylamine, was not detected at any of the reference locations (Table 4-18). The maximum detected concentration of the majority of the SVOCs at reference sites were located at station 24.

Twenty pesticides and three PCB congeners (Aroclor-1248, -1254, and -1260) were also detected in study area sediments. The pond at Davidson Park (station 07) had many of the maximum pesticide detections. In contrast, maximum PCB levels were found in the Aberjona River and adjacent wetlands, north of Salem Street (stations 10 and JY). Pesticides and PCBs with maximum detected levels exceeding screening criteria included 4,4'-DDD, 4,4'-DDE, 4,4'-DDT, aldrin, alpha-chlordane, Aroclor-1248, Aroclor-1254, Aroclor-1260, beta-BHC, delta-BHC, endosulfan I, and gamma-chlordane. These compounds were selected as COPCs. There were no criteria available for endosulfan sulfate, endrin aldehyde, or endrin ketone. Of these, only endrin ketone was eliminated from further consideration since it was detected in only 4% of all samples (4 of 93). Of the eleven pesticides and three PCB congeners selected as COPCs, all were detected at reference locations, with the exception of delta-BHC and Aroclor-1254. The maximum sediment concentration of six of the pesticides (4,4'-DDD, 4,4'-DDE, 4,4'-DDT, alpha-chlordane, endosulfan I, and gamma-chlordane) and two PCBs (Aroclor-1248 and -1260) exceeded screening criteria at one or more reference locations.

A total of twenty inorganics were detected in site-related sediments, with the majority detected in over 90% of the samples. Maximum levels of every analyte exceeded screening criteria. Inorganics selected as COPCs included aluminum, antimony, arsenic, barium, beryllium, cadmium, chromium, cobalt, copper, cyanide, iron, lead, manganese, mercury, nickel, selenium, silver, thallium, vanadium, and zinc.

Among the sediment contaminants eliminated from further consideration based on detection frequency alone, there were no special circumstances identified that would warrant inclusion in the BERA. In general, infrequently detected sediment contaminants were also present at low concentrations. Among the twenty inorganic COPCs, all except cyanide were detected at reference locations. Twelve of the twenty inorganic COPCs (antimony, arsenic, cadmium, chromium, copper, iron, lead, manganese, mercury, nickel, silver, and zinc) had maximum detected concentrations at one or more reference location that exceeded the sediment screening criteria.

4.1.4 Chemicals of Potential Concern Summary

COPCs for the Aberjona River Study area BERA are the sum of the COPCs selected for surface water and sediment. The effects-based screening resulted in the selection of nine COPCs in surface water (all inorganics; Tables 4-1 through 4-3) and fifty-eight COPCs in sediment (VOCs, SVOCs, pesticides/PCBs, and inorganics; Table 4-16). Inorganics COPCs selected in surface water were a subset of those selected in sediment.

4.2 PROBLEM FORMULATION

Problem Formulation, which is the first step in the risk assessment process, includes toxicity literature reviews for COPCs, descriptions of site resources, identification of complete exposure pathways, selection of assessment and measurement endpoints, and preparation of a site conceptual model. General fate and transport characteristics of site-related contaminants are discussed in Section 2.3 of this report. The potential for site-related contaminants to accumulate in the tissues of exposed organisms is addressed below.

4.2.1 Ecotoxicity Literature Review

An ecotoxicity literature review has been performed and is discussed in the following subsections.

4.2.1.1 Volatile Organic Compounds. VOCs were detected in many of the sediment samples collected at the study area. In surface water, VOCs were detected in the main channel of the Aberjona River 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.

Acetone. The lowest chronic value (LCV) reported for daphnids in freshwater is 1,560 µg/L (Suter and Tsao, 1996). The estimated LCV for fish is approximately 510,000 µg/L. Using the equilibrium partitioning (EqP) approach to develop a sediment quality criterion, Jones *et al*. (1997) calculated a SCV of 8.7 µg/kg for freshwater aquatic organisms, based on 1% sediment organic carbon content. For acetone, which is a polar organic compound, Jones *et al*. (1997) indicates that the EqP approach is likely to result in a conservative estimate of exposure (*i.e.*, the acetone SCV may be lower than the level which would be associated with an impact to ecological receptors).

2-Butanone. As estimated by Suter and Tsao (1996), LCVs for daphnids and fish in freshwater are approximately 1,400,000 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.

cis-1,2-Dichloroethene. The estimated freshwater LCV for fish exposed to 1,2-DCE is approximately 9,500 µg/L (Suter and Tsao, 1996). The sediment SCV for 1,2-DCE is 400 µg/kg

(Jones *et al.*, 1997). Research conducted with rats orally exposed to cis-1,2-DCE in the diet indicated a no observed adverse effect level (NOAEL) for respiratory function of 870 mg/kg-day (ATSDR, 1994b).

Tetrachloroethene. LCVs reported for freshwater daphnids and fish are 750 and 840 µg/L, respectively (Suter and Tsao, 1996). Jones *et al.* (1997) present a sediment SCV for PCE of 410 µg/kg, based on 1% sediment organic carbon content.

Trichloroethene. The LCV reported for fish in freshwater is 11,100 µg/L (Suter and Tsao, 1996). For daphnids, the estimated LCV is approximately 7,300 µg/L. For sediment, the SCV was calculated to be 220 µg/kg, based on 1% sediment organic carbon content (Jones *et al.*, 1997).

4.2.1.2 Semi-Volatile Organic Compounds (Polycyclic Aromatic Hydrocarbons). 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. In sediment toxicity tests with the tubificid, *Limnodrilus hoffmeisteri*, Lotufo and Fleeger (1996) observed a median lethal phenanthrene level of 298 mg/kg (sediment organic carbon content = 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 control] of 40.5 mg/kg and 59.1 mg/kg for phenanthrene and pyrene, respectively).

Sediment-associated PAHs can be accumulated by bottom-dwelling invertebrates and fish (Eisler, 1987a). Great Lakes sediments contaminated with elevated levels of PAHs were reported by Eadie *et al.* (1983 *cited in* Eisler, 1987a) to be the source of body burdens in bottom-dwelling invertebrates. Lake *et al.* (1985 *cited in* Eisler, 1987a) 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.

In aquatic environments, exposure to ultraviolet light can result in photomodification of 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.

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; avoidance of areas with high PAH contamination has been demonstrated in some fish species (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. The fact that mussels are ranked last may be because mussels 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, 1987a). USEPA has classified benzo(a)anthracene, benzo(b)fluoranthene, benzo(k)fluoranthene, benzo(a)pyrene, dibenz(a,h)anthracene, chrysene, 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 by USEPA (ATSDR, 1995b).

Carbazole. AQUIRE [Aquatic Toxicity Information Retrieval Database] (USEPA, 1998c) presents the following endpoints for aquatic receptors exposed to carbazole: an EC_{50} (concentration at which 50% of the individuals are affected) of 3,350 µg/L for the water flea, *Daphnia magna* (Brooke, 1991); and LC_{50}s ranging from 930 to <1,500 µg/L for the fathead minnow, *Pimephales promelas* (Brooke, 1991). Carbazole bioconcentration factors (BCFs) reported in AQUIRE (USEPA, 1998c) for *D. magna* and *Daphnia pulex* are 113.4 (Newsted and Giesy, 1987) and 65 (Southworth, 1979), respectively.

Data on the toxicity of carbazole to wildlife are limited. Dermal treatment with benzo(a)carbazole at a dose of 250 mg/kg resulted in significant reductions in maternal body weight gain and food consumption in pregnant Sprague-Dawley rats (Dutson *et al.*, 1997).

Dibenzofuran. The LCV estimated for daphnids is approximately 1,000 µg/L (Suter and Tsao, 1996). Using the EqP approach to develop a sediment quality criteria, Jones *et al.* (1997) calculated an LCV of 110,000 µg/kg for daphnids, based on 1% sediment organic carbon content.

N-nitrosodiphenylamine. The estimated LCV for daphnids is approximately 1,000 µg/L (Suter and Tsao, 1996). In studies by LeBlanc (1980 *cited in* AQUIRE [USEPA, 1998c]), LC_{50}s for *D. magna* were 7,800 and >46,000 µg/L. LC_{50}s of 5,800 and 44,000 µg/L have been reported for bluegill sunfish (*Lepomis macrochirus)* (Buccafusco *et al.*, 1981 *cited in* AQUIRE [USEPA, 1998c]). A BCF of 9.21 has also been reported for bluegill sunfish (Barrows *et al.*, 1980 *cited in* AQUIRE [USEPA, 1998c]).

4.2.1.3 Pesticides/PCBs - DDD, DDE, and DDT. LC_{50} values between 0.2 and 1,230 µg/L have been reported for aquatic invertebrates exposed to DDT and its breakdown products, DDD and DDE (USEPA, 1980). 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, 10 and 7 µg/L for mosquito larvae (*Culex fatigans* and *Anopheles albimanus*) and stonefly (*Pteronarcys california*), respectively. The most sensitive freshwater invertebrate reported by Mayer and Ellersieck (1986) was the water flea, *D. pulex*, with a 48-hr EC_{50} of 0.36 µg/L, based on immobilization.

In water, 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 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, 5.0 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, 1980).

Sediment ERLs for DDT, DDD, DDE and total DDT are 1, 2, 2.2, 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, 20, 27, and 46.1 µg/kg, respectively (Long and Morgan, 1990; Long *et al.*,1995).

Median lethal dietary concentrations in the range of 651 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. 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 induce death in organisms by interfering with central nervous system transmission through the disruption of sodium ion passage (Connell and Miller, 1984).

Acute median lethal dosages for birds include LD_{50}s of >2,240 mg/kg for mallard ducks and 841 mg/kg for Japanese quail (Hudson *et al.*, 1984). Following chronic exposures to DDT dietary concentrations of 100 mg/kg, 50% of exposed adult mallards died in about one year. DDE has been found to cause eggshell thinning in birds consuming a diet containing DDT and its breakdown products. Weimeyer *et al.*, (1970) found 14 to 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).

Aldrin. Aldrin has been demonstrated to be acutely toxic to several fish and aquatic invertebrate species. LC_{50}s range from 5.6 to 53 µg/L (Mayer and Ellersieck, 1986). The most sensitive species reported in Mayer and Ellersieck (1986) was the bluegill sunfish with a 24-hour LC_{50} of 9.3 µg/L and a 96-hour LC_{50} of 5.6 µg/L. The black bullhead (*I. melas*) was slightly less sensitive with a 24-hr LC_{50} of 22 µg/L and a 96-hr LC_{50} of 19 µg/L. The channel catfish was the least sensitive of the warmwater species evaluated with 24-hr and 96-hr LC_{50}s of 53 µg/L. The toxicity of aldrin to the water flea (*D. pulex*) was also measured, with a resulting 48-hr EC_{50} of 28 µg/L.

Information on the toxicity of aldrin to benthic invertebrates is limited. The OMEE LEL for aldrin is 2 µg/kg (Persaud *et al.*, 1993).

Aldrin can be acutely toxic to mammals. Median lethal dietary concentrations for 12 to 18 month old male mule deer were in the range of 18.8 to 37.5 mg/kg (Hudson *et al.*, 1984). Aldrin has been demonstrated to be acutely toxic to several species of birds. Lethal endpoints for birds exposed to aldrin in their diet include LD_{50}s of 6.59 mg/kg for bobwhite, 16.8 mg/kg for pheasant, 29.2 mg/kg for fulvous whistling duck, and 520 mg/kg for mallard (Hudson *et al.*, 1984). Necropsies of birds revealed liver adhesions to the parietal peritoneum in some individuals (Hudson *et al.*, 1984).

A chronic NOAEL for reproduction of 0.2 mg/kg-day was developed by Treon and Cleveland (1955 *cited in* Sample *et al.*, 1996). In that study, rats were fed aldrin in their diet over three generations.

Chlordane (alpha and gamma). Chlordane was formerly used as a pesticide in the United States. It is very persistent in the environment and bioaccumulates in aquatic and terrestrial organisms (USEPA, 1985). Aquatic LCVs for chlordane include 1.6, 16, and 1.09 µg/L for fish, daphnids, and non-daphnid invertebrates, respectively (Suter and Tsao, 1996). EqP-based sediment LCVs, based on 1% sediment organic carbon content, were calculated at 26,000, 260,000, and 18,000 µg/kg for fish, daphnids, and non-daphnid invertebrates, respectively (Jones *et al.*, 1997).

BHC (beta and delta). The daphnid LCV, for BHC isomers other than gamma-chlordane (*i.e.*, Lindane), was estimated as 95 µg/L (Suter and Tsao, 1996). Using the EqP approach, Jones *et al.* (1997) calculated a daphnid sediment LCV of 5,200 µg/kg.

A study conducted with rats orally exposed to beta-BHC in their diet over a 13-week period resulted in a NOAEL of 0.4 mg/kg-day and a lowest observed adverse effect level (LOAEL) of 2 mg/kg-day for effects on growth, blood chemistry, and organ histology (Van Velson *et al.*, 1986 *cited in* Sample *et al.*, 1996).

Endosulfan I. Endosulfan is a pesticide that exists in alpha (endosulfan I) and beta (endosulfan II) forms. The aquatic toxicity of endosulfan has been summarized by Mayer and Ellersieck (1986). The toxicity of endosulfan to aquatic invertebrates ranges from 9.2 to 24 µg/L (24-hour

LC_{50}s). 96-Hour LC_{50}s reported for the amphipods, *Gammarus fasciatus* and *G. lacustris,* and the stonefly, *P. californica*, were 6.0, 5.8, and 2.3 µg/L, respectively.

In general, fish are more sensitive to endosulfan exposure than are many other aquatic invertebrates. The fathead minnow, channel catfish, and bluegill sunfish demonstrated similar response thresholds to endosulfan, with 24-hour LC_{50}s of 2.4, 1.8 and 3.3 µg/L, respectively (Mayer and Ellersieck, 1986). 96-hour LC_{50}s for these organisms ranged from 1.2 to 1.5 µg/L. A sediment SCV of 5.5 µg/kg was calculated for all endosulfan isomers by Jones *et al.* (1997).

Hudson *et al.* (1984) reported LD_{50}s, in mg/kg, for mallards and pheasant of 31.2 and 190, respectively. Endosulfan is lipid soluble, and as a result, may be eliminated from the body in the eggs of a laying hen (Hudson *et al.*, 1984). A study conducted with rats exposed to endosulfan via oral intubation for 30 days indicated a subchronic NOAEL for reproductive effects of 1.5 mg/kg-day (Dikshith *et al.*, 1984 *cited in* Sample *et al.*, 1996). Abiola (1992 *cited in* Sample *et al.*, 1996) conducted a study in which the reproductive NOAEL for gray partridge orally exposed to endosulfan during a 4-week period was 10 mg/kg-day.

Endosulfan Sulfate. Information on the environmental toxicology of endosulfan sulfate is limited. Records in AQUIRE (USEPA, 1998c) include an EC_{50} of 21 µg/L for the sand dollar, *Dendraster excentricus,* (Dinnel *et al.*, 1982) and an LC_{50} of 756 µg/L for the water flea, *Daphnia carinata* (Barry *et al.*, 1995).

Endrin Aldehyde. Since information on the environmental toxicology of endrin aldehyde is limited, toxicity information on endrin, which is similar to endrin aldehyde, is provided in this section. Acute and chronic NAWQC for endrin are 0.095 and 0.061 µg/L, respectively. The USEPA EqP-based sediment quality criterion for endrin is 20 µg/kg, assuming 1% sediment organic carbon content (USEPA, 1993c). The NOAA ERL and ERM values are 0.02 and 45 µg/kg, respectively (Long and Morgan, 1990).

Acute oral LD_{50}s for endrin exposure to mule deer and domestic goat ranged from 6.25 to 50 mg/kg (Hudson *et al.*, 1984). Of the birds discussed in Hudson *et al.* (1984), the sharp-tailed grouse was the most sensitive to endrin, with an acute LD_{50} of 1.06 mg/kg. Other birds which were less sensitive included California quail (1.19 mg/kg), pheasant (1.78 mg/kg), rock dove (2 to 5 mg/kg), and mallard (5.64 mg/kg) (Hudson *et al.*, 1984). A study conducted with mallards orally exposed to endrin in their diet for more than 200 days indicated a NOAEL for reproduction of 0.3 mg/kg-day (Spann *et al.*, 1984 *cited in* Sample *et al.*, 1996).

Polychlorinated Biphenyls. PCBs have been shown to cause reproductive failure, birth defects, skin lesions, tumors, liver disorders, and death in fish and wildlife (Eisler, 1986a). Due to their high lipid solubility, PCBs bioaccumulate and biomagnify within the food chain. At the study area, the maximum detected concentrations of two PCB congeners, Aroclor-1248 and Aroclor-1260, were greater than screening criteria. Of the two congeners, more information is available on the environmental toxicity of Aroclor-1260.

Eisler (1986a) reports LC_{50} values for freshwater and marine organism exposed to various Aroclors from 0.1 to 10 µg/L, with crustaceans and younger developmental stages being the most sensitive. For Aroclor-1260, the LCV for fish is <1.3 µg/L (Suter and Tsao, 1996). Based on the EqP approach and 1% sediment organic carbon content, the LCV for fish exposed to Aroclor-1260 in sediment is <63,000 µg/L (Jones *et al.*, 1997).

Fish are a major source of PCBs to wildlife (O'Hara and Rice, 1996). Mink, which consume fish, have been found to be very sensitive to PCBs (Fuller and Hobson, 1986 *cited in* O'Hara and Rice, 1996). A LOAEL for reproductive effects of 3.425 mg/kg-day was observed in mink exposed to Aroclor-1016 in the diet for 18 months (Aulerich and Ringer, 1980 *cited in* Sample *et al.*, 1996). As in mammals, PCBs can severely affect the reproduction of avian piscivores (O'Hara and Rice, 1996).

Waterfowl may also be impacted by PCB contamination. In a study by Heath *et al.* (1972 *cited in* Eisler, 1986a), LD_{50}s for mallards fed Aroclor-1248 and Aroclor-1260 were associated with dietary concentrations of 2,798 mg/kg and 1,975 mg/kg, respectively.

4.2.1.4 Inorganics. Numerous metals of interest detected at the site 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, 1988 *cited in* Suter and Tsao, 1996). 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*) and BCFs for water fleas (*D. magna*) exposed to aluminum chloride ranged from 320 to 1,020 (Cleveland *et al.,* 1991; Havas, 1985 *cited in* AQUIRE [USEPA, 1998c]).

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). Significant accumulation in tissues of mice required dietary doses in excess of 200 mg/kg-day (Scheuhammer, 1987). Oral LD_{50} values for several animal species range from 380 to 780 mg/kg (USEPA, 1985).

There is some evidence of potential toxicity of aluminum in soil to plants, particularly tree seedlings and crops, at low pH (< 5.0) (Kelly *et al.,* 1990). High concentrations of calcium and magnesium and a high organic carbon content in soils have been documented to decrease aluminum toxicity through buffering and complexation, respectively (Kelly *et al.,* 1990; Andersson, 1988).

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 a 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* AQUIRE [USEPA, 1998c]).

Antimony can be toxic to mammals. Testing by Shroeder *et al.* (1968 *cited in* Sample *et al.*, 1996) showed a chronic oral dose of 5 mg/L in drinking water caused a reduction in the median life span of female mice.

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).

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 (USEPA, 1980 *cited in* Eisler,1988a); a 96-hour As (III) LC_{50} for *D. pulex* of 1,300 µg/L (USEPA, 1980 *cited in* 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, 1985 *cited in* Eisler, 1988a).

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, *Pimephales 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).

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 not greater than 17. For As

(V), the BCFs were not greater than 6, and the maximum BCF for organoarsenicals was 9 (USEPA, 1980; USEPA, 1985 *cited in* Eisler, 1988a).

Sediment ERL and ERM values for arsenic are 8.2 and 70 mg/kg, respectively (Long *et al.,* 1995). The OMEE LEL and Severe Effect Level (SEL) for arsenic are similar, at 6 and 33 mg/kg, respectively (Persaud *et al.,* 1993).

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 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 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 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).

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).

Barium. Barium readily forms insoluble carbonate and sulfate salts which have low toxicity, but soluble barium salts may be toxic (USEPA, 1985). The Tier II SCV calculated by Suter and Tsao (1996) is 4.0 µg/L. In seawater, barium concentrations ranging from 0.1 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, 1992c). 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, 1992c). 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, 20-60, and >60 mg/kg as non-polluted, moderately polluted, and heavily polluted, respectively (USEPA, 1977 *cited in* Beyer, 1990).

Oral LD_{50}s for barium (as barium carbonate) are reported as 418 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 *et al.*, 1983 *cited in* Sample *et al.*, 1996).

Beryllium. LCVs for freshwater daphnids and plants are 5.3 and 100,000 µg/L, respectively (Suter and Tsao, 1996). Bluegill sunfish have been shown to bioconcentrate beryllium (Barrows *et al.*, 1980 *cited in* AQUIRE [USEPA, 1998c]). A NOAEL for longevity and weight loss in rats of 0.66 mg/kg-d 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.

Cadmium. The literature review of cadmium effects by Eisler (1985) concluded that freshwater organisms were the most sensitive biota. 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 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 Cd (II)/L arrested development in 100% of exposed individuals.

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).

Chromium. Chromium has not been observed to biomagnify, and concentrations are usually highest at lower trophic levels (Eisler, 1986b). The toxicity of chromium varies widely between organisms and is dependent on form. 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, 1986b). 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, 1986b). These data support the generalization drawn by Eisler that Cr (VI) is more toxic to freshwater species and mammals than Cr (III).

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, 1986b). Some salts of chromium are carcinogenic in rats and Cr (VI) is a teratogen in hamsters (USEPA, 1985).

Cobalt. Cobalt is an essential element that can be accumulated by plants and animals (USEPA, 1985). 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, 1985). The LCV for daphnids is 5.1 µg/L (Suter and Tsao, 1996). Estimated soil-to-plant BAFs range from 0.007 to 0.02 (Bysshe, 1988).

Copper. 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, 1985). Chronic toxicity values for freshwater species range from 3.9 µg/L for brook trout to 60.4 µg/L for northern pike (USEPA, 1985).

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 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. However, experimentation has shown that chronic exposure to dietary copper can impact both sheep and swine (USEPA, 1985). Aulerich *et al.* (1982 *cited in* Sample *et al.*, 1996) determined a NOAEL for reproductive effects in mink of 11.7 mg/kg-day.

Iron. The NAWQC for iron is 1,000 µg/L. The LCV for fish is 1,300 µg/L (Amelung, 1981 *cited in* Suter and Tsao, 1996). This concentration caused 100% morality in an embryo-larval test with rainbow trout exposed to dissolved iron salts. The LCV for daphnids (158 µg/L) is a threshold for reproductive effects from a 21-day test of iron chloride with *D. magna* (Dave, 1984 *cited in* Suter and Tsao, 1996). Pentreath (1973 *cited in* AQUIRE [USEPA, 1998c]) measured an iron BCF of 9.53 for the mussel, *Mytilus edulis*.

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 to 5.1 µg/L.

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).

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.

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 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, 1985). 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, 1972 *cited in* Lewis *et al.*, 1979). The LCV for daphnids is <1,100 µg/L (Suter and Tsao, 1996).

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, 5.59, and 8.68 mg/L at hardness levels of approximately 30, 150, 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* AQUIRE [USEPA, 1998c]). Litzke and Hubel (1993 *cited in* AQUIRE [USEPA, 1998c]) 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, 300-600, and >600 mg/kg as non-polluted, moderately polluted, and heavily polluted, respectively (USEPA, 1977 *cited in* Beyer, 1990). The OMEE LEL and SEL are 469 and 1,100 mg/kg, respectively (Persaud *et al.*, 1993).

Manganese is an essential mineral for birds and mammals (USEPA, 1985). It is a cofactor for a number of enzymatic reactions (Klaassen *et al.*, 1996). Chickens showed reduced growth and 52% mortality at dietary concentrations of 4,800 mg/kg (Heller and Penquite, 1973 *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).

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).

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 be bioconcentrated in organisms and biomagnified through food chains (Wolfe *et al.*, 1998; Eisler, 1987b).

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), 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), 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).

Nickel. LCVs for daphnids, non-daphnid invertebrates, and aquatic plants are <5, 128.4, and 5 µg/L, respectively (Suter and Tsao, 1996). Nickel is not significantly accumulated by aquatic organisms (USEPA, 1985). 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 3 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. 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, 91.65 and 100 µg/L, respectively.

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 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 (Cumbie and Van Horne, 1978 *cited in* 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).

Based on biological effects data compiled from the literature, sediment selenium concentrations of 2.5 mg/kg would be a threshold based on predicted effects, and concentrations of 4.0 mg/kg would be the observed threshold for fish and wildlife toxicity (Van Derveer and Canton, 1997).

Silver. 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, 1985). Chronic toxicity values ranging from 2.6 to 29 µg/L have been reported for *D. magna* (USEPA, 1985). 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 *cited in* AQUIRE [USEPA, 1998c]). Excess silver in the diets of mammals and birds has been reported to induce selenium, vitamin E, and copper deficiency symptoms (USEPA, 1985).

Thallium. Information on the toxicity and biological fate of thallium is limited. LCVs for fish, daphnids, and plants are 57, 130, and 100 µg/L, respectively (Suter and Tsao, 1996). 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 *cited in* AQUIRE [USEPA, 1998c]).

Vanadium. Information on the toxicity and biological fate of vanadium is limited. Suter and Tsao (1996) report LCVs of 80 µg/L for fish and 1,900 µg/L for daphnids. 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). BCF values ranged from 107 to 1,130 for insects and from 51 to 432 for freshwater fish (USEPA, 1980 *cited in* 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 (sensitive species) and soil invertebrates at soil concentrations of >100 mg/kg and from 470 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).

4.2.2 Ecosystems Potentially at Risk

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.

4.2.2.1 Aquatic Resources. The ecological characterization of study area aquatic habitats was primarily based on field investigations conducted by USEPA, USFWS, and Ebasco (now Foster Wheeler) in 1991 and 1995; and information from previous studies, including the *Upper Mystic Lake Watershed Urban Runoff Project Main Report* (DEQE, 1982) and the *Wells G&H Wetlands Assessment Final Report* (PRC, 1986). Fish species reported for different areas of the study area were primarily identified during field investigations conducted in 1995 by USFWS and Foster Wheeler in support of the BERA for the Wells G&H Superfund Site OU3. Fish were collected via electroshocking, seines, and trot line. Crayfish were also collected with seines during the 1995 investigation from several of the study area reaches. Species information obtained from electroshock surveys conducted by Dr. Armin Peter (Swiss Federal Institute for Environmental Science and Technology) in September and October 1995 and species information provided in DEQE (1982) were also used to prepare lists of species inhabiting each of the reaches of the Aberjona River.

Additional habitat data were collected during the June 2001 sediment triad sampling. Using Rapid Bioassessment Protocols (Plafkin, *et al.*, 1989) habitat assessments were conducted at each of the 15 stations within the Aberjona River study area and at 5 reference locations. These data were collected primarily to characterize habitat for benthic invertebrates using rapid bioassessment evaluations for low gradient stream habitats. A summary of the habitat characterization at the 20 triad sampling locations is presented in Appendix D.1 to D.3. Habitat assessment and characterization field data sheets are provided in Appendix A.5.

The Aberjona River is an urban stream corridor within a densely developed urban watershed. As such, aquatic habitats of the river and associated waterbodies are affected by a number of factors including low flow; neglect; indiscriminate disposal of debris; and local and upstream runoff, including non-point and point source discharges. The aquatic habitats of the study area were divided into six reaches (Section 1.0, Figure 1-3), based on similarity of habitat, species presence or absence, and application of fishing methods. Aquatic physical and chemical parameters, measured within the sampling period extending from August 15 to September 7, 1995, were highly variable among the reaches (Table 4-19).

The Aberjona River throughout reach 1 is generally shallow, slow-moving, and turbid. The river meanders through the Wells G&H 38-acre emergent wetland for most of this reach. Aquatic macrophytes common in reach 1 include floating and emergent species, typically: duckweed (*Lemna minor*), arrow arum (*Peltandra virginica*), arrowhead (*Sagittaria sp.*), and spikerush (*Eleocharis* sp.). Broad-leaved cattail (*Typha latifolia*), common reed (*Phragmites* sp.) and purple loosestrife (*Lythrum salicaria*) are common in the shallows along the river banks. Filamentous green and blue-green algae densely cover submerged rocks. Invertebrates observed in the Aberjona River (all reaches) during the course of BERA field investigations represent a relatively diverse assemblage of organisms which include flatworms, nematodes, oligochaetes, gastropods, copepods, isopods, amphipods, crayfish, and adult and larval aquatic insects (Table 4-20). Bullfrog tadpoles are also common throughout the shallow waters. Species assemblages vary with bottom substrate, water velocity, and oxygen concentration. Fish species which have been observed in reach 1 include brown bullhead (*Ameiurus nebulosus*), pumpkinseed (*Lepomis gibbosus*), shiner (*Notropis* sp.), and white sucker (*Catostomus commersoni*).

Below Salem Street, the Aberjona River in reach 2 flows through the former cranberry bogs. The majority of the main channel was channelized, with numerous small channels entering from the adjacent emergent wetland. The remainder of the main channel in reach 2 is characterized by relatively faster-flowing water than reach 1and benthic substrates consisting of imbedded gravel and cobbles. In sections, vegetation along the banks provides greater stream cover in this reach. Redfin pickerel (*Esox americanus*), pumpkinseed, and white sucker have been observed in reach 2. Reach 3 is similar to reach 2, and includes a 1-acre pond at Davidson Park. Extensive portions of the stream banks have been stabilized with rip-rap in this reach. Fish species observed in reach 3 include largemouth bass (*Micropterus salmoides*), brown bullhead, pumpkinseed, yellow perch (*Perca flavescens*), and white sucker.

The river is culverted at Swanson Street, the start of reach 4, and follows an underground course for approximately 0.3 miles after which it discharges into Judkins Pond located in downtown Winchester. The river in reach 5 is similar to that in reaches 2 and 3, however, park land is the dominant land use along this reach. Fish species observed in reaches 4 and 5 include largemouth bass, pumpkinseed, and white sucker.

Reach 6 consists of the upper and lower forebays of Upper Mystic Lake as well as the main basins of the Upper Mystic and Lower Mystic Lakes. The upper and lower forebays of the lake are depositional areas characterized by soft, fine grained sediments. The mouth of the Aberjona River, and upper and lower forebays support a heavy growth of aquatic vegetation. The forebays are dominated by pondweeds (*Potamogeton richardsonii* and *P. crispus*) and water lilies (*Nymphea odorata* and *Nymphea* sp.), while the shoreline of the southern basin is dominated by the pondweeds and waterweed (*Elodea canadensis*). Other plants occurring in the lake include pickerelweed (*Pontederia* sp.), arrowhead (*Sagittaria latifolia*), broad-leaved cattail, and purple loosestrife. The heavy aquatic vegetation in the lake provides habitat for fish. Fish species commonly found in the lake include bluegill (*Lepomis macrochirus*), common carp (*Cyprinus carpio*), largemouth bass, pumpkinseed, brown bullhead, yellow perch, black crappie (*Pomoxis nigromaculatus*), golden shiner (*Notemigonus cyrsoleucas*), American eel (*Anguilla rostrata*), alewives (*Alosa pseudoharengus*), white sucker, and chain pickerel (*Esox niger*).

4.2.2.2 Terrestrial/Wetland Resources. No formal surveys of fauna or flora were conducted throughout the study area. However, Habitat Assessments for Low Gradient Streams were performed for the 20 stations (5 reference and 15 non-reference) included in the triad sampling conducted in June 2001. The following descriptions of the habitats and typical species compositions are based on qualitative observations made by field biologists during sampling in the study area. Differences in species observed in the study area from previous studies or similar habitats may only reflect the limited nature of the field observations. More detailed evaluation of the 20 stations sampled for benthic invertebrate triad sampling is presented in Section 4.3.2.3.

Terrestrial and wetland habitats associated with the river and water bodies within the study area are generally restricted to a relatively narrow corridor which includes the river's floodplain, and undeveloped and developed upland either adjacent to the river or immediately upgradient of its floodplain. The width of this corridor varies from approximately 20 feet to 0.3 mile. At several locations, development encroaches to the waters edge. Habitats along the river include emergent, scrub/shrub and forested wetlands, fragmented upland forests, sub-mature woodlots, grassy meadows, and maintained park land. As with aquatic habitats, value of terrestrial habitats is affected by over use, neglect, and encroachment of adjacent development.

The most significant wetland habitat associated with the river corridor is the Wells G&H 38-acre wetland at the northern end of the study area. Parts of the outer perimeter of this wetland complex are bordered by upland forest. The wetland complex consists of emergent, scrub/shrub, and forested areas. Emergent wetlands occupy the largest portion of this complex and consist of an association of broad-leaved cattail, purple loosestrife, tussock sedge (*Carex stricta*), and other sedges (*Carex* spp.). Common reed (*Phragmites* sp.) is the dominant species in scattered stands throughout the marsh. Other common plants in the marsh include jewelweed (*Impatiens capensis*), arrowhead, spikerush (*Eleocharis* sp.), rush (*Scirpus* sp.), buttonbush (*Cephalanthus occidentalis*), elderberry (*Sambucus canadensis*), steeplebush (*Spiraea latifolia*), and swamp dodder (*Cuscuta gronovii*). Table 4-21 lists plant species encountered within the Aberjona River Study area during field investigations.

Forested wetlands occupy the western and eastern edges of the Wells G&H 38-acre complex including a 3-acre forested wetland which forms the extreme northeastern boundary of the study area. Red maple (*Acer rubrum*) is the dominant tree species and an association of arrowwood (*Viburnum dentatum*), highbush blueberry (*Vaccinium corymbosum*), silky dogwood (*Cornus amomum*), buckthorn (*Rhamnus frangula*), swamp azalea (*Rhododendron viscosum*), shadbush (*Amelanchier canadensis*), and poison sumac (*Rhus vernix*) are found in the shrub layer.

Due to the vast acreage and the diversity of structure and food provided by the varied plant communities, the Wells G&H 38-acre wetland complex is valuable for wildlife. A variety of migratory and non-migratory species utilize this wetland, as well as other habitats from Salem Street extending south to Upper Mystic Lake. At least three amphibian, three reptile, twenty-four bird, and six mammal species have been observed on-site during field investigations (Table 4-22).

4.2.2.3 Rare Species. Based on correspondence with the Massachusetts Natural Heritage and Endangered Species Program (MANHESP, 1998) and USFWS (1998), no federal or state-listed rare species are known to inhabit the study area. However, the Mystic Valley Amphipod (*Crangonyx aberrans*) is known to occur within the Aberjona River immediately to the north of Route 128. The Aberjona River flows under Route 128 shortly before entering the Wells G&H Superfund Site (Section 1.0, Figure 1-2).

4.2.3 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 site COPCs are those whose activities frequently bring them into direct contact with sediment and surface water, that directly consume aquatic plants and/or detritus (dead plant and animal material), 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 on-site, and those that may be most likely to receive potential exposures to site COPCs. Species were selected as indicators for exposure evaluation to represent various components of the food chain in the river/wetland ecosystem.

4.2.3.1 Plants. 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 may also contain COPCs and be consumed by detritivores. Herbivores and detritivores may, in turn, become a source of COPC exposure for secondary consumers.

4.2.3.2 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 site 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. The potential for the dermal and inhalation pathways to be complete and potentially significant is discussed below, primarily with reference to fish and aquatic mammals.

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 main channel of the river and Upper Mystic Lake 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.

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 found in all six reaches during field investigations, 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; McClane *et al.*, 1978). Brown bullhead, which was collected in reaches 1, 3, and 6 only, 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 all but reach 4. 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 reach 3 (Davidson Park) south into Upper Mystic Lake. 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 *cited 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, and raccoon. 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 molluscs (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 at the northern end of the study area. Beavers are herbivores. Plants present at the site 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 in a few areas within the Wells G&H 38-acre wetland where sediment VOC concentrations are high, muskrat and beaver may potentially be impacted via inhalation 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 most of the other habitat that exists along the Aberjona River. Raccoon home ranges vary between 0.6 and 1.8 miles in diameter (400 to 1,200 acres) (Kaufmann, 1982; DeGraaf and Rudis, 1983). The percentage of time spent on-site and the percentage of food obtained on-site 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.

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. 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, herring gull, 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). Of all these species, herring gulls may have the most varied diet, consisting of various terrestrial and aquatic invertebrates and vertebrates (USEPA, 1993d).

There are habitats at various points along the entire stretch of the Aberjona River being evaluated that may be utilized by the five species discussed above. Mallards and herring gulls 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 five bird species, exposure to COPCs is likely to be lowest for Canada geese and herring gulls. Canada geese primarily feed on vegetation, often in upland grassy areas, where the concentration of COPCs would be relatively low. The diet of herring gulls is varied and, in an urban area, does not likely consist entirely of prey items of natural origin.

4.2.3.3 Terrestrial Receptors. Animals that inhabit the drier areas of the Aberjona River floodplain 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.

4.2.3.4 Site Conceptual Model and Selected Receptor Species

The complete exposure pathways are summarized in the Site Conceptual Model (Figure 4-1) 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 Aberjona River Study area:

- 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 likely occur across the entire study area, including Upper Mystic Lake. In contrast, beavers appears to be limited to the northwestern portion of the Wells G&H 38-acre wetland. 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. Muskrat houses, which were observed in the wetland complex at the northern end of the study area, are also important because they may be utilized as nesting sites by other species (Weller, 1981). Muskrats have been observed in the Wells G&H 38-acre wetland at the northern end of the study area (north of Salem Street), and likely occur in all of the open water habitats within the study area.

- 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 Aberjona River 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 site.

- 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 from Davidson Park (reach 3) into Upper Mystic Lake (reach 6).

- White sucker (*Catostomus commersoni*) was selected to represent epibenthic fish species present at the study area. White sucker was collected in all six reaches, while brown bullhead was collected in reaches 1, 3, and 6 only. White sucker feed on insect larvae, crustaceans, and worms via a method of foraging that involves substantial contact with sediments. During spring spawning runs (Harlen *et al.*, 1987), white sucker may be exposed to COPCs in both the main channel of the Aberjona River and in side channels at the northern end of the study area.

- Pumpkinseed (*Lepomis gibbosus*) was selected to represent a small foraging fish. Pumpkinseeds 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 reaches of the study area, but were not collected in reach 4.

- 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 this aquatic system.

- 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 border the Aberjona River. 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.

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 4-1). 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.

4.2.4 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 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 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 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 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	

Assessment Endpoints	Measurement Endpoints
Sustainability (survival, growth, reproduction) of local populations of predatory fish	• compare tissue concentrations of COPCs measured in largemouth bass caught within OU3 to published fish tissue benchmarks which are indicative of potential impairment • compare those same tissue concentrations to largemouth bass caught at reference locations
Sustainability (survival, growth, reproduction) of local populations of bottom-feeding fish	• compare tissue concentrations of COPCs measured in white suckers caught within OU3 to published fish tissue benchmarks which are indicative of potential impairment • compare those same tissue concentrations to white suckers 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 OU3 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 throughout OU3 and reference locations to conservative sediment benchmarks which are indicative of potential impairment • Compare acute and chronic toxicity of sediment samples collected from OU3 to samples collected from reference locations using *Hyalella azteca* and *Chironomus tentans* laboratory bioassays • compare the tissue concentrations of COPCs measured in crayfish caught at OU3 and reference locations to published invertebrate tissue benchmarks indicative of potential impairment • quantify the *in-situ* benthic community composition using sediment samples collected at OU3 and reference locations

For each of the individual indicator species, the assessment endpoint references an impact on survival, growth, or reproduction of a population. 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.

4.3 ANALYSIS OF ECOLOGICAL EXPOSURES AND EFFECTS

4.3.1 Exposure Characterization

Exposure characterization for each of the receptor species is presented in the following subsections. Table 4-23 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, and shrew populations were evaluated using food-chain exposure models.

- Fish species exposures were based on evaluation of tissue concentrations COPCs compared to concentrations in tissue of the same species collected at reference locations and comparison to tissue residue benchmarks for other freshwater species (Section 4.3.1.3).

- 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 crayfish tissue to reference concentrations and tissue residue benchmarks, 3) evaluation community composition, and 4) results of sediment toxicity testing analyses.

To assist in exposure estimation for wildlife indicator species (muskrat, heron, mallard), fish, crayfish, and plants were collected from the study area and analyzed for SVOCs, pesticides/PCBs, 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. Notes and analytical results for the field effort, which was conducted by USFWS and Foster Wheeler in 1995, are provided in *Preliminary Data Compendium, Wells G&H RI/FS OU III, Aberjona River Study Area* (Foster Wheeler, 1996). Field methods, sampling locations, and analytical results for fish, crayfish, and plants are discussed in Section 2.0 of this report. Surface water and sediment COPCs detected in plants, crayfish, and fish are presented in Tables 4-24 through 4-26, respectively. Average COPC concentrations were calculated using rules presented in Section 2.1.5.

Thirty-one COPCs were detected in one or more of the six composite plant tissue samples collected at the study area (Table 4-24). With the exception of fluoranthene and pyrene, average PAH concentrations were greater than maximum concentrations, indicating elevated non-detected values. The COPCs in one or more of the tissue samples collected in the study area also included seven pesticides, Aroclor-1260, and fifteen inorganics.

The reference sample differed from study area samples in several ways (Table 4-24). No SVOCS were detected in the reference samples. Aroclor-1260, mercury, and selenium were not detected in the reference samples, and only four of the seven pesticide COPCs found in study area plant tissue were present at detectable concentrations in the reference samples. The mean concentration for the on-site plant tissue samples were higher than that for the reference tissue samples for all inorganic COPCs except barium and manganese.

Plant tissue samples were collected at three on-site stations and one reference station. In order to utilize the plant tissue data for all stations, uptake factors were calculated for all COPCs in plant tissues. Uptake values were calculated from ratios of average COPC concentration in plant tissue to average concentrations in sediments for each of four sites where plant data are available (18, 20, 21, and 23). A mean uptake factor was computed from these four values and used to estimate plant tissue concentrations based on the sediment COPC concentration for each station. The estimated values for tissue concentrations were used in the exposure calculations for the muskrat and the mallard for a portion of the daily exposure from the ingestion of plant tissue. The average uptake values were compared to literature in Table 4-27.

Twenty-six COPCs were detected in crayfish tissue collected at the study area (Table 4-25). The detected COPCs included nine pesticides, Aroclor-1254 and -1260, and fifteen inorganics; no SVOCs were detected in crayfish tissue.

Fewer COPCs were detected in reference tissue from the Shawsheen River (Table 4-25). Of the pesticide/PCBs detected in on-site tissue, only three, 4,4'-DDD, 4,4'-DDE, and Aroclor-1260, were detected in reference samples. Among inorganics, antimony, arsenic, beryllium, cyanide, lead, mercury, nickel, and thallium were not detected in either of the two reference samples.

Twenty-seven COPCs were detected in small fish tissue collected within the study area (Table 4-26). Detected COPCs included eleven pesticides, Aroclor-1248 and -1260, and fourteen inorganics; no SVOCs were detected.

As was the case for crayfish, fewer COPCs were detected in reference tissue collected from Wright's Pond (Table 4-26). Heptachlor epoxide was detected in reference tissue, but not in study area tissue The mean and maximum concentrations for the study area small fish tissue samples were higher than the reference tissue samples for all organic and inorganic COPCs except manganese, mercury, silver, and zinc. The maximum detected concentration of mercury in reference tissue for small fish (0.098 mg/kg) was almost twice the maximum detected concentration for study area tissue (0.054 mg/kg).

4.3.1.1 Exposure Estimation for Mammalian and Avian Species. For muskrat, 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:

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

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:

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

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.

It is important to note that 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 that bioavailability is similar to that of the bioassay from which the toxicity reference value (TRV) is derived. Use of a factor of 1 also assumes that there is no difference in uptake of a chemical between that of the receptor species and the species from which the TRV was derived. The only exception to this assumption was for the bioavailability of arsenic from incidental sediment ingestion to the mammals (muskrat and shrew).

As seen from the swine study conducted in conjunction with this risk assessment, only approximately 50% of the arsenic in sediment fed to young swine was bioavailable (section 3.4.4). In the study, data were collected to calculate the relative bioavailability (RBA) of arsenic from site 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. The most conservative RBA value determined for study area sediments (51%) was selected as the most appropriate to evaluate the oral toxicity of arsenic in sediments at all stations within the study area for mammals (muskrat and shrew). The site-specific RBA value of 51% was used to adjust the incidental sediment ingestion dose for each of the mammal models of arsenic. The dose from plant material was not adjusted by this RBA, since no RBA for plants was derived

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

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

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, heron, mallard, and shrew are provided in Tables 4-28 through 4-31, respectively. Sets of surface water, sediment, plant, crayfish, and small fish data used to estimate COPC exposures for muskrat, heron, mallard, and shrew are described below for the Aberjona River study area and for reference locations for each indicator species Table 4-23. Samples utilized in the development of exposure estimates for each avian and mammalian indicator species in the study area are presented in Table 4-32. 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 most locations within the Aberjona River 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 along a linear area of river or wetland. These groups of up to 20 samples (*e.g.*, station WG, reach 1, Figure 2-3) are separated by up to 1,000 linear feet along the edge of the wetland. Due to the difference in the number of samples and the geographic area of some of the stations, the stations represent an area of habitat from a few square feet to up to approximately 20,000 square feet (1,000 foot-long x 20 foot-wide area).

The exposure concentrations for each receptor were calculated based on the foraging area of the species. Sediment, surface water, and tissue samples used to estimate mammalian and avian COPC doses are listed in Table 4-23. The two mammalian species, muskrat and shrew, have smaller foraging areas (less than 1 acre). Consequently, exposure for these receptors were calculated on a station by station basis, since the size of a station corresponds approximately to the size of their foraging areas. Since surface water data were not collected at each station, average data for the reach in which a station was located were used to represent surface water concentrations for muskrat and shrew (Tables 4-33 through 4-38).

Crayfish and small fish were collected by reach rather than on a station-specific basis. Crayfish collected within a reach (Tables 4-39) were used to represent COPC concentrations at stations collected within the same reach. No crayfish were collected in reaches 4 and 6. Crayfish data from reaches 3 and 5 were used for reaches 4 and 6, respectively.

For each receptor, two exposure models were calculated, an average case scenario and a maximum case scenario. The average case scenario was a dietary exposure model based on mean concentrations of each COPC calculated for sediment, surface water, and animal tissue (plant tissue concentrations were based on sediment concentrations), 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 media (sediment, surface water, and 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.

Heron

Heron exposures were calculated for all samples collected within suitable habitat, throughout the Aberjona River Study area to compute a "site-wide" scenario (Table 4-23) since the foraging ranges of this species is relatively large. Home range for great blue heron are estimated at 1.44 acres with foraging distances of 2 to 15 miles (EPA, 1993d), which corresponds to the size of the study areas (7 miles long). However, home range of the smaller green heron is lacking (Sample et. al, 1997). 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 urban areas or with forested canopies and little open water were excluded from the heron model, as well as stations with depths greater than 3 feet of water. COPC data from all selected sediment stations

(Table 4-40) were used to calculate incidental sediment exposure (1% of diet, Table 4-29). All small fish data collected site-wide were combined for the heron evaluation (Table 4-26) in the study area (45% of diet). Similarly, all crayfish samples collected within the study area (Table 4-25) were used to estimate exposure for heron invertebrate ingestion (55% of diet). All surface water stations were used in the site-wide model for estimating dietary ingestion of water (Table 4-41). Both an average and maximum 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 03-IP (Phillips Pond), since samples were taken at a depth of 9 to 13 feet, and were too deep to represent incidental sediment ingestion (Table 4-42). All small fish data collected from Lubber Brook were used for the reference heron model and samples from the Shawsheen River were used in the estimate of reference crayfish tissue concentrations (Table 4-25). All reference surface water samples except samples collected at station 03-IP were used to calculate dietary exposure from water (Table 4-43).

<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). Two exposure scenarios were evaluated for mallard: the 38-acre wetland scenario and the site-wide scenario. The 38-acre wetland scenario involved the estimation of exposure concentrations associated with sampling stations located within reach 1, which is the Wells G&H 38-acre wetland. The site-wide exposure scenario involved estimation of exposure concentrations similar to heron, based on sampling stations across the entire study area, including the Wells G&H 38-acre wetland. The sets of sampling stations used to estimate exposure concentrations for mallard for each scenario are presented in (Tables 4-23 and 4-32).

Sediment samples with water depths less than 3 feet were used for estimation of incidental sediment ingestion (3.3% of diet) for mallard (Tables 4-44 and 4-45) and for estimation of plant tissue concentrations (using site-specific uptake factors). Dietary exposure for mallard was based on 33% plant tissue and 67% invertebrates (represented by crayfish tissue). All crayfish samples collected within the study area were used to estimate site-wide exposure for mallard invertebrate ingestion (Table 4-25) and samples from reach 1 were used for the 38-acre scenario (Table 4-39). Exposure from surface water ingestion was based on the average COPC concentration in surface water for all samples site-wide (Table 4-41) or from reach 1 (38-acre scenario, Table 4-33). Both average and maximum exposure case scenarios were calculated for mallard for both the study area and the reference models.

Data used in the reference models included sediment data from all reference locations, except station 03-IP (Phillips Pond) since samples were taken at a depth of 9 to 13 feet (Table 4-42). Sediment data were used to estimate plant tissue concentrations of COPCs and incidental sediment ingestion. Crayfish samples from the Shawsheen River were used in the estimate of invertebrate tissue concentrations for the reference model (Table 4-25). All reference surface water samples except samples collected at station 03-IP were used to calculate dietary exposure from water (Table 4-43).

Muskrat

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 case scenario was calculated for all COPCs for the muskrat. The maximum exposure scenario for muskrat was calculated for a limited set of COPCs and is discussed in Section 4.5.3. Average station COPC concentration in sediment was used to estimate incidental sediment ingestion (3.3% of diet) and to estimate plant tissue concentrations (Tables 4-46 to 4-88). Plant tissue concentrations were estimated for each station from station sediment concentrations and uptake factors calculated from on-site plant tissue data. Exposure from surface water ingestion was based on the average COPC concentration in surface water for the reach in which the station was located (Tables 4-33 to 4-38). Similarly, crayfish collected within a reach were used to represent COPC concentrations at stations collected within the same reach (Table 4-39). No crayfish were collected in reaches 4 and 6. Crayfish data from reaches 3 and 5 were used for reaches 4 and 6, respectively.

Total dose estimates for muskrat at reference locations were calculated separately for wetland, pond, and river habitats. Data from similar habitats were pooled to estimate exposure at reference locations in order to have more data to calculate exposures than were available at individual reference stations (only 1 or 2 samples at several reference locations, Table 4-32). Plant tissue concentrations used for the muskrat reference models for each habitat were estimated based on average station sediment COPC concentration (Tables 4-89 to 4-91) for each habitat (pond, wetland or river) multiplied by the site-wide uptake factors (Table 4-27). Exposure from surface water ingestion for muskrat was based on the COPC concentrations in surface water for the samples from the three habitat types (pond, river, or wetland, Tables 4-92 to 4-94).

<u>Shrew</u>

The home range of white-tailed shrew is small, on the order of less than one acre (EPA, 1993d). Similar to muskrat, the risk evaluation for shrew populations was conducted on a station by station basis.

The majority of on-site 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 (19 on-site, 3 reference) were relevant to the evaluation of shrew exposure (Table 4-32). Stations selected for shrews included those in saturated areas with little standing water and those 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 (the pond at Davidson Park, Judkins Pond, and Upper Mystic Lake), were not used to estimate COPC exposures to shrews. Several stations were selected to represent potential depositional areas immediately adjacent to the main stream channel or open water wetland area. These locations were above normal high water marks, and labeled as riparian habitats. In these locations, the substrate samples were labeled as “soils” and may represent wetland soils (seasonally saturated or inundated) or slightly drier upland soils. These drier stations, including: CB-05, DA, NRSO, KFSO, and WSS, were used only for the estimation of exposure to shrew, and not used for any of the aquatic or semi-aquatic receptors (Table 4-32). Among the 19 shrew stations, analytical data for organics COPCs were collected at 6 stations and only at 1 of the three reference stations. Inorganic data was collected at all 21 stations.

In contrast to muskrat, heron, and mallard, site-specific tissue data were not collected for the evaluation of COPC exposures to shrew. Site-specific sediment data (Tables 4-77 to 4-88 and 4-95 to 4-101) were used to estimate body burdens of prey for shrew. The concentration of COPCs in shrew prey (*i.e.*, earthworms) were estimated using different methods for inorganic and organic COPCs. For organic COPCs, an equilibrium partitioning model was used to estimate earthworm body burdens. The basic assumption underlying this equilibrium partitioning model, presented in Sample *et al.* (1997), is that invertebrates are in equilibrium with the aqueous phase of soil.

For inorganic COPCs, regression equations relating contaminant concentrations in soil and earthworm tissue (Sample *et al.*, 1998) were used to estimate burdens of arsenic, cadmium, chromium, copper, lead, manganese, mercury, nickel, selenium, and zinc in earthworms at the study area. Concentration factors for aluminum (0.34), barium (0.36), and iron (0.38) (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, beryllium, cobalt, silver, thallium, and vanadium. 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 maximum sediment COPC concentrations are presented in Tables 4-102 through 4-139. Exposure from surface water ingestion for shrew was based on the average COPC concentration in surface water for the reach in which each station was located (Tables 4-33 to 4-38).

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 4-94) 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, Table 4-32). The concentration of COPCs in shrew prey (*i.e.*, earthworms) were estimated using the COPC concentrations in sediments, using the methods described above (Tables 4-140 and 4-141). 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 4-94).

4.3.1.2 Exposure Assessment for Fish. Risks to fish populations were evaluated via a comparison of COPC body burdens in three fish species collected from the study area to both reference station samples and tissue residue benchmarks from the literature. Species-specific average COPC concentrations for largemouth bass, white sucker, and pumpkinseed were calculated for five of the six Aberjona River reaches in the study area and for reference station samples (collected from Horn Pond and Wright's Pond), along with lipid-normalized concentrations for SVOCs, pesticide/PCBs, and mercury. No largemouth bass or white sucker were collected in reach 1. No pumpkinseeds were collected in reach 2. Separate averages were derived for whole body, fillet, and offal, as available, for individual fish species.

4.3.1.3 Exposure Assessment for Benthic Invertebrates. Risk to the benthic invertebrate community was evaluated via sediment toxicity testing, comparison of COPC concentrations in sediment to benchmarks for benthic invertebrate, and comparison of COPC body burdens in crayfish collected from the study area to both reference station sample data and tissue residue benchmarks.

The sediment toxicity testing program is discussed in Section 4.3.2.3. The benthic invertebrate evaluation utilizing sediment benchmarks was conducted for each of the six reaches. By reach, the average station concentration, maximum (95% UCL) station concentration, and maximum detected concentration for each COPC were calculated. The reference sites were combined into habitats, and the average station concentration, maximum station concentration, and maximum detected concentration for wetland, pond, and river reference sediments for each COPC are presented for comparison.

For the crayfish evaluation, COPC concentrations detected in the eight samples collected within the study area were averaged (Table 4-25). The study area averages and lipid-normalized concentrations (for pesticides/PCBs and mercury) are compared to tissue concentrations in reference samples and to available tissue residue benchmarks.

4.3.2 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.

4.3.2.1 Mammalian and Avian Indicator Species. Mammalian and avian TRVs for COPCs were obtained from the literature (Tables 4-142 through 4-145). 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. TRVs for SVOCs and VOCs for heron and mallard were based on tests with mammalian species, since studies with avian species for these compounds were not available. 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 4.5.3.2.

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 indicator species. HQs and HIs for mammalian and avian indicator species are discussed below. Model results are presented in Appendix E.1.

Muskrat. At every station identified as muskrat habitat (Table 4-32), HQs for VOCs, SVOCs, and pesticide/PCBs were all less than 1 (Tables 4-146 through 4-189). HIs were also less than 1 for these chemical classes at each station evaluated, except stations 04 and 06, which had HIs values for SVOCs of slightly greater than 1. In contrast, every station evaluated had several inorganic COPCs with HQs in excess of 1. As a consequence, the HI values for inorganics were also greater than 1 at each station evaluated. HQs in excess of 1 are summarized in Table 4-189. HQs were greater than 1 for aluminum, antimony, arsenic, barium, cadmium, chromium, copper, iron, lead, manganese, mercury, selenium, and vanadium. HQs for arsenic and iron were greater than 1 at all muskrat stations, including reference locations. The highest HQs were associated with arsenic (range = 6-1275), and iron (range = 100-2259). HIs for inorganics ranged from 110 at station 16 to 3560 at station 19, and the majority were greater than 200.

The three separate models used for reference stations were for pond, river, and wetland habitats. Similar to the study area data, the highest HQs were associated with arsenic (range = 5-10), and iron (range = 269-274). The other compounds with HQs greater than 1 at two or more reference sites for muskrat were aluminum, lead, manganese, and vanadium. The HIs for inorganics in reference models ranged from 285 to 292 (Tables 4-190 to 4-192).

For inorganic COPCs, exposure related to the ingestion of plant material generally dominated HQs (*i.e.*, the percent contribution to the HQ from crayfish, sediment, and surface water was far less than for plants). There were several notable exceptions. Antimony and beryllium in sediment contributed heavily to the HQs for these COPCs at many stations, as did silver and thallium in both crayfish and sediment samples.

Green Heron. For VOCs, SVOCs, and pesticide/PCBs, HQs and HIs were all less than 1, for both the average and maximum site-wide exposure cases for green heron (Tables 4-193 to 4-197). For inorganics, iron was the only COPC with an HQ in excess of 1 (average exposure case HQ = 8; maximum exposure case HQ = 12) (Table 4-197). In both exposure cases (average and maximum) at the reference locations, the HQ for iron was also greater than 1 (average exposure case HQ = 3; maximum exposure case HQ = 3) (Table 4-197). Iron in prey (fish and crayfish) contributed over 70% of the exposure dose on-site and 50-60% of the dose for the reference models.

Mallard. For both site-wide and 38-acre scenarios, average and maximum exposure case HQs and HIs were all less than 1 for VOCs, SVOCs, and pesticide/PCBs (Tables 4-198 through 4-203). As for muskrat and heron, several inorganic HQs were greater than 1. Average and maximum exposure case HQs for chromium, iron, lead, and mercury were greater than 1 for both the site-wide and 38-acre scenarios (Table 4-197). The maximum HQs for iron ranged from 90 to 112, and maximum HQs for chromium were 9 (38-acre) and 4 (site-wide). The remainder of the COPC-specific HQs for mallard were less than 10. Plants contributed the majority of the dietary exposure for those COPCs with HQs greater than 1.

At the reference sites, the HQs for iron and lead were also greater than 1 (Table 4-197). Estimated consumption of plant tissue contributed over 80% of the exposure of both lead and iron at the reference sites.

Shrew. For both average and maximum exposure case scenarios, HQs were all less than 1 at all 19 stations having data for VOCs, SVOCs, and pesticide/PCBs (Tables 4-204 through 4-241), with the exception of Aroclor-1254 at station JY, which had average and maximum case HQs of 5 and 7, respectively. For both the average and maximum exposure cases, HIs for SVOCs slightly exceeded 1 at station 20, but were lower than the HI for SVOCs at reference wetland, station 24.

HQs were greater than 1 for several inorganic COPCs (Tables 4-244 to and 4-245). Both average and maximum scenario case HQs were greater than 1 for aluminum, antimony, arsenic, cadmium, chromium, iron, lead, mercury, thallium, and vanadium at one or more stations. The highest HQs for both the average case and maximum case scenarios were for arsenic (average range = 1-59) and iron (average range = 14-378). With the exception of antimony, which had a maximum exposure case HQs of 60 at station 22/TT-22 and 11 at station BW, aluminum with an HQ of 11 at station 20, and lead with an HQ of 29 at station 22/TT-22, all other HQs were below 10 excluding iron and arsenic.

The reference scenario was calculated based on data from three wetland reference stations for inorganics (stations 24, HB, and SA). However, sediment data for organics COPCs were available only from wetland station 24. Consequently the average and maximum sediment concentrations for organics are based on the 4 samples collected at station 24 (Table 4-242 and 4-243). All HQs for organic compounds were less than 1 at reference station 24 (Maple Meadow Brook) for the average case scenario, however, similar to the on-site results, the HI for SVOCs was slightly greater than 1 (HI = 2.9) due to the detection of PAHs in sediment. Five individual PAHs (benzo(a)anthracene, benzo(a)pyrene, benzo(b)flouranthene, benzo(k)flouranthene, and chrysene) had maximum HQs of greater than 1.0 (range 1.5 to 3.0) at station 24, contributing to a maximum HI of 12. The reference exposure also calculated high HQs for iron (HQ = 53, average; HQ = 77, maximum). The HQs for aluminum, arsenic, and vanadium also exceeded 1 for shrew at the wetland reference site.

The majority of the daily dose of inorganics with HQs in excess of 1 came from ingestion of prey (*i.e.*, earthworms). The only exceptions were arsenic, for which approximately 65-90% was from incidental ingestion of sediment. At the reference location, approximately 68% of the daily dose of arsenic was from sediment, and the contribution of sediment to the daily dose for aluminum and iron was greater than 50%.

4.3.2.2 Fish Species. Pesticide/PCBs, inorganics, and one SVOC (benzo[g,h,i]perylene) were detected in fish tissue from largemouth bass, white sucker, and pumpkinseed (Tables 4-246 to 4-251). COPC concentrations in fish tissues were evaluated in two ways. First, COPC concentrations in tissue (whole body, fillet, and offal) from each of the river reaches were compared to concentrations detected in samples collected from reference locations for white sucker and largemouth bass. For the pumpkinseed, COPC concentrations were also evaluated by comparing tissue concentrations in fish collected from all reaches in the study area to 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. COPC concentrations were also evaluated by comparing tissue concentrations in fish collected from the study area 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/USEPA, 2002) contained entries for seven of the thirteen organic COPCs and seven of the sixteen inorganic COPCs detected. Benchmarks were available for some of the COPCs contributing to risks for other indicator species (*e.g.*, arsenic, mercury, and lead), but unavailable for others (*e.g.*, iron and chromium). Although benchmarks were not available for all COPCs detected in fish, the benchmark comparison 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 pumpkinseed were the same as for largemouth bass and white sucker, with the exception of selenium. A selenium value for bluegill was used for the pumpkinseed comparison, and the value for largemouth bass was used for the other species.

Largemouth bass, white sucker, and pumkinseed sunfish were selected to represent fish species with different feeding strategies as receptor endpoints in the BERA. However, the same fish tissue residue evaluations were also conducted for brown bullhead and eel so that all data collected for the Aberjona River Study would be fully utilized. Additional data on fish tissue for species collected in the study area that were not selected as measurements endpoints (brown bullhead and American eel) are presented in Appendix E.2.

There were several trends apparent from comparison of study area and reference COPC tissue concentrations (Tables 4-246 to 4-251). These were as follows:

- In the majority of comparisons for largemouth bass and pumpkinseeds, the concentrations of pesticide/PCBs were greater in tissue collected from the study area. The opposite trend was true for white sucker.

- Among inorganics, approximately 50% of the comparisons for all species indicated higher concentrations in study area tissue.

- Lipid normalization for pesticide/PCBs and mercury did not substantially alter the relationship between reference and study area tissue concentrations.

- There was a slight trend of fewer exceedances of reference inorganic concentrations moving from reach 3 south through reach 6, particularly in largemouth bass. The same pattern was not evident for organic COPCs.

- For organic COPCs, the highest ratios of study area to reference data were found in largemouth bass collected from reach 6 (Upper Mystic Lake). The difference between study area and reference data was also substantial in largemouth bass collected from reaches 3 and 5. For inorganics, the absolute differences between study area and reference data were similar across all reaches.

Tissue residue benchmarks and study area tissue concentrations are also presented in Tables 4-246 through 4-251. There were no exceedances of benchmarks for organic COPCs and few exceedances of benchmarks for inorganic COPCs. In reaches 3, 4, 5, and 6 average concentrations of mercury in largemouth bass fillets, exceeded the respective benchmark (yellow perch, whole body, no observed effects dose). The reference average concentration for mercury in largemouth bass fillets also exceeded the benchmark.

In general, there were few exceedances and the difference between study area tissue concentrations and benchmarks were less than an order of magnitude. The importance of these exceedances relative to ecological effects is questionable, because exceedences for mercury were also observed at reference locations. Results of the evaluation suggest that, overall, COPC concentration in fish collected from the study area are lower than concentrations which would be associated with adverse effects.

4.3.2.3 Benthic Invertebrate Community. The ecological effects evaluation for the benthic invertebrate community involved three components: comparison of sediment COPC concentrations to sediment benchmarks; comparison of the concentration of COPCs in crayfish tissue to both reference crayfish tissue and tissue residue benchmarks, and toxicity testing.

Sediment Benchmark Comparison. By reach, the average station concentration (average case), 95% UCL concentration (maximum case), and maximum detected concentration (maximum detection case) were compared to sediment benchmarks for freshwater benthic invertebrates. SCVs (Jones *et al.*, 1997) and OMEE Severe Effect Levels (SELs) (Persaud *et al.*, 1993) were utilized for the comparison. These benchmarks were selected since they are adjustable for reach-specific organic carbon content. When SCVs and SELs were unavailable, NOAA ERMs (Long *et al.*, 1995), NOAA Threshold Effects Level (TEL) (Buchman, 1999) or NOAA sediment background levels (Buchman, 1999) were used. Exposure concentrations, benchmarks, and HQs are presented by reach in Tables 4-252 through 4-257. HIs, along with HQs, are summarized in Tables 4-261 through 4-263.

For comparison purposes, the average case, maximum case (95% UCL) and the maximum detection case were also calculated for reference locations (Tables 4-258 through 4-260). The reference locations were divided to three habitats pond (4 stations), wetland (3 stations), and river (5 stations).

Acetone was the only VOC with HQs in excess of 1. HQs were greater than 1 in all reaches for the average case and in all but reach 5 for the maximum case comparisons (Tables 4-261 and 4-262). When benchmarks were compared to maximum detected concentrations, the HQ for acetone was greater than 1 in reaches 1, 2, and 6, as well as 2 of the reference groups (Table 4-263). This indicates that the HQs in the average case and maximum case evaluations were driven by non-detected values with high detection limits, rather than detected values.

There were no SVOCs in any reach with average HQ values greater than 1.0, although in reaches 3, 4, 5, and 6, average case HIs were between 1.3 and 4.5 (Table 4-261). Benzo(k)fluoranthene had an HQ of 1.3 in reach 3 for the maximum detected concentration. The maximum detected concentration of acenaphthylene in the wetland reference also exceeded the benchmarks (HQ = 1.3).

Among the pesticides/PCBs, there were no HQs for on-site or reference samples greater than 1 for the average or maximum detected case, with the exception of gamma-chlordane in reach 1 with a maximum detected HQ value of 1.1. HIs for the average case comparison exceeded 1.0 only in reach 3 and reach 5. Two reference groups (pond and river) had HIs slightly greater than 1 (1.6 and 1.4, respectively) for the maximum detected comparison.

Due to some high non-detected values, average case and maximum case HQs are likely less than those reported. With these few, relatively low magnitude exceedances, it is likely that the ecological effects of VOC, SVOC, and pesticide/PCB COPCs on benthic invertebrate populations is limited.

Results for inorganics showed more exceedances of benchmarks. In the average case, maximum case, and maximum detection comparisons, all reaches and reference sites had HIs greater than 10 (Tables 4-261 through 4-263). For the average case comparison, the HIs in the three reference habitats (pond, river, and wetland) ranged from 12 to 14.

In the average case comparison, aluminum, antimony, cadmium, iron, manganese, nickel, and silver were among the inorganic COPCs with no HQs over 1. The highest HQs and corresponding HI for inorganics were in reach 1. HQs for arsenic, chromium, and selenium were 13, 18, and 16 in reach 1, respectively. The benchmark for cyanide was very conservative as it represented a benchmark for free cyanide. The highest HQs for cyanide (10 to17) were observed in reaches 1, 3, and 6. No other HQs exceeded 10 in the average case comparison.

In the maximum case (95% UCL) comparison (Table 4-262), aluminum, iron, nickel, and silver had HQs less than or equal to 1 for all reaches and reference habitats. Again, the highest HQs and associated HIs were observed in reach 1. HIs among the six reaches ranged from 15 to 128. The HIs in the reference habitats ranged from 19 to 24.

The average case comparison is the most relevant with respect to evaluating risks to benthic invertebrate populations throughout entire reaches. The results of the comparison to sediment quality benchmarks suggests that the benthic invertebrate communities may be impaired by inorganic contamination. However, in reach 5, only the HQs for cyanide and selenium were greater than 1 and were less than those observed at references stations. The maximum case (95% UCL) in Table 4-262 and maximum detection case comparisons in Table 4-263 indicated that there are stations and individual sampling locations, in all reaches except reach 5, at which the comparison to benchmarks indicate the potential for adverse effects. Reach 5 consists largely of the Aberjona River channel with limited depositional areas (*i.e.*, wetlands or impoundments). The depositional area of reach 5, near the mouth of the Aberjona River was addressed separately by samples collected at station AJRW and is addressed in Appendix E.5. The Wells G&H 38-acre wetland of reach 1, the Cranberry Bog in reach 2, the pond at Davidson Park (reach 3), Judkins Pond (reach 4), and Upper Mystic Lake (reach 6) are all sinks for inorganic contamination.

Acid Volatile Sulfide/Simultaneously Extracted Metals. The bioavailability of metals in sediment can significantly affect the 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, 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):

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

Where $umol/g_{oc}$ is the concentration of the metal in micro moles per gram of orgainc 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 site, August-September 1995, November 1997, and June 2001. A total of 28 stations and 12 reference locations were samples in the summer sampling rounds; 12 stations on-site and 2 reference stations were sampled in the November 1997 (winter) sampling round.

Normalized comparisons for data collected in summer (June - September), as presented in Table 4-264, indicate that at approximately 49 percent (65 of 132) of all sample sites, AVS exceeds SEM, thus divalent metals are likely to be biologically unavailable to benthos. Approximately 31 percent (41 of 132) of all sample sites were between 130 $umol/g_{OC}$ and zero, and thus are categorized as unlikely to be toxic. Approximately 17 percent (22 of 132) are of uncertain toxicity. Only 3 percent (4 of 132) exceed 3,000 $umol/g_{OC}$; these exceedences occurred at locations where TOC or TCO was reported as the detection limit.

At no station did all samples exceed 3,000 umol/g_{OC} to indicate likely toxicity. At five locations (stations 05, 08, 09, 14, and 16), a majority of samples exceeded 130 umol/g_{OC}; at stations 08 and 16 all samples exceeded 130 umol/g_{OC}, indicating uncertain toxicity. Stations 08, 09, and 16 are located in reach 2, station 05 is in reach 5, and station 14 is in reach 1. At eight locations (stations 01, 02, 06, 07, 10, 11, 20, and 21), a majority of samples were below 130 umol/g_{OC}, but some samples exceeded this threshold, indicating unlikely toxicity. At the remaining sites for which AVS-SEM data was collected (stations 03, 04, 12, 13, 15, 18, 19, 22, WH, UF, AO, TT-30, TT-32, WW, and all reference stations), all samples were below 130 umol/g_{OC}, indicating unlikely toxicity due to divalent metals.

Normalized comparisons for data collected in winter (November 1997) indicated that all locations sampled on-site and reference stations were below 130 umol/g_{OC} (Table 4-265), suggesting that during the winter months, divalent metals are likely not bioavailable.

Crayfish. As presented in Table 4-266, with the exception of four inorganics, COPC concentrations were greater in crayfish collected at the study area than those collected from reference stations (*i.e.*, ratios of study area to reference concentrations were generally greater than 1). Study area to reference ratios for barium, cadmium, manganese, and silver ranged from 0.2 to 0.9, indicating the reference samples contained higher body burdens. Ratios for lipid normalized organics and mercury were all greater than 1. The ERED (USACE/USEPA, 2002) contained entries for five of the twenty-five COPCs. Due to the limited amount of data, residue benchmarks from freshwater and marine decapods were used in the evaluation (crayfish, lobsters, and crabs). Comparison of these no observed effect and lowest observed effect concentrations to study area tissue concentrations suggests that COPC body burdens in site crayfish are lower than those which would be associated with adverse effects. A possible exception is copper, which had an average concentration of 49.7 mg/kg in crayfish, approximately twice the laboratory test concentration at which no effects were observed. The tissue concentration of copper from on-site samples was 2.5 times higher than the reference samples.

Toxicity Testing. Three rounds of toxicity testing with benthic invertebrates were conducted to support the BERA. Complete reports for the 1995 and 1997 rounds may be found in Appendices A.1 and A.2, respectively. Results from the third round in June, 2001, are found in Appendix D.4 to D.9. Sample locations and habitat categories for each round are presented in Sections 2.1.2. The purpose of toxicity testing was to determine whether survival, growth, or other sublethal endpoints of organisms exposed to study area sediment would be significantly different from similar endpoints in organisms exposed to reference sediments. As part of the 2001 sediment quality triad analysis the community compositions of the benthic invertebrates were also examined.

1995 Results. In first round of toxicity testing (1995), the survival percentage for *H. azteca* in the control was 16.6%, which was lower than the performance criterion for an acceptable test (*i.e.*, 80%). Survival in the two reference samples SD-23-01-FW and SD-24-01-FW was 0.8% and 78.1%, respectively. The only statistical comparisons conducted were for growth and survival between SD-24-01-FW (the wetland reference), SD-18-01-FW, SD-19-01-FW, and SD-19-02-FW. Statistical testing indicated that both growth and survival were lower at SD-18-01-FW than at SD-24-01-FW. However, this result must be cautiously interpreted since the performance criterion for this test was not met.

The survival percentage for *C. tentans* in the control (70%) met the performance criterion for an acceptable test (*i.e.*, 70%). Statistical testing indicated no difference in survival between the control and any of the study area samples. However, all study area samples showed a slight but statistically significant lower growth than the control at the end of the ten-day test period. The on-site riverine sample SD-16-01-FW showed less growth than was observed in the riverine reference sample (SD-23-01-FW). Likewise, growth in the three study area wetland samples (SD-18-01-FW, SD-19-01-FW, and SD-19-02-FW) was statistically lower than in the reference wetland sample (SD-24-01-FW).

In summary, results from the first round of toxicity testing were as follows:

- conclusions about growth and survival of *H. azteca* were questionable due to poor survival rates in some reference and control samples.

- growth of *C. tentans* at SD-16-01-FW (riverine), SD-18-01-FW (wetland), SD-19-01-FW (wetland), and SD-19-02-FW (wetland) was statistically lower than at respective reference stations, however the survival in the wetland reference was low (36%).

- survival of *C. tentans* was not significantly different between study area sediments and reference sediments.

1997 Results. In the second round of toxicity testing (1997), the survival percentage for *H. azteca* in the control was 77%, which was marginally lower than the performance criterion for an acceptable test (*i.e.*, 80%). Statistical tests indicated no significant difference in survival between reference and on-site sediments.

The survival percentage for *C. tentans* in the control (94%) met the performance criteria for an acceptable test (*i.e.*, 70%). As for *H. azteca*, statistical tests on *C. tentans* data indicated no significant difference in survival between reference and study area sediments. There were, however, statistically significant differences in the growth of *C. tentans* between study area and reference samples. Among the riverine samples, individuals exposed to SD-10-02-ME sediment grew less than those exposed to the reference sediment (Fowle Brook). Likewise, individuals exposed to the pond/wetland samples SD-06-03-ME and SD-19-01-ME exhibited small but significantly less growth than was observed in the reference sample (SD-25-02-ME). Growth in the other three study area samples (SD-12-03-ME, SD-18-02-ME, and SD-07-10-ME) was not significantly different than growth observed in respective reference samples.

In summary, the results from the second round of toxicity testing were as follows:

- survival was not significantly different for *H. azteca* or *C. tentans* between reference and on-site sediments

- growth of *C. tentans* showed small but significant reduction at some stations as compared to reference location.

The two rounds of toxicity testing (1995, 1997) indicated small, but statistically significant lower growth of *C. tentans* in study area samples compared to reference samples. For each study area and reference sample used in the toxicity tests, the average weight of surviving individuals was calculated at the end of the test. To evaluate whether growth results were related to the magnitude and composition of contamination in sediment, ratios of mean individual weight in study area samples and mean individual weight in reference samples were calculated and compared to several characteristics of the sediment. Growth ratios were compared to the following parameters: SEM/AVS ratio, aluminum, iron, arsenic, total metals, total metals (without aluminum, iron, and arsenic), and TCO (Figures 4-2 through 4-8). Growth ratios were also compared to concentrations of PCBs, pesticides, SVOCs, VOCs, and the sum of organic contaminants (Figures 4-9 through 4-13).

In the development of growth ratios for the first round of toxicity testing, station 23 was used as the reference station for stations 07, 12, and 16, and station 24 was used as the reference for stations 18 and 19. For the second round of toxicity testing data, station 25 was used as the reference for all stations, since sediment collected from Fowle Brook (the other reference station) was not analyzed for COPCs. The growth ratio for reference locations was set at 1.

Figures 4-2 through 4-13 suggest that no one parameter clearly correlates to the rate of growth of individuals exposed to study area sediment. However, for every study area sample, the growth ratio was less than 1, indicating that, relative to reference locations, the growth of *C. tentans* was impaired when exposed to study area sediments. No clear relationship between reduced growth rates and concentration of contaminants was detected.

2001 Results. A third, more comprehensive round of toxicity testing was conducted in June 2001. 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. This round included benthic community composition analyses at each station, as well as including sublethal endpoints in longer-term sediment exposures for both the amphipod, *H. azteca* and the midge, *C. tentans*.

Twenty stations, 15 stations selected in the Aberjona River Study Area and 5 reference locations, were selected to represent a cross-section of habitat types and sediment metals concentrations (Appendix D.1, Table D.1-1). Descriptions of the triad sampling locations are given in Appendix D.2. Habitat assessments were conducted at each sampling location (Appendix D.3; see Appendix A.5 for field data sheets). Sampling stations included three evaluated for sediment toxicity in 1995 (stations 12, 18, and 19) and five sampled in 1997 (stations 06, 10, 12, 18, and 19). Results of sediment chemistry analyses collected on simultaneously collected sediments used in the laboratory toxicity tests are presented in Appendix D.4.

Habitat characterizations are presented in Appendix D.3 (see Appendix A.5 for field data sheets). Among all habitats samples, the stream habitats had the highest ratings for the Low Gradient Stream Habitat Assessment (LGSHA) scores. Among the streams, only station 10 had a LGSHA value that was much lower than the reference locations, mainly due to the conditions of the bank and riparian vegetation zone. Also, station 10 was a wide area of the main channel of the Aberjona River, just above the Salem Street bridge. For these reasons the habitat conditions for pool variability, sediment deposition, and channel sinuosity were ranked rather low.

Among the wetland stations (TT-32, TT-33, WW, WH, 13, 19, and 22), station 13 had the lowest rating for habitat quality. This sample was collected in an area of a side channel on the western side of the 23-acre wetland. This location received lower rankings, similar to station 10 because of disturbed banks, and the presence of a pool of uniform depth, appearing to be an area of sediment deposition.

The lowest habitat ratings were assigned to the stream/pond habitats, since the assessment was designed predominately for low gradient streams. The conditions at station 06 were rated lower than the other ponds due to the conditions of the riparian zone, consisting of mowed grass, buildings, and roadways.

<u>2001 toxicity testing results</u>.

Endpoints for each test were compared to results in laboratory controls (artificial sediment) according to *Methods for Measuring the Toxicity and Bioacummulation of Sediment-associated Contaminants with Freshwater Invertebrates* (USEPA, 2000). In addition, the endpoints were also compared to the results in the corresponding field reference site. In general, many of the tests had better results on the natural reference sediments than on the laboratory controls. 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.

The 10-day acute toxicity tests for *H. azteca* showed no acute toxicity (reduced growth or survival) at any station except the pond control site, station 03-IP (Phillips Pond) as compared to laboratory controls. The 10-day acute tests for *C. tentans* showed a decrease in survival at station HB (reference wetland) and a decrease in growth, as compared to laboratory controls at stations 12, 13, and 04 (Table 4-267, Appendix D.5 and D.6).

The 42-day chronic toxicity tests for *H. azteca* showed no significant reduction in survival at 28, 35, or 42 days at any station, as compared to laboratory controls (Table 4-267, Appendix D.7). Similarly, there was no reduction in growth of *H. azteca* at 28 days at any station, and reduction of growth at station 12, only, at 42 days as compared to laboratory controls. For the reproduction endpoint, there was a decrease ($p < 0.05$) in neonates per female at station 18 as compared to laboratory controls.

The life-cycle chronic toxicity tests for *C. tentans* showed no significant reduction in survival (20-day) at any station, except stations HB (stream reference) and station 06 (Judkins Pond), as compared to laboratory controls (Table 4-267, Appendix D.8). There was no reduction in growth of *C. tentans* at 20 days at any station as compared to laboratory controls. There were two emergence endpoints for *C. tentans* (percent emerged and days to emergence). The percent of the midges emerged was lower than laboratory controls ($p<0.05$) at stations HB, 06, and 04. The number of days to emerge was higher that controls ($p<0.05$) at stations 13, 18, TT-32, and TT-33 as compared to laboratory controls. For the reproduction endpoint (percent hatched), there was a reduction as compared to laboratory controls ($p< 0.05$), at stations WH, 19, TT-32, and TT-33). However, the mean proportion emerged in one laboratory control group was very low (0.21).

A summary of the toxicity endpoints from the 2001 sampling round are presented in Figures 4-14 to 4-31. Similar to the earlier sampling rounds, the endpoints are expressed as a ratio of each sites' endpoint (mean of 8 replicates) to the result observed for the corresponding field reference location. For the development of ratios, station SA was used as the wetland reference location, station 04-IP was used as the stream reference location, and station 03-IP (Phillips Pond) was used as the pond reference location.

Statistical comparisons were made (using raw data) to evaluate observed differences in the endpoints as compared to the corresponding field reference sites. For most of the tests, there were eight replicate treatments for each sediment sample. Analysis of variance (ANOVA) with post-hoc analyses (Bonferoni adjustment) was used to compare the mean of each treatment to the mean of the reference from the same habitat. Prior to the ANOVA, the data with skewed distributions which failed tests for normality or homogeneity of variance were transformed. The survival data were transformed using an arcsine square root transformation and computing the ANOVA on the transformed values.

The 10-day acute toxicity tests for *H. azteca* showed no significantly reduced survival at any of the stations tested as compared to the corresponding reference locations (Table 4-268, Figures 4-

14, 4-17, and 4-20). However, at wetland sites 13, 19, TT-32, and TT-33 the growth of *H. azteca* was statistically lower (ANOVA, p <0.05, post-hoc comparisons) than at either of the two wetland reference locations (SA and HB). Among the stream sites (Figure 4-18) growth at station 12 was lower (ANOVA, p <0.05, post-hoc comparisons) than at reference 01-IP, but not statistically lower than the second stream reference, 04-IP. Among the lake stations (UF, 06, and 04), neither 10-day survival nor growth were significantly different from the reference pond sample (Phillips Pond, 03-IP).

None of the endpoints of the chronic tests for *H. azteca* (42-day growth and survival, reproduction) showed statistically significant differences from the corresponding reference locations at either the stream, wetland, or pond habitats (Table 4-267).

The 10-day acute tests for *C. tentans* showed no reduction in survival at wetland stations as compared to reference sites. The survival of *C. tentans* at both reference locations was low, particularly at HB with a 10-day survival of 25%. For *C. tentans*, both the 10-day acute and 20-day chronic tests showed reduced growth at all wetland stations as compared to the wetland reference locations (Figure 4-24). Statistical analysis of these results indicated that the growth of *C. tentans* on sediments from stations 13, 19, TT-32, and TT-33 was significantly lower (ANOVA, p <0.05, post-hoc comparisons) than the wetland reference station, SA, for both the 10-day and 20-day tests. Due to low survivorship, station HB was not used in the growth comparison.

For the stream stations, the 10-day acute tests for *C. tentans* showed a reduction in survival at station 10 (arcsine of square root transformation, ANOVA, p <0.05, post-hoc comparisons) as compared to reference sites. In addition, growth of *C. tentans* on sediments from stream stations 10, 12, and TT-30 had statistically lower values (ANOVA, p <0.05, post-hoc comparisons) as compared to the stream reference stations, 01-IP and 04-IP, for the 10-day tests and growth of *C. tentans* was lower than both stream references at station TT-29 for the 20-day tests (Figure 4-27). Growth at station 12 was also low for the 20-day tests, however, there were no surviving midges after 20 days in one of the four replicate test chambers. This reduced the number of growth replicates from four to three, thereby deceasing the sensitivity of the statistical test.

The 10-day acute tests for *C. tentans* showed a reduction in growth at stations 04 and 06 (ANOVA, $p < 0.05$, post-hoc comparisons) as compared to the reference site (03-IP).

No consistent differences were observed among the on-site stations (wetland, stream, or pond) as compared to the corresponding reference locations in the percent of emergence of midges or in the percent of midges hatching (Figures 4-25, 4-28, and 4-31).

<u>2001 community composition results.</u>
As part of the sediment quality triad sampling, three replicate sediment samples were collected for identification and enumeration of benthic macroinvertebrates from each triad sampling location (Appendix D.9). A number of community indices were calculated to evaluate the community composition of the stations (Table 4-269 and Appendix D.9).

Total abundance of organisms was highest among the triad sampling locations at station 04 (Upper Forebay, Mystic Lake) and station 18 (stream station) (Table 4-269). Very low invertebrate abundance was observed at the pond control station (03-IP, Phillips Pond) and at station 10. The number of different taxa (most identified to the species level) observed at each station ranged from 8 to 39. The majority of the stations had more than 30 taxa represented, with the exception of station 03-IP, four of the stream stations (stations 10, 12, 18, and TT-29), and one wetland station, TT-33. The diversity index values (Shannon-Weiner Index, Appendix D.9) at these stations were also low.

The majority of the 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 are not unexpected, since these taxa are frequently found in fine sediments. 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).

Among the wetland stations, station 13, 19, TT-32, and TT-33, the percent of the community consisting of Oligochaetes plus Chironomids was more than 70% and greater than either of the reference locations. Whereas at the wetland reference location HB, and the other three wetland stations (WW, WH, and 22), the percent Oligochaetes plus Chironomids was less than 55%. The second reference wetland, SA, however, was dominated by Chironomids (83%), with an Oligochaetes plus Chironomids value of 95%. All of the stream stations, including the reference locations, had Oligochaetes plus Chironomids values of greater than 80%, with all of the locations dominated by Oligochaetes. All of the pond stations (04, 06, and UF) had Oligochaetes plus Chironomids values of greater than 95%, while the lake/pond reference (which had very low total macroinvertebrate density) was 42%.

Pielou's Evenness and percent dominance 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 at the site. 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). Wetland and stream stations with low evenness (below corresponding reference values) included 12, 18, TT-29, TT-30, TT-32, and TT-33. Stations with high dominance values included these same stations plus stations 10 and 13 (Table 4-269).

Community Loss Index (CLI) was calculated on a station basis (using the three replicate samples combined). The CLI is computed as the loss of benthic species between a reference station and the sampling station (Table 4-269). Station HB was used for the wetland reference, station 04-IP for the stream reference, and station 03-IP for the pond/lake reference. By definition, the value of CLI for the reference station used for comparison is 0. The value of CLI increases with the degree of dissimilarity with the composition of the reference community (Plafkin *et al.*, 1989). Among the wetland stations, the CLI for the second reference location (SA) was 0.93, indicating a relatively high similarity between the stations. Among the other wetland locations, stations TT-33 and WH had CLI values higher than reference location SA.

In the stream samples, the second reference also had a low CLI (0.64), indicating the benthic communities in the reference streams were similar. All of the stream stations, except station TT-30, had CLI values greater than the reference locations. Station 10, in particular, showed a high dissimilarity to the reference communities with a CLI of 4.13.

Among the wetland stations, those dominated by pollution-tolerant species included stations 13, 19, TT-32, and TT-33. The other three wetland stations (WW, WH, and 22) did not have high percentages of Oligochaetes and Chironomids, and were not dominated by species with such high tolerance values, and in general, did not share the community characteristics of the other stations that indicted impaired benthic communities.

Among the stream stations, all were dominated by highly tolerant species of Oligochaetes (Table 4-269). All stream stations 10, 12, 18, TT-30, and TT-29 all had several community characteristics indicative of highly tolerant benthic invertebrate communities. Among the stream stations, TT-30 showed the fewest indicators of benthic community impairment, with higher diversity and lower CLI values.

Among the pond/lake stations, the reference location showed the most serious characteristics of community impairment. The total invertebrate abundance (total of three replicates) of 68 organisms at the reference pond organisms was very low. The toxicity testing did not detect any toxicity at station 03-IP. The low invertebrate abundance at this station may be a result of seasonally low oxygen concentrations in the pond. At the time of the sample, the dissolved oxygen concentration above the sediments at the 03-IP sampling station was 0.7 mg/L. For this reason, use of this station has limited value for use as a reference comparison of benthic community composition. The three on-site stations (4, 6, and UF) were all dominated by Oligochaetes. Station UF was dominated by a relatively less tolerant Oligochaete species (*Aulodrilus pigueti*) (Table 4-269).

In order to assess the weight of evidence from the toxicity tests and benthic invertebrate community data from the SQT, two simple indices were calculated. The results of all of the toxicity tests were summarized by assigning a value of 1 point to each toxicity endpoint from either the *H. azteca* or the *C. tentans* tests (chronic or acute), that had statistically different mean values than the corresponding reference location (Table 4-268). A Toxicity Index was computed for each station which was simply a total of the number of tests exceeding the reference values. With the exception of the survival of *C. tentans* at station 10, all of the endpoints contributing to the Toxicity Index (TI) were reduced growth of either *H. azteca* or *C. tentans* as compared to reference locations. The highest values observed (TI = 3) were at four wetland stations in reach 1 or 2 (stations 13, 19, TT-32, and TT-33).

Secondly, a Community Index (CI) was calculated in a similar manner using several different community characteristics as part of the weight of evidence. Each of the community indices (Table 4-269) that showed impairment in community characteristics, as compared to the corresponding reference samples, was assigned a value of one. The highest Community Index values indicate a weight of evidence for impaired community characteristics. The stations with the highest observed values were 10, 12, 18, TT-29, and TT-33 (CI = 6 or 7). The CI values at wetland stations 13, 19, and TT-32 were underestimated using this method, since the reference station SA had such a high dominance of highly tolerant Chiromonid species.

The abundance of invertebrates was very low at the pond reference station (03-IP), indicating a potential for the influence of a different physical or chemical stressor influencing the benthic community (possibly low oxygen concentrations). The comparisons of the other pond/lake stations to this as a reference sample makes these comparisons less reliable. Consequently, a CI value was not computed for the pond stations.

Comparisons to Sediment Chemistry.

A correlation analysis using the Pearson correlation procedure was used to identify relationships among chemical variables and measures of benthic invertebrate community structure or sediment toxicity. The chemistry variables selected were those analytes that had the highest HQs in reaches 1 and 2 as compared to effects-based sediment benchmarks. These included arsenic, chromium, copper, lead, mercury, and zinc. Other chemical variables, important in assessing bioavailability of the inorganics, were used in the analysis, including TOC, SEM, and SEM/AVS. The chemistry data used were the analytical results from the simultaneously collected sediment chemistry samples during the triad sampling, with the exception of the data for station TT-29. No chemistry data were available for this sample, so mean sediment chemistry data for this station from the 2000 sampling round were used for arsenic, chromium, copper, mercury, and zinc. All correlations were based on log-transformed chemistry data. The Pearson correlations coefficients are presented in Table 4-270.

The total abundance of invertebrate taxa and the number of taxa at each station did not closely correlate to any of the selected chemistry variables. Evenness of the community (Pielou's evenness), diversity, lower dominance values correlated to stations with higher TOC. This association indicates that stations with higher TOC generally support more diverse benthic communities. Dominance was increased and diversity decreased at stations with higher concentrations of arsenic, copper, and mercury in the sediments. Similarly, the 10-day acute growth of both *C. tentans* and *H. azteca* was reduced at stations with higher SEM, as well as higher arsenic, chromium, copper, and zinc concentrations.

The Community Index was highly correlated to elevated SEM, arsenic, chromium, copper, mercury, and zinc concentrations, indicating that the stations with the most evidence of community impairment were those with high metal concentrations. The Community Index was most highly correlated to copper and arsenic concentrations in sediment. The Toxicity Index was highest at stations with high arsenic concentrations.

2001 triad sampling summary.

- Toxicity testing results from 2001 confirmed the earlier toxicity results, detecting evidence of acute or chronic toxicity of benthic invertebrates exposed to sediment from several stations.

- Stations showing two or more endpoints indicating toxicity as compared to reference locations included stations 10, 13, 19, TT-32, and TT-33 (Table 4-268).

- Analysis of the benthic community composition at the triad sampling locations indicated stations 10, 12, 18, TT-29, and TT-33 showed characteristics of highly impaired benthic invertebrate communities, dominated by pollution-tolerant species. Stations TT-30 and TT-32 showed a moderate amount of impairment. While stations 22, WW, WH, 13, and 19 had fewer indicators of impaired benthic community structure.

- Evaluation of the sediment chemistry indicated that both the sites with evidence of reduced growth of benthic invertebrates in toxicity tests, and those stations with evidence of impacted natural communities were correlated with those with higher sediment concentration of arsenic and copper, and also correlated to high concentrations of chromium, mercury, and zinc.

4.4 REFINEMENT OF COPCS

Based on the results of the ecological effects characterization, a number of the contaminants found at concentrations above screening-level concentrations and selected as COPCs, can be eliminated from further consideration. There were no indications of significant ecological risk from VOCs or SVOCs to any of the ecological receptors. Among the pesticide/PCBs, Aroclor-1254, had an HQ >1 at station JY in reach 1 for dietary exposure to shrew only. Due to the negligible ecological risk to receptor species, ecological risk from VOCs, SVOCs, and pesticide/PCBs will not be evaluated further.

Among the inorganics identified as COPCs in surface water and/or sediment in the Aberjona River Study area, beryllium, cobalt, cyanide, nickel, silver, and zinc did not have HQs greater than 1 for any mammalian or avian receptor species. In addition, these inorganics were not indicated to be associated with high HQs or potential toxicity to invertebrate communities. No significant ecological effects appear to be associated with these COPCs in the study area and will not be considered further for contribution to ecological risk.

Several inorganics, including selenium, thallium, and vanadium, had low HQs (HQs < 3 or HQs < 2 times the HQ at the reference location) for all of the mammalian and avian receptor species. Due to the limited risk of these metals compared to reference toxicity values and similar exposures to receptors at reference locations, these COPCs will not be considered further for contribution to ecological risk.

The dietary exposure models for shrew and muskrat resulted in HQs greater than 1 for aluminum, iron, and manganese at most of the stations in the study area and also the reference locations. The aluminum concentration in sediment and plant tissue from the study area did not greatly exceed those from the reference locations. In general, for mammals, the evidence for direct toxic potential for aluminum is low compared to other inorganics, because mammals typically limit the absorption of aluminum and effectively excrete any excess (Sheuhammer, 1987). In addition, the aluminum did not exceed the benthic invertebrate benchmarks for either the average reach concentrations or the 95% UCL reach concentrations in sediment. Based on these results, there is little evidence for risk due to exposure to aluminum in the study area.

Iron and manganese are both essential nutrients that may be toxic to some organisms at very high concentrations. Neither the concentration of iron or manganese in sediment exceeded sediment benchmark values for either the average or 95% UCL case concentrations. Based on the generally low toxicity of these metals, the normally high concentrations in urban aquatic environments, iron and manganese will not be considered further for contribution to ecological risk.

The nine remaining inorganic COPCs that will be further evaluated for potential risk to one or more of the receptor species include: antimony, arsenic, barium, cadmium, chromium, copper, lead, mercury, and zinc.

4.5 RISK CHARACTERIZATION

4.5.1 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 Aberjona River 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. There were no indications of significant ecological risk from VOCs, SVOCs, and pesticide/PCBs. Several inorganic COPCs were determined to have negligible risk to the receptor species, including: aluminum, beryllium, cobalt, cyanide, iron, manganese, nickel, selenium, silver, thallium, and vanadium. The following evaluation for each endpoint addresses risk from exposure to metals, including: antimony, arsenic, barium, cadmium, chromium, copper, lead, mercury, and zinc.

4.5.1.1 Semi-aquatic Mammals. The assessment endpoint for was:

Sustainability (survival, growth, reproduction) of local populations of 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 site-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. All of the stations at which muskrats would be expected to be present had HQs in excess of 1 for several inorganic COPCs (Table 4-189). Among these inorganics, antimony, arsenic, barium, cadmium, chromium, copper, lead, and mercury are further evaluated below, along with a characterization of the confidence in the risk calculation.

Antimony. For antimony, HQs were between 1 and 1.7 at only 3 stations in reach 1 (Table 4-189). Based on these low HQ values and the limited number of stations at which they occurred, the risk to muskrat from exposure to antimony is negligible.

Arsenic. In comparing HQs at reference locations to those on-site, the values for arsenic greatly exceeded values observed at reference locations. The HQs for arsenic were greater than 10, and higher than at the corresponding reference sites, for all but two sampling stations in the Aberjona River Study area (Table 4-189). The highest HQs were observed in reach 1, where over half of the stations had values greater than 200. The highest HQ for arsenic of 1275 was at station 19. The HQs at the reference locations ranged from 5 to 10.

The TRV for arsenic is based on a chronic (reproductive) LOAEL in a mouse of 1.93 mg/kg-d. The NOAEL of 0.1 mg/kg-d, derived from the LOAEL (dividing by a factor of 10) and adjusting for body size is a conservative benchmark. In addition, the toxicity of arsenic depends on its chemical speciation, occurring in various oxidation states an 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 on-site. 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 reach 1.

Barium. For barium, HQs were between 1 and 5 at 50% or fewer of the stations within each reach (Table 4-189). The highest HQ for barium (5) was observed in the Wells G&H 38-acre wetland at station WW, near Wildwood. All other HQs for barium were 2 or less. Based on these low HQ values and the limited number of stations at which they occurred, the risk to muskrat from exposure to barium is negligible.

Cadmium. For cadmium, HQs were between 1 and 2 at only 4 (21%) of the stations within reach 1 and were below 1 at all other stations (Table 4-189). Based on these low HQ values and the limited number of stations at which they occurred, the risk to muskrat populations from exposure to cadmium is low.

Chromium. In reach 1, 53% of the stations had an HQ of greater than 1 for chromium. Chromium HQs also exceeded 1 at two stations along the Aberjona River in the upper extent of reach 2 (station TT-30 and TT-33), and at station AS in reach 3. The HQs for chromium were generally low (1-5) with the exception of station WW (Wildwood) in reach 1, which had an HQ of 20. The TRV for chromium (7 mg/kg-d) is based on a subchronic NOAEL for chromium (III) in a rat. For other wildlife species, adverse effects have been reported at 5.1 and 10 mg of Cr (VI) and Cr (III), respectively per kilogram of diet (Eisler 1986b). These values are consistent with the derived TRV of 7 mg/kg-d for muskrat.

The results of the HQ analysis indicates a potential for effects on muskrat survival at a majority of stations slightly above the NOAEL TRV. Based on the number of stations affected, the data indicate a low risk to muskrat populations in reach 1, and this conclusion is associated with a moderate amount of uncertainty associated with the modeling, the forms of chromium in the diet as compared to the TRV, and the derivation of the TRV from a different species.

Copper. In reach 1 and reach 2, 58% and 21% of the stations, respectively, had HQ values greater than 1 for copper (Table 4-189). However, the exceedences of the TRVs for copper were low for muskrat at all of the stations, with the HQs ranging from 1.1 to 4. All of the reference habitats had HQs for copper less than 1. The HQ values in reach 6 (Mystic Lakes), in particular were low, with the highest observed HQ at 1.3. The TRV for copper was based on a reproductive NOAEL for mink. Only in reach 1, where more than 60% of the stations' HQs indicated a potential for reproductive failure, would there be low risk to muskrat populations from exposure to copper in diet. However, as this is based on a NOAEL from a different species (mink), confidence in this risk conclusion is only moderate.

Lead. The HQs for the pond and wetland reference sites were 1.6 and 2.5, respectively, for lead (Table 4-189). HQs for lead exceeded reference values for the corresponding habitat types at 50% or more of the stations in all reaches, except reach 2. The HQ values, however, were generally low (less than 7) with the exception of station WH, in reach 1, which had an HQ of 11. The TRV for lead was based on a chronic (reproductive) NOAEL in a rat. The measurement endpoint is not met for lead, but the confidence in the impacts on reduction of muskrat populations is low due to uncertainty in the TRV and relatively low values of the HQs at all but one location.

Mercury. In comparing HQs at reference locations to those on-site, the values for mercury greatly exceeded values observed at reference locations at three stations (Table 4-189). HQs for mercury were 8, 8, and 19, respectively, at stations 10, 12, and TT-30, in reaches 1 and 2. In contrast, none of the three reference habitat types had HQs for mercury that exceeded 1. Although the magnitude of these HQs are high, relatively few stations were indicated to have risks from mercury. In addition, the TRV selected for mercury was a conservative value of 0.032 mg/kg-d, based on a NOAEL (reproductive) in a rat. If an alternative TRV of 1.0 mg/kg-d (NOAEL) in mink (Sample *et al*., 1996) was used instead, the HQs would have all been less than 1. Consequently, the potential adverse effects on populations due to mercury exposure is low, since impacts are predicted at relatively few locations, and uncertainty associated with the TRV is moderate.

The exposure related to the ingestion of plants contributed the largest proportion of the HQs for the majority of the inorganic contaminants for the muskrat. Since the plant tissue concentrations were derived from the average sediment concentration of each COPC (multiplied by a site-specific uptake factor), the estimated risk is strongly weighted by the sediment concentrations of the COPCs.

Observations from the study area indicated that a muskrat population is present at the site. The calculations of HQs greater than 1 are an indication of chronic exposure which may affect reproductive capacity or survival of an exposed individual. Consequently, the potential reduction in population due to contaminant exposure (reduction in reproductive success or survivorship, for example) may not be observable from qualitative data on presence or absence of the test species on-site. Therefore, the observation of muskrats in the study area does not support nor contradict the results of the exposure estimations and characterized risks. Muskrat populations may be present on the site whether or not the conditions at a portion of the sampled stations present a risk to survival or reproduction of individuals.

Overall, the exposure analysis indicates that survival or reproduction for muskrat may be impaired in the study area due to exposure to inorganics in diet. The assessment endpoint was not met for a number of inorganic compounds, which indicates that habitat at those locations are impacted. Based on NOAEL TRVs, the evidence for risk to muskrat populations is highest from exposure to arsenic, chromium, copper, lead, and mercury, particularly in reach 1. There is a moderate level of uncertainty in each of these risk estimates due to estimates made in the ingestion models, the uncertainty in the TRVs and the lack of information on the form and toxicity of the metals found in the sediment as compared to the forms used in laboratory tested used to generate the TRVs.

4.5.1.2 Piscivorus Birds. The assessment endpoint was:

Sustainability (survival, growth, reproduction) of local populations of piscivorus birds.

Dietary exposures of green heron to COPCs were used to evaluate risk to piscivorus bird populations foraging on-site. Exposures were evaluated using the HQ approach, whereby daily dose, estimated from site-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. There were no COPCs with HQs greater than one for green heron for any COPC. The exposure analysis indicates that there is no evidence of negative impacts on the survival, growth, or reproduction of green heron populations, or other piscivorus birds, resulting from the exposure to COPCs in the study area.

4.5.1.3 Waterfowl. The assessment endpoint for waterfowl was:

Sustainability (survival, growth, reproduction) of local populations of waterfowl.

Dietary exposures of mallard ducks to COPCs were used to evaluate risk to omnivorous waterfowl populations inhabiting the site. Exposures were evaluated using the HQ approach, whereby daily dose, estimated from site-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 assessment endpoint for waterfowl was not met based on risks to mallards from exposure to inorganic COPCs. For both the site-wide and 38-acre scenarios, average and maximum exposure case HQs for chromium, lead, and mercury were greater than 1 (Table 4-197). The average exposure case is more relevant than the maximum exposure case since the mallard forages over wide areas. There is fairly high confidence in daily exposure doses due to the collection of site-specific data on plants, which were responsible for the majority of the exposure via ingestion.

Since the ingestion of plants and sediment contributed heavily to the estimated daily dose, estimates of risk were again closely related to sediment concentrations of the contaminants.

Estimated exposure using the same model at reference sites indicated a risk from exposure to lead, but not for chromium or mercury for either the maximum or average exposure scenarios. This indicates that the exposure calculations for lead are conservative for an urban watershed.

The HI of 76 associated with the mallard site-wide scenario for the average case (Table 4-198) is generated primarily by the high HQ for iron of 68 (approximately 90%). Chromium (HQ=2.9), lead (HQ=2.4), and mercury (HQ=1.5) had relatively small contributions to the HI. For the average case, 38-acre wetland scenario (Table 4-200), the HI for inorganic contaminants was 110. Again, the majority of the HI was contributed by a high HQ for iron of 97 (approximately 88%). Chromium (HQ=5.2), lead (HQ=3.6), and mercury (HQ=2.0) accounted for a relatively small proportion of the HI. In contrast, the HI for the average case scenario at the reference locations was 39, with the majority (95%) of the HI contributed by iron and with lead as the only other compound with an HQ greater than 1 (Table 4-202).

There is relatively high confidence in the mallard TRVs used for arsenic and mercury since they were based on the same species for a chronic exposure. There is still uncertainty associated with the form of the compounds fed to laboratory test animals as compared to the toxicity of the metals ingested on-site. The sediment concentration, site-wide, that would correspond to an HQ of 1 using a NOAEL TRV for arsenic was approximately 1,000 mg/kg. For both the site-wide and 38-acre scenarios, the stations that had average arsenic concentration in sediment greater than 1,000 mg/kg (corresponding to HQ greater than 1) were located only within reach 1, at stations BW, 12, 13, and 19. The exposure analysis indicates that a portion of the potential mallard habitat may be impacted within the Wells G&H 38-acre wetland. However, the limited area of arsenic above 1,000 mg/kg is not sufficient to represent a threat to mallard populations within the wetland, even if the ducks limited foraging to this wetland exclusively.

The sediment concentration, site-wide, that would correspond to an HQ of 1 for mallard using a NOAEL TRV for mercury was between 1 and 2 mg/kg. The majority of stations in reach 1 and reach 2 had average sediment mercury concentrations greater than 1.5 mg/kg which would have corresponded to a HQ greater than 1 for mallard using a NOAEL TRV. This indicates that mercury concentrations correspond to levels that may lead to reductions in mallard populations. Since this conclusion is based on the NOAEL TRV, the probability of risk is considered to be low.

The highest station averages for lead and chromium that resulted in a calculation of risk to mallard (maximum case scenarios) resulted from high sediment COPC values in reach 1 at stations 22/TT-22 and WW, respectively. The TRVs for lead and chromium were based on NOAELs (reproductive) for other species of birds (quail and black duck, respectively). This reduces the confidence in the observed HQ calculations.

The interpretation of the results of the measurement endpoint indicated that chromium, lead, and mercury presented potential for impacting the reproduction and survival of mallard populations site-wide and within the Wells G&H 38-acre wetland. The HQs for the average case scenarios were not high, ranging from 2-5. However, the extent of the exceedences of each of these on an areal basis indicates the potential for impacts to waterfowl populations, based on a NOAEL TRV. Due the limited area of very high concentrations (>1,000 mg/kg) sampled in the site area, arsenic can not be concluded to be likely to impact the sustainability of mallard populations.

4.5.1.4 Small Terrestrial Mammals. The assessment endpoint was:

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

Dietary exposures of short-tailed shrew to COPCs were used to evaluate risk to small terrestrial mammal populations inhabiting the site. Exposures were evaluated using the HQ approach, whereby daily dose, estimated from site-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.

As for other mammalian and avian indicator species, the assessment endpoint for shrew was not met because of risk from exposure to inorganic contaminants. All of the stations at which shrew would be expected to be present had HQs in excess of 1 for several inorganic COPCs (Table 4-244). Among these inorganic were antimony, arsenic, cadmium, chromium, lead, and mercury. In addition, the HQ for Aroclor-1254 at station JY was 3.

Among the metals, cadmium, chromium, lead, and mercury had HQs greater than 1 at only 1 or 2 stations out of 11 sampled in reach 1. The limited number of stations with HQs exceeding 1 indicates that the risk to populations in the habitat suitable to shrew is negligible from exposure to these metals.

The HQ calculated for antimony exceeded 1 at four stations in reach1 (Stations BW, 22/TT-22, WG, and WH), with the highest HQ of 13 observed at station 22/TT-22. Antimony concentrations may pose a risk to reproduction or survival of shrew at a limited number of stations in reach 1, however, the results indicate that the risk to populations in the habitat suitable to shrew is negligible from exposure to antimony.

The HQ for arsenic for the wetland reference was 1.3. At all 19 stations in the study area, except station WSS, the HQ for arsenic was greater than 1.0 (Table 4-244). The highest HQ was at station BW which had an average sediment arsenic concentration of 1,239 mg/kg. The results indicate a potential for impairment of reproduction or survival of individuals at a majority of the stations, indicating potential impacts on small mammal populations from exposure to arsenic.

Confidence in exposure estimates for shrews are lower than for the other indicator species, because the concentration of COPCs in prey (*i.e.*, earthworms) were either modeled or uptake factors were obtained from the literature. Approximately 50% of the estimated daily exposure was derived from the ingestion of soil, for which site-specific data were available. The sediment values used were for wetland or shallow open water sediments in areas potentially exposed for foraging for prey by the shrew. For part of the year, the sediments in depositional areas may be saturated and may not be used by small mammals like shrew for feeding. The assumption of foraging in wetland sediments is likely to somewhat over-estimate exposure. Conversely, the assumption that the remainder of the shrew diet consists of organisms with no detectable body burden of inorganics is conservative and likely to underestimate exposure. The resulting exposure scenarios are reasonable, but have a higher level of uncertainty than for other receptors.

The TRVs for antimony and arsenic were based on chronic LOAELs for a mouse. The TRV for cadmium was based on a chronic NOAEL in a rat. For lead and mercury, the TRVs were based on chronic (reproductive) NOAELs in a rat. A moderate degree of uncertainty is associated with the use of a different species for the TRV, and this is compounded for antimony and arsenic, for which a factor of 10 was used to convert a LOAEL to a NOAEL.

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. The highest HQs for average case scenarios were seen with arsenic. Aroclor-1254, antimony, cadmium, chromium, mercury, and lead concentrations may pose a risk to reproduction or survival of shrew at a limited number of stations, in reach 1, but due to the limited extent of habitat affected, the risk to small mammals is negligible.

4.5.1.5 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 populations.

Comparisons of COPC concentrations in fish tissue (largemouth bass, white sucker, and pumpkinseed) collected from the study area against COPC concentration in tissue collected from reference stations indicated that largemouth bass and pumpkinseed within the Aberjona River Study area carry higher body burdens of organic COPCs than do fish from reference stations (Tables 4-246 to 4-251). Comparisons are made separately for fillet, offal, and whole body tissue concentrations for largemouth bass and white sucker. For these species, the highest ratios of study area to reference organic COPC body burdens were found in fish collected from Upper Mystic Lake (reach 6). Other portions of the study area in which tissue concentrations exceeded reference concentrations to a substantial degree were reaches 3 and 5. Although this evaluation did not address the ecological effects of COPCs, it did indicate that bioaccumulation of organic contaminants is greater within the Aberjona River Study area than at reference stations.

There were few exceedances of available fish tissue residue benchmarks obtained from the literature, and exceedances were less than an order of magnitude (Tables 4-246 to 4-250) for largemouth bass. The mercury concentration in fillets of largemouth bass from reaches 3, 4, 5, and 6 exceeded a tissue residue benchmark (no observed effect dose) for whole body yellow perch. In reach 3, offal tissue concentrations of arsenic and lead in brown bullheads exceeded benchmark values (Appendix E.2). Results of the ecological effects evaluation indicated COPC concentrations in fish collected from the study area are, in most cases, lower than those that would be expected to be associated with adverse effects. Confidence in this evaluation is limited to the extent that benchmarks were not available for all COPCs and the evaluation did not consider the potential additive and synergistic effects of multiple COPCs in fish.

The ten pumpkinseed samples used for this analysis were collected from reach 1 (1 sample), reach 3 (3 samples), reach 4 (3 samples), and reach 5 (3 samples). The highest ratios of study area to reference organic COPC body burdens were found for 4, 4'-DDD, alpha-chlordane, and Aroclor-1254. These data indicate that the exposure of fish to organic contaminants is greater within the Aberjona River Study area than at reference stations, resulting in higher tissue concentrations. The comparisons of COPC concentrations in tissue to reference locations for pumpkinseeds also indicated that study area fish have higher body burdens of aluminum, arsenic, cadmium, cobalt, copper, lead, and selenium than at reference locations. Again, based on a limited number of benchmarks (arsenic, cadmium, copper, lead, mercury, and selenium), none were exceeded, indicating the observed tissue levels in pumpkinseeds do not pose risk to individuals' survival or reproduction due to exposure to each contaminant separately.

Results of the ecological effects evaluation indicated that COPC concentrations in fish collected from the study area are, in most cases, lower than those that would be expected to be associated with adverse effects. Confidence in this evaluation is limited to the extent that benchmarks were not available for all COPCs and the evaluation did not consider the potential additive and synergistic effects of multiple COPCs in fish. However, the assessment did not indicate any impacts on the local populations of predatory fish, bottom-feeding fish, and small foraging fish populations.

4.5.1.6 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 three separate lines of evidence evaluated. The weight of evidence from the benchmarks screening, crayfish tissue concentrations, and toxicity testing are each discussed below.

The comparison of sediment concentrations to benchmarks for benthic invertebrates indicated that concentrations of several inorganic compounds in the sediment exceeded SELs, ERMs, TELs, and NOAA SBs under the average station comparison (Table 4-252 to 4-260). The screening value used for cyanide was an LEL for free cyanide (OME, 1996). This value is very conservative, as the total sediment concentrations used will greatly over-estimate free cyanide. Since there is little confidence in the screening benchmark, the concentrations of cyanide above this level are not interpreted as an indication of risk.

For inorganic compounds other than cyanide, only arsenic, chromium, and selenium in reach 1 showed HQs greater than 10 for the average case scenario (Table 4-261). Excluding cyanide and the inorganic COPCs without effects-based screening criteria (barium, beryllium, cobalt, selenium, thallium, and vanadium), there were no inorganic HQs greater than 1 in reach 5. Fewer exceedences in reach 5 were likely explained by the lack of samples from depositional areas in this reach (see Appendix E.4). The average case HQs for arsenic, chromium, and copper were greater than 1.4 in all other reaches (reaches 1-4 and 6). The average case HQs for lead, mercury, and zinc were between 1.6 and 3.5 for reach 1. Mercury had no HQs greater than 1 in any other reach. Lead had HQs of 1.4 and 1.7 in reaches 4 and 6, respectively. Among these inorganics with effects-based benchmarks, chromium (HQ=1.2) and lead (HQ=1.3) had HQs greater than 1 at the reference wetland locations.

Evaluations involving comparison of the maximum case and maximum detected concentrations to benchmarks indicated there are many stations and individual sampling locations at which the benthic invertebrate community may be impaired, particularly in reaches 1 (the Wells G&H 38-acre wetland) (Tables 4-262 and 4-263).

The greatest source of uncertainty involves the magnitude of ecological effects associated with exceedances of sediment background levels. NOAA sediment background values (Buchman, 1999) were used in the comparison for analytes with no available effects-based benchmarks (cobalt, selenium, and vanadium). Excluding these analytes, there were still wide-spread exceedances of effects-based benchmarks for arsenic, chromium, copper, lead, mercury, and zinc. For several of these compounds (chromium, copper, and lead), the maximum (95% UCL concentration) and maximum detected concentrations at the reference sites also exceeded the benchmarks.

The overall results from the benchmark analysis indicates potential effects on benthic communities from inorganics, especially arsenic, chromium, copper, lead, mercury, and zinc in reaches 1 and 2. Since the benchmarks used for each of these metals was the SEL (Persaud, *et al.*, 1993), 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 site-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 4.3.2.3, AVS-SEM data collected on-site 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. In order to investigate site-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. The areas of impaired benthic invertebrate communities correspond to areas of elevated concentrations of arsenic, chromium, copper, mercury, and zinc in sediment.

Two derived indicators of community impairment, the Toxicity Index (TI, Table 4-268), and the Community Index (CI, Table 4-269), were computed to summarize the weight of evidence for the benthic invertebrate endpoints at wetland and stream locations. The CI index for the pond locations was not computed since the reference location at 03-IP had a dramatically reduced invertebrate community, possibly due to low oxygen conditions. Evidence for toxicity at the three lake/pond stations (stations 04, 06, and UF) was limited, with only two stations (04 and 06) showing reduction in a single toxicity endpoint. Based on these results, the risk to benthic communities from exposure to sediments is low at the lake stations.

Several stream and wetland stations had TI values 2 or 3, including stations 10,13, 19, TT-32, and TT-33. These stations had multiple toxicity endpoints showing reduced growth of either *C. tentans* or *H. azteca* as compared to reference locations. For stations with multiple growth endpoints showing impairment (TI values of 2 or 3) there is a strong indication of toxicity of sediments to benthic invertebrates at these stations (Figures 4-32 to 4-34). Stations with less evidence of toxicity, with TI values of 1, included 12, TT-29, and TT-30. Stations with CI values of 3 or more have a moderate to high likelihood of impaired benthic invertebrate communities. These stations included: 10, 12, 18, TT-29, TT-30, TT-32, and TT-33.

Among the metals exceeding effects-based sediment screening benchmarks (arsenic, chromium, copper, lead, mercury, and zinc), the CI value was highly correlated to arsenic, copper, and zinc, whereas the TI value correlated most highly to arsenic concentrations.

The third measurement endpoint for the benthic invertebrate community involved the comparison of COPC burdens in crayfish collected from the study area to both body burdens in reference crayfish and to tissue residue benchmarks for various decapods. With the exception of four inorganics (barium, cobalt, manganese, and silver), COPC concentrations in study area crayfish exceeded those detected in reference crayfish (Table 4-266). However, as for fish, the comparison to tissue residue benchmarks suggested that body burdens observed in crayfish collected from the study area were not elevated enough to be associated with adverse effects to individuals. Again, the tissue residue comparison carries some uncertainty in that there were few available tissue residue benchmarks, and those available were associated with both marine and freshwater decapods (lobster, crayfish, and crabs). The crayfish results indicate elevated body burdens of contaminants. These concentrations were not at levels known to be associated with adverse effects on individuals, however, the potential for sub-lethal effects on individuals (impairment in reproduction) is not addressed and may result in adverse impacts on populations. However, evidence from the tissue data alone are not sufficient to make these conclusions.

Consideration of the evidence among the three benthic invertebrate measurement endpoints indicates that there are impacts from inorganic contaminants on invertebrate communities within the Aberjona River Study area. The toxicity testing demonstrated adverse effects on 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 crayfish tissue data indicated elevated concentrations of contaminants, but these were not associated with known biological effects on individuals. The comparison of sediment concentrations to effects-based benchmarks indicate that there are depositional areas, particularly in reach 1, which have inorganic contaminant concentrations that could impair benthic communities. Metals with average station concentrations in reaches 1 and 2 above effects-based criteria included arsenic, chromium, and copper. Lead, mercury, and zinc concentrations in reach 1 also exceeded effects-based criteria.

None of the stations with arsenic concentrations less than or equal to 44.5 mg/kg (observed at station 04-IP) showed indications of toxicity (Figure 4-37). Indication of impairment of benthic invertebrate communities associated with indications of chronic toxicity and/or impairment of community composition characteristics were associated with wetland and stream stations having sediment concentrations of arsenic greater than 220 mg/kg. For the site-specific data, this concentration of 220 mg/kg arsenic represents a upper threshold effects level since all stations in the triad analysis with arsenic concentrations at or above this level consistently showed evidence of toxicity and/or community impairment. There were no indications of toxicity or impairment for stations with sediment arsenic concentrations of less than or equal to 44.5 mg/kg, which represents lower threshold effects level.

Sample SD-MC-04-TR (station 04-IP) had a sediment chromium concentration of 512 mg/kg which greatly exceeded the SEL of 110, however, no toxicity was detected on these sediments. The sample collected at WW-06 had very high concentration of chromium (6,550 mg/kg), and showed little evidence of community impairment and no evidence of sediment toxicity compared to either laboratory controls or reference locations. This station also had a very high organic carbon content (76%), which may influence the bioavailability of chromium. None of the toxicity endpoints were significantly different from the laboratory controls or reference station for station WW. The invertebrate abundance was relatively low at WW, but none of the other community indices indicated impairment. These observations indicate that high concentrations of total chromium are not consistently associated with impacts on benthic communities.

Chromium (+6) is known to have greater toxic effects than chromium (+3) (Eisler, 2000). Additional data collected in October 2002 to evaluate the presence of chromium (+6) in sediments in reaches 1 and 2 (see Appendix C.4). Among the six stations sampled, the total chromium concentrations ranged from 244 mg/kg at WS-08, to 13,400 mg/kg at WW-06. Among these samples, Chromium (+6) was detected only at the station WW-06 at 17.3 mg/kg or approximately 0.13% of the chromium (+3). These chromium results indicate that most of the total chromium on-site is likely in the form of the less toxic chromium (+3).

Copper, mercury, and zinc concentrations also correlated with higher CI index values and reduced growth of *C. tentans* (Table 4-270). The relationships of copper and zinc to CI and TI values were similar (Figure 4-33). All stations with indicators of impaired communities (10, 12, 13,18, 19, TT-29, TT-30, TT-32, and TT-33) also had copper concentrations above the SEL of 110 mg/kg (Figure 4-33). However, station WW and 04-IP showed no evidence of sediment toxicity and very low evidence of community impairment, but also had relatively high concentrations of copper (210 mg/kg at station WW-06 and 344 mg/kg at 04-IP). These data indicate that the association of the observed toxicity with higher copper concentrations is not consistent. This conclusion is supported by AVS-SEM data which indicates that divalent metals are unlikely to be bioavailable at either station WW or 04-IP.

Zinc concentrations were the highest at several triad stream or wetland stations with the highest CI values and relatively lower TI values (stations 10, 12, 18, and TT-29). With the exception of station TT-33, all stations having impaired benthic communities had zinc concentrations greater than the SEL value of 820 mg/kg. However, the association of high zinc concentrations with evidence of biological effects is inconsistent with the lack of toxicity observed at station WW (Zn - 888 mg/kg) and indications of impairment at station TT-33 with a zinc concentration of 581 mg/kg. The evidence for the contribution of zinc to the observed toxicity is not strong. Again, the lack of toxicity of zinc at these stations maybe attributed to bioavailability.

Although the concentration of a number of metals co-vary with the elevated concentration of arsenic at the triad sampling sites, both the TI and CI indices are most closely associated with arsenic concentrations at levels greater than 220 mg/kg. There were no apparent effects on benthic communities at arsenic concentrations less than or equal to 44.5 mg/kg, and the concentration corresponding to the lowest observed adverse effects threshold was 220 mg/kg.

4.5.2 Risk Description

The main purposes of risk description are 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. For the Aberjona River study area, TELs are estimated for sediment concentrations of COPCs, as these values and the associated plant tissue concentrations are the media responsible for the majority of the COPC exposure for all of the receptors evaluated. TELs for surface water are not calculated since surface water was determined to be a minor exposure pathway for all of the receptors considered.

The risk estimates based on NOAEL values represent the lower bound of the threshold for adverse ecological effects for each assessment endpoint. In the preceding section, the COPCs were identified for each receptor for COPCs above the lower threshold for adverse effects with a majority of the habitat for the receptor. Heron is not considered here, since there was no indication of ecological effects associated with exposure to COPCs on site. The majority of the habitat on the site was defined > 50% of the stations for muskrat or shrew. Majority of the habitat was defined as the average case for mallard sitewide. Where the majority of the habitat 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 endpoints exceeding the lower threshold effects level (NOAEL HQs > 1 for majority of the habitat), included: muskrat (arsenic, chromium, copper, lead, and mercury), mallard (chromium, lead, and mercury) and shrew (arsenic).

Using less conservative LOAEL-based TRVs and 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, shrew, and mallard are presented in Appendix E.3. 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 significant risk to receptor populations. The location and extent of the risk can be evaluated by identifying the stations with COPC concentrations in sediment 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 route (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 were directly derived from sediment concentrations, 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 site or foraging area. The assumption of uniform contamination to back-calculate effects levels has been used for the development of water-quality criteria (USEPA, 1995a) 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 site. The distribution of contaminants in sediments determined to be risk-drivers for indicator species (arsenic, chromium, lead, and mercury) were evaluated across the site on a station by station basis in Figures 4-35 to 4-38. All station averages are presented, however, not every station was used to represent habitat for all species.

For muskrat, mallard, and shrew, upper TELs were calculated for the COPCs indicating risk for the population. These COPCs were those that exposure estimates showed the majority of the stations with HQs above 1 for NOAEL TRV. For muskrat, the risk-driver COPCs were arsenic, chromium (reach 1), copper (reach 1), lead, and mercury. For shrew, arsenic was the only risk-driver COPC. Upper TELs were calculated for mallard for chromium, lead, and mercury since the average case scenario indicated risk to populations from exposure to these metals.

For the muskrat, using NOAEL TRVs, most of the stations exceeded an HQ of 1 for arsenic, including the reference locations. This lower TEL corresponds to a sediment arsenic concentration of 2 mg/kg. Applying a LOAEL TRV (Table 4-271), the majority of the stations in reach 1 still indicated a risk due to ingestion of arsenic by muskrat from plants and sediment. Using the food-chain model for muskrat, the upper TEL for arsenic was calculated to be 150 mg/kg arsenic in sediment. This upper TEL value was based on a LOAEL TRV of 4.0 mg/kg-d (chronic, reproductive, mouse). The TRVs from various studies ranged from 0.03 to 4.0 mg/kg-d (Appendix E.3-1). Using the highest reported LOAEL (4.0 mg/kg-d) for arsenic, the estimated upper indicates risk to populations in reach 1, with 14 of the 19 muskrat stations having average sediment arsenic concentrations above 150 mg/kg.

Although the majority of the stations representing muskrat habitat exceeded NOAEL TRVs for mercury, only 6 stations (5 in reach 1, 1 in reach 2) exceeded the LOAEL TRV (0.13 mg/kg-d, rat, chronic, reproductive). Using the food-chain model for muskrat, the lower TEL for mercury was calculated as 1.3 mg/kg mercury in sediment. The upper TEL for mercury was calculated to be 6.5 mg/kg mercury in sediment based on the LOAEL TRV. The concentrations of mercury are at levels high enough to indicate possible reduction in reproduction or sublethal effects at individual stations within reaches 1 and 2, but there is not strong evidence for population effects site-wide at the upper TEL, since only 6 stations exceeded the upper TEL concentration.

The majority of the stations representing muskrat habitat also exceeded NOAEL TRVs for lead, but only 1 station (WH), exceeded the LOAEL TRV. Using the food-chain model for muskrat, the lower TEL for lead was calculated as 135 mg/kg lead in sediment. The upper TEL for lead was calculated to be 1,340 mg/kg lead in sediment based on the LOAEL TRV. The concentration of lead exceeded this upper TEL at only 2 stations (WH and 22/TT-22), however, station 22/TT-22 was not considered to be muskrat habitat. This indicates possible reduction in reproduction or sublethal effects at individual stations within limited areas of reach 1, but there is not strong evidence for population effects site-wide at the upper TEL, since the majority of the stations did not exceed the upper TEL for lead.

The majority of the stations representing muskrat habitat in reach 1 also exceeded NOAEL TRVs for copper, however, less than half (9 of 19) in reach 1 exceeded the LOAEL TRV. Using the food-chain model for muskrat, the lower TEL for copper was calculated as 399 mg/kg copper in sediment. The upper TEL for copper was calculated to be 525 mg/kg copper in sediment based on the LOAEL TRV. The concentration of copper exceeded this upper TEL at 9 stations in reach 1 and one station (TT-30) stations in reach 2. This indicates possible reduction in reproduction or sublethal effects at individual stations within limited areas of reaches 1 and 2, but there is not strong evidence for population effects site-wide at the upper TEL, since the majority of the stations did not exceed the upper TEL for copper. Available LOAEL TRVs for copper in rodents ranged from 2 mg/kg-day (mouse) to 104 mg/kg-d (rat). The selected LOAEL TRV in mink

(15.83 mg/kg-d) represents a moderately conservative value on which to set the upper TEL. There is, however, a relatively high uncertainty associated with this TRV.

The majority of the stations representing muskrat habitat also exceeded NOAEL TRVs for chromium in reach 1, but not site-wide. Two stations (TT-29 and WW), exceeded the LOAEL TRV. Using the food-chain model for muskrat, the lower TEL for chromium was calculated as 616 mg/kg chromium in sediment. The upper TEL for chromium was calculated to be 2,800 mg/kg in sediment based on the LOAEL TRV. The concentration of chromium exceeded this upper TEL at only 3 stations (TT-28, TT-29, and WW). This indicates possible reduction in reproduction or sublethal effects at individual stations within limited areas of reach 1, but there is not strong evidence for population effects site-wide at the upper TEL, since the majority of the stations did not exceed the upper TEL for chromium.

Acute and chronic effects of chromium on mammals are usually associated with chromium (+6) (Eisler, 2000). Studies of the forms of chromium within the study area at stations with high total chromium concentrations (including station WW) indicate that the chromium is almost entirely (>99%) in the less toxic chromium (+3) form in sediments. The LOAEL TRVs for muskrat available for chromium ranged from 2 mg/kg-d to 32 mg/kg-d (Appendix E.3-1) for laboratory rodents. Using the lowest laboratory LOAEL the upper TEL would be equivalent to <200 mg/kg chromium in sediment, and using the highest LOAEL, the upper TEL would be equivalent to 2,800 mg/kg chromium in sediment. This range indicates that the uncertainty associated with the TRV is two orders of magnitude.

The HQs based on LOAEL TRVs for mallard are greater than 1 for only chromium (Table 4-272) for the average case scenarios. The LOAEL TRV exceedences for chromium were low, with HQs of 1.0 and 1.8 for chromium in the average case and maximum case (95%UCL) scenarios, respectively, in the Wells G&H 38-acre wetland (reach1). Using the food-chain model for mallard, the lower TEL (based on NOAEL TRVs) for chromium in sediment was 390 mg/kg. The upper TEL for chromium was calculated to be 2,200 mg/kg total chromium in sediment.

This upper TEL value was based on a LOAEL TRV of 5.0 mg/kg-d (chromium III, black duck, reproductive). Sediment concentrations at five stations within mallard habitat exceeded this upper TEL value site-wide. Based on the site-wide model, risks to mallards exposed to conditions across all of the stations would not pose a risk above the LOAEL TRV under average case conditions. Confidence in this risk estimation is reduced since it is based on TRV from a different species (black duck) and the laboratory dose used a different form of chromium (chromic potassium).

The HQs based on LOAEL TRVs for mercury and lead were less than 1 for mallard. As stated in section 4.5.1.3, the majority of stations in reach 1 and reach 2 had average sediment mercury concentrations greater than 1.5 mg/kg which would have corresponded to an HQ greater than 1 for mallard using a NOAEL TRV, representing the lower TEL Using the food-chain model for mallard, an upper TEL was calculated as 17 mg/kg mercury in sediment, based on the LOAEL TRV. Only the sediment concentration at station TT-30 exceeds this value.

Using the food-chain model for mallard, the lower TEL was calculated as 140 mg/kg of lead in sediment. An upper TEL for lead was calculated to be 750 mg/kg lead in sediment, based on the LOAEL TRV. The average sediment concentrations at several stations exceeded this level, however, several of these stations were deeper pond stations (UM, UF, LF, and LM) and were not considered mallard habitat. The three stations within mallard habitat with lead concentrations above the mallard upper TEL were stations WH, 13, and 20 in reach 1. Based on the site-wide model, however, risks to mallards exposed to average conditions across all of the stations would not pose a risk above the LOAEL TRV (Table 4-272). The LOAEL TRV was based on a chronic, reproductive LOAEL using Japanese quail fed lead acetate. However, other studies (Douglas-Stoebel, et. al., 2001) have shown sublethal effects in mallard ducklings in response to ingestion of contaminated sediments with 3449 mg/kg lead. The only station that had lead concentrations in excess of this value was at station 22, near the rifle range in the northern part of the 23-acre wetland. The probability of risk to waterfowl due to the ingestion of lead in the vicinity of 22 is high, however, station 22/TT-22 was not used in the mallard model since the canopy cover and lack of open water in most of this area made is less desirable mallard habitat. The risks from exposures to lead are limited to isolated stations which may not represent a substantial risk to the mallard population utilizing the entire site. Using the NOAEL TRV, there was risk even on the site-wide basis. Consequently, the risk site-wide (to mallard populations) is considered to be low since the high concentrations of lead are not widely distributed.

The evaluation of LOAEL TRVs for shrew (Table 4-273) indicated that only arsenic concentrations exceeded the LOAEL TRVs. The average case LOAEL HQs above 1.0 for arsenic ranged from 1.4 to 5.9 at only 3 stations. Using the food-chain model for shrew, a lower TEL, based on the NOAEL TRV was calculated as 16 mg/kg arsenic in sediment. An upper TEL was calculated to be 200 mg/kg arsenic in sediment, based on the LOAEL TRV. Sediment concentrations at three stations in shrew habitat (stations 20, BW, and CB-03) exceeded this upper TEL value. There is evidence in reach 1 and reach 2 for possible impacts to shrew or other small mammals due to exposure to arsenic in diet at a limited number of stations, although the uncertainty associated with this risk is high.

Evaluation of data collected for the BERA indicates that inorganic contamination in the Aberjona River Study area may be associated with current and future impacts on populations of semi-aquatic mammals, water fowl, small terrestrial mammals, and benthic invertebrates (Table 4-274). Exposure analysis indicates that risk to local populations of piscivorus birds, like green herons, are unlikely.

For fish, evidence did not suggest wide-spread and direct adverse effects of COPCs on individuals. Fish tissue analysis indicated exposure to inorganics result in higher body burdens of COPCs than fish collected at reference stations. However, the results of comparisons to tissue residue benchmarks indicated COPC concentrations in fish collected from the study area are lower than those that would be expected to be associated with adverse effects, with the exception of mercury concentrations in largemouth bass. Tissue concentrations in fillet and offal from largemouth bass from reference locations exceeded those from the study area.

Comparisons to available effects-based sediment benchmarks identified arsenic, 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 site 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 site-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 individual contaminants on-site. The concentration of arsenic was most closely correlated to the benthic invertebrate endpoints. To a lesser degree, concentrations of copper, chromium, mercury, and zinc were elevated among the stations either showing sediment toxicity or impaired community composition, but the relationship of adverse affects 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.

Based on the biological effects data for benthic invertebrates, site-specific threshold concentrations were established as based on lowest observed affects levels from the triad sampling data. For arsenic, the site-specific TEL was 220 mg/kg for benthic invertebrates. The bioavailability and toxicity of arsenic to benthic organisms may vary station to station depending on other variables including AVS, TOC, and other metals. However, based on the site-specific toxicity information and its close correlation with sediment arsenic concentration, the threshold value of 220 mg/kg arsenic consistently represents an upper TEL.

The arsenic sediment benchmark for benthic invertebrates (33 mg/kg) was exceeded at a majority of stations in each reach, except reach 5 (one station, station 05) based on average station concentrations. The highest average sediment arsenic concentration observed at the reference locations was just above this benchmark (38 mg/kg) at station 04-IP, a stream reference site. As summarized in section 4.5.1.6, the toxicity testing data support the conclusion that there are impacts on benthic invertebrate communities that are associated with high sediment arsenic concentrations. The upper TEL for benthic invertebrates, based on-site-specific data was 220 mg/kg. Ten stations in reach 1 (10, 11, 12, 13, 18, 19, 20, BW, TT-28, and TT-29), four stations in reach 2 (TT-30, TT-32, TT-33, and CB-03), and two in reach 6 (04 and UM) had average station arsenic concentrations above the invertebrate upper TEL (Figure 4-37).

Lead distributions across the site showed high concentrations (above 500 mg/kg) observed in reach 1 at stations 22/TT-22, BW, JY, WH, 10, 13, 20, and 21. Levels of lead at five stations in reach 6 were at or above those observed at reference locations. The sediment benthic invertebrate benchmark for lead (250 mg/kg) was exceeded in reach 1 at every station except NRSE, 14, and 15 (Figure 4-35). Although the benchmarks indicate exposure to lead above threshold values, as summarized in section 4.5.1.6, the toxicity testing data do not strongly support conclusion of significant effects on benthic invertebrate communities associated with lead.

On a station by station basis, the benthic invertebrate sediment benchmark for mercury of 2 mg/kg was exceeded in reach 1 (11 of 20 stations), at three stations in reach 2, at station AS in reach 3, and at station 04 in reach 6 (Figure 4-36). Although the benchmarks indicate exposure to mercury above threshold values, as summarized in section 4.5.1.6, the toxicity testing data do not clearly support conclusion of significant effects on benthic invertebrate communities associated with mercury.

Chromium distributions across the site showed high concentrations observed in reach 1 and reach 2, with the highest concentration at station WW (12,365 mg/kg) (Figure 4-38). The sediment benchmark for chromium (110 mg/kg) was exceeded in reach 1 every station except 14 and 15 (Figure 4-38). Although the benchmarks indicate exposure to chromium above threshold values, as summarized in section 4.5.1.6, the toxicity testing data do not strongly support the conclusion of significant effects on benthic invertebrate communities associated with chromium.

4.5.3 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 4.3.1, 4.5.1, and 4.5.2, important sources of uncertainty associated with the BERA for the Aberjona River Study area are addressed below, and summarized in Table 4-275.

Uncertainty associated with the conceptual model (Figure 4-1) 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. In an urban watershed, the source of contaminants such as inorganics in sediments can not be distinguished from site-related compounds or other local run-off or upstream source. However, the number of sediment samples collected within the study area provides high confidence in the estimation of exposure from sediment.

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 site and will influence actual presence or exposure of species or communities in the different reaches. For example, the short-tailed shrew was selected as a small mammal that is likely to inhabit the drier areas of the Aberjona River riparian habitat. Using wetland sediment 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.

4.5.3.1 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, sediment ingestion rate) were based on literature values, not site-specific data. For instance, it was assumed, based on other studies, that approximately 30% of the shrew diet is comprised of earthworms. 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.

There is also 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. For example, plant tissue was collected from only the Wells G&H 38-acre wetland and from one reference location. These data were used to calculate uptake factors applied to the entire site. Using uptake factors derived for the site and for each specific COPC decreased the uncertainty over using literature values. However, there were several sources of uncertainty associated with calculations and use of these parameters. First, multiple plants were collected for each tissue sample over a limited area near the sediment sampling station. These samples contained more than one species of plant, which combined the potential variation of different species to uptake or concentrate various COPCs into a single sample. These tissue concentrations were then paired with average site sediment concentration. Due to variation in the concentration of COPCs within the site, the average site sediment concentration may not reflect precisely the conditions in which the individual plants were growing. Uptake factors were calculated for tissue samples collected at four locations.

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. In order to evaluate the influence of the use of derived uptake factors for plants on the dietary exposure of muskrats, a comparison was made to the HQs calculated using the uptake factors and station-specific sediment data compared to plant tissue data collected at a specific station. HQs using uptake factors for the muskrat (inorganics only) for stations 18, 20, 21, and 23 were compared to HQs recalculated using actual plant tissue concentrations available for each of these four stations (Table 4-276). In general, the uptake factors presented a good estimate of the plant tissue concentration compared to site-specific tissue concentration. In most cases, using the model with uptake factors and site-specific sediment concentrations over-estimated the plant tissue concentration. This reflects a portion of the uncertainty in the uptake parameter, because it assumes the ratio of sediment to plant tissue concentration will be linear across sediment concentrations.

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.

Dietary exposure to invertebrate prey species was based on crayfish tissue data for muskrat, heron, and mallard models. This was another source of uncertainty in exposure estimates since a single species may either overestimate or underestimate the dose of a particular COPC compared to the actual mixed diet of benthic invertebrates species ingested by a consumer.

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. As noted above, the selection of relevant stations is most uncertain for the shrew. Some stations were selected which had one or more samples located in areas of standing water or near the shoreline of the Aberjona River, but still within the channel. It was assumed that these locations could be utilized by shrews during periods of drier weather when water levels in the river recede. 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.

To evaluate the magnitude of this source of uncertainty, exposures were re-calculated for some COPCs based on geometric means for the muskrat model. Using four inorganic COPCs (arsenic, copper, lead, and mercury), the data were evaluated using the statistical program Pro-UCL (ver

2.1), to determine the distribution of the samples. For those stations with sediment COPC concentrations having log-normal distributions, the geometric mean was calculated and compared to the arithmetic mean for the same samples (Table 4-277). Only a sub-set of the data are shown, since the analysis was only done for COPCs approximating a log-normal distribution. All of the geometric mean values were lower than the corresponding arithmetic mean value for the same data set. These data were then used in the muskrat exposure model to calculate adjusted HQs based on geometric mean values for sediment (Table 4-278). Again, all of the re-calculated HQs were lower than the corresponding HQs based on arithmetic mean sample values. These HQs are not adjusted for concentrations of the COPC in invertebrate tissue, however, this is a relatively small error in the model, since 80 to 90% of the daily dose for muskrat was derived from the exposure to sediment and plants (plant concentrations were estimated from sediment concentrations).

The results indicate that HQs would have been lower for some stations if geometric means of sediment concentrations had been used. However, the relative magnitude of the changes were small, and would not have significantly changed the risk conclusions.

The uncertainty associated with use of arithmetic means for the calculation of the sediment concentrations used in the benthic invertebrate benchmark comparisons were also evaluated for seven inorganic compounds (Table 4-279). Similar to the discussion above for muskrat, the average sediment concentrations at any station with a log-normal distribution for a particular COPC was re-calculated as a geometric mean. The results of this comparison is similar to the muskrat results. At two stations an HQ value would have dropped below 1. For arsenic and chromium in reach 1, the geometric means would have been below 5, instead of 13, and 18, respectively. These minor changes in the HQ values resulting from the calculation of geometric means would not have changed the conclusions of the BERA.

In general, there is high confidence that data collected for the BERA represent the types and distributions of contaminants within the Aberjona River 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 for most of the models are worst-case scenarios that tend to overestimate exposure.

For muskrat, the risk evaluation was based on average case scenarios at 43 stations site-wide and three reference scenarios. In order to evaluate the potential risk for muskrat using the maximum exposure, the five metals determined to have potential risk (arsenic, chromium, copper, lead, and mercury) to muskrat were also modeled (Appendix E.1-97 to E.1-141) using maximum exposure estimates using both NOAEL HQs TRVs (Table 4-280) and the less conservative LOAEL TRVs (Table 4-281). The maximum exposure estimates were based on 95% UCLs for sediments (which were also used to estimate plant dose using an uptake factor). If there were insufficient numbers of samples available to calculate a 95% UCL (less than 4), or the 95% UCL was greater than the maximum detected value, the maximum was used (Table E.1-142, Appendix E, or Tables 4-47 to 4-91). Prey dose for each station was based on maximum concentrations for crayfish in the reach (Table 4-25 and Appendix E.1-143) in which the station was located. Maximum tissue values were used as there were insufficient numbers of samples available to calculate a 95% UCL (less than 4). Similarly, maximum surface water concentrations were based on 95% UCLs (or maximum, if less than the 95%UCL) in reaches 1 and 6. For reaches 2, 3, and 4, there were insufficient numbers of samples available to calculate a 95% UCL and the maximum values were used.

The results of the dietary exposure analysis for muskrat using the maximum case scenario was similar to the average case scenario for each of these metals. As would be expected, all of the resulting HQ values were higher for the maximum case (Table 4-280) as compared to the average case (Table 4-189). All of the 43 stations, and the three reference habitats had HQs above 1 for arsenic. All of these maximum HQs (Table 4-280) were also above 10, with the exception of station 08. Using the LOAEL TRV for arsenic (4.02 mg/kg-d), all but one station in reach 1 exceeeded LOAEL TRV. In reach 2, the number of stations exceeding the LOAEL TRV increased from 2 (CB-03 and TT-30) for the average case to 5 (CB-03, TT-30, CB-04, TT-32, and TT-33) using the maximum (Table 4-281). There was no risk to muskrat from arsenic in any of the other reaches (3 - 6) based on the LOAEL TRV, even using the maximum exposure estimates.

Similarly, for chromium, copper, and lead, using the maximum exposure case in the LOAEL TRV comparison, HQs were higher, and a greater number of stations exceeded an HQ of 1. However, this model is overly conservative in assessing risk. There was one or more HQ above 1 for every COPC, with the exception of copper, for the three reference scenarios. Using the LOAEL TRVs (Table 4-281) and the maximum exposure estimates, the reference HQs were all less than 1. There was no risk exceeding the LOAEL TRV for muskrat at any station in reaches 3-6 for chromium, lead, and mercury, and fewer than 50% of the stations in reaches 1 and 2 had HQs above 1. For copper, using the maximum case and the LOAEL TRV increased the number of stations exceeding an HQ of 1 to 53%, and there were two stations, one in reach 3 (AS) and one in reach 6 (station 01) with HQs slightly above 1. This slight increase in risk from copper exposure to muskrat is the only difference in the conclusions derived from the average case scenario as compared to the maximum case. The analysis dietary exposure is more sensitive to the selection of TRVs than the calculations of the exposure concentrations in each medium.

4.5.3.2 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 Aberjona River 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 site 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.

Based on the review of available studies for which possible LOAEL TRV values were given (Appendix E.3), 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. 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. As seen from the swine study conducted in conjunction with this risk assessment, only approximately 50% of the arsenic in sediment fed to young swine was bioavailable (see section 4.4.4). 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. The most conservative RBA value determined for study area sediments (51%) was selected as the most appropriate to evaluate the oral toxicity of arsenic in sediments at all stations within the study area for mammals (muskrat and shrew). The site-specific RBA value of 51% was used to adjust the incidental sediment ingestion dose for each of the mammal models of arsenic. The dose from plant material was not adjusted by this RBA, since no RBA for plants was derived, and is a source of uncertainty for the muskrat model. If the RBA of 51% has been applied to plant ingestion, the upper TEL for muskrat would have been increased from 150 mg/kg arsenic in sediment to 270 mg/kg. At this higher TEL, ten stations in reach 1 and one station in reach 2 would still have exceeded the upper TEL, resulting in the same conclusion of potential risk to muskrat. No RBA factors will used for any other COPC or receptor species, and this contributes a large proportion of the uncertainty in the dietary models.

4.5.4 Summary of Ecological Risks

Based on data collected between 1995 and 2001, the effects-based screening resulted in the selection of nine COPCs in surface water (all inorganics) and fifty-eight COPCs in sediment

(VOCs, SVOCs, pesticides/PCBs, and inorganics) for evaluation in the BERA. Recently collected surface water, sediment, and sediment core data were qualitatively evaluated in Appendices E.4 and E.5. Eight 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 Aberjona River 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.

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 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 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 4.5.3, 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. 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 eight selected indicators/endpoints in the BERA, there were no indications of significant ecological risk associated with VOC, SVOC, and pesticide/PCB contamination within the Aberjona River Study area. However, the strength of evidence suggests that inorganic contaminants including arsenic, chromium, copper, lead, and mercury may pose current and future risks to one or more of the indicator populations.

Risk to survival or reproduction of green heron populations or other semi-aquatic predatory avian species was determined to be negligible since none of the HQs for COPCs were greater than the target HQ of 1 (Table 4-197) using the NOAEL TRV. The confidence in this evaluation is high due to the use of site-specific tissue data, and use of conservative NOAEL TRV values.

Results of the ecological effects evaluation indicated COPC concentrations in fish tissue collected from the study area are, in most cases, lower than those that would be expected to be associated with adverse effects (Tables 4-246 to 4-251). Evaluation of fish tissue residues suggested there is no direct and widespread impacts of inorganic COPCs on individual fish, or to fish populations. Comparisons of COPC concentrations in fish tissues to tissue benchmarks indicative of impairment showed no elevated tissue levels for large predatory fish (largemouth bass), large bottom-feeding foraging fish (white sucker), or small foraging fish (pumpkinseed), with the exception of concentrations of mercury in largemouth bass tissue. However, in all of these cases, the concentrations of mercury in reference fish tissues also exceeded tissue residue benchmarks. Confidence in this evaluation is limited to the extent that benchmarks were not available for all COPCs and the evaluation did not consider the potential additive and synergistic effects of multiple COPCs in fish.

Analysis of the exposure assessment for muskrat indicated HQs greater than 1, at a majority of stations based on NOAEL TRVs, for arsenic, chromium (in reach 1), copper (reach 1), lead, and mercury. A majority of the stations representing muskrat habitat exceeded NOAEL TRVs for mercury, only six stations (10, 12, 13, 21, TT-29, and TT-30) exceeded the LOAEL TRV (Table 4-271) and the associated upper TEL in sediment of 6.5 mg/kg. Similarly, the majority of the stations representing muskrat habitat also exceeded the NOAEL TRV for lead, but only one station (WH), slightly exceeded the LOAEL TRV, and associated upper TEL for lead of 1,340 mg/kg lead in sediment. The majority of the stations representing muskrat habitat also exceeded the NOAEL TRV for chromium, but only 3 stations (WW, TT-28, and TT-29), exceeded the LOAEL TRV, and associated upper TEL, of 2,800 mg/kg chromium in sediment. The majority of the stations representing muskrat habitat in reach 1 also exceeded NOAEL TRVs for copper, however, less than half (9 of 19) of the stations in reach 1 exceeded the LOAEL TRV and the upper TEL of 525 mg/kg copper in sediment. These results indicate that the risk from exposure to chromium, copper, lead, and mercury is high (above upper TEL) only at a few stations, and based on the endpoint criterion, exposure to chromium, copper, lead, and mercury presents a low risk (above lower TEL) in reach 1.

For arsenic, however, the majority of the stations reach 1 had HQ values for muskrat greater than 1 using both NOAEL and LOAEL TRVs (Tables 4-189 and 4-271). Due to the high HQs, and the number of stations exceeding the threshold values, the magnitude of the risk for muskrat exposure to arsenic is high. Fourteen of 19 stations in reach 1 and four stations (CB-03, TT-30, TT-32, and TT-33) in reach 2 exceeded the upper TEL of 150 mg/kg arsenic in sediment for muskrat. Stations below the TEL in reach 1 included 14, 16, WG, WH, and WW. The uncertainty associated with these estimates is moderately high, and associated mainly with the relative toxicity of the forms of arsenic in the reference doses as compared to field conditions. These results indicate a potential impacts on survival or reproduction of mammal populations such as muskrat or beaver exposed to arsenic in the diet while foraging in the Aberjona River study area in reach 1. The risk in reach 2 is lower due to the limited number of stations (4 of 14) above the upper TEL.

The mallard was used to represent waterfowl having relatively high exposure to sediments. Based on NOAEL TRVs, chromium, lead, and mercury had HQs greater than 1 for the site-wide and 38-acre average case and UCL scenarios (Table 4-197). Based on LOAEL TRVs, the only HQ above 1 was for the UCL model for chromium (Table 4-272). The upper TEL calculated for chromium in sediment for mallard was 2,200 mg/kg total chromium. Sediment concentrations at five stations within mallard habitat exceeded this upper TEL value site-wide (four in reach 1, one in reach 2).

For mercury and lead, the HQs above 1 for the mallard endpoint occurred for the NOAEL TRVs only. The concentrations of mercury and lead were indicated low risk for reduction in reproduction or sublethal effects to populations, but there was not evidence for high risk populations since LOAEL HQs were less than 1. Site-wide risk from exposure to lead were only slightly greater than those calculated for reference locations. Only three mallard stations (13, 20, and WH) had sediment concentrations exceeding the upper TEL of 750 mg/kg lead and one station (TT-30, reach 2) had a sediment mercury concentration above the upper TEL of 17 mg/kg.

With respect to arsenic, none of the stations had HQ values above 1 for the LOAEL TRV for mallard. The HQ of less than 1, using a NOAEL TRV, corresponded to relatively few areas of very high concentrations of arsenic (>1,000 mg/kg) at three stations in reach 1. Consequently, arsenic can not be concluded to be likely to impact the reproduction or survival of mallard populations on a site-wide basis.

The assessment of the waterfowl endpoint indicates a low risk to populations, site-wide from exposure to chromium, lead, and mercury. Although confidence in the estimate of risk to mallard was high, the magnitude of the risk to mallard populations were low, since it is based on the NOAEL TRV value, which represents the threshold for effects for potential impacts to populations.

Short-tailed shrew exposure models were used to evaluate potential risk to small mammal populations living in and near the wetlands bordering the Aberjona River. Analysis of the exposure assessment for shrew indicated HQs greater than 1, at a majority of stations based on NOAEL TRVs, for arsenic. The concentrations of arsenic indicated low risk to populations, but there was not evidence for high risk to populations since LOAEL HQs were less than 1 at all but three stations in shrew habitat. These three stations (stations 20, BW, and CB-03) had sediment arsenic concentrations which exceeded the upper TEL value of 200 mg/kg arsenic in sediment. Since the number of stations exceeding the upper TEL is small, the risk to small mammal populations, due to exposure to sediment arsenic, is low.

The evidence for the benthic invertebrate endpoints suggest that there were impacts from inorganic contaminants on invertebrate communities. The strength of the evidence was based on exceedances of sediment-effects benchmarks, which were supported by toxicity testing and evaluation of community composition data. The results of comparisons of the crayfish tissue residues to benchmark values did not strongly support the other benthic invertebrate endpoints.

Overall, the benchmark analysis indicated potential effects on benthic communities from inorganics, especially arsenic, chromium, copper, lead, mercury, and zinc with the highest HQs in reach 1. Since the benchmarks used for each of these metals was the SEL (severe effect level), they represent contaminant levels that potentially eliminate most of the benthic organisms (Persaud, *et al.*, 1993).

Biological effects evaluation, in the form of sediment quality triad analysis, was conducted. The results of the toxicity testing indicated that stations in reach 1 and reach 2 had reduced growth of *H. azteca* and/or *Chiromomus tentans* as compared to reference stations.

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 individual contaminants on-site. Two indices were computed to represent the strength of evidence for impairment of invertebrate communities at the stations sampled for the triad analysis. The TI (toxicity index), with a values ranging from 0 to 3, provided a relative indicator of the toxicity of the sediment to benthic invertebrates. Similarly, an index representing the weight of evidence for impairment of natural benthic invertebrate communities, CI (community index), was calculated for the stream and wetland habitats. The majority of the stations with CI values of greater than 1 also had corresponding TI values of 1 or more. This shows a strong correlation between stations having sediment toxicity in the laboratory showing impaired benthic communities in the field (Figure 4-32).

A number of variables can influence sediment toxicity and the resulting benthic community structure at any given location. However, a strong association was seen among the triad sampling stations between sediment chemistry and the CI and TI indices. The concentration of arsenic was most closely correlated to the benthic invertebrate endpoints (Figure 4-32, Table 4-270). To a lesser degree, concentrations of copper, chromium, mercury, and zinc were elevated at the stations showing sediment toxicity or impaired community composition, but the relationship of adverse affects to higher concentrations of these metals were less consistent. Based on the biological effects data for benthic invertebrates, a threshold was used to establish a site-specific upper TEL for arsenic. Based on the effects levels from the triad sampling data, the site-specific upper TEL for benthic invertebrates was calculated as 220 mg/kg of arsenic in sediment.

In summary, there was evidence of impairment of benthic invertebrate communities which was associated with high concentrations of arsenic, and to a lesser degree, high concentrations of copper, chromium, mercury, and zinc in reaches 1 and 2 of the Aberjona River Study area. However, the endpoints for impacts on the survival or reproduction of populations of largemouth bass, white sucker, and pumpkinseed showed no evidence for significant ecological risk. Since the major contaminants of concern for invertebrates are inorganics, which do not tend to have high trophic transfer rates, the lack of detectable effects on fish is consistent with the invertebrate results. Also consistent with these results was a lack of potential risk to piscivorus birds (represented by green heron) that feed in the study area.

There was evidence of potential risk of impacts on the survival or reproduction of muskrat due to the exposure to arsenic (primarily from ingestion of plants and sediment) in reaches 1 and 2. There was low risk to mallard populations site-wide from exposure to mercury. A small number of stations contained sediment concentrations of chromium, lead, and arsenic high enough to pose a risk to mallards on a limited spatial scale (maximum case scenarios), but the site-wide models (average case) did not result in a conclusion of probable effects on mallard populations. The risk to shrew populations from exposure to inorganics in sediment and diet was highest for arsenic, however, confidence in this conclusion is low, due to the high uncertainty associated with the estimation of these risks.

Additional surface water data were collected after completion of the draft BERA. These data are qualitatively evaluated in Appendix E.4 to supplement the surface water data set evaluated in the BERA. Among the nine inorganics identified as surface water COPCs in the BERA, seven were detected at concentrations above screening values in the supplemental data set (Appendix E.4), including: barium, cadmium, copper, iron, lead, mercury, and silver. Three of the metals with NAWQC criteria available (cadmium, iron, and lead) were above the NAWQC values in only one of over 100 samples, with the cadmium and lead values occurring during storm events. Copper exceeded the NAWQC in four of the 133 samples, three of which occurred during a storm event. Based on the low magnitude and frequency of these exceedences, the risk to aquatic organisms in the study area from exposure to metals in surface water is low. These conclusions are consistent with those based on the original surface water data, but confidence in the conclusion is increased, since the supplemental data set was larger, based on both storm event and baseflow data, and included dissolved (filtered) metals results.

Additional sediment sampling was also conducted after completion of the draft BERA. These data were evaluated separately and compared to risk conclusions in Appendix E.5. Additional sediment data included 15 samples collected at station AJRW, and nine sediment cores throughout the study area. Based on the data collected at station AJRW, there is no significant ecological risk associated with exposure of receptors exceeding thresholds established in the BERA. If these data had been incorporated into the BERA calculations, there would have been no changes in the conclusions of sitewide risks calculated for heron or mallard, nor would there have been risk associated with exposure to receptors at this station for semi-aquatic mammals (muskrat) or benthic invertebrates.

Sediment core data collected throughout the study area indicated that sediment metals concentrations generally decreased with depth. The metals concentrations in the sediment core data were consistent with sediment data previously collected in the Aberjona River Study Area in support of the BERA. The sediment core sample results for the 0-1' interval were qualitatively assessed (Appendix E.5). Threshold effects levels were exceeded for SC05 through SC08, located in Reach 1, primarily for arsenic, chromium, and mercury. These results are consistent with results presented for nearby Reach 1 sediment stations assessed as part of the BERA.

SECTION 5.0
SUMMARY AND CONCLUSIONS

This section summarizes the findings and conclusions of the field investigation activities and baseline human health and ecological risk assessments conducted for the 3rd operable unit (OU-3) of the Wells G&H Superfund Site, Woburn, Massachusetts (i.e., the Aberjona River Study). The purpose of this report was to assess contamination within the Aberjona River Study area and evaluate human health and ecological risks related to the contamination. The environmental setting, geology, surface hydrology, hydrogeology, nature and extent of contamination, fate and transport, and human health and ecological risk are summarized in the following text.

5.1 SUMMARY

The study area includes a six-mile reach of the nine-mile-long Aberjona River, extending from Route 128 in Woburn to the Lower Mystic Lake in Arlington. The Wells G&H 38-acre wetland south of Route128 is also included as part of the study area. The Aberjona River Valley was historically a base for the leather and tanning industry, and continues to support industrial and commercial businesses. There are also two NPL Superfund Sites in the valley: the Wells G&H Superfund Site, located within the study area boundaries, and the Industri-Plex Site, located immediately upstream of the study area. These sites have been the focus of extensive studies since the 1980s, and an investigation of the impact of these potential source areas on the Aberjona River was required by the Consent Decrees for the sites. This report presents the findings of the field investigations and human health and ecological risk assessment conducted as part of the Aberjona River Study.

5.1.1 Environmental Setting

The Aberjona River originates in wetlands in Reading, Massachusetts. At Route 128, the river is culverted under the highway and flows south. Immediately south of Route128, the Aberjona River is channelized to Olympia Avenue. The river continues to flow south and enters the Wells G&H 38-acre wetland characterized by shallow, slow moving, meandering surface water. Approximately ¼ mile south of Olympia Avenue, and north of Well H, the river appears to split into an east and west channel. Based on aerial photographs, the eastern channel appears to be the main flow channel, and continues to have well-developed meanders. The western channel appears to have a less defined channel and more stagnant conditions. The connection between the main (east) channel and the west channel may be seasonally enhanced, potentially influenced by periodic beaver activity, and more well developed during periods of peak flow. Approximately ¼ mile south of where they split, the east and west channels reconverge west of Well G and north of Salem Street.

Downstream of the confluence, the combined channel is faster moving, and more well defined. South of Salem Street, the river flows through a former cranberry bog containing lateral irrigation channels, and continues south as a well defined channel. Between Montvale Avenue and Winchester Center, the river remains well channelized flowing through a series of small urban bermed impoundments, and approximately 1,125 feet of culvert under the grounds of Winchester High School, and then discharges into Judkins Pond and Mill Pond. The Aberjona River then flows south through mostly open channels before discharging into Upper Mystic Lake.

The river passes through a mix of park land, residential, urban, and light industrial areas as it passes from its headwaters to the Mystic Lakes system. Land use is highly developed along the entire length of the study area. Future land use is not expected to significantly change.

Wildlife habitat within the study area is generally restricted to a narrow urban stream corridor varying from approximately 20 feet to 0.3 miles within a densely developed urban watershed.

Development encroaches to the waters edge at several locations. Habitats along the study area include emergent, scrub/shrub and forested wetlands, fragmented upland forests, sub-mature woodlots, grassy meadows, and maintained park land. The Wells G&H 38-acre wetland in the northern portion of the study area is vegetatively diverse and provides significant wildlife habitat. For the purpose of this report, the study area has been divided into six reaches, defined based on similarity of habitat, species presence or absence, and accessibility. The six reaches are as follows:

- Reach 1 Just south of Rt 128 south to Salem Street including the Wells G&H 38-acre wetland;
- Reach 2 Salem Street south into Winchester to the area where the river crosses Washington Street (including Cranberry Bog Conservation Area);
- Reach 3 Washington Street south to Swanton Street (including Davidson Park);
- Reach 4 Swanton Street south to Mill Pond located in downtown Winchester;
- Reach 5 Mill Pond outlet south to Upper Mystic Lake Inlet; and
- Reach 6 Upper Mystic Lake's upper and lower forebays and the Upper and Lower Mystic Lakes.

5.1.2 Geology, Surface Hydrology, and Hydrogeology

Bedrock underlying the study area is primarily comprised of gabbro-diorite and granite. The depth to bedrock from land surface ranges from zero along the east and west sides of the Aberjona River Valley to approximately 140 feet in the center of the valley beneath the river. The bedrock valley is filled with glacial outwash deposits and recent alluvial sediments. The unconsolidated valley fill deposits consist of interbedded sands, silts, clays and gravels. Glacial till is found primarily in the uplands on either side of the Aberjona River Valley.

The Aberjona River flows from north to south in the valley, and drains an area of approximately 25 square miles. The Wells G&H 38-acre wetland in the northern portion of the study area serves important hydrogeological functions such as flood control and attenuation capacities. Upper Mystic Lake, in the southern portion of the study area, is a 166-acre glacial kettle pond comprised of two shallow northern basins and one southern basin. The river discharges into the northernmost basin, referred to as the upper forebay, and through the second northern basin, the lower forebay, before entering into the southern basin. The lake discharges at the outlet dam on the southern shoreline to Lower Mystic Lake. Surface water elevation in the river ranges from approximately 100 feet in the headwaters area to approximately 13 feet at the entry to Upper Mystic Lake.

Groundwater recharge is largely from precipitation which infiltrates through the bedrock in the upland area. A downward vertical component of flow exists in the uplands of the Aberjona River Valley, and an upward vertical component of flow exists in the vicinity of the wetlands near the river. Therefore, groundwater being recharged in the uplands migrates through bedrock and ultimately discharges into the wetlands and river through the shallow peat layer. Depth to groundwater ranges from zero feet in the wetland areas to 10 to 15 feet in the hilly areas along the east and west boundaries of the study area.

5.1.3 Field Investigation

Field investigations were conducted in 1995 for the collection of surface water, sediment, fish, crayfish, and plant samples. Supplemental sediment sampling was conducted in 1997. In 1999, one location was sampled for surface water and sediment in support of the Industri-Plex Superfund Site Investigation. The 1999 data have been used to supplement the study area surface water and sediment data collected in 1995 and 1997. In 2000, 2001, and 2002, additional sediment and surface soil sampling was conducted throughout the study area. In response to comments provided on the May 2003 draft risk assessment report, the following supplemental data were collected:

- Sediment and floodplain surface soil data collected in 2004 from the Aberjona River south of Bacon Street in Winchester (station AJRW);
- Baseflow and storm event surface water data from six gauging stations (SW-05 through SW-10) along the length of the study area, collected between July 2001 and October 2002; and
- Sediment core data collected in February 2003 from nine locations (SC05 through SC13) along the length of the study area.

Field investigations also included the collection of 2002 groundwater metals data from within the Wells G&H 38-acre wetland. These data were included in a screening-level evaluation (i.e., comparison to benchmarks) to provide preliminary information on the potential impact of sediments contaminated with metals on groundwater quality. The groundwater-sediment interaction will be discussed more fully in the comprehensive Remedial Investigation (RI) Report.

Within the six reaches of the study area, a total of 21 surface water and 53 sediment and/or soil sampling stations were identified. An additional twelve stations were identified from areas outside the influence of the potential source areas to provide reference for the study area with regard to contaminant distribution. Over 80 fish samples were collected throughout the study area and analyzed as either whole body (small or large fish), fillet, or offal samples. In addition, five crayfish samples were collected from reaches 1, 2, 3, and 5, and six plant samples were collected from reach 1.

The 1995 surface water and sediment samples and the 1997 sediment samples were analyzed for VOCs, SVOCs, pesticides, PCBs, and inorganics. In addition, surface water samples were analyzed for TDS and general water quality parameters, and sediment samples were analyzed for TOC or TCO, and AVS/SEM. Sediments were also collected for bioassays with benthic invertebrates. Biota were analyzed for SVOCs, pesticides, PCBs, inorganics, and percent moisture, with fish and crayfish samples also analyzed for percent lipids.

Analytes for the 1999 sampling round included: VOCs, SVOCs, pesticides, PCBs, metals, TOC, grain size, percent moisture, and AVS/SEM analyses for sediment, and VOCs, SVOCs, pesticides, PCBs, dissolved metals, and total metals for surface water.

Following the 1997 sampling round, metals were determined to be the primary contaminants of concern for human health and ecological risk. Therefore, the 2000/2001 sediment and surface soil data, and the 2003 sediment core data were analyzed for inorganics only (metals, cyanide, sulfides (sediments only), and hexavalent chromium (selected sediments only)). The 2001/2002 baseflow and storm event surface water data were analyzed for total metals, dissolved metals, and TSS.

Also, in June 2001, triad sampling was conducted to provide additional information for the ecological risk assessment. Sediment samples were collected at 20 locations for sediment chemistry (VOCs, SVOCs, pesticides, PCBs, metals, TOC, grain size, and AVS/SEM), benthic invertebrate community analysis, and laboratory sediment toxicity testing (10 day and chronic 28 and 42 day testing).

In 2002 and 2004, sediment samples were collected at nine new stations and surface soils were collected at six new stations for metals, PCBs (selected sediments), and TOC analyses.

5.1.4 Nature and Extent of Contamination

The analytical data collected for surface water, sediment, crayfish/fish, plants, soil, and groundwater from 1995 through 2004 are summarized below.

Surface Water. At the 21 study area surface water stations sampled in 1995, no SVOCs or PCBs were detected, and only four chlorinated VOCs at concentrations below 5 Wg/L and five pesticides below 0.015 Wg/L, were measured. Elevated concentrations of inorganics, in comparison to reference samples, were detected throughout the study area, with the sample from station 10 (Aberjona River near Salem Street Bridge) containing a number of the maximum detected concentrations. Inorganics present at elevated concentrations include antimony, arsenic, beryllium, cadmium, chromium, copper, lead, mercury, and zinc.

Baseflow and storm event surface water data were collected in 2001/2002 from six surface water gauging stations throughout the study area. The 1995 and 2001/2002 data sets display overall consistent results for total metals concentrations. Surface water gauging station 05 by the Salem Street Bridge in Woburn displayed the highest maximum baseflow concentration of arsenic. Storm event data indicated slightly elevated average concentrations in comparison to baseflow samples for aluminum, antimony, arsenic, cadmium, chromium, cobalt, copper, iron, lead, mercury, nickel, vanadium, and zinc. In general, storm event samples collected from surface water gauging station 08 by the USGS gauging station in Winchester displayed the highest maximum storm event concentrations of metals.

Sediment. At the up to 50 study area stations sampled in 1995, 1997, 2000/2001, 2002, and 2004, chlorinated and petroleum-related VOCs (benzene, ethylbenzene, toluene, and xylenes) were detected in sediments within study area primarily from samples collected in the Wells G&H 38-acre wetland. SVOCs were detected at numerous locations, with most compounds identified as PAHs. Davidson Park (station 07) in Winchester contained the maximum levels for the majority of the PAHs detected in the sediments. Pesticides were frequently detected throughout the study area. Aroclor-1248 and -1260 were also widely detected across the study area at levels ranging from 8.1 Wg/kg to 560 Wg/kg.

Numerous inorganics were detected in sediments, many at levels greater than the reference sample levels. Arsenic, a historical contaminant in the study area, was ubiquitously present with a maximum level of 4,550 mg/kg (station 12; Aberjona River near Well H). Other inorganics detected at significantly elevated levels included antimony, cadmium, chromium, copper, lead, mercury, and zinc. Generally, the highest concentrations of metals were detected in the vicinity of the Wells G&H 38-acre wetland.

Thirty-six sediment core samples were collected including four samples at each of nine locations from throughout the study area. The four samples collected at each location included one sample from each of the following depth intervals: 0 - 1 foot, 1 - 2 foot, 2 - 3 foot, and 3 - 4 foot. Average metal concentrations decreased with increasing depth, except at SC05, which displayed higher arsenic concentrations with increasing depth, and SC11 and SC12, which had higher lead concentrations in deeper intervals. Samples collected from the Wells G&H 38-acre wetland displayed the highest concentrations of arsenic, antimony, barium, beryllium, cadmium, cobalt, copper, iron, manganese, mercury, nickel, and zinc.

Surface Soil. Metals were measured in sixty-five surface soil samples collected at Normac Road, the Cranberry Bog (CB-05), Salem Street (WSS), Danielson Park, Davidson Park, Kraft Foods, and the area south of Bacon Street in Winchester (designated AJRW). Generally, metal concentrations were lower in soils than in nearby sediment samples.

Biota. For the fish and crayfish samples collected in 1995, SVOCs were either not detected or only infrequently detected at low levels. In contrast, pesticides, PCBs (Aroclor-1254 and -1260) and inorganics were frequently detected. Maximum detected levels of pesticides ranged up to 820 Wg/kg (4,4'-DDE), and maximum PCB levels ranged up to 1,100 Wg/kg (Aroclor-1260). Plants within the study area contained over 20 organic compounds including PAHs, pesticides, and Aroclor-1260, as well as elevated levels of a number of inorganics.

Groundwater. Groundwater sampling data were collected from thirteen monitoring wells within the Wells G&H 38-acre wetland. Shallow overburden data were available from all thirteen monitoring well locations. Medium and deep overburden data were available from six of the thirteen monitoring well locations. In order to evaluate the potential impact of sediments contaminated with metals on groundwater quality, a screening-level evaluation was conducted which consisted of a comparison of groundwater metals data to USEPA primary and secondary Maximum Contaminant Levels (MCLs; USEPA, 2003). Groundwater is not currently used as a source of potable water. Only groundwater metals data were included in the screening because metals are the primary contaminants within the 38-acre wetland sediments. Other less significant contaminants present within the wetland sediments (e.g., volatile organic compounds) are likely the result of impacts from the Wells G&H source area properties (OU-1) and, if necessary, will be addressed by the remediation of the source areas (OU-1) and the central area (OU-2) of the Wells G&H Superfund Site.

Shallow overburden arsenic concentrations in eight of the thirteen monitoring wells exceeded the arsenic MCL (10 Wg/L). No other exceedances of primary MCLs were noted in shallow overburden wells. No exceedances of primary MCLs were noted in the medium and deep overburden wells included in the sampling program. Secondary MCLs for aluminum, iron, and manganese were exceeded in shallow, medium, and deep overburden groundwater. Manganese secondary MCL exceedances occurred most frequently.

Overall, there appears to be a decreasing trend of groundwater metals concentrations with increasing depth of groundwater. This trend is most noticeable with arsenic, suggesting a possible impact of sediment arsenic contamination on groundwater quality. The groundwater-sediment interaction for metals will be discussed more fully in the comprehensive RI Report.

5.1.5 Contaminant Fate and Transport

Potential migration pathways at the study area include migration of contaminants from potential source areas to environmental media within the study area, migration of contaminants into and within surface waters, migration of contaminants into air, and migration of contaminants from sediments during high flow and/or storm events. The low concentrations of VOCs detected in the surface waters and sediments indicate that migration of contaminants into air would be relatively insignificant for the study area.

Migration of contaminants from source areas into surface waters within the study area most likely occurs through stormwater surface runoff and to some extent groundwater discharge. The groundwater discharge pathway is likely for highly soluble constituents. Dissolved constituents as well as fine particulates with associated contaminants can enter the surface water system through the surface runoff pathway. Upon entering the Aberjona River, contaminants will migrate with the local flow.

The migration of contaminants within the surface waters will be a principal environmental fate and transport mechanism for the study area. Surface water transport can occur in two ways: through transport of dissolved components and through transport of contaminated material in the sediment load. The sediment load migration pathway is especially important for constituents with moderate to low aqueous solubility and high affinity for sorption to fine particulates (e.g., PAHs, pesticides, PCBs, and inorganics). Soluble inorganic species, and to a lesser extent VOCs, can be transported as dissolved components.

5.1.6 Baseline Human Health Risk Assessment

The baseline human health risk assessment was performed to evaluate the potential for adverse health effects to human populations who may come into contact with contaminants present in environmental media within the study area. Exposures were evaluated for the following media: baseflow and storm event surface water, sediment, floodplain surface soil, fish fillet tissue, and sediment cores. Chemicals of Potential Concern (COPCs) for the environmental media were identified for the thirty-seven exposure areas of concern that were quantitatively evaluated under current and/or future land-use conditions: stations NR, 14, 22/TT-22, 13/TT-27, WH, NT-1, NT-2, NT-3, WG, WW, JY, WS/WSS, TT-30, TT-31, CB-01 through CB-07, 16/TT-33, 09, DA, AM, KF, 08, 07/DP, LP, AS, 06, 05, AJRW, 04, 03, 02, and 01. For fish fillet tissue, COPCs were identified for the four reaches within the study area where fillet samples had been collected (reaches 3, 4, 5, and 6).

Prior to completion of the risk assessment, an arsenic bioavailability study was performed to assist in the quantification of sediment risks. This site-specific bioassay determined that arsenic is absorbed less efficiently from sediment than from a water medium. The most conservative relative bioavailability estimate determined in the study was used to quantify sediment ingestion risks at the study area. In addition, site-specific hexavalent chromium data for sediments were collected and used in the risk assessment to more accurately characterize sediment risks at the study area.

Possible recreational human exposure to the selected COPCs was characterized through exposure pathways for current and future land use. Under current land use, stations WS/WSS and CB-01 through CB-06 were evaluated as having the highest exposure potential (based on their proximity to nearby residences), stations 16/TT-33, DA, 09, 08, 07/DP, AJRW, and 06 through 01 as having "typical" exposure potential for recreational areas, and stations NR, 14, 22/TT-22, WH, WG, CB-07, TT-30, AM, and AS as having the lowest exposure potential (partially isolated or in industrial locations). It was assumed that future land use would be the same as current land use except for stations NR, 14, 22/TT-22, WG, WH, and AS, the exposure potential of which was assumed to increase in the future as the area near these stations becomes more developed. In addition, stations 13/TT-27, JY, WW, and TT-31 are evaluated under a future land use scenario assuming that the physical barriers limiting current access are removed. Stations NT-1, NT-2, and NT-3 are also evaluated under a future land use scenario due to potential redevelopment plans of the City of Woburn within the Wells G&H wetland that may include the construction of a nature trail (station NT-3) with a possible boardwalk (station NT-1) or pier (station NT-2) extending out into the wetland. Other stations of the study area have not been quantitatively evaluated in the human health risk assessment because they are difficult to access due to the presence of dense vegetation, deep surface water and/or their distance from the shoreline.

Potential exposures to COPCs in sediment core samples collected from nine locations throughout the study area were evaluated to determine risk to workers during future flood mitigation projects involving dredging of contaminated sediments. Exposures to residents in the vicinity of stations WS/WSS, CB-05, KF, 07/DP, and AJRW were also evaluated, under the assumption that periodic flooding in these areas has resulting in the migration of contamination to residential yards. Residential exposures to COPCs were conservatively evaluated using storm event surface water data and flood plain surface soil data.

Whenever possible, Upper Confidence Limits (UCLs) on the mean concentration, calculated using USEPA's software ProUCL version 3.0, were used as exposure point concentrations. Exposures to surface water, sediment, and floodplain surface soil were quantitatively evaluated by exposure area (i.e., stations) while exposures to fish fillet tissue were evaluated by river reach (reaches 3 through 6). Only those sediment samples collected from below two feet or less of standing water were used in the quantitative evaluation. Worker exposures to sediment cores were evaluated by sampling location under the assumption that dredging would occur at individual areas along the river and would result in worker exposures to all sediment depth intervals at that location. Residential exposures to contaminants that may have migrated from the river to yards during flooding events were evaluated using floodplain surface soil, by station, and storm event surface water data, by gauging station.

Two sets of quantitative exposure estimates were prepared, corresponding to sets of exposure assumptions designated as central tendency (CT) and reasonable maximum exposure (RME) scenarios. The potential for adverse health effects was evaluated by comparing the calculated incremental lifetime cancer risks (ILCRs) to the target risk range of 10^{-6} to 10^{-4} and the calculated hazard indices (HIs) for noncarcinogenic effects to the target risk level of 1. Table 5-1 presents a study area risk summary for recreational exposures, with HIs and ILCRs summarized by station for surface water, sediment and surface soil, and by reach for fish fillet tissue, for each of the receptors evaluated. Risks presented in Table 5-1 are based on 1995 surface water data. Table 5-2 provides a comparison of surface water risks, by station, for the 1995 surface water and 2001/2002 baseflow surface water data.

Table 5-3 provides a summary of worker risks associated with dredging exposures that may occur during flood mitigation activities. Risks are presented by sediment core sampling location. Table 5-4 provides a summary of potential risk, based on residential exposures to floodplain surface soil at stations WS/WSS, CB-05, KF, 07/DP, and AJRW. Risks to residents associated with surface water exposures during flooding events are also included on Table 5-4.

When risks were estimated for a child and adult receptor (Tables 5-1, 5-2, and 5-4), the child HIs are presented as the most conservative, while ILCRs presented are the sum of the child and adult risks (i.e., a total receptor cancer risk). Surface water, sediment, and surface soil recreational risks, presented by station, have been summed together under the assumption that each receptor is exposed to all three media during recreational activities. Fish fillet ingestion risks, presented by reach, are summarized separately. In cases where the total pathway HI exceeded 1, COPCs having similar systemic effects were summed for each pathway and medium. Table 5-1 also summarizes the primary risk contributors for those receptors with estimated ILCRs greater than the target range of 10^{-6} to 10^{-4} and target organ-specific HIs greater than 1.

Study area media for which all evaluated pathways were within or below USEPA's target risk range for carcinogens (ILCR of 10^{-6} to 10^{-4}) and risk criterion for noncarcinogens (HI of 1) are:

- surface water (at all stations quantitatively evaluated)
- floodplain surface soil (at stations NR, WS/WSS, CB-05, DA, KF, 07/DP, and AJRW)

- sediment (at stations NR, 14, 22/TT-22, WG, WW, JY, WS/WSS, TT-30, TT-31, CB-01, CB-02, CB-04, CB-05, CB-06, CB-07, 16/TT-33, 09, AM, KF, 08, 07/DP, LP, AS, 06, 05, AJRW, 04, 03, 02, and 01)
- sediments cores (at locations SC07, SC09, SC10, SC11, SC12, and SC13)
- fish fillet tissue (at reaches 3, 4, 5, and 6)

The media for which RME exposures for one or more pathways were above USEPA's target risk range for carcinogens and/or noncarcinogens are:

- sediment (current use at stations WH and CB-03)
- sediment (future use at stations 13/TT-27, WH, NT-1, NT-2, NT-3, and CB-03)
- sediment cores (future exposures at locations SC05, SC06, and SC08)

The contaminants responsible for driving risk above the target risk ranges in sediment and/or surface soil under current and/or future use are:

- arsenic (at stations 13/TT-27, WH, NT-1, NT-1, NT-3, and CB-03 and sediment core locations SC05, SC06, and SC08)
- benzo(a)pyrene (at stations 13/TT-27, WH, NT-1, and NT-2)

It should be noted that arsenic was present in all sediment samples collected from the ten reference stations. However, detected levels at the reference stations ranged from 3.8 mg/kg to 44.5 mg/kg (see Appendix C.5; Table C.5–2.5) compared to a range at station 13/TT-27 of 15.9 mg/kg to 4,210 mg/kg (Table 3-2.2.4), station WH of 4.7 mg/kg to 3,230 mg/kg (Table 3-2.2.5), station NT-1 of 6.6 mg/kg to 4,250 mg/kg (Table 3-2.2.6), station NT-2 of 6.6 mg/kg to 3,230 mg/kg (Table 3-2.2.7), station NT-3 of 6.6 mg/kg to 3,230 mg/kg (Table 3-2.2.8) and station CB-03 of 9.1 mg/kg to 1,410 mg/kg (Table 3-2.2.17).

It should further be noted that the sediment RME EPC for arsenic at some of these stations is uncertain due to one or a small number of arsenic detects compared to the remainder of the data set. This uncertainty is specifically applicable to stations 13/TT-27 (samples SD-13-01-FW and SD-13-02-FW; 4210 mg/kg and 2480 mg/kg, respectively), WH (sample SD-12-01-ME; 3230 mg/kg), and CB-03 (sample CB-03-11; 1410 mg/kg).

Benzo(a)pyrene was detected in 12 of 16 sediment samples collected from the reference stations. Detected levels at the reference stations ranged from 0.13 mg/kg to 5.5 mg/kg (see Appendix C.3; Table C.3–2.5) compared to 1 mg/kg at station WH (Table 3-2.2.5) and a range at station 13/TT-27 of 0.15 mg/kg to 1.7 mg/kg (Table 3-2.2.4), station NT-1 of 0.16 mg/kg to 1 mg/kg (Table 3-2.2.6), and station NT-2 of 0.16 mg/kg to 1 mg/kg (Table 3-2.2.7). The benzo(a)pyrene sediment concentrations at these study area stations fall within the range of concentrations detected at the reference stations.

Lead in sediment, sediment cores, floodplain surface soil, and surface water was evaluated through the use of recommended USEPA models. The evaluation indicated that exposures to lead under both current and future land-use conditions were not estimated to result in adult or childhood blood lead levels in excess of blood lead level goals. Average lead concentrations in fish fillet tissue are consistent with the published average background lead level. Therefore, lead was determined not to be of concern for human receptors at the study area.

5.1.7 Baseline Ecological Risk Assessment

The baseline ecological risk assessment was performed to evaluate the potential for contaminants in surface water, sediment, and biota to impact ecological receptor populations present within the Aberjona River Study area. Study area COPCs, 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.

Maximum concentrations of analytes detected in the surface waters of Upper Mystic Lake, the main channel of the Aberjona River, and the Aberjona River wetlands were each compared to USEPA freshwater chronic National Ambient Water Quality Criteria (NAWQC). Among the nine inorganics identified as surface water COPCs, seven were detected at concentrations above the NAWQC, including aluminum, cadmium, copper, iron, lead, mercury, and zinc. The NAWQC for aluminum, iron, and lead were also exceeded at a majority of the 10 reference locations. These exceedences of NAWQCs were based upon limited sampling rounds, consequently, exceedence of an NAWQC, alone, was not concluded as necessarily representing ecological risk.

After completion of the Baseline Ecological Risk Assessment (BERA), additional surface water data were collected to address the fate and transport of contaminants through the Aberjona River Study area. The data collected within the Aberjona River Study area (South of Route 128) are qualitatively evaluated in this Appendix E.4 to supplement the surface water data set evaluated in the BERA. Among the nine inorganics identified as surface water COPCs in the BERA, seven were detected at concentrations above screening values in the supplemental data set (Appendix E.4), including: barium, cadmium, copper, iron, lead, mercury, and silver. Three of the metals with NAWQC criteria available (cadmium, iron, and lead) were above the NAWQC values in only one of over 100 samples, with the cadmium and lead values occurring during storm events. Copper exceeded the NAWQC in four of the 133 samples, three of which occurred during a storm event. Based on the low magnitude and frequency of these exceedences, the risk to aquatic organisms in the study area from exposure to metals in surface water is low. These conclusions are consistent with those based on the original surface water data, but confidence in the conclusion is increased, since the supplemental data set was larger, based on both storm event and baseflow data, and included dissolved (filtered) metals results.

There was no individual endpoint for exposures to any organisms to surface water, alone, as part of the BERA. However, each of the inorganics exceeding NAWQCs were evaluated in the exposure calculations for mammalian and avian indicator species as part of the total exposure to the COPC.

To select sediment COPCs, maximum detected levels of chemicals in sediment samples were compared to sediment quality criteria. Over 90 analytes were detected in study area sediments, including a number of VOCs, SVOCs, pesticides/PCBs, and inorganics. All sediment contaminants that exceeded conservative ecological screening benchmarks, and were detected in more than 5% of the samples, were selected as COPCs to be further evaluated in the risk assessment. Background concentrations and concentration at reference locations were not used to screen COPCs. The screening process identified 58 COPCs in sediment. Among these were 5 VOCs, 19 SVOCs, 14 pesticides/PCBs, and 20 inorganics. Inorganic COPCs selected in surface water were a subset of those selected in sediment.

Indicators species selected for the evaluation included muskrat, green heron, mallard, and short-tailed shrew. These species were selected to represent aquatic mammals, piscivorus birds, waterfowl, and small terrestrial mammals, respectively. Additional indicators species/communities selected for the evaluation included largemouth bass (to represent predatory fish), pumpkinseed (to represent small foraging fish), white sucker (to represent bottom-feeding fish), and the benthic invertebrate community.

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. Fish populations were evaluated by reach, and heron and mallard, which have the largest foraging areas, were assessed on a site-wide scale.

For mammalian and avian receptor species, the potential for risk was evaluated using the Hazard Quotient (HQ) approach, whereby estimates of total exposure from the ingestion of food, water and sediment were divided by toxicity reference values (TRVs) and compared to a target HQ of 1. Both average case and maximum case scenarios were evaluated for heron, mallard, and shrew exposure models. Maximum case exposures were based, whenever possible, on UCLs, calculated using USEPA's software program, ProUCL version 3.0.

For fish, risks were evaluated by comparing average COPC concentrations in fish tissue in each reach to both reference samples and tissue residue benchmarks obtained from the literature. Risks to the benthic invertebrate community were evaluated via laboratory toxicity tests, benthic invertebrate community analysis, comparison of COPC concentrations in sediment to sediment benchmarks, and comparison of COPC concentrations in crayfish tissue to both reference samples and tissue residue benchmarks.

A summary of ecological receptor risks is presented in Table 5-5. There were no indications of significant ecological risk associated with VOC, SVOC, and pesticide/PCB contamination to any ecological receptor within the Aberjona River Study area. Evaluations of tissue residues indicated there was no evidence for impacts of inorganic COPCs on fish represented by receptor species. Based on exposure modeling results, using site-specific fish and invertebrate tissue data and no observed adverse effects level (NOAEL) TRVs, the risk to survival or reproduction of green heron, or other semi-aquatic predatory birds, was negligible.

Among the other selected ecological receptors, risk were identified at depositional locations within the Aberjona river/wetland system, primarily in reaches 1 and 2. Risks were identified for muskrat, mallard, shrew, and the benthic invertebrate community in the ecological risk assessment for the following reasons:

- The highest risks to muskrats, based on lowest observed adverse effects level (LOAEL) TRVs (Table 4-271), were for arsenic in reach 1 and reach 2. Sediment arsenic concentrations exceeded the upper TEL (150 mg/kg-d) at 14 of 19 muskrat stations in reach 1 and four stations (CB-03, TT-30, TT-32, and TT-33) in reach 2. These results indicated a likelihood of potential impacts on survival or reproduction of mammal populations such as muskrat exposed to arsenic in the diet while foraging in the Aberjona River Study area.

- Using LOAEL TRVs, for muskrat, HQs above 1 for the muskrat receptor occurred for arsenic, and also chromium (reach 1), copper (reach1), lead, and mercury. The results indicated that the risk from exposure to chromium, copper, lead, and mercury is high (above upper TEL) only at a few stations, and based on the endpoint criterion, exposure to chromium, copper, lead, and mercury presents a low risk (above lower TEL) in reach 1.

- Using NOAEL TRVs for mallard (Table 4-197), HQs above 1 for the mallard receptor occurred for chromium, lead, and mercury on a site-wide basis and within the Wells G&H 38-acre wetland. These higher HQs were a result of high sediment concentrations of these metals in reaches 1 and 2. However, because mallard LOAEL HQs were less than 1 for mercury and lead, risk to this receptor was low for these metals. Based on LOAEL TRVs, the only HQ above 1 was for the UCL model for chromium in reach 1. The upper TEL calculated for chromium in sediment for mallard was 2,200 mg/kg total chromium in sediment. Sediment concentrations at five stations within mallard habitat exceeded this upper TEL value site-wide, at stations 21, TT-28, TT-29, and WW in reach 1 and station TT-30 in reach 2. Although chromium may present a risk at individual stations above the upper TEL, because of the limited areal extent of these concentrations, the risk to waterfowl such as mallard on a site-wide basis is low.

- The evaluation of average case HQs for shrew indicated that arsenic concentrations exceeded the LOAEL TRVs in reaches 1 and 2 (Table 4-273). Three stations (stations 20 and BW in reach 1, and CB-03 in reach 2) had sediment arsenic concentrations which exceeded the upper TEL value of 200 mg/kg arsenic in sediment. Since the number of stations exceeding the upper TEL is small, site-wide risk to small terrestrial mammals (represented by the shrew), due to exposure to sediment arsenic, is low.

- For the benthic invertebrate endpoint, comparison of sediment metals concentration to effects-based sediment benchmarks (SELs) indicated potential effects on benthic communities from arsenic, chromium, copper, lead, mercury, and zinc (Table 4-261), with the highest HQs observed in reach 1. The results of the toxicity testing and benthic invertebrate community analysis indicted that stations with evidence of sediment toxicity (mainly reduced growth of *Hyalella azteca* or *Chironomus tentans*) also had benthic invertebrate communities with evidence of impaired community structure (impairment is based on comparison to reference samples for several ecological indices based on organism abundance, species richness, or dominance of tolerant organisms). The weight of evidence from a combination of sediment chemistry compared to ecological benchmarks, benthic invertebrate community analysis, and toxicity testing (triad study) supports the conclusion that there are adverse ecological effects occurring at the benthic invertebrate community-level associated with high concentrations of metals in the sediment. The areas of impaired benthic invertebrate communities most closely corresponded to areas of high concentrations of arsenic, and to a lesser extent with high chromium, copper, mercury, and zinc in sediment. Based on the site-specific toxicity information, the concentration 220 mg/kg arsenic consistently represented an upper threshold effects level for sediment. None of the stations with arsenic concentrations less than or equal to 44.5 mg/kg showed indications of toxicity to invertebrates, representing the lower threshold effects level for sediment. There were 10 stations in reach 1 (10, 11, 12, 13, 18, 19, 20, TT-28, TT-29, and BW), three stations in reach 2 (CB-03, TT-30, and TT-33), and two stations in reach 6 (4 and UM) that exceeded the upper TEL for arsenic. Based on the limited number of stations in reach 6 (2 of 8 stations) above the upper TEL, the risk to benthic invertebrates in this reach is low.

Toxicity testing results showed reductions in growth of laboratory test organisms that closely corresponded to the concentration of arsenic in sediments. Indications of reduced growth or survival of laboratory test organisms were associated with indicators of community impairment for all stations tested with sediment arsenic concentrations above 220 mg/kg. Although the concentration of a number of metals (including copper and zinc) co-varied with the elevated concentration of arsenic at the triad sampling sites, both the toxicity results and the impairment in the community structure were most consistently associated with arsenic. AVS/SEM data, normalized for organic carbon concentration of the sediments, also supported the conclusion that divalent metals, including those most closely correlated with arsenic concentrations (copper and zinc), were unlikely to contribute to impacts on benthic communities.

Additional sediment sampling was conducted after the completion of the BERA. These data were evaluated separately and compared to risk conclusions in Appendx E.5. Additional sediment data included 15 samples collected at station AJRW, and nine sediment cores throughout the study area. Based on the data collected at station AJRW, there is no significant ecological risk associated with exposure of receptors exceeding thresholds established in the BERA. If these data had been incorporated into the BERA calculations, there would have been no changes in the conclusions of sitewide risks calculated for heron or mallard, nor would there have been risk associated with exposure to receptors at this station for semi-aquatic mammals (muskrat) or benthic invertebrates.

Sediment core data collected throughout the study area indicated that sediment metals concentrations generally decreased with depth. The metals concentrations in the sediment core data were consistent with sediment data previously collected in the Aberjona River Study Area in support of the BERA. The sediment core sample results for the 0-1' interval were qualitatively assessed (Appendix E.5). Threshold effects levels were exceeded for SC05 through SC08, located in Reach 1, primarily for arsenic, chromium, and mercury. These results are consistent with results presented for nearby Reach 1 sediment stations assessed as part of the BERA.

5.2 CONCLUSIONS

Conclusions for the human health risk assessment and the ecological risk assessment are presented below. As indicated in the Executive Summary and Introduction sections, as well as publicized in USEPA's previous Spring and June 2002 Fact Sheet, USEPA Region I intends to expand this risk assessment to include environmental data collected immediately upstream of the study area along the Halls Brook Holding Area (HBHA). The final risk assessment will be included in a comprehensive Remedial Investigation (RI) report documenting all the data collected along the Aberjona River and Halls Brook Holding Area from North Woburn to the Mystic Lakes and further explain the nature and extent of contaminants and their fate and transport mechanisms. The groundwater-sediment interaction for metals will also be discussed more fully in the comprehensive RI Report. The comprehensive RI is expected to be completed in the Fall of 2004.

5.2.1 Human Health

Under current and/or future use conditions, carcinogenic and/or noncarcinogenic risk estimates exceed target risk levels for recreational exposure to sediments at stations 13/TT-27, WH, NT-1, NT-2, NT-3, and CB-03, and for dredging exposures at sediment core locations SC05, SC06, and SC08. All stations noted are located within the Wells G&H 38-acre wetland, north of Salem

Street in Woburn, with the exception of CB-03 which is located in the former cranberry bog immediately south of Salem Street. The risk exceedances were due primarily to the presence of arsenic. The sediment EPC for arsenic at stations 13/TT-27, WH, and CB-03 is uncertain due to one or a small number of arsenic detects compared to the remainder of the data set. Benzo(a)pyrene was a minor risk contributor at stations 13/TT-27, WH, NT-1, and NT-2. The risk associated with benzo(a)pyrene at these stations was between 2×10^{-6} and 3×10^{-6}. The benzo(a)pyrene sediment concentrations at these study area stations fall within the range of concentrations detected at the reference stations.

Estimated risks are within or below the target risk levels for exposure to all media at stations NR, 14, 22/TT-22, WG, WW, JY, WS/WSS, TT-30, TT-31, CB-01, CB-02, CB-04, CB-05, CB-06, CB-07, 16/TT-33, 09, DA, AM, KF, 08, 07/DP, LP, AS, 06, 05, AJRW, 04, 03, 02, and 01. Estimated risks are also within or below the target risk level for worker exposures to sediment cores during possible future dredging activities at locations SC07, and SC09 through SC13, and for residents along the river whose properties may have been or may in the future be impacted by periodic flooding of the river.

5.2.2 Ecological Risk

The assessment produced evidence of impaired benthic invertebrate communities which was associated with high concentrations of sediment arsenic, and to a lesser degree, high concentrations of sediment copper and zinc in reaches 1 and 2 of the Aberjona River Study area. Within reach 1 (stations 10, 12, 13, 19, and TT-29) and reach 2 (stations TT-30, TT-32, and TT-33), there was evidence of concentrations exceeding ecological benchmarks, benthic invertebrate community impairment, and sediment toxicity. Station 18 had strong evidence for community impairment, but no evidence of toxicity. Each of these nine stations in reaches 1 or 2 had sediment arsenic concentrations greater than 220 mg/kg, which provided field verification of its use as an upper threshold effects level for benthic invertebrate community impairment. Other stations within reaches 1 or 2, which were not used for the triad study, but also had sediment arsenic concentrations above this threshold effects level for invertebrates included stations 11, 20, BW, TT-28 in reach 1, and CB-03 in reach 2.

There was no evidence of significant ecological risk to the survival or reproduction of largemouth bass, white sucker, or pumpkinseed. Since the contaminants of concern for invertebrates were inorganics, which do not tend to transfer up the food chain (have low trophic transfer rates), the lack of significant risk to fish was consistent with the results for the benthic invertebrate community. Also consistent with this finding was a lack of potential risk to piscivorus birds (*e.g.*, green heron) that feed on small fish in the study area.

The assessment provided evidence of potential risk to the survival or reproduction of muskrat due to the exposure to arsenic (primarily from ingestion of plants and sediment) in reaches 1 and 2. There was low risk to muskrat, mainly in reach 1, from exposure to chromium, copper, mercury, and lead. There was low risk to mallard site-wide from exposure to chromium, lead, and mercury, which resulted from high sediment concentrations of these metals in reaches 1 and 2. A small number of stations contained sediment concentrations of chromium, lead, and mercury high enough to pose a risk to muskrat and mallards on a limited spatial scale above the upper TEL. The risk to shrew from exposure to inorganics in sediment and diet was highest for arsenic, however based on the limited number of stations above an upper threshold effects level of 220 mg/kg arsenic, the risk to small mammals such as shrews is low.

SECTION 6.0

REFERENCES

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Aurillio, A.C., R.P. Mason, and H.F. Hemond. 1994. Speciation and fate of arsenic in three lakes of the Aberjona Watershed. *Environ. Sci. Technol.* 28: 577-585.

City of Woburn. 2001. (Personal Communication with William Campbell, Town Clerk, Town of Woburn , October 2001.

Department of Environmental Quality Engineering (DEQE). 1975. *Upper Mystic Lake water quality study, April 1974-April 1975. Report #8470-75-100-10-75-CR.* Division of Water Pollution Control, Water Quality Section. September 1975.

Department of Environmental Quality Engineering (DEQE). 1982. *Upper Mystic Lake watershed urban runoff project main report.* Office of Planning and Program Management. October 1982.

Durant, J., J.J. Zemach, and H.F. Hemond. 1990. The history of leather industry waste contamination in the Aberjona Watershed: A mass balance approach. *Civil Engineering Practice* 5: 41-66.

Durant, J. 1991. *Industrial history, mutagenicity, and hydrologic transport of pollutants in the Aberjona watershed.* Master Thesis, MIT. February 1991.

Ebasco Corporation (Ebasco). 1988a. *Final supplemental remedial investigation for feasibility study, Wells G&H Site, Woburn, Massachusetts.* Report prepared for the U.S. Environmental Protection Agency. Contract No. 68-01-7250. December 1988.

Ebasco Corporation (Ebasco). 1988b. *Draft final risk assessment for the Wells G&H Site, Woburn, Massachusetts.* Report prepared for the U.S. Environmental Protection Agency. Contract No. 68-01-7250. December 1988.

Foster Wheeler Environmental Corporation (Foster Wheeler). 1996. *Preliminary data compendium, Wells G&H RI/FS, Operable Unit III, Aberjona River Study, sampling period July 31, 1995 to September 12, 1995.* Report prepared for the U.S. Environmental Protection Agency. Contract No. 68-W9-0034.

Hemond, H.F. 1995. Movement and distribution of arsenic in the Aberjona watershed. *Environ. Health Perspect.* 103 (Suppl. 1): 35-40.

Massachusetts Department of Environmental Protection (MADEP). 1996 and 1997. Code of Massachusetts Regulations, Title 314. Massachusetts Surface Water Quality Standards. Division of Water Pollution Control. Sections 4.02 and 4.053b. 12/27/96 and 5/30/97.

Massachusetts Department of Environmental Protection (MADEP). 2004. MADEP Groundwater Use and Value Determination, Wells G&H Superfund Site (operable unit 2), Woburn, Massachusetts. June 21, 2004.

Planning Research Corporation (PRC). 1986. *Wells G&H wetlands assessment final report.* Prepared for the U.S. Environmental Protection Agency. Contract No. 68-01-7037. March 1986.

Spliethoff, H.M. and H.F. Hemond. 1996. History of toxic metal discharge to surface water of Aberjona watershed. *Environ. Sci. Technol.* 30 (1): 121-128.

Tarr, J.A. 1987. History of Pollution in Woburn, Massachusetts. In: *Review of EPA Report "Wells G&H Site Remedial Investigation Report, Part I, Woburn, Massachusetts" Vol. 2.* GeoTrans Inc. Boxborough, Massachusetts.

Town of Winchester. 2001. (Personal Communication with Kelly, Davis, Town Clerk's Office), October 2001

U.S. Environmental Protection Agency (USEPA). 1988. *Site analysis: Wells G&H, Woburn, Massachusetts*. TS-PIC-8808S. November, 1988.

U.S. Environmental Protection Agency (USEPA). 1989. *Wells G&H Superfund Site record of decision*. USEPA Region I. Boston, Massachusetts. September 14, 1989.

U.S. Geological Survey (USGS). 1987. *Area of influence and zone of contribution to Superfund Site Wells G and H, Woburn, Massachusetts*. Water-Resources Investigations Report 87-4100. Boston, Massachusetts.

U.S. Geological Survey (USGS). 1990. *Water resources data for Massachusetts and Rhode Island, water year 1989*. USGS Water Data Report MA-RI-89-1.

SECTION 2.0 REFERENCES

Ebasco Corporation (Ebasco). 1995. *Operable Unit III final field operations plan, Aberjona River Study, Wells G&H Superfund Site, Woburn, Massachusetts*. Prepared for U.S. Environmental Protection Agency. Contract No. 68-W9-0034.

Foster Wheeler Environmental Corporation (Foster Wheeler). 1996. *Preliminary data compendium, Wells G&H RI/FS, Operable Unit III, Aberjona River Study, sampling period July 31, 1995 to September 12, 1995.* Prepared for the U.S. Environmental Protection Agency. Contract No. 68-W9-0034.

Lyman, W.J., W.F. Reehl and D.H. Rosenblatt. 1982. *Handbook of chemical property estimation methods: Environmental behavior of organic compounds.* McGraw-Hill Book Co. New York, NY.

Menzie Cura. (1999) *Toxicological Surface Water Sediments Sampling and Fish Sampling Work Plan and Quality Assurance Project Plan for the Industri-Plex Site, Woburn Massachusetts.* July 1999.

Metcalf and Eddy, Inc. (M&E). 1997. *Final quality assurance project plan for Wells G&H Superfund Site, Aberjona River Study (Operable Unit 3), Appendix B: DAS specifications.* Prepared for the U.S. Environmental Protection Agency. Contract No. 68-W6-0042. October 1997.

Metcalf and Eddy, Inc. (M&E). 1998. *Supplemental data compendium for Wells G&H Superfund Site, Aberjona River Study (Operable Unit 3), Woburn, Massachusetts.* Prepared for the U.S. Environmental Protection Agency. Contract No. 68-W6-0042. May 1998.

TRC. 2002a. *Field Sampling Plan for Pre-Design Investigation and Site Characterization. Wells G&H Superfund Site, Operable Unit 1, Olympia Property, Woburn, Massachusetts.* Prepared for the U.S. Environmental Protection Agency. Prepared by: Metcalf & Eddy, Inc., Team Subcontractor: TRC Environmental Corporation. Contract No. 68-W6-0042. February 2002.

TRC. 2002b. *Data Trend Evaluation. Wells G&H Superfund Site, Operable Unit 1, Olympia Property, Woburn, Massachusetts.* Prepared for the U.S. Environmental Protection Agency. Prepared by: Metcalf & Eddy, Inc., Team Subcontractor: TRC Environmental Corporation. Contract No. 68-W6-0042. November 2002.

U.S. Environmental Protection Agency (USEPA). 1979. *Water-related environmental fate of 129 priority pollutants*. Washington, D.C. EPA 440/4-79-029. December 1979.

U.S. Environmental Protection Agency (USEPA). 1985. *Chemical, physical, and biological properties of compounds present at hazardous waste sites - final report*. Prepared by Clement Associates, Inc. for the U.S. Environmental Protection Agency. September 1985.

U. S. Environmental Protection Agency (USEPA). 1988. *EPA Region I laboratory data validation functional guidelines for evaluating organics analyses.* USEPA New England. February 1998 (modified November 1988).

U.S. Environmental Protection Agency (USEPA). 1989a. *Ecological assessment of hazardous waste sites: A field and laboratory reference.* Environmental Research Laboratory. Corvallis, OR. EPA/600/3-89/013.

U.S. Environmental Protection Agency (USEPA). 1989b. *Exposure factors handbook.* Office of Health and Environmental Assessment. Washington, DC. July 1989.

U. S. Environmental Protection Agency (USEPA). 1989c. *EPA Region I laboratory data validation functional guidelines for evaluating inorganics analyses.* USEPA New England. June 1998 (modified February 1989).

U. S. Environmental Protection Agency (USEPA). 1994. *Methods for measuring the toxicity and bioaccumulation of sediment-associated contaminants with freshwater invertebrates.* Office of Research and Development. Washington, D.C. EPA/600/R-94/024.

U. S. Environmental Protection Agency (USEPA). 1996a. *Region I, EPA-NE data validation functional guidelines for evaluating environmental analyses.* USEPA New England. December 1996.

U. S. Environmental Protection Agency (USEPA). 1996b. *Wells G&H sediment toxicity test report, Hyallela azteca, Chironomus tentans.* USEPA New England. Office of Ecosystem Assessment. February 1996.

U.S. Environmental Protection Agency (USEPA). 1998a. *Risk assessment guidance for Superfund. Volume I: Human health evaluation manual. Part D, Standardized planning, reporting, and review of Superfund risk assessments. Interim.* Office of Solid Waste and Emergency Response. Washington, D.C. OSWER Directive 9285.7-01D. January 1998.

U. S. Environmental Protection Agency (USEPA). 1998b. *Sample collection activities and sediment toxicity testing results for the Wells G&H Superfund Site, Woburn, Massachusetts.* New England Regional Laboratory. Office of Environmental Measures and Evaluation. Office of Ecosystem Assessment. March 1998.

U. S. Environmental Protection Agency (USEPA). 2000. *Methods for measuring the toxicity and bioaccumulation of sediment-associated contaminants with freshwater invertebrates,* Second Edition. Office of Research and Development. Washington, D.C. EPA/600/R-99/064.

U. S. Environmental Protection Agency (USEPA). 2002. *Soil and Sediment Sampling for the Industri-Plex OU-2 and the Wells G&H OU-3 Superfund sites QAPP.* July 2002.

U.S. Environmental Protection Agency (USEPA). 2003. *Drinking water standards.* Office of Water. June 2003.

SECTION 3.0 REFERENCES

Agency for Toxic Substances and Disease Registry (ATSDR). 2002. *Toxicological profile for copper.* U.S. Department of Health and Human Services. Public Health Service. December 2002.

Chen, C.J., C.W. Chen, M.M. Wu, and T.L. Kuo. 1992. Cancer potential in liver, lung bladder and kidney due to ingested inorganic arsenic in drinking water. *Br. J. Cancer* 66:888-892.

Foster Wheeler Environmental Corporation (Foster Wheeler). 1996. *Preliminary data compendium, Wells G&H RI/FS, Operable Unit III, Aberjona River Study, sampling period July 31, 1995 to September 12, 1995.* Prepared for the U.S. Environmental Protection Agency. Contract No. 68-W9-0034.

Lewis, D.R., Southwick, J.W., Ouellet-Hellstrom, W., Rench, J., and Calderon, R.L. 1999. Drinking water arsenic in Utah: A cohort mortality study. *Environ. Health Perspect.* 107(5):359-365.

Metcalf and Eddy, Inc. (M&E). 1998. *Supplemental data compendium for Wells G&H Superfund Site, Aberjona River Study (Operable Unit 3), Woburn, Massachusetts.* Prepared for the U.S. Environmental Protection Agency. Contract No. 68-W6-0042. May 1998.

Metcalf and Eddy, Inc. (M&E). 2000. Personal communication from J. Barry, Winchester Board of Health, to K. Scott, Metcalf & Eddy, Inc. regarding consumption of aquatic species from the Aberjona River. June 29, 2000.

Moore, L.E., Lu, M., and Smith, A.H. 2002. Childhood cancer incidence and arsenic exposure in drinking water in Nevada. *Arch. Environ. Health* 57:201-206.

Tseng, W.P., H.M. Chu, S.W. How, J.M. Fong, C.S. Lin, and S. Yeh. 1968. Prevalence of skin cancer in an endemic area of chronic arsenicism in Taiwan. *J. Natl. Cancer Inst.* 40:453-462.

Tseng, W.P. 1977. Effects and dose-response relationships of skin cancer and blackfoot disease with arsenic. *Environ. Health Perspect.* 19:109-119.

U.S. Environmental Protection Agency (USEPA). 1989. *Risk assessment guidance for Superfund. Volume I: Human health evaluation manual. Part A.* Interim Final. EPA/540/1-89/002. December 1989.

U.S. Environmental Protection Agency (USEPA). 1991. *Role of baseline risk assessment in Superfund remedy selection decisions.* Office of Solid Waste and Emergency Response. Washington, D.C. OSWER Directive 9355.0-30. Memo from Don R. Clay. April 22, 1991.

U.S. Environmental Protection Agency (USEPA). 1992. *Supplemental guidance to RAGS: calculating the concentration term.* Office of Solid Waste and Emergency Response. Washington, D.C. Publication 9285.7-081. May 1992.

U.S. Environmental Protection Agency (USEPA). 1993. *Provisional guidance for quantitative risk assessment of polycyclic aromatic hydrocarbons.* Office of Research and Development. Washington, DC. EPA/600/R-93/089. July 1993.

U.S. Environmental Protection Agency (USEPA). 1994a. *Revised interim soil lead guidance for CERCLA sites and RCRA corrective action facilities.* Office of Solid Waste and Emergency Response. Washington, D.C. OSWER Directive 9355.4-12. July 1994.

U.S. Environmental Protection Agency (USEPA). 1994b. *Risk updates, no. 2.* USEPA Region I. August 1994.

U.S. Environmental Protection Agency (USEPA). 1994c. *Guidance manual for the integrated exposure uptake biokinetic model for lead in children.* Office of Solid Waster and Emergency Response. Washington, D.C. EPA/540/R-93/081. February 1994.

U.S. Environmental Protection Agency (USEPA). 1995. *Risk updates, no. 3.* USEPA Region I. August 1995.

U.S. Environmental Protection Agency (USEPA). 1996. *Risk updates, no. 4.* USEPA Region I. November 1996.

U.S. Environmental Protection Agency (USEPA). 1997a. *Exposure factors handbook.* Office of Research and Development. Washington, D.C. August 1997.

U.S. Environmental Protection Agency (USEPA). 1997b. *Health effects assessment summary* *tables, FY-1997 annual (HEAST).* Office of Health and Environmental Assessment. Environmental Assessment and Criteria Office and Office of Solid Waste and Emergency Response. Washington, D.C. July 1997.

U.S. Environmental Protection Agency (USEPA). 1998. Personal communication from M. McDonough, USEPA, to D. Silverman, Metcalf & Eddy, Inc. regarding iron, cobalt and aluminum as COPCs for Wells G&H OU3. April 21, 1998.

U.S. Environmental Protection Agency (USEPA). 2001. *Risk assessment guidance for Superfund. Volume I: Human health evaluation manual. Part D, Standardized planning, reporting, and review of Superfund risk assessments. Final.* Office of Emergency and Remedial Response. Washington, D.C. OSWER Directive 9285.7-47. December 2001.

U.S. Environmental Protection Agency (USEPA). 2002a. *Preliminary remediation goals table. Region 9 technical support team.* Office of Superfund Programs. Region IX. October 1, 2002.

U.S. Environmental Protection Agency (USEPA). 2002b. *User's guide for the integrated exposure uptake biokinetic model for lead in children.* Off ice of Emergency and Remedial Response. Washington, D.C. EPA 9285.7-42. May 2002.

U.S. Environmental Protection Agency (USEPA). 2002c. *Calculating upper confidence limits for exposure point concentrations at hazardous waste sites.* Off ice of Emergency and Remedial Response. Washington, D.C. OSWER 9285.6-10. December 2002.

U.S. Environmental Protection Agency (USEPA). 2002d. *National recommended water quality criteria.* Federal Register. Decemeber 27, 2002.

U.S. Environmental Protection Agency (USEPA). 2003a. *Recommendations of the Technical Review Workgroup for lead for an approach to assessing risk associated with adult exposures to lead in soil.* Technical Review Workgroup for Lead. United States Environmental Protection Agency. January 2003.

U.S. Environmental Protection Agency (USEPA). 2003b. *Drinking water standards.* Office of Water. June 2003.

U.S. Environmental Protection Agency (USEPA). 2004a. *Integrated risk information system (IRIS)*. Environmental Criterion and Assessment Office. August 2004.

U.S. Environmental Protection Agency (USEPA). 2004b. *Risk-based concentration table.*

Region III technical guidance manual, risk assessment. Hazardous Site Cleanup Division. Office of Superfund Programs. Region III. April 14, 2004.

U.S. Environmental Protection Agency (USEPA). 2004c. *ProUCL version 3.0 statistical software to compute upper confidence limits on the unknown population mean.* Technical Support Center. Las Vegas, Nevada. http://www.epa.gov/nerlesd1/tsc/tsc.html. April 2004.

U.S. Environmental Protection Agency (USEPA). 2004d. *Risk assessment guidance for Superfund Volume I: Human health evaluation manual (Part E, Supplemental guidance for dermal risk assessment), Final.* Office of Superfund Remediation and Technology Innovation. Washington, D.C. EPA/540/R/99/005. OSWER 9285.7-02EP. PB99-963312. July 2004.

SECTION 4.0 REFERENCES

Agency for Toxic Substances and Disease Registry (ATSDR). 1990. *Toxicological profile for silver*. U.S. Department of Health and Human Services. Public Health Service. December 1990.

Agency for Toxic Substances and Disease Registry (ATSDR). 1992a. *Toxicological profile for 2-butanone*. U.S. Department of Health and Human Services. Public Health Service. July 1992.

Agency for Toxic Substances and Disease Registry (ATSDR). 1992b. *Toxicological profile for antimony*. U.S. Department of Health and Human Services. Public Health Service. December 1992.

Agency for Toxic Substances and Disease Registry (ATSDR). 1992c. *Toxicological profile for barium*. U.S. Department of Health and Human Services. Public Health Service. July 1992.

Agency for Toxic Substances and Disease Registry (ATSDR). 1992d. *Toxicological profile for cobalt*. U.S. Department of Health and Human Services. Public Health Service. July 1992.

Agency for Toxic Substances and Disease Registry (ATSDR). 1993. *Toxicological profile for N-nitrosodiphenylamine*. U.S. Department of Health and Human Services. Public Health Service. April 1993.

Agency for Toxic Substances and Disease Registry (ATSDR). 1994a. *Toxicological profile for* *acetone*. U.S. Department of Health and Human Services. Public Health Service. May 1994.

Agency for Toxic Substances and Disease Registry (ATSDR). 1994b. *Toxicological profile for 1,2-dichloroethene*. U.S. Department of Health and Human Services. Public Health Service. August 1994.

Agency for Toxic Substances and Disease Registry (ATSDR). 1994c. *Toxicological profile for 4,4'-DDT, 4,4'-DDE, and 4,4'- DDD*. U.S. Department of Health and Human Services. Public Health Service. May 1994.

Agency for Toxic Substances and Disease Registry (ATSDR). 1995a. *Toxicological profile for naphthalene*. U.S. Dept. of Health and Human Services. Public Health Service. August 1995.

Agency for Toxic Substances and Disease Registry (ATSDR). 1995b. *Toxicological profile for polycyclic aromatic hydrocarbons (PAHS)*. U.S. Dept. of Health and Human Services. Public Health Service. August 1995.

Agency for Toxic Substances and Disease Registry (ATSDR). 1999a. *Toxicological profile for aluminum*. U.S. Dept. of Health and Human Services. Public Health Service. Updated July 1999.

Agency for Toxic Substances and Disease Registry (ATSDR). 1999b. *Toxicological profile for cadmium*. U.S. Dept. of Health and Human Services. Public Health Service. Updated July 1999.

Agency for Toxic Substances and Disease Registry (ATSDR). 1999c. *Toxicological profile for lead*. U.S. Dept. of Health and Human Services. Public Health Service. Updated July 1999.

Agency for Toxic Substances and Disease Registry (ATSDR). 2000a. *Toxicological profile for arsenic*. U.S. Dept. of Health and Human Services. Public Health Service. Updated September 2000.

Agency for Toxic Substances and Disease Registry (ATSDR). 2000b. *Toxicological profile for chromium*. U.S. Dept. of Health and Human Services. Public Health Service. Updated September 2000.

Agency for Toxic Substances and Disease Registry (ATSDR). 2000c. *Toxicological profile for polychlorinated biphenyls*. U.S. Dept. of Health and Human Services. Public Health Service. Updated November 2000.

Agency for Toxic Substances and Disease Registry (ATSDR). 2002. *Toxicological profile for copper*. U.S. Dept. of Health and Human Services. Public Health Service. Updated September 2002.

Allen, A.W. and R.D. Hoffman. 1984. *Habitat suitability index models: Muskrat*. U.S. Fish and Wildlife Service. FWS/OBS-82/10.46.

Andersson, M. 1988. Toxicity and tolerance of aluminum in vascular plants. *Water, Air and Soil Pollution* 39:439-463.

Beyer, W.N. 1990. Evaluating soil contamination. *U.S. Fish and Wildlife Service Biological Report* 90:25.

Beyer, W.N. and C. Stafford. 1993. Survey and evaluation of contaminants in earthworms and in soils derived from dredged material at confined disposal facilities in the Great Lakes region. *Environ. Monitor. Assessment.* 24:151-165.

Beyer, W.N., E.E. Connor, and S. Gerould. 1994. Estimates of soil ingestion by wildlife. *J. Wildlife Management* 58:375-382.

Blus, L.J. 1978. Short-tailed shrews: toxicity and residue relationships of DDT, dieldrin, and endrin. *Arch. Environ. Contam. Toxicol*. 7:83-98.

Buchman, 1999. NOAA Screening Quick Reference Tables, NOAA HAZMAT Report 99-1, Seattle, WA, Coastal Protection and Restoration Division, National Oceanic and Atmospheric Admistration, 12 pp.

Bysshe, S.E. 1988. Uptake by biota. In: Bodek, I., W.J. Lyman, W.F. Reehl, and D.H. Rosenblatt (eds.). *Environmental Inorganic Chemistry*. Pergamon Press. New York. Chapter 4.

Camardese, M.. B., D.L. Hoffman, L.J. LeCaptain and G.W. Pendleton. 1990. Effects of arsenate on growth and physiology in mallard ducklings. *Environ. Toxicol. Chem*. 9:785-795.

Canfield, T.J, N.E. Kemble, W.G. Grumbaugh, F.J. Dwyer, C.G. Ingersoll, and J.F. Fairchild. 1994. Use of benthic community structure and the sediment quality triad to evaluate metla-contaminated sediment in the uper Clark Fork River, Montana. *Environ Toxicol. Chem*., Vol. 13, pp 1999-2012.

Chapman, P.M., F. Wang, C. Janssen, G. Persoone and H.E. Allen. 1998. Ecotoxicology of metals in aquatic sediments: binding and release, bioavailability, hazard, rrisk and remediation. *Can. J. Aquatic Sci*. Vol. 55, pp. 221-2243.

Connell, D.W. and G.J. Miller. 1984. *Chemistry and ecotoxicology of pollution*. John Wiley and Sons. New York, New York. pp. 162-227.

Davies, P.H. 1980. *Investigations on the toxicity of metals to fish*. Colorado Division of Wildlife. Fort Collins, CO. Water Pollution Studies: Job 2. Job Progress Report F-33-R-15.

DeGraff, R.M. and D.D. Rudis. 1983. *New England wildlife: habitat, natural history, and distribution*. U.S. Department of Agriculture. Northeastern Forest Experiment Station. General Technical Report NE-108.

Department of Environmental Quality Engineering (DEQE). 1982. *Upper Mystic Lake watershed urban runoff project main report*. Office of Planning and Program Management. October 1982.

Dobbs, M.G., D.S. Cherry, and J. Cairns, Jr. 1996. Toxicity and bioaccumulation of selenium to a three-trophic level food chain. *Environ. Toxicol. Chem.* 15:340-347.

Domingo, J.L. 1994. Metal-induced developmental toxicity im mammals: a review. *J. Toxicol.& Environ. Health.* 42:123-141.

Doudoroff, P. and M. Katz. 1953. Critical review of literature on the toxicity of industrial wastes and their components to fish. II. The metals, as salts. *Sewage and Industrial Wastes* 25:802-839.

Dutson, S.M., G.M. Booth, G.B. Schaalje, R.N. Castle, and R.E. Seegmiller. 1997. Comparative developmental dermal toxicity and mutagenicity of carbazole and benzo[*a*]carbazole. *Environ. Toxicol. Chem.* 16:2113-2117.

Duxbury, C.L., D.G. Dixon, and B.M. Greenburg. 1997. Effects of simulated solar radiation on the bioaccumulation of polycyclic aromatic hydrocarbons by the duckweed *Lemna gibba*. *Environ. Toxicol. Chem.* 16:1739-1748.

Ebasco Corporation (Ebasco). 1995. *Operable Unit III final field operations plan, Aberjona River Study, Wells G&H Superfund Site, Woburn, Massachusetts*. Prepared for the U.S. Environmental Protection Agency. Contract No. 68-W9-0034.

Eisler, R. 1985. Cadmium hazards to fish, wildlife, and invertebrates: a synoptic review. *U.S. Fish and Wildlife Service Biological Report* 85:46.

Eisler, R. 1986a. Polychlorinated biphenyl hazards to fish, wildlife and invertebrates: a synoptic review. *U.S. Fish and Wildlife Service Biological Report* 85:72.

Eisler, R. 1986b. Chromium hazards to fish, wildlife, and invertebrates: a synoptic review. *U.S. Fish and Wildlife Service Biological Report* 85:60.

Eisler, R. 1987a. Polycyclic aromatic hydrocarbon hazards to fish, wildlife, and invertebrates: a synoptic review. *U.S. Fish and Wildlife Service Biological Report* 85: 81.

Eisler, R. 1987b. Mercury hazards to fish, wildlife, and invertebrates: a synoptic review. *U.S. Fish and Wildlife Service Biological Report* 85:90.

Eisler, R. 1987c. Selenium hazards to fish, wildlife, and invertebrates: a synoptic review. *U.S. Fish and Wildlife Service Biological Report* 85:57.

Eisler, R. 1988a. Arsenic hazards to fish, wildlife, and invertebrates: a synoptic review. *U.S. Fish and Wildlife Service Biological Report* 85:92.

Eisler, R. 1988b. Lead hazards to fish, wildlife, and invertebrates: a synoptic review. *U.S. Fish and Wildlife Service Biological Report* 85:134.

Eisler, R. 1993. Zinc hazards to fish, wildlife, and invertebrates: a synoptic review. *U.S. Fish and Wildlife Service Biological Report* 26:106.

Eisler, R. 2000. *Handbook of chemical risk assessment: health hazards to humans, plants and animals*, Volume I: Metals. Lewis Publishers, Boca Raton, FL.

Foster Wheeler Environmental Corporation (Foster Wheeler). 1996. *Preliminary data compendium, Wells G&H RI/FS, Operable Unit III, Aberjona River Study, sampling period July 31, 1995 to September 12, 1995*. Prepared for the U.S. Environmental Protection Agency. Contract No. 68-W9-0034.

Hall, R.J. 1980. Effects of environmental contaminants on reptiles: a review. *U.S. Fish and Wildlife Service Special Scientific Report - Wildlife* 228:12.

Harlan, J.R., E.B. Speaker, and J.M. Mayhew. 1987. *Iowa fish and fishing*. Iowa Department of Natural Resources.

Hazardous Substances Data Bank (HSDB), updated 2000. Toxicology and Environmental Health Information Program, National Library of Medicine, http://toxnet.nlm.nih.gov.

Herkovits, J., P. Cardellini, C. Pavanati, and C.S. Perex-Coll. 1997. Susceptibility of early life stages of *Xenopus laevis* to cadmium. *Environ. Toxicol. Chem.* 16:312-316.

Howard, P.H. 1990. *Handbook of environmental fate and exposure data for organic chemicals. Volume II: Solvents*. Lewis Publishers, Chelsea, MI.

Hudson, R.H., Tucker, R.K., and M.A. Haegle. 1984. *Handbook of toxicity of pesticides to wildlife*. U.S. Department of the Interior. Fish and Wildlife Service. Washington, D.C. Resource Publication 153.

Ireland, D.S., G.A. Burton, and G.G. Hess. 1996. In situ toxicity evaluations of turbidity and photoinduction of polycyclic aromatic hydrocarbons. *Environ. Toxicol. Chem.* 15:574-581.

Jones, D.S., G.W. Suter, II, and R.N. Hull. 1997. *Toxicological benchmarks for screening contaminants of potential concern for effects on sediment-associated biota, 1997 revision*. Risk Assessment Program. Health Sciences Research Division. Oak Ridge, TN. Prepared for U.S. Department of Energy, Office of Environmental Management. November 1997.

Jones, J.K., Jr. and E.C. Birney. 1988. *Handbook of mammals of the north-central states*. University of Minnesota Press, Minneapolis.

Kaufmann, J.H. 1982. Raccoons and Allies. In: Chapman, J.A. and Feldhamer, G.A. (eds). *Wild Mammals of North America: Biology, Management, Economics*. John Hopkins University Press, Baltimore, MD. pps. 567-585.

Kelly, J.M., M. Schaedle, F.C. Thornton, and J.D. Joslin. 1990. Sensitivity of tree seedlings to aluminum: II. red oak, sugar maple, and European beech. *J. Environ. Quality* 19:172-180.

Klaassen, C.D. 1996. *Casarett and Doull's toxicology: The basic science of poisons, fifth edition*. McGraw-Hill, New York.

Lehman, A.J. 1954. Appraisal of the study of chemicals in food, drugs, and cosmetics. *U.S. Association of Food and Drug Officials*. 18:66.

Lewis, M.A., D.W. Evans, and J.G. Wiener. 1979. Manganese. In: Thurston, R.V., R.C. Russo, C.M. Ferrerolf, Jr., T.A. Edsall, and Y.M. Barber, Jr. (eds.). *A Review of the USEPA Red Book: Quality Criteria for Water*. American Fisheries Society. Bethesda, MD. pps. 137-144.

Locke, L.N. and N.J. Thomas. 1996. Lead poisoning of waterfowl and raptors. In: A. Fairbrother, L.N. Locke, and G.L. Hoff (eds.). *Noninfectious Diseases of Wildlife, Second Edition*. Iowa State University Press. Ames, Iowa.

Long, E.R., D.D. McDonald, S.L. Smith, and F.D. Calder. 1995. Incidence of adverse biological effects within ranges of chemical concentrations in marine and estuarine sediments. *Environmental Management* 19:81-97.

Long, E.R. and L.G. Morgan. 1990. *The potential for biological effects of sediment-sorbed contaminants tested in the National Status and Trends Program*. NOAA Technical Memorandum NOS OMA 52, Seattle, WA.

Lotufo, G. R. and J. W. Fleeger. 1996. Toxicity of sediment-associated pyrene and phenanthrene to *Limnodrilus hoffmeisteri* (Oligochaeta: Tubificidae). *Environ. Toxicol. Chem.* 15:1508-1516.

Martin, A.C., H.S. Zim, and A.L. Nelson. 1961. *American wildlife and plants, a guide to wildlife food habitats*. Dover Publications, Inc. New York.

Massachusetts Department of Environmental Protection (MADEP). 1995. *Guidance for disposal site risk characterization*. Bureau of Waste Site Cleanup and Office of Research and Standards. BWSC/ORS-95-141. July 1995.

Massachusetts Natural Heritage and Endangered Species Program (MANHESP). 1998. Letter dated 23 April 1998 from Andrea Arnold, MANHESP, to Jonathon Weier, Metcalf & Eddy, Inc. regarding the presence of rare species at the Wells G&H Superfund Site.

Mayer, F.L. and M.R. Ellersieck. 1986. *Manual of acute toxicity: Interpretation and data base for 410 chemicals and 66 species of freshwater animals*. U.S. Department of the Interior. Fish and Wildlife Service. Washington, D.C. Resource Publication 160.

McClane, A.J. 1978. *McClane's field guide to freshwater fishes of North America*. Henry Holt and Company. New York.

Metcalf & Eddy, Inc. (M&E). 1998. *Supplemental data compendium for Wells G&H Superfund Site, Aberjona River Study (Operable Unit 3), Woburn, Massachusetts*. Prepared for the U.S. Environmental Protection Agency. Contract No. 68-W6-0042. May 1998.

National Academy of Science (NAS). 1980. *Mineral tolerance of domestic animals*. Subcommittee on mineral toxicity in animals. National Research Council. Washington, D.C.

Neff, J.M. 1985. Polycyclic Aromatic Hydrocarbons. In: Rand, G.M. and S.R. Petrocelli (eds). *Fundamentals of Aquatic Toxicology, Methods, and Applications*. Hemisphere Publishing Corporation. Washington, D.C.

Novak, M., 1987. Beaver. In: M. Novak, J.A. Baker, M.E. Obbard, and B. Malloch (eds). *Wild Furbearer Management and Conservation in North America*. Ministry of Natural Resources. Ontario, Canada. pps. 283-312.

O'Hara, T.M. and C.D. Rice. 1996. Polychlorinated Biphenyls. In: A. Fairbrother, L.N. Locke, and G.L. Hoff (eds.). *Noninfectious Diseases of Wildlife, Second Edition*. Iowa State University Press. Ames, Iowa.

Ontario Ministry of the Environment (OME). 1996. Guidlines for Use at Contaminated Sites in Ontario. ISBN 0-778-4052-9. PIBS 3161E01. June 1996 (Appendix 2, Table E).

Opresko, D.M, B.E. Sample, and G.W. Suter, II. 1994. *Toxicological benchmarks for wildlife, 1994 revision*. Health Science Research Division and Environmental Sciences Division. Oak Ridge National Laboratory. ES/ER/TM-86/R1. September 1994.

Pastorok, R.A., D.C. Peek, J.R. Sampson, and M.A. Jacobson. 1994. Ecological risk assessment for river sediment contaminated by creosote. *Environ. Toxicol. Chem.* 13:1929-1941.

Pelletier, M.C., R.M. Burgess, K.T. Ho, A. Kuhn, R.A. McKinney, and S.A. Ryba. 1997. Phototoxicity of individual polycyclic aromatic hydrocarbons and petroleum to marine invertebrate larvae and juveniles. *Environ. Toxicol. Chem.* 16:2190-2199.

Persaud, D., R. Jaagumagi, and A. Hayton. 1993. *Guidelines for the protection and management of aquatic sediment quality in Ontario*. Ontario Ministry of Environment and Energy. August 1993.

Peterson, R.T. 1980. *Eastern birds*. Houghton Mifflin Company. Boston.

Plafkin, J.L, M.T. Barbour, K.D. Porter, S.K. Gross and R.M. Hughes. 1989. Rapid Bioassessment Protocols for Use in Strams and Rivers: Benthic Macroinvertebrates and Fish. USEPA Office of Water Regulations and Standards, Washington, DC. EPA/444/4-89-001.

Planning Research Corporation (PRC). 1986. *Wells G&H wetlands assessment final report*. Prepared for the U.S. Environmental Protection Agency. Contract No. 68-01-7037. March 1986.

Puls, R. 1988. *Mineral levels in animal health*. Sherpa International. Clearbrook, British Columbia.

Riedel, G.F. and J.G. Sanders. 1996. The influence of pH and media composition on the uptake of inorganic selenium by *Chlamydomonas reinhardtii*. *Environ. Toxicol. Chem.* 15:1577-1583.

Roberts, B.L. and H.W. Dorough. 1985. Hazards of chemicals to earthworms. *Environ. Toxicol. Chem.* 4:307-323.

Sample, B.E., M.S. Alpin, R.A. Efroymson, G.W. Suter, II, and C.J.E. Welsh. 1997. *Methods and tools for the estimation of the exposure of terrestrial wildlife to contaminants.* Environmental Science Division. Oak Ridge National Laboratory. ORNL/TM-13391. October 1997.

Sample, B.E., J.J. Beauchamp, R.A. Efroymson, G.W. Suter, II, and T.L. Ashwood. 1998. *Development and validation of bioaccumulation model for earthworms.* Lockheed Martin Environmental Restoration Program. Oak Ridge National Laboratory. ES/ER/TM-220. February 1998.

Sample, B.E., D.M. Opresko, and G.W. Suter, II. 1996. *Toxicological benchmarks for wildlife, 1996 revision*. Environmental Sciences Division. Oak Ridge National Laboratory. ES/ER/TM-86/R3. June 1996.

Sample, B.E. and G.W. Suter, II. 1994. *Estimating exposure of terrestrial wildlife to contaminants*. Environmental Sciences Division. Oak Ridge National Laboratory. ES/ER/TM-125. September 1994.

Sax, N.I. and L.J. Lewis, Sr. 1989. *Dangerous properties of industrial materials, seventh edition. Vol. 1.* New York, NY.

Scheuhammer, A.M. 1987. The chronic toxicity of aluminum, cadmium, mercury, and lead in birds: a review. *Environ. Pollut.* 46:263-295.

Schmidt-Nielsen, K. 1983. *Animal physiology: adaptation and environment.* Cambridge University Press. London.

Scotts-Fordsmand, J.J., P.H. Krogh, and J.M. Weeks. 1997. Sublethal toxicity of copper to a soil-dwelling springtail (*Folsomia fimetaria*) (Collembola: Isotomidae). *Environ. Toxicol. Chem*. 16:2538-2542.

Spangenberg, J.V. and G.N. Cherr. 1996. Developmental effects of barium exposure in a marine bivalve (*Mytilus californianus*). *Environ. Toxicol. Chem.* 15:1769-1774.

Stendell, R.C., Beyer, W.N., and R.A. Stehn. 1989. Accumulation of lead and organochlorine residues in captive American kestrels fed pine voles from apple orchards. *J. Wildlife Disease* 25:388-391.

Douglas-Stoebel, E.K., G.L. Brewer and D.J. Hoffman. 2001. Effects of lead-contaminated sediment and nutrient level on mallard duckling brain growth and biochemistry. SETAC Annual Meeting Abstract, p. 198.

Stubblefield, W.A., S.F. Brinkman, P.H. Davies, T.D. Garrison, J.R. Hockett, and M.W. McIntyre. 1997. Effects of water hardness on the toxicity of manganese to developing brown trout (*Salmo trutta*). *Environ. Toxicol. Chem.* 16:2082-2089.

Stuber, R.J., G. Gebhart, and O.E. Maughan. 1982a. *Habitat suitability index models: Bluegill*. U.S. Department of Interior. Fish and Wildlife Service. FWS/OBS-82/10.8.

Stuber, R.J., G. Gebhart, and O.E. Maughan. 1982b. *Habitat suitability index models: Largemouth bass*. U.S. Department of Interior. Fish and Wildlife Service. FWS/OBS-82/10.16.

Suter, G.W., II. 1993. *Ecological risk assessment*. Lewis Publishers. Ann Arbor, Michigan.

Suter, G.W., II. and C.L. Tsao. 1996. *Toxicological benchmarks for screening potential contaminants of concern for effects on aquatic biota, 1996 revision*. Environmental Sciences Division. Oak Ridge National Laboratory. ES/ER/TM-96/R2.

Swartz, R.C., S.P. Ferraro, J.O. Lamberson, F.C. Cole, R.J. Ozretich, B.L. Boese, D.W. Schults, M. Behrenfeld, and G. Ankley. 1997. Photoactivation and toxicity of mixtures of polycyclic aromatic hydrocarbon compounds in marine sediment. *Environ. Toxicol. Chem*. 16:2151-2157.

U.S. Army Corps of Engineers and U.S. Environmental Protection Agency (USACOE/USEPA). 2002. *Environmental residue-effects database (ERED)*. Last updated October 2002.

U.S. Environmental Protection Agency (USEPA). 1980. *Ambient water quality criteria for DDT*. Office of Water Regulations and Standards. Washington, D.C. EPA/440/5-80/038.

U.S. Environmental Protection Agency (USEPA). 1984. *Health effects assessment for iron (and compounds)*. Environmental Criteria and Assessment Office and Office of Solid Waste and Emergency Response. Washington, D.C. EPA/540/1-86/054. September 1994.

U.S. Environmental Protection Agency (USEPA). 1985. *Chemical, physical, and biological properties of compounds present at hazardous waste sites - final report*. Prepared by Clement Associates, Inc. for the U.S. Environmental Protection Agency. September 1985.

U.S. Environmental Protection Agency (USEPA). 1986a. *Quality criteria for water*. Office of Water Regulations and Standards. Washington, D.C. EPA 440/5-86-001. May 1986.

U.S. Environmental Protection Agency (USEPA). 1986b. *Update #1 to quality criteria for water*. Office of Water Regulations and Standards. Washington, D.C.

U.S. Environmental Protection Agency (USEPA). 1987. *Update #2 to quality criteria for water*. Office of Water Regulations and Standards. Washington, D.C.

U.S. Environmental Protection Agency (USEPA). 1989. *Risk assessment guidance for Superfund. Volume I: Human health evaluation manual. Part A.* Interim Final. EPA/540/1-89/002. December 1989.

U.S. Environmental Protection Agency (USEPA). 1992a. Water quality standards: Establishment of numeric criteria for priority toxic pollutants; State's compliance. *Fed. Regist*. 57:50848-60919.

U.S. Environmental Protection Agency (USEPA). 1992b. *Supplemental guidance to RAGS: calculating the concentration term.* Office of Solid Waste and Emergency Response. Washington, D.C. Publication 9285.7-081. May 1992.

U.S. Environmental Protection Agency (USEPA). 1993a. Water quality guidance for the Great Lakes System and correction: proposed rules. *Fed. Regist.* 58:20802-21047

U.S. Environmental Protection Agency (USEPA). 1993b. *Sediment quality criteria for the protection of benthic organisms: Dieldrin*. Washington, D.C. EPA-822-R-93-015.

U.S. Environmental Protection Agency (USEPA). 1993c. *Sediment quality criteria for the protection of benthic organisms: Endrin*. Washington, D.C. EPA-822-R-93-016.

U.S. Environmental Protection Agency (USEPA). 1993d. *Wildlife exposure factors handbook, Volume I of II*. Office of Research and Development. EPA/600/R-93/187a. December 1993.

U.S. Environmental Protection Agency (USEPA). 1994. *Guidance manual for the integrated exposure uptake biokinetic model for lead in children.* Office of Solid Waster and Emergency Response. Washington, D.C. EPA/540/R-93/081. February 1994.

U.S. Environmental Protection Agency (USEPA). 1995. Great Lakes Water Quality Initiative Criteria Documents for the Protection of Wildlife. USEPA Oddice of Water. Washington, D.C., EPA 820/b-95/008.

U.S. Environmental Protection Agency (USEPA). 1996. *Eco update, ecotox thresholds*. Office of Solid Waste and Emergency Response. EPA 540/F-95/038. January 1996.

U.S. Environmental Protection Agency (USEPA). 1997. *Ecological risk assessment guidance for Superfund: Process of designing and conducting ecological risk assessments, interim final*. Office of Solid Waste and Emergency Response. EPA 540-R-97-006. June 1997.

U.S. Environmental Protection Agency (USEPA). 1998a. *Guidelines for ecological risk assessment.* Risk Assessment Forum. Washington, D.C. EPA/630/R-95/002F. April 1998.

U.S. Environmental Protection Agency (USEPA). 1998b. *National recommended water quality criteria.* Federal Register. Decemeber 10, 1998.

U.S. Environmental Protection Agency (USEPA). 1998c. *AQUIRE - Aquatic toxicity information retrieval database*. Office of Research and Development. National Health and Environmental Effects Laboratory. Mid-Continent Ecology Division. Duluth, MN. Last updated on 4/29/98.

U.S. Environmental Protection Agency (USEPA). 1999. Draft Equilibrium-Partitioning Sediment Guidelines (ESGs) for the Protection of Benthic Organisms: Metal Mixtures (Cadmium, copper, lead, nickel ,silver and zinc). Office of Science and Technology and Office of Research and Development. EPA 822-R-02-045.

U. S. Environmental Protection Agency (USEPA). 2000. *Methods for measuring the toxicity and bioaccumulation of sediment-associated contaminants with freshwater invertebrates*. Second Edition. Office of Research and Development. Washington, D.C. EPA/600/R-99/064.

U.S. Fish and Wildlife Service (USFWS). 1998. Letter dated 24 April 1998 from Michael Amaral, USFWS, to Jonathon Weier, Metcalf & Eddy, Inc. regarding the presence of rare species at the Wells G&H Superfund Site.

Van Derveer, W.D. and S.P. Canton. 1997. Selenium sediment toxicity thresholds and derivation of water quality criteria for freshwater biota of western streams. *Environ. Toxicol. Chem.* 16:1260-1268.

Weller, M.W. 1981. *Freshwater marshes - ecology and wildlife management.* University of Minnesota Press. Minneapolis.

Wiemeyer, Stanley N., and Richard D. Porter. 1970. DDE thins eggshells of captive American kestrels. Nature. 227:737-738.

Wolfe, M.F., S. Schwarzbach, and R.A. Sulaiman. !998. Effects of mercury on wildlife a comprehensive review. *Environ. Toxicol. Chem.* 17:146-160.