BASELINE ECOLOGICAL RISK ASSESSMENT
NUCLEAR METALS, INC. SUPERFUND SITE
CONCORD, MASSACHUSETTS

by

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de maximis
Environmental Project Management

EXECUTIVE SUMMARY

This Baseline Ecological Risk Assessment (BERA) was performed as part of the Remedial Investigation (RI) at the Nuclear Metals, Inc. (NMI) Superfund Site (Site) located in Concord, Massachusetts (Figure 1-1). The objective of this BERA is to estimate the risk of ecological harm associated with Site-related Contaminants of Potential Concern (COPCs) which consist primarily of metals, polychlorinated biphenyls (PCBs) and polynuclear aromatic hydrocarbons (PAHs). Risks from VOCs are negligible as they were detected infrequently and at low concentrations.

The ecological risk assessment process at the Site follows the United States Environmental Protection Agency (USEPA) *Ecological Risk Assessment Guidance for Superfund, Process for Designing and Conducting Ecological Risk Assessment, 1997* herein referred to as the ERA Guidance. This BERA was also prepared in accordance with the approach described in the Remedial Investigation/Feasibility Study (RI/FS) Work Plan (*de maximis*, 2005).

The RI/FS Work Plan specified the RI data collection activities and methods under the RI. The Phase 1A and Phase 1B field work performed in 2005 and 2006 focused on field investigations to support a draft Screening Level Ecological Risk Assessment (SLERA) that was submitted in June 2006 (*de maximis, 2006*a). The SLERA concluded that risk of adverse radiological effects to ecological receptors is negligible and could be eliminated from further evaluation. Therefore, radiological risk to ecological receptors following exposure to radionuclides is not evaluated as part of this BERA.

Data gaps identified by the SLERA were addressed in 2006, 2007 and 2008 as part of the Phase 1C field investigations. This BERA evaluates data collected by *de maximis* during the Phase 1A, Phase 1B, and Phase 1C RI field programs (*de maximis, 2010)*. The BERA also addresses agency comments received June 10, 2006 and comments from Citizens Research and Environmental Watch (CREW) received on November 17, 2006.

SITE DESCRIPTION

The Site is a 18.8-hectare (46.4 acre) specialty metals research and manufacturing facility operating at 2229 Main Street in the western portion of the Town of Concord, Middlesex County, Massachusetts (Figure 1-1) from 1958 to the present. Past facility operations involved research and development in fundamental metallurgy, physical metallurgy, chemical metallurgy, engineering and product development, fuel element development and manufacture, and high temperature materials (NMI, 1961). The facility also manufactured depleted uranium products, metal powders, and products made of beryllium, beryllium alloy, and titanium.

As currently configured, the facility includes eight interconnected buildings, several smaller outbuildings, paved parking areas, a cooling water recharge pond, a former waste holding basin, and areas of fill and waste materials. A 1.5-hectare (3.7 acre) sphagnum bog located on the eastern portion of the facility property is another predominant feature. The Assabet River flows west to east approximately 300 feet (ft) north of the Site. Though it is not part of the facility, an approximately 6,000 foot reach of the Assabet River lies within the RI study area.

PROBLEM FORMULATION AND CONCEPTUAL SITE MODEL

As described in the "Delineation of Soil, Sediment, and Surface Water Contamination, Nuclear Metals, Inc. Superfund Site" (MACTEC, January 16, 2009) (Delineation Memorandum), the nature and extent of potential contaminants in surface water, sediment, and soil have been sufficiently characterized to proceed with the risk assessments. PAHs, PCBs, and metals, including uranium, are considered the primary COPCs, although volatile organic compounds (VOCs) were also detected but at low frequency or low concentrations.

Waste at the facility was historically generated through several manufacturing and laboratory processes, including the following:

- Depleted Uranium Operations – copper jacket removal
- Depleted Uranium Operations – machining and casting operations
- Machining Operations – dust collection systems and air handler units
- Research and Development – metal fabrication in support of Department of Defense (DOD) initiatives
- Thorium Operations – manufacturing of thoriated tungsten rods
- Beryllium Operations – beryllium fabrication
- Facility Support Processes – operation and maintenance of the facility
- On-Site electrical transformers.

The waste from these processes may have contaminated facility property via roof drain runoff, leaky subsurface drain lines, septic systems, buried drums, underground storage tanks (USTs) or transformer pads. Waste was also emitted into the air through the facility's air handling systems. After initial disposal, COPCs migration to other media would have occurred via leaching, overland flow, and particulate resuspension and deposition.

Substances on the pavement (e.g., metal particulates from airborne deposition) could be washed off during precipitation events, channeled to the storm water outfalls, and then transported to points down gradient from the storm water discharge outfalls. The catch basins that drain the facility's parking areas discharges to a slope above an embayment of the Assabet River floodplain via a culvert beneath Route 62, providing a possible pathway to the Assabet River main channel.

The following ecologically relevant areas of interest (AOIs) were identified during the Phase 1A, Phase 1B, and Phase 1C investigations:

- AOI 4 – Cooling Water Recharge Pond
- AOI 6 – Sphagnum Bog
- AOI 10 – Northeast Wetland
- AOI 18A – Assabet River Main Channel
- AOI 18B – Assabet River Embayment Area
- Site-Wide Surface Soil (principally includes AOI-14 and AOI-8 but also includes parts of AOIs 1, 2, 3, 7, 8, 9, 10, and 11 that are not located below pavement or buildings, and parts of AOI 4 that are not considered inundated sediments).

Site-specific data were obtained through analysis of surface water, sediment (including Simultaneously Extracted Metals/Acid Volatile Sulfide (SEM/AVS normalized to TOC) and soil. Data was also generated from sediment toxicity tests, benthic macroinvertebrate community surveys, and analysis of amphibian and benthic macroinvertebrate tissue for COPCs. Surface water was also collected to evaluate if COPCs might affect amphibian larvae (using FETAX, or Frog Embryo Teratogenesis Assay - Xenopus). The pH limits for the assay, however, were exceeded (Appendix I). This measurement endpoint was therefore replaced with the evaluation of tissue COPCs in amphibians of AOI 6 (Sphagnum Bog) to the reference bog. For some sediment locations, collocated data were obtained for sediment chemistry, sediment bioassays, and benthic macroinvertebrate surveys, an approach commonly referred to as a "sediment quality triad."

Although a SLERA was previously performed using Phase 1A and Phase 1B data, COPCs were re-selected using the pooled Phase 1A, 1B, and 1C data sets (Appendix A) per agreement with all parties. In accordance with USEPA policy, a SLERA can be sufficient to document risk in areas where a known remedy will be implemented. Based on the concentrations of PCBs and uranium in the cooling water recharge pond (AOI-4) documented in the Delineation Memorandum (MACTEC 2009), the need for a remedy at AOI 4 has already been identified, meaning that remedial measures for AOI-4 will be presented in the Feasibility Study. As a result, additional evaluation of ecological risk at AOI 4 is no longer necessary because risk associated with potential exposure to ecological receptors will be addressed by the presumptive remedy. Therefore, while the natural communities inhabiting the pond were described and COPCs were selected at AOI 4, effects were not assessed and risk at AOI 4 was not characterized in the BERA.

EXPOSURE ASSESSMENT AND EFFECTS ASSESSMENT

Assessment and measurement endpoints (Table ES-1) were evaluated for the following four wetland and aquatic study areas:

- AOI 6 – Sphagnum Bog,
- AOI 10 – Northeast Wetland,
- AOI 18A – Assabet River Main Channel; and
- AOI 18B – Assabet River Embayment Area

and one terrestrial study area:

- Site-Wide soil.

The specific methods used to assess each measurement endpoint are explained in Section 3.0, but generally included:

- Comparison of COPC exposure point concentrations (EPCs) in surface water, sediment, and surface soil to screening and effects benchmarks derived from the scientific literature.
- Evaluation of SEM AVS data and sediment equilibrium partitioning benchmark (ΣPAH) models.
- Sediment toxicity tests with an analysis of co-located sediment data to try to relate a dose response relationship to observed effects.
- An evaluation of tissue COPCs in amphibians of AOI 6 (Sphagnum Bog) to the reference bog.
- Quantitative benthic macroinvertebrate surveys.

- Food chain modeling for terrestrial and wetland birds and mammals by which modeled doses were compared to toxicity reference values (TRVs) based on both no-observable-adverse-effects-levels (NOAELs) and lowest-observable-adverse-effects levels) LOAELs.

Assessment populations evaluated in wetland and aquatic AOI food chain models included:

- Mallard ducks, representing omnivorous birds,
- Great blue heron, representing piscivorous birds,
- Osprey, representing piscivorous birds,
- Short-tailed shrew, representing small mammal invertivores; and
- Raccoon, representing omnivorous mammals.

The AOI 18B – Assabet River Main Channel also included freshwater mussels as an assessment population because the Massachusetts Natural Heritage and Endangered Species Program (MANHESP) indicated the presence of three species of concern including the eastern pondmussel (*Ligumia nasuta*), triangle floater (*Alasmidonta undulata*), and creeper (*Strophitus undulates*) (Appendix B).

Assessment populations evaluated in site-wide soil food chain models included:

- American robin, representing invertivorous songbirds,
- Cardinal, representing omnivorous songbirds,
- Meadow vole, representing a small herbaceous mammal; and
- Red fox, representing omnivorous mammals.

Reference locations were identified and paired with Site AOIs based on similar habitat characteristics including hydrology, plant community characteristics, and surrounding land use.

Both reasonable maximum exposures (RME) and central tendency exposures (CTE) were considered when assessing and characterizing risk. CTE represents the most likely concentration to which a population of receptors would be exposed. CTE EPCs were calculated as the lower of the arithmetic mean or the maximum detected concentration, assuming a one-half detection limit for non-detects. RME EPCs were calculated as the lower of the 95 percent upper confidence limit (UCL) or the maximum detected concentration.

Exposure pathways that were evaluated included direct contact (aquatic and terrestrial plants and invertebrates) and ingestion (wildlife). Dermal contact and inhalation were not evaluated as an exposure pathway in wildlife because of the lack of toxicity data and that the physicochemical nature of the COPCs at the Site render these exposure routes insignificant compared to other pathways (e.g. metals are poorly absorbed by skin; VOCs were rarely detected).

RISK CHARACTERIZATION

Risk characterization involves the integration of exposure and effects data to determine the likelihood of adverse effects (USEPA, 1997). A weight-of-evidence approach was used to make conclusions regarding risk of harm for assessment endpoints with more than one measurement endpoint (Menzie et al., 1996). Measurement endpoints were each assigned an inference weight based upon how closely they represented the analogous assessment endpoints. Conclusions regarding risks to an assessment endpoint were reached by considering the inference weight for each measurement endpoint, i.e., the overall weight of evidence. Risk was characterized using a two- or a four-way decision matrix (Pauwels, 2008) that incorporates all four benchmark combinations to derive a risk conclusion and an associated confidence level (i.e., low, moderate or high confidence) based on the completeness of the available exposure and toxicity data.

Measurement endpoints involving food chain models and comparison of media concentrations to benchmarks were assessed using a hazard quotient (HQ) approach. When HQs were calculated as part of the Effects Assessment (e.g. comparison to benchmarks, food chain models), the likelihood of adverse population effects was determined using a four-way matrix that incorporated all four combinations using RME and CTE EPCs and NOAEL and LOAEL TRVs (or screening and effects benchmark) combinations. Where only screening benchmarks were used to assess effects, a two-way table based (similar in concept to the four-way table) was used to characterize risk. In addition to estimation of total risk, incremental risk HQs, which subtracted the background risk contribution from the Site risk, was used as the basis for the risk characterization.

Other assessment endpoints which were not based on HQs, such as quantitative benthic community surveys, sediment toxicity tests, and evaluation of amphibian tissue COPCs, were evaluated by comparing Site conditions to background reference areas. SEM and AVS data were compared to threshold concentrations established by USEPA (USEPA, 2005).

CONCLUSIONS

AOI 6 – Sphagnum Bog

The CTE HQs for cadmium, copper, lead and uranium in surface water were substantially above unity for both total and incremental risk. Two other metals in surface water (beryllium and silver) also exceeded unity for both total and incremental risk, but to a lesser extent. The sediment benchmark evaluations for Aroclor-1254 and Aroclor-1260 suggest that risk to the benthic community from these two COPCs is possible in the mineral fraction of sediment in the Southwest Corner of the Sphagnum Bog. The sediment benchmark evaluations identified molybdenum, mercury and uranium as metals for which risk to the benthic community was characterized as possible. The HQs are generally more elevated in the Southwest Corner; however, the SEM/AVS evaluation indicated that divalent metals would not be bioavailable. Although the low pH of Sphagnum bog would render divalent metals more soluble than a neutral pH, it is also known that bogs are rich in dissolved organic matter (humic and fulvic acids) that bind metals rendering them less available for uptake into aquatic organisms.

Based on the weight-of-evidence, eight of the thirteen mineral sediment samples from AOI 6 - Sphagnum Bog demonstrate that there is no adverse impact to the benthic community (Table ES-2). Adverse effects in *C. dilutus* sediment toxicity tests at SD-RI-0600100R and SD-RI-0600900R appear to be explained by elevated sediment metal and PCB concentrations at those specific locations, which are within the Southwest Corner of the Sphagnum Bog. Benchmark evaluations were the only line of evidence used to evaluate risk to benthic invertebrates in moss and peat, and identified possible risk from molybdenum and uranium (low confidence). Medium weighted food chain models characterized risk to mallard duck and raccoon at AOI 6 – Sphagnum Bog as unlikely. Comparison of amphibian tissue COPC concentrations between AOI 6 and the reference bog revealed that PCBs and uranium are elevated in frog tissue, although the distribution was skewed (i.e. several samples were below the COPC detection limit in AOI 6). All sample concentrations were well below published Critical Body Burden values for copper, cadmium, lead and zinc for amphibian tissue.

Food chain models suggest that risk to great blue heron is possible from beryllium (high confidence) in mineral, peat and moss sediment fractions. Food chain models also suggest that in the mineral and peat fractions, risk to shrew is possible from molybdenum (low to moderate confidence) in all three sediment fractions. The highest concentrations of beryllium and molybdenum were detected in sediment samples SD-RI-06001 and SD-RI-06009 (which, as stated above, also had elevated of other trace metals and PCBs), and sample SD-RI-06032 which are located in the Southwest Corner. The food chain models, however, assume that 100% of the metals ingested are absorbed. Beryllium is poorly absorbed by animals (ATSDR, 2002) and molybdenum is not known to biomagnify. It can be concluded, therefore, that any risk of harm to the bog is overestimated due to the conservative nature of the food chain models. Notwithstanding these modeled results, the accessible (open water area) mineral sediment portion of the Southwest Corner of the Sphagnum Bog will be evaluated for remediation in the Feasibility Study using Preliminary Remediation Goals proposed by EPA.

AOI 10 – Northeast Wetland

Low/medium weighted benchmark comparisons characterized risk to benthic invertebrates and amphibians at AOI 10 - Northeast Wetland as possible in surface water due to copper, lead, and manganese and as unlikely in sediment (Table ES-2). Medium-weighted food chain models characterized risk to wetland birds and mammals as unlikely. Based on the weight of evidence and confidence and uncertainties in the data, ecological risk at AOI 10 – Northeast Wetland is unlikely.

AOI 18A - Assabet River Main Channel

Low/medium weighted benchmark comparisons characterized risk to benthic invertebrates, amphibians, and fish at AOI 18A – Assabet River Main Channel as unlikely for surface water and sediment (Table ES-2). Medium weighted food chain models characterized risk to wetland birds and mammals as unlikely. The lack of "rare" mussels is likely more attributable to the many impoundments that lie on the Assabet River that impede or prevent movement of fish (which act as hosts so are important in completing the mussel life cycle) upstream. Based on the weight of evidence and confidence and uncertainties in the data, ecological risk at AOI 18A – Assabet River Embayment is unlikely.

AOI 18B - Assabet River Embayment Area

Low/medium weighted benchmark comparisons characterized risk to benthic invertebrates, amphibians, and fish at AOI 18B – Assabet River Embayment as unlikely for surface water and sediment (Table ES-2). Medium weighted food chain models characterized risk to wetland birds and mammals as unlikely. Based on the weight of evidence and confidence and uncertainties in the data, ecological risk at AOI 18B – Assabet River Embayment is unlikely.

Site-Wide Surface Soil

Low/medium weighted benchmark comparisons characterized risk to plant and terrestrial invertebrates in site-wide soils as highly unlikely for all COPCs except uranium which was characterized as possible (high confidence) (Table ES-2). Medium weighted food chain models characterized risk to mammals as unlikely; food chain models for cardinal and American robin characterized risk as possible for Aroclor-1254 (low to moderate confidence), Aroclor-1260 (low to moderate confidence), and zinc (low confidence). Based on the weight of evidence and confidence and uncertainties in the data, ecological risk in site-wide soils is possible from uranium (plants and soil invertebrates), Arcolor-1254 (birds), and Aroclor-1260 (birds).

Overall, based on the marginal risks posed by the Site, it can be concluded that, other than a presumptive remedy that has already been decided for the cooling water pond (AOI 4), and sediment risk in the Southwest Corner of the Sphagnum Bog, there is no need for further remedial action to mitigate risks to the environment at any of the AOIs as a result of Site-related COPCs.

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ACRONYMS

AET	Apparent Effects Threshold
ANOVA	Analysis of Variance
AOC	Administrative Order by Consent
AOI	Areas of Interest
AQUIRE	Aquatic Information Retrieval
ARCS	Assessment and Remediation of Contaminated Sediments
ASTM	American Society for Testing and Materials
AVS	Acid Volatile Sulfide
AWQC	Ambient Water Quality Criteria
BAF	Bio-Accumulation Factors
BERA	Baseline Ecological Risk Assessment
bgs	below ground surface
BSAF	Biota-Sediment Accumulation Factors
BTAG	Biological Technical Assistance Group
$CaCO_3$/L	Calcium Carbonate per Liter
CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act
CISP	Community Involvement Support Plan
CLI	Community Loss Index
COPC	Contaminants of Potential Concern
CREW	Citizens Research and Environmental Watch
CSM	Conceptual Site Model
CTE	Central Tendency Exposure
CV	Consensus Value
DOD	Department of Defense
DQO	Data Quality Objective
ECOSAR	Ecological Structure Activity Relationship
EcoSSL	Ecological Soil Screening Level
EDI	Estimated Daily Intake
EE/CA	Engineering Evaluation/Cost Analysis
EPC	Exposure Point Concentration
EqP	Equilibrium-Partitioning
ER-L	Effects Range-Low
ER-M	Effects Range-Median
ESB	Equilibrium partitioning Sediment Benchmark
ESBTU	Equilibrium Sediment Benchmark Toxicity Unit
ESG	Equilibrium-partitioning Sediment Guideline
EPH	Extractable Petroleum Hydrocarbons
ET	Ecotox Threshold
FETAX	Frog Embryo Teratogenesis Assay – *Xenopus*
FSP	Field Sampling Plan
foc	Fraction Organic Carbon
ft	feet

GIS	Geographic Information System
GLEC	Great Lakes Environmental Center
GMP	Groundwater Modeling Plan
HASP	Health and Safety Plan
HBI	Hilsenhoffs Biotic Index
HQ	Hazard Quotient
LCV	Lowest Chronic Value
LEL	Lowest Effect Level
LOAELs	Lowest-Observable-Adverse-Effects Levels
MACTEC	MACTEC Engineering and Consulting, Inc.
MANHESP	Massachusetts Natural Heritage and Endangered Species Program
MassDEP	Massachusetts Department of Environmental Protection
μmole/gOC	Micromole per gram of Organic Carbon
mg	Milligrams
mg/kg BW-day	Milligrams per Kilogram Body Weight Day
NMI	Nuclear Metals, Inc.
NOAA	National Oceanic and Atmospheric Administration
NOAELs	No-Observable-Adverse-Effects Levels
NPL	National Priorities List
NRCS	Natural Resources Conservation Service
NWI	National Wetlands Inventory
OMOE	Ontario Ministry of the Environment
ORNL	Oak Ridge National Laboratory
PAET	Probable Apparent Effects Threshold
PAH	Polynuclear Aromatic Hydrocarbons
PAL	Project Action Level
PCB	Polychlorinated Biphenyl
PEC	Probable Effects Concentration
PSOP	Project Summary and Operations Plan
PSS1E	Palustrine, Scrub-Shrub, Broad-Leaved Deciduous, Seasonally Flooded/Saturated Area
PSS3Ba	Palustrine, Scrub-Shrub, Broad-Leaved Evergreen, Saturated, and Acidic
PUBHx	Palustrine, Unconsolidated Bottom, Permanently Flooded, Excavated Area
QAPP	Quality Assurance Project Plan
RAGS	Risk Assessment Guidance for Superfund
RAP	Risk Assessment Plan
RI	Remedial Investigation
RI/FS	Remedial Investigation/Feasibility Study
RME	Reasonable Maximum Exposure
SAP	Sampling and Analysis Plan

SARA	Superfund Amendments and Reauthorization Act
SAV	Secondary Acute Value
SCV	Secondary Chronic Value
SEL	Severe Effect Levels
SEM AVS	Simultaneously Extracted Metals/Acid Volatile Sulfide
SFF	Site Foraging Frequency
SLERA	Screening Level Ecological Risk Assessment
SMP	Site Management Plan
SQB	Sediment Quality Benchmark
SQC	Sediment Quality Criteria
SQG	Sediment Quality Guideline
SQL	Sample Quantitation Limit
SQV	Sediment Quality Value
SVOC	Semi-Volatile Organic Compound
TEC	Threshold Effect Concentrations
TOC	Total Organic Carbon
TRV	Toxicity Reference Value
UCL	Upper Confidence Limit
USDA	United States Department of Agriculture
USEPA	United States Environmental Protection Agency
USFWS	United States Fish and Wildlife Service
USGS	United States Geological Survey
UST	Underground Storage Tank
VOC	Volatile Organic Compound

1. INTRODUCTION

On June 14, 2001, the Nuclear Metals, Inc. (NMI) Site (the Site) located at 2229 Main Street (Route 62) in Concord, Massachusetts (Figure 1-1) was added to the United States Environmental Protection Agency's (USEPA's) National Priorities List (NPL) (66 Federal Register 32235, 32241), established under section 105(a)(8)(B) of the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA), as amended by the Superfund Amendments and Reauthorization Act (SARA).

Following the inclusion of the Site on the NPL, USEPA Region 1, in agreement with the Massachusetts Department of Environmental Protection (MassDEP), issued an Administrative Order by Consent (AOC) on June 13, 2003, requiring the completion of a Remedial Investigation/Feasibility Study (RI/FS) and, if directed, an Engineering Evaluation/Cost Analysis (EE/CA) for the Site. The RI/FS is conducted to assess Site conditions, investigate the nature and extent of Contaminants of Potential Concern (COPCs) and evaluate remedial alternatives and costs to the extent necessary to select a remedy for the Site. The Site is defined as a 18.8 hectare property (46.4 acre) that houses facilities formerly used for specialty metals research and manufacture, and off-property areas potentially affected by historic discharges. Eighteen Areas of Interest (AOIs) have been identified at the Site (Figure 1-2).

A draft Screening Level Ecological Risk Assessment (SLERA) for the NMI Site was submitted in June 2006 (*de maximis,* 2006a). The SLERA evaluated potential risk associated with exposure to chemicals in Site surface soil, surface water, and sediment and identified chemicals at the following ecologically relevant AOIs during the Phase 1A and Phase 1B field programs:

- AOI 4 – Cooling Water Recharge Pond
- AOI 6 – Sphagnum Bog
- AOI 10 – Northeast Wetland
- AOI 18A – Assabet River Main Channel
- AOI 18B – Assabet River Embayment Area
- Site-Wide Soil (principally includes AOI-14 and AOI-8 but also includes parts of AOIs 1, 2, 3, 7, 8, 9, 10, and 11 that are not located below pavement or buildings, and parts of AOI 4 that are not considered inundated sediments).

The June 2006 SLERA concluded that risk of adverse chemical effects to ecological receptors could not be screened out, and that these AOIs should be further evaluated as part of a Baseline Ecological Risk Assessment (BERA).

The June 2006 SLERA (*de maximis,* 2006a) also evaluated potential ecological risk associated with exposure to radionuclides in all Site media. The SLERA provided a demonstration that radioactivity levels reported in surface water, sediment, and soil samples collected at the Site were well below screening benchmarks, indicating that there is no significant risk to ecological receptors from radiation exposures and, therefore, radionuclides and/or radiation exposure could be eliminated from further evaluation. Therefore, radiological risk to ecological receptors is not discussed further in this BERA.

Data gaps identified by the SLERA were addressed in 2006, 2007, and 2008 as part of the Phase 1C field investigation. Per agreement with all parties, COPCs were re-screened using the pooled Phase 1A, 1B, and 1C data sets (Appendix A, Table A-14 through A-16). Table 1-1 through Table 1-3 identify which COPCs were carried forward through the BERA.

This BERA was prepared by Haley and Aldrich using tables, figures, and calculations that were completed by MACTEC Engineering and Consulting and provided by *de maxims*. The BERA was completed in accordance with the approach described in Volume V (the Risk Assessment Plan (RAP) of the RI/FS Work Plan (*de maximis*, 2005a). The specific objective of this BERA is to evaluate the risk of ecological harm associated with Site-related contaminants. This BERA evaluates data collected by *de maximis* during the Phase 1A, Phase 1B, and Phase 1C of the Remedial Investigation (RI) field programs (Table 1-4, Table 1-5, and Figure 1-3 through Figures 1-10). The BERA also incorporates USEPA comments received June 10, 2006 and February 25, 2011, and April 5, 2012; as well as comments from Citizens Research and Environmental Watch (CREW) received on November 17, 2006 and August 19, 2011.

The remainder of this section discusses the following:

- Approach and Regulatory Framework (Section 1.1)
- Site Location and History (Section 1.2)
- Investigations (Section 1.3)
- Report Organization (Section 1.4)

1.1 Approach and Regulatory Framework

The ecological risk assessment process at the NMI Site follows that prescribed in the USEPA *Ecological Risk Assessment Guidance for Superfund: Process for Designing and Conducting Ecological Risk Assessments* (USEPA, 1997), herein referred to as the ERA Guidance. The following documents will also be used as guidance for conducting the BERA:

- Framework for Ecological Risk Assessment. USEPA, Risk Assessment Forum, Washington, DC, EPA/630/R-92/001. February 1992 (USEPA, 1992);
- Risk Assessment Guidance for Superfund (RAGS), Volume II: Environmental Evaluation Manual (USEPA, 1989b);
- Supplemental Guidance to RAGS: Region 4 Bulletins, Ecological Risk Assessment (USEPA, 2001a);
- Guidelines for Ecological Risk Assessment (USEPA, 1998a);
- The Role of Screening-Level Risk Assessments and Refining Contaminants of Concern in Baseline Ecological Risk Assessments, ECO Update (USEPA, 2001b); and
- ECO Updates published by USEPA between 1991 and 2008 (USEPA, 1991-2008).

The ERA Guidance outlines an eight-step tiered approach to ecological risk assessment. The eight-step approach consists of two tiers. The first tier includes Step 1 (Screening-Level Problem Formulation) and Step 2 (Screening-Level Exposure Estimate and Risk Calculation), which comprise the SLERA.

The second tier is a BERA that evaluates COPCs in greater detail and in the context of Site-specific factors. The second tier includes Step 3 through Step 8 of the ERA Guidance. Step 3 (Problem Formulation) and Step 4 (Study Design and Data Quality Objectives (DQO) Process) were incorporated into the BERA Study Design. Step 5 (Verification of Field Sampling Design) and Step 6 (Site Investigation and Data Analysis) were completed as part of the field sampling program. Step 7 of the ERA Guidance (Risk Characterization) is principally addressed in the BERA. Step 8 as described in the ERA Guidance is Risk Management. This BERA will assist EPA in making risk management decisions at the NMI Site.

1.2 Site Location and History

An 18.8-hectare (46.4 acre) specialty metals research and manufacturing facility has operated at 2229 Main Street in the western portion of the Town of Concord, Middlesex County, Massachusetts (Figure 1-1) from 1958 to the present. Starmet Corporation was the most recent and current owner. The facility is bordered by Main Street (Route 62) and several commercial and residential properties to the north, residential properties to the east, woodland and commercial/ industrial properties to the west, and woodland and residential properties to the south. A day camp is located southwest of the facility. As currently configured, the facility includes eight interconnected buildings, several smaller outbuildings, paved parking areas, a cooling water recharge pond, a former waste holding basin, and areas of fill and waste materials. A 1.5-hectare (3.5 acre) sphagnum bog located on the eastern portion of the facility property is another predominant feature. The Assabet River runs approximately 300 feet (ft) north of the Site.

Past facility operations involved research and development in fundamental metallurgy, physical metallurgy, chemical metallurgy, engineering and product development, fuel element development and manufacture, and high temperature materials (NMI, 1961). Most of the operations at the Site were for the United States Atomic Energy Commission and the Department of Defense (DOD). Materials for missiles, airframes, and other components for private industry were also investigated and developed.

The focus of facility operations shifted from research and development to large scale production in the mid-1970s. The facility manufactured depleted uranium products (including shields, counter weights, and armor penetrators), metal powders, and products made of beryllium, beryllium alloy, and titanium. A more detailed description of facility operations is discussed in the Site's Project Summary and Operations Plan (PSOP) (*de maximis* 2005b).

1.3 Investigations

To address concerns regarding groundwater quality and other environmental issues, a series of investigations was conducted at the site throughout the 1980s and 1990s, as described in further detail in the PSOP (*de maximis* 2005b). Data from these previous investigations are not included in this BERA as data quality is questionable (*de maximis* 2005b).

In December 2003, *de maximis* submitted a draft RI/FS Work Plan which consisted of the PSOP, Site Management Plan (SMP), Sampling and Analysis Plan (SAP), Risk Assessment Plan (RAP), Groundwater Modeling Plan (GMP), Health and Safety Plan (HASP), and Community Involvement Support Plan (CISP). Reviewer comments were incorporated and a revised SAP, consisting of a Field Sampling Plan (FSP) and Quality Assurance Project Plan (QAPP), HASP, and SMP were re-issued in September 2004 (*de maximis*, 2004a,b,c,d). Other components of the RI/FS Work Plan (RAP, PSOP, GMP, CISP) were re-issued in April 2005 (*de maximis*, 2005a,b,c,d).

The first phase of the RI field program (Phase 1A) occurred during an initial field mobilization from September 2004 to December 2004, and a second field mobilization from March 2005 to May 2005. Phase 1A activities focused on gathering data across various site media (surface and subsurface soil, sediment, surface water, and groundwater) to characterize the nature and extent of contamination from historic Site activities at each of the 18 Site AOIs. Samples were analyzed for chemical constituents including volatile organic compounds (VOCs), semi-volatile organic compounds (SVOCs), polychlorinated biphenyls (PCBs), and inorganic compounds. Samples were also analyzed for radionuclides (*de maximis,* 2004b). In May and June of 2004, on-Site habitats were qualitatively assessed and a portion of the wetland boundary along the on-Site Cooling Water Recharge Pond was delineated. Candidate background locations for Site wetland and upland areas were identified using public information, and field reconnaissance of those areas were conducted (MACTEC, 2004).

Based on results of the Phase 1A investigation, a Phase 1B field program was initiated in October to November 2005 to further characterize the extent of contamination at the Site. The Phase 1B field event included additional sampling in surface water, sediment, soil, and groundwater for both chemicals (VOCs, SVOCs, PCBs, inorganic compounds) and radionuclides. The validated results from the Phase 1A and Phase 1B RI field programs were uploaded into a database and evaluated in the SLERA.

The Phase 1C Field investigation was undertaken in 2006 and 2007, with minor re-sampling activities in 2008. Phase 1C included activities to address data gaps and regulator comments as identified in the SLERA, and as outlined in the Risk Assessment Study Design (*de maximis,* 2006b). Phase 1C Field activities included collection of soil, surface water, and sediment samples for chemical analysis of SVOCs, PCBs, and metals. Sediment samples were collected for toxicity tests using *Chironomus dilutus*. Aquatic invertebrate and frog tissue was collected and analyzed for SVOCs, PCBs, metals, and lipids. A benthic community survey was also performed as part of the Phase 1C investigation.

1.4 Report Organization

The remainder of this BERA is organized as follows:

- Section 2.0 presents the Problem Formulation and Ecological Conceptual Site Model (CSM);
- Section 3.0 presents the Exposure Assessment and Effects Assessments;
- Section 4.0 presents the Risk Characterization;
- Section 5.0 presents the Uncertainty Analysis; and
- Section 6.0 presents Conclusions.

Figures and tables are respectively included in "Figures" and "Tables" sections of the report. Supporting information is provided in the following appendices:

A COPC Selection

B Agency Consultations

C United States Geological Survey (USGS) Powdermill Dam Sediment Study

D Validated Data Used in the BERA

E 95th Percentile Upper Confidence Limit (UCL) calculations

F Total Polycyclic Aromatic Hydrocarbon (ΣPAH) Calculations

G Simultaneously Extracted Metals/Acid Volatile Sulfide (SEM/AVS) Calculations

H Sediment Toxicity Test Laboratory Report *(Chironomus dilutus)*

I FETAX Laboratory Report

J Benthic Community Survey Report

K Food Chain Models

L Sediment Toxicity Dose-Response Curves

M FETAX Toxicty Test Interpretation from EPA, April 22, 2014

N Assabet River Background & Site Statistical Comparisons

O FETAX Toxicity Test Interpretation and Surface Water Criteria Development

2. PROBLEM FORMULATION

2.1 Environmental Setting

This section describes the regional and local setting, dominant habitats, and natural communities at or bordering the Site and background (reference) areas. The environmental setting was initially investigated in May 2004 as documented in the RI/FS PSOP (*de maximis,* 2005b) and in a technical memorandum titled *Wetland Delineation and Identification of Background Sampling Areas* (MACTEC, 2004).

2.1.1 Regional and Local Setting

General information relative to topography, geology, and hydrogeology at the regional and local levels is provided in the following subsections. More specific details are provided in the PSOP (*de maximis,* 2005b).

2.1.1.1 Topography

The topography of the Site is characterized by typical glacial kame and kettle features that produce irregular steep-sided hills and closed depressions. The surface elevation of the Site varies from approximately 137 ft to 213 ft above mean sea level, rising generally from north to south.

2.1.1.2 Geology

The surface geology of the Site consists of unconsolidated glacial deposits (glacial overburden) that varies generally from 30 ft to over 150 ft in thickness. The overburden is primarily a loose mixture of silt, sand, and gravel and varies from directly fine-grained outwash silt to coarse sand and gravel with boulders. Below the overburden and above the bedrock is till, a denser ice-contact deposit that generally consists of a broad spectrum of grain sizes. Bedrock is found from approximately 40 ft to 150 ft beneath the ground surface and the surface of the bedrock slopes generally from southeast to northwest beneath the Site. Drill cores indicate that the bedrock consists of metamorphic gneiss.

2.1.1.3 Hydrogeology

The only on-property surface water body that pre-dates Site development is a sphagnum bog located in the eastern-central portion of the Site. A man-made cooling water pond, filled principally with storm water runoff from the facility, occupies a natural depression to the west of the bog. Water surfaces of both the bog and the cooling water recharge pond are perched above the permanent water table and neither water body has an overland drainage outlet. North of the facility, the Assabet River flows in an easterly direction and empties into the Sudbury River approximately 3.5 miles from the Site. Several impoundments (dams) are located along the Assabet River upstream and downstream of the Site.

Groundwater is present in both the overburden and in bedrock fractures. Overburden groundwater flows generally northward, towards the Assabet River. Depth to groundwater at the Site ranges from 5 ft below ground surface (bgs) near the Assabet River to the north, to 60 ft bgs in the upland portions of the Site to the south and northwest. The saturated thickness of the overburden varies from less than 10 ft at the northwestern edge of the property to over 80 ft along the eastern property boundary. In the underlying bedrock, the piezometric gradient is from the southeast towards the northwest, indicating that groundwater migrates northwest.

2.1.2 Natural Communities

The natural communities present at or in the vicinity of the Site were initially documented in the RI/FS PSOP (*de maximis*, 2005b) and in a technical memorandum, titled *Wetland Delineation and Identification of Background Sampling Areas* (MACTEC, 2004). The following sections describe the natural communities that inhabit various aquatic and wetland AOIs (Section 2.1.4.1) and terrestrial AOIs (Section 2.1.4.2) located at or in the vicinity of the Site.

2.1.2.1 Aquatic and Wetland Communities

Cooling Water Recharge Pond (AOI 4)

The Cooling Water Recharge Pond (AOI 4) is a roughly oval-shaped permanent water body with no outlets. It occupies approximately 0.24 hectares (0.59 acre) in the central portion of the facility property and is classified under the National Wetlands Inventory (NWI) nomenclature (Cowardin *et al.*, 1979) as a palustrine, unconsolidated bottom, permanently flooded, excavated area (PUBHx). The Cooling Water Recharge Pond is not naturally occurring and was constructed by human activity. The pond is located in a natural topographic depression and was created by placing a sand dam across the swale. Soils in the vicinity of the Cooling Water Recharge Pond have been mapped by the United States Department of Agriculture (USDA) Natural Resources Conservation Service (NRCS) as Hinckley loamy sand, with a 25 to 35% slope (Soil No. 35E) (NRCS, 1995).

Topography around AOI 4 slopes gently to the water's edge in areas south and southeast of the pond, and is moderately to steeply sloping in surrounding upland areas located north, northeast and west of the pond. An approximately 15-ft high gabion retaining wall is located north of the pond. During a Site visit in May 2004, MACTEC Engineering and Consulting, Inc. (MACTEC) personnel observed minimal vegetation in the pond, including approximately two willow (*Salix sp.*) shrubs, which were growing in areas of shallow water (*de maximis*, 2006a). Vegetation growing along the shores of the pond includes bindweed (*Polygonum sp.*) and smartweed (*Polygonum sp.*).

Upland vegetation surrounding the pond is dominated by a canopy of red oak (*Quercus rubra*), eastern white pine (*Pinus strobus*), red maple (*Acer rubrum*), and white oak (*Quercus alba*), with an understory of black birch (*Betula lenta*), red maple (*Acer rubrum*), and eastern hemlock (*Tsuga canadensis*). The shrub layer is dominated by black birch and glossy buckthorn (*Rhamnus frangula*); and groundcover dominated by Virginia creeper (*Parthenocissus quinquefolia*), Canada mayflower (*Maianthemum canadense*), and seedlings of maple-leaved viburnum (*Viburnum acerifolium*). A thick layer of leaf litter covers the ground surface in upland areas near the pond.

During field activities conducted in 2004 and 2005, numerous bullfrogs (*Rana catesbeiana*) were observed and heard chorusing in the pond. Other wildlife observed in the pond included an eastern garter snake (*Thamnophis sirtalis sirtalis),* and aquatic invertebrates, including water boatmen and whirligig beetles (*de maximis*, 2006a).

Sphagnum Bog (AOI 6)

The Sphagnum Bog (AOI 6) is located in the eastern portion of the facility property, immediately north of the Old Landfill (AOI 3). It is roughly square-shaped with a triangular projection along its northern side and occupies approximately 1.5 hectares (3.7 acres). Topography slopes moderately toward the bog from the south and west. Topography slopes gently towards the bog in areas to the north and east. The NWI classifies the Sphagnum Bog as palustrine, scrub-shrub, broad-leaved evergreen, saturated, and acidic (PSS3Ba) (Cowardin *et al.*, 1979). The NRCS mapped soils in the Sphagnum Bog as Freetown muck, ponded (Soil No. 99) (NRCS, 1995). Water in the bog is elevated approximately 10 ft to 13 ft above the underlying groundwater. This condition is likely due to the relatively low vertical permeability of the sediments beneath the bog.

Highbush blueberry (*Vaccinium corymbosum*), red maple shrubs, and black chokeberry (*Photinia melanocarpa*) are found along the perimeter of the bog. Clumps of sphagnum moss (*Sphagnum* sp.) are present among the shrubs. Low vegetation predominates in the center of the pond. The central area contains extensive sphagnum mats with low shrubs including leatherleaf (*Chamaedaphne calyculata*) and rhodora (*Rhododendron canadense*). Scattered throughout this central area are occasional shrubs and saplings of red maple, eastern white pine, and larch (*Larix laricina*). Occasional snags are present along the bog perimeter.

Heavy leaf litter is present in upland areas surrounding the bog. Plant species observed growing in upland areas surrounding the bog include eastern hemlock, red maple, and eastern white pine in the canopy; black birch and red maple in the understory; and dangleberry (*Gaylussacia frondosa*), black birch, and gray birch (*Betula populifolia*) in the shrub layer. Virtually no groundcover was observed in the surrounding upland area due to the heavy leaf litter.

During field activities conducted in 2004 and 2005, spotted salamander (*Ambystoma maculatum*) egg masses, woodfrog (*Rana sylvatica*) egg masses, and woodfrog tadpoles were observed in the bog, and spring peepers were heard chorusing from the bog. Deer have been observed along the edge of the bog and woodpecker cavities were observed in surrounding snags and trees.

Northeast Wetland (AOI 10)

The Northeast Wetland (AOI 10) is located in the northeastern portion of the facility property, immediately south/southeast of Main Street. The Northeast Wetland is a seasonally flooded isolated wetland, extends in a northeast/southwest direction, occupies approximately 0.28 hectares (0.7 acres), and exhibits a relatively open canopy. The NWI classifies the Northeast Wetland as a palustrine, scrub-shrub, broad-leaved deciduous, seasonally flooded/saturated area (PSS1E). The NRCS mapped soils in the Northeast Wetland as Swansea muck (Soil No. 45) (NRCS, 1995).

Numerous wood frog tadpoles were observed by MACTEC personnel in May 2004 within the northeastern portion of the wetland, which was flooded at that time. Deer have been observed in the wetland. In June 2004, the Northeast Wetland was observed to be completely dry (*de maximis*, 2006a).

The central portion of the wetland contains primarily herbaceous vegetation, including false nettle (*Boehmeria cylindrica*), purple loosestrife (*Lythrum salicaria*), and sedges (*Carex sp.*). Clumps of highbush blueberry and red maple shrubs are scattered throughout.

Fallen branches and logs are present through the Northeast Wetland. Vegetation located along the wetland perimeter is generally dominated by a canopy of red maple and red oak with an understory of red maple, American elm (*Ulmus Americana*), and eastern hemlock. The shrub layer includes highbush blueberry, red maple, glossy buckthorn, and American elm, intertangled with grape (*Vitis* sp.). Groundcover species observed along the edges of the wetland include jewelweed (*Impatiens capensis*), sensitive fern (*Onoclea sensibilis*), and sedges (*de maximis*, 2006a).

Assabet River Main Channel (AOI 18A)

The Assabet River is located approximately 400 ft north of Main Street/Route 62. The river adjacent to the facility property is approximately 50 ft wide and flows west to east. Red maple dominates the canopy. Shrub layer species include glossy buckthorn, Japanese barberry (*Berberis thunbergii*), black cherry (*Prunus serotina*), and Morrow's honeysuckle (*Lonicera morrowii*). Virginia creeper and American elm seedlings are present in the groundcover. During field activities conducted in 2004, MACTEC personnel observed raccoon (*Procyon lotor*) prints along the southern riverbank (*de maximis*, 2006a). The nearest impoundment is located approximately 0.9 miles upstream of the Site (Zimmerman and Sorenson, 2003).

Assabet River Embayment Area (AOI 18B)

AOI 18B is an approximately 0.06 hectare (0.15 acre) backwater embayment area of the Assabet River Main Channel (AOI 18A). The embayment is located between the river and Route 62. Runoff from AOI 9 (Pavement Drain Outfalls) discharges through a three-foot diameter storm onto 30 ft of rip-rap located within AOI 18B.

AOI 18B is characterized by an emergent wetland, dominated by herbaceous vegetation including tussock sedge (*Carex stricta*), fringed sedge (*Carex crinita*), cattail (*Typha latifolia*), false nettle, sensitive fern, arrow arum (*Peltandra virginica*), purple loosestrife, Jack-in-the-pulpit (*Arisaema triphyllum*), enchanter's nightshade (*Circaea quadrisulcata*), wild oats (*Uvularia sessilifolia*), wood fern (*Dryopteridaceae* sp.), and deer-tongue grass (*Panicum cladestinum*). Shrubs observed in this area include multiflora rose (*Rosa multiflora*), elderberry (*Sambucus canadensis*), black cherry, American elm, glossy buckthorn, poison ivy (*Toxicodendron radicans*), and Virginia creeper. Red maple trees were observed along the wetland perimeter.

2.1.2.2 Terrestrial Communities

Upland communities at the Site were characterized in June 2004 as part of a qualitative ecological assessment, as described in the technical memorandum, titled *Wetland Delineation and Identification of Background Sampling Areas* (MACTEC, 2004). Upland habitat at the Site is principally comprised of AOI 14 at the northwestern, northeastern, and southwestern corners of the facility property, and AOI 8 at the southeastern corner of the facility property. Other incidental upland habitat that includes portions of AOIs 1, 2, 3, 7, 8, 9, 10, and 11 that are not located below pavement or buildings, and portions of AOI 4 that are not considered inundated sediments, comprised a very small proportion of Site terrestrial habitats, so were not characterized in detail and are not discussed below.

Soil in the northwestern corner of the facility is mapped as Hinckley loamy sand, 3 to 8% slopes (Soil No. 35B) (NRCS, 1995). This portion of the uplands has a canopy dominated by red maple, oak, and eastern white pine. Black birch trees are present along the eastern edge of this area. Black oak (*Quercus velutina*) and white oak saplings were observed in the understory. The shrub and groundcover layers are generally sparse. A thick leaf litter covers the ground surface. Species observed in the shrub layer included early-low blueberry, black oak, red oak, eastern white pine, black cherry, highbush blueberry, and multiflora rose. Canada mayflower, wild oats, striped wintergreen (*Chimaphila maculata*), bracken fern, and seedlings of red maple, red oak, eastern white pine, and nannyberry (*Viburnum lentago*) were observed in the groundcover.

The northeastern corner of AOI 14, located south and east of the Northeast Wetland, contains soils that are mapped as Hinckley loamy sand, 25 to 35% slopes (Soil No. 35E) (NRCS, 1995). Vegetation located along the slope bordering the Northeast Wetland and north of paved areas is dominated by a canopy of red maple, eastern white pine, red oak, white birch, and black birch, with saplings of American elm and white birch. Shrubs observed in this area included American elm, multiflora rose, black raspberry, tree-of-heaven (*Ailanthus altissima*), black cherry, glossy buckthorn, poison ivy, and dense entanglements of oriental bittersweet (*Celastrus orbiculatus*). Common mullein (*Verbascum thapsus*) and butter-and-eggs (*Linaria vulgaris*) were observed in the groundcover near the edge of pavement. The forested area east of the Northeast Wetland, in the vicinity of the on-site Pavement Drain Outfall (AOI 9), contains a canopy of black birch and black oak trees, with shrubs of eastern white pine, glossy buckthorn, and eastern hemlock.

The southwestern corner consists of a mixed deciduous-coniferous forest. NRCS mapped soils predominantly as Hinckley loamy sand, 15 to 25% slopes (Soil No. 35D) and as Windsor loamy sand, 3 to 8% slopes (Soil No. 67B) (NRCS, 1995). Red oak and eastern white pine are dominant in the canopy, with scattered individuals of black birch, white birch (*Betula papyrifera*), and white oak. Eastern hemlock saplings were observed in the understory. The shrub layer consists of scattered patches of black birch, late-low blueberry (*Vaccinium angustifolium*), and early-low blueberry (*Vaccinium pallidum*). Groundcover species include Canada mayflower, sedges, starflower (*Trientalis borealis*), hay-scented fern (*Dennstaedtia punctilobula*), bracken fern (*Pteridium aquilinum*), wood sorrel (*Oxalis* sp.), whorled loosestrife (*Lysimachia quadrifolia),* and bedstraw (*Galium sp*.). The southwestern portion of AOI 14 contains a thick leaf litter consisting of pine needles and leaves. Evidence of red squirrel feeding was observed in the southwestern area of AOI 14.

The southeastern corner of the facility property is characterized by a relatively open canopy and disturbed, mounded soils. The native soils are mapped as Hinckley loamy sand, 15 to 25% slopes (Soil No. 35D) (NRCS, 1995). The mounds of fill in this area reportedly include dredged (excavated) substrate from the Cooling Water Recharge Pond (*de maximis,* 2004a). Cottonwood (*Populus deltoides*) and black locust (*Robinia pseudoacacia*) saplings are present in the canopy along the edges of this area. Shrubs are sparsely scattered throughout this area. Species observed in the shrub layer included highbush blackberry (*Rubus allegheniensis*), red oak, glossy buckthorn, and eastern white pine. The groundcover contains large patches of crown vetch (*Coronilla varia*) with scattered patches of silvery cinquefoil (*Potentilla argentea*). Other species observed in the groundcover included sedges, bindweed, jewelweed, common mullein, common cinquefoil (*Potentilla simplex*), sheep sorrel (*Rumex acetosella),* common mugwort (*Artemisia vulgaris*), hay-scented fern, and deer-tongue grass. Evidence of deer grazing was observed.

2.1.3 Wildlife Observed On Site

According to Starmet Corporation personnel, mammalian wildlife observed on-site include white-tailed deer (*Odocoileus virginianus*), eastern cottontail (*Sylvilagus floridanus*), coyote (*Canis latrans*), fox (*Vulpes vulpes*), and woodchuck (*Marmota monax*). On May 20 and 21, 2004, MACTEC personnel observed numerous deer signs (tracks, scat, and grazed plants) across the Site, a woodchuck south of Building E, an eastern cottontail north of the Holding Basin, and scat, belonging to fox or possibly coyote, in the wooded area south of Building E. Red squirrel (*Tamiasciurus hudsonicus*), gray squirrel (*Sciurus carolinensis*), and chipmunk (*Tamias striatus*) were also observed on site.

The following avian species have been identified on site based on visual observation and/or vocalizations: mourning dove (*Zenaida macroura*), eastern wood-pewee (*Contopus virens*), northern cardinal (*Cardinalis cardinalis*), eastern phoebe (*Sayornis phoebe*), black-capped chickadee (*Parus atricapillus*), American crow (*Corvus brachyrhynchos*), European starling (*Sturnus vulgaris*), Baltimore oriole (*Icterus galbula*), song sparrow (*Melospiza melodia*), American goldfinch (*Carduelis tristis*), blue jay (*Cyanocitta cristata*), tufted titmouse (*Parus bicolor*), red-shouldered hawk (*Buteo lineatus*), American robin (*Turdus migratorius*), and woodpecker (*Picidae* sp.).

Reptiles and amphibians observed on site include garter snakes (observed in the Sweepings and Fill Area and Cooling Water Recharge Pond), bullfrogs (observed in the Cooling Water Recharge Pond and Sphagnum Bog), green frogs (observed in the Sphagnum Bog), and wood frogs (observed in the Northeast Wetland and Sphagnum Bog). In addition, movements strongly indicative of tadpoles were observed along the western edge of the Sphagnum Bog. Spotted salamander and woodfrog egg masses were observed in the Sphagnum Bog.

Based on information received from the Massachusetts Division of Fisheries and Wildlife (MANHESP, 2009b), the Assabet River supports a wide variety of fish species. Fisheries surveys have yielded 23 species in the Assabet River: American eel (*Anguilla rostrata*), golden shiner (*Notemigonus crysoleucas*), banded sunfish (*Enneacanthus obesus*), largemouth bass (*Micropterus salmoides*), black crappie (*Pomoxis nigromaculatus*), pumpkinseed (*Lepomis* *gibbosus*), blacknose dace (*Rhinichthys atratulus*), rainbow trout (*Onchorhynchus mykiss*), bluegill (*Lepomis macrochirus*), redbreast sunfish (*Lepomis auritus*), brook trout (*Salvelinus fontinalis*), redfin pickerel (*Esox americanus*), brown bullhead (*Ameiurus nebulosus*), spottail shiner (*Notropis hudsonius*), brown trout *(Salmo trutta*), tessellated darter (*Etheostoma olmstedi*), common carp (*Cyprinus carpio*), tiger trout (*Salmo trutta x Salvelinus fontinalis*), chain pickerel (*Esox niger*), white sucker (*Catastomus commersoni*), creek chubsucker (*Erimyzon oblongus*), yellow bullhead (*Ameiurus natalis*), fallfish (*Semotilus corporalis*), and yellow perch (*Perca flavescens*). Rainbow trout, brook trout, and brown trout may also be annually stocked in the spring.

2.1.4 Rare, Threatened, or Endangered Species

The Massachusetts Natural Heritage and Endangered Species Program (MANHESP, 2009a) and United States Fish and Wildlife Service (USFWS) were consulted regarding the presence of state- and federal-listed rare, threatened, or endangered species and priority habitat at and in the vicinity of the Site. The USFWS indicated that there are no federally listed species known to occur in the project area (USFWS, 2009). The MANHESP indicated that three species of freshwater mussels including the eastern pondmussel (*Ligumia nasuta*), triangle floater (*Alasmidonta undulata*), and creeper (*Strophitus undulates*) have been historically observed in surface waters in the vicinity of the Site and are listed as species of special concern (MANHESP, 2009b). Responses from both agencies and fact sheets for these three state-listed special concern species are presented in Appendix B. Based on the habitat and substrate preferences for these species, it is likely that their potential presence at the Site would be limited to the main channel of the Assabet River.

2.1.5 Background Areas

Potential background locations were identified during the May 2004 qualitative ecological assessment and identified in *Technical Memorandum – Wetland Delineation and Identification of Background Sampling Areas* (MACTEC, 2004). Background locations were selected based on similarity to on-site areas for parameters including soil type, dominant vegetation, NWI classification, hydrology, topography, size, location, and surrounding land use.

Geographic Information System (GIS) software was used to initially identify prospective areas based on NWI classification, topography, hydrology, location, and size. Mapped soils data obtained from the NRCS Middlesex County Interim Soil Survey Report (1995) were used to identify locations with similar soil types. Candidate locations were preferentially selected from public lands, if a suitable match could be found. Areas east of the Site were generally avoided since this was inferred to be the predominant downwind direction from the Site. Locations with characteristics similar to site conditions were then visited by MACTEC personnel to further characterize potential background areas, including vegetative structure and composition, hydrology, and other location-specific observations. On November 11, 2004, USEPA visited and subsequently approved the proposed background locations (Figure 1-3, and Figure 1-6 through Figure 1-10).

Background locations corresponding to AOI 4 (Cooling Water Recharge Pond), AOI 6 (Sphagnum Bog), AOI 10 (Northeast Wetland), AOI 18A (Assabet River Main Channel), and AOI 18B (Assabet River Embayment Area) are discussed below.

Background Pond

The Maynard High School Pond was selected as the corresponding background area for the Cooling Water Recharge Pond. The Maynard High School Pond is located northeast of Maynard High School, south of Route 117 and upwind of the Site. The southwestern corner of the pond is bordered by urban land. The pond is surrounded by moderately sloping topography and soils that are mapped as loamy sand, and has vegetation that closely matches Cooling Water Recharge Pond vegetation. Additionally, the Maynard High School Pond and the Cooling Water Recharge Pond are similar in size and are both partially shaded by vegetation. Both ponds support populations of bullfrogs (MACTEC, 2004).

Background Bog

The Westford Bog was selected as the corresponding background area for the Sphagnum Bog. The Westford Bog is located near Westford Station north of the Springfield Terminal railroad tracks in Westford, MA. It is situated west of an unnamed road that is west of Depot Street, and is southeast of Burgess Pond. The bog is located on property owned by East Boston Camps. The Westford Bog has the same mapped soil type (Freetown muck) as the Sphagnum Bog, and is similar to the Sphagnum Bog with respect to size, vegetative structure, moat structure, and vegetative species. The Westford Bog is 1.6 hectares (4 acres) in size and has tall shrubs along its perimeter. There is a sphagnum mat and low shrubs in the center of the Westford bog with occasional scattered saplings and tall shrubs (MACTEC, 2004).

Background Wetland (Background for AOI 10 – Northeast Wetland)

A wetland located north of the Conant Well property in Acton, MA was selected as the corresponding background area for the Northeast Wetland (Figure 1-5). The Conant Well Property wetland is located over two miles upwind from the Site and near a roadway, similar to the Northeast Wetland. The background wetland is located east of Route 27 on property owned by the Town of Acton's Water Department, and is situated immediately north of the water department's driveway. The topography is relatively flat or gently sloping to the wetland. The background wetland has characteristics that closely resemble the Northeast Wetland, including plant species, vegetative structure, and soils mapped as Swansea muck. There are fallen branches and logs in the water, shrubs in the background wetland's center, and shrubs, ferns, and trees along its edge (MACTEC, 2004).

Background River Channel Locations

The background area selected for the Assabet River Main Channel is an approximately 2,800-ft-long stretch of the Assabet River immediately upstream of the Site and downstream of the Powdermill Dam.

Background Embayment Area

Two background areas were selected to represent AOI 18A – Embayment Area. The first background area is another embayment located on the south bank of Assabet River approximately 0.6 miles upstream of the Site. The background embayment is physically and hydrologically similar to the Site embayment area (i.e., a small backwater channel).

A second background embayment area is located at the Powdermill Impoundment, situated approximately 0.9 miles upstream of the Site Embayment. Sediment deposited behind the impoundment characterizes decades of non-facility related industrial inputs to the Assabet River which may have been deposited in the river's backwater embayments. Both data sets were combined to characterize the background embayment.

Background Soil

The Gardner Hill Town Forest in Stow, MA was identified as a background area for Site soils. The Gardner Hill Town Forest is located 3.7 miles southwest and upwind of the Site. The Gardner Hill Town Forest contains vegetation that closely matches the upland Site vegetation and has the same soil type (Hinckley loamy sand) as the Site (MACTEC, 2004).

2.2 COPC Sources, Transport, and Fate

This section summarizes historical releases that have previously been identified (Section 2.2.1), then discusses primary receiving media (Section 2.2.2), and finally discusses migration pathways and secondary receiving media (Section 2.2.3).

2.2.1 Historical Releases

Waste at the facility was historically generated through several manufacturing and laboratory processes, including the following:

- Depleted Uranium Operations – copper jacket removal
- Depleted Uranium Operations – machining and casting operations
- Machining Operations – dust collection systems and air handler units
- Research and Development – metal fabrication in support of DOD initiatives
- Thorium Operations – manufacturing of thoriated tungsten rods
- Beryllium Operations – beryllium fabrication
- Facility Support Processes – operation and maintenance of the facility
- On-Site electrical transformers.

2.2.2 Receiving Media

The wastes from the processes listed in Section 2.2.1 were discharged to the following receiving media across the Site (Figure 1-2):

- <u>AOI 1 - Holding Basin Soil.</u> Neutralized nitric acid solution containing dissolved copper and uranium was discharged to an unlined Holding Basin between 1958 and 1985. Various facility drain lines from the facility buildings also appear to have discharged to the Holding Basin. The primary receiving medium was vadose zone and saturated soil below, adjacent, and surrounding the Holding Basin, and groundwater below the Holding Basin.

- <u>AOI 2 - Buried Drum Area Soil.</u> Drums containing beryllium and possibly other materials were found in a buried trench located between the Cooling Water Recharge Pond and the Holding Basin. Soil and groundwater would be the primary receiving media from drums that may have leaked or been damaged.

- AOI 3 - Old Landfill Soil. The Old Landfill was reportedly used for disposal of solid waste that could have included materials from the research and development laboratories, drummed material containing various metals including uranium and beryllium, and municipal and office waste. The primary receiving medium from drums that may have leaked or been damaged was soil and groundwater in and beneath the landfill.

- AOI 4 - Cooling Water Recharge Pond Surface Water, Sediment, and Bank Soil. Building floor drains and roof drains discharged to the Cooling Water Recharge Pond. Roof drains are a potential source of dusts deposited from machine exhaust vents. The Cooling Water Recharge Pond also received direct discharge from the Holding Basin on at least two occasions. Non-contact cooling water pumped from on-Site wells that contained uranium was discharged into the Cooling Water Recharge Pond. The primary receiving media include surface water and sediment in the Cooling Water Recharge pond, and groundwater below the pond. In addition, sediments from the Cooling Water Recharge Pond may have been dredged and placed on the banks surrounding the pond in an effort to increase the pond's capacity. Therefore, soil surrounding the Cooling Water Recharge Pond may also be a primary receiving medium.

- AOI 5 - Septic Systems Soil. On-Site septic disposal has been used since the facility start-up in 1958. Septic systems could therefore have received Site-related chemical or radiological wastes. The primary receiving medium was soil and groundwater in and beneath the leach fields.

- AOI 6 - Sphagnum Bog Surface Water and Sediment. Supernatant liquid from the Holding Basin was reportedly discharged to the Sphagnum Bog between 1958 and possibly as late as the 1970s. In addition, sink and floor drains from laboratories located in Building A discharged to the Sphagnum Bog between 1958 and approximately 1975. The primary receiving media were surface water, sediment, and peat in the Sphagnum Bog.

- AOI 7 - Former Waste Handling Area Soil. Prior to the construction of Building E, an area located to the south of and beneath Building E was used for waste handling and storage. During that period, the disposal area was not paved. The primary receiving medium for material that may have been spilled or disposed of would have been soil.

- AOI 8 - Sweepings Area Soil. An area southwest of the main parking lot contains piles that reportedly include sweepings from building floors. The deposited material has soil-like characteristics (e.g., sand and gravel). Information collected during the RI suggests that at least some of the piles in this area may also consist of dredging spoils from the Cooling Water Recharge Pond.

- AOI 10 - Northeast Wetland Sediment. One historical aerial photograph (1981) indicates that a pipe existed in the Cooling Water Recharge Pond. Although it is not clear what the function of the pipe was or to where it may have discharged, the pipe may have controlled the water level and may have discharged to the Northeast Wetland. If this assumption is true, COPCs present in the recharge pond surface water could have been discharged to soils in the wetland area.

- AOI 11 - Drain Lines Soil. Drain lines carried process wastes, cooling water and storm water from the facility buildings to the Holding Basin, Sphagnum Bog, and Cooling Water Recharge Pond. If contaminated liquids leaked from underground piping, they would have been released to soil beneath the pipes, and potentially to groundwater. Drain lines are generally located beneath the area of land east of Buildings C and D.

- AOI 12 - Underground Storage Tanks Soil. The facility maintains two 10,000-gallon underground storage tanks (USTs) that were used to store heating oil, located north of Building B. If these tanks leaked, the oil would have migrated to soil (and potentially groundwater) around and beneath the tanks.

- AOI 14 - Down-Wind Surface Soils. Particulate emissions from the air handlers and stacks on the facility buildings may have migrated to ambient air and been deposited on surficial soils on-Site down-wind of the buildings.

- AOI 15 - Transformer Pads Soil. A transformer pad with one transformer located adjacent to Building B dates from facility start-up in 1958. A second pad with three transformers located east of Building D dates from construction of that building in 1978. Additional electrical units are located on the former switchgear pad located in the paved yard behind Buildings C and D. It is not known if the transformers ever contained petroleum-based dielectric fluid or if the fluid contained PCBs. If dielectric fluid spilled, the fluid would have been released to surface soil around the transformers.

2.2.3 Migration Pathways and Secondary Receiving Media

After deposition in the primary receiving media, COPCs may migrate to other media by three principal mechanisms: leaching, overland flow, and particulate re-suspension and deposition. Each of these mechanisms is discussed below, followed by a summary of secondary receiving media.

2.2.4 Leaching

Leaching refers to the dissolution of soluble constituents from a medium through percolation. Leaching can result in the movement or transport of a chemical within an environmental medium or to the transfer of a contaminant from one environmental medium to another. Leaching through soil or sediment is driven by infiltration from precipitation or discharge, whereas the migration of chemical constituents through groundwater or surface water is driven by the flow of the water.

COPCs released to the Sphagnum Bog migrated in surface water and adsorbed to sediment and peat within the bog. Infiltration due to precipitation and leaching could have caused additional migration of COPCs into deep sediments and possibly groundwater.

2.2.4.1 Overland Flow

Overland flow is the erosional process by which contaminants that are bound to surface particles such as soil are carried across the land surface by storm water runoff. Constituents from the Holding Basin were transported to the Cooling Water Recharge Pond via overland flow during at least one documented event when the soil berm that separates the Holding Basin and Cooling Water Recharge Pond gave way (*de maximis*, 2005b). In addition, if the Cooling Water Recharge Pond ever overflowed, the runoff could have breached the low point between the Pond and the Sphagnum Bog.

Overland flow could also have transported COPCs into catch basins located along paved facility surfaces. The three storm water outfalls associated with these catch basins discharge to:

- Surface soils near AOI 8 - Sweepings Areas;
- Surface soils to the north of the Septic System (ST2) leach field located in the northeast portion of the Site; and
- Surface soils on a slope above the Assabet River floodplain via a culvert beneath Route 62.

Substances on the pavement (e.g., metal particulates from airborne deposition, PAHs from parked automobiles) could be washed off during precipitation events, channeled to the storm water outfalls, and then transported to points down gradient from the storm water discharge outfalls. The topography of the storm water outfall near Septic System ST2F suggests that storm water runs toward the Sphagnum Bog, rather than the wetland area on the south side of Route 62. The storm water discharge from the culvert located beneath Route 62 flows through a 30-foot long area of rip-rap that extends north from the road. Storm water debris and sediment could accumulate in the rip-rap until a heavy-flood event occurs, which could potentially wash COPCs further down slope into the Embayment Area and then into the main channel of the Assabet River. COPCs that are transported to surface soils via overland flow could also potentially migrate through the soil column as a result of continued storm water infiltration in those areas.

The Holding Basin, Cooling Water Recharge Pond, and Sphagnum Bog are located in low lying areas of the Site. COPCs in surface soils on areas that slope toward these water bodies can migrate with storm water runoff to the water bodies.

2.2.4.2 Particulate Re-Suspension and Deposition

Soil particulates can be suspended into the air, transported by wind, and then resettle as dust. Constituents (generally non-volatile chemicals) that are adsorbed to soil particulates can be transported via this mechanism. This process may occur as a result of wind eroding exposed dry soil, or during the mechanical disturbance of dry soil through construction, excavation, or contact with vehicles. Generally, soil that is moist, vegetated, or has high organic carbon content is not likely to be suspended in air. Under the existing conditions at the Site, very little unvegetated or unpaved soil exists and no excavation activities are presently occurring; therefore, this is not a primary migration pathway to ecological habitats at the Site.

2.2.4.3 Secondary Receiving Media

Based on the information concerning sources of contamination and migration pathways, the following media are, or may have been, affected by releases from the Site:

- Residual soil/sludge at the bottom and beneath the Holding Basin;
- Surface and subsurface soil surrounding the Holding Basin;

- Surface water and sediment in the Cooling Water Recharge Pond;
- Surface and subsurface soils surrounding the Cooling Water Recharge Pond;
- Overburden and bedrock groundwater at the Site;
- Surface water, sediment, peat, and sphagnum in the Sphagnum Bog;
- Surface water and sediment in the Assabet River in proximity to the Site;
- Subsurface soil in the vicinity of the Drain Lines, beneath the building floors, in the Septic System Leachfield areas, and the UST Area (located north of Building B);
- Soil in the vicinity of the Old Landfill, Sweepings Area, Drum Burial Area, Hazardous Waste Area, Transformer Pads, and Parking Lot Outfalls;
- Surface water and sediment in the Northeast Wetland, south of Route 62; and
- Surface soils that may have received deposition from stack emissions from the Site.

2.3 Complete Exposure Pathways

COPCs may move from environmental media to ecological receptors through several major biological exposure pathways:

- Uptake of COPCs from sediment, surface water, shallow groundwater, and soil through roots (vegetation);
- Ingestion of COPCs bound to soil (terrestrial invertebrates, birds, mammals);
- Ingestion of COPCs bound to sediment (benthic invertebrates, fish, aquatic and wetland birds, mammals)
- Ingestion of dissolved and particulate COPCs in surface water (aquatic invertebrates, fish, semi-aquatic and wetland birds, mammals);
- Ingestion of COPCs through consumption of contaminated plants (herbivores, omnivores); and
- Ingestions of COPCs through consumption of contaminated prey (all predators).

Although inhalation and dermal absorption pathways are possibly complete for some receptors, these pathways are considered to be minor compared to dietary ingestion and are not evaluated.

The exposure pathway is considered incomplete for media located below pavement, buildings or other impervious surfaces that are considered inaccessible to ecological receptors. In addition, since groundwater does not directly discharge to the ground surface (e.g., through seeps), there are no direct exposures to groundwater by environmental receptors.

2.4 Nature and Extent

As described in the RI Report (*de maximis*, 2010), the nature and extent of potential contaminants have been sufficiently delineated. Sample locations, dates, and laboratory methods used to analyze samples are also fully described in the RI Report. In general, surface water samples from Site study areas and background areas were analyzed for total and dissolved metals (Method 6020/7470A) and hardness (Method SM 2340B). Sediment samples were analyzed for SVOCs (Method 8270C and Method 8310), PCBs (Method 8082), metals (Method 6020/7471A), and total organic carbon (TOC) (Method 9060 modified). Soil samples were analyzed for SVOCs (Method 8270C), polynuclear aromatic hydrocarbons (PAHs) (Method 8310), and metals (Method 6020/7471A).

Based on visual observations of patterns of TOC in the analytical data from AOI 6 - Sphagnum Bog, bog substrate was divided into three separate categories: mineral sediment, moss, and peat. In the context of AOI 6, mineral sediment refers to the granular mineral substrate. Moss refers to dense mats of plant material largely characterized by living *Sphagnum* sp. that dominate some sections of the bog. Peat refers a highly organic substrate consisting of decaying *Sphagnum* moss. The term sediment when used alone refers collectively to mineral sediment, moss, and peat.

As stated in Section 2.1.5, two background embayments were identified. Sediment samples collected in the first embayment background area located 0.6 miles upstream of the Site were analyzed as described in the preceding paragraphs. Sediments behind the Powdermill Impoundment were sampled by the USGS during a sampling and analysis program of several impoundments along the Assabet River (Zimmerman and Sorenson, 2005; see Appendix C). The USGS collected 12 sediment samples at the Powdermill Impoundment from the surface interval (0-12 inches) and analyzed them for metals (Method 6020). These data are incorporated into the BERA as shown on Table 1-5.

2.5 Selection of Potential Receptors

The types and numbers of ecological receptors are generally defined by the size and quality of the available habitat. AOI 4 - Cooling Water Recharge Pond, AOI 6 - Sphagnum Bog, AOI 10 - Northeast Wetland, AOI 18A - Assabet River Main Channel, and AOI 18B - Assabet River Embayment Area provide habitat for aquatic organisms including plants, benthic macroinvertebrates, fish, and wetland wildlife. Terrestrial portions of the Site provide habitat for a variety of terrestrial receptors, including plants, soil invertebrates, and wildlife. This section identifies ecological receptors for aquatic and wetland habitats (Section 2.4.1) and for terrestrial habitats (Section 2.4.2).

2.5.1 Aquatic and Wetland Receptors

<u>Aquatic Plants</u>
Aquatic plants growing in sediment may be exposed to (and possibly accumulate) COPCs from sediment porewater through root surfaces during periods of water and nutrient uptake. Emergent and submerged aquatic plants may also accumulate COPCs directly from surface water (Duxbury *et al.*, 1997). Plants may accumulate COPCs in roots, stems, leaves, or fruits, which get transferred to herbivores when consumed. Detritus may also contain COPCs, which may be consumed by detritivores. Herbivores and detritivores may, in turn, become a source of COPC exposure for secondary consumers.

<u>Benthic Macroinvertebrates</u>
Aquatic invertebrates, such as oligochaetes, amphipods, and aquatic life stages of some terrestrial insects may be exposed to and accumulate COPCs in sediment and surface water. Benthic macroinvertebrates, in particular, may have substantial exposure to COPCs in sediment. Exposure could result from direct contact between sediment and outer membranes and respiratory surfaces, from the direct ingestion of sediments during feeding activities, and from the consumption of affected prey or detritus, depending upon specific feeding habits. Benthic macroinvertebrates may, in turn, become a source of COPC exposure for secondary consumers, including fish, birds, and mammals.

<u>Fish</u>
Adult and juvenile fish in wetlands may be exposed to (and possibly accumulate) COPCs in surface water (absorption across gills), via the consumption of contaminated vegetation and prey and/or the inadvertent ingestion of sediment.. Fish may also be exposed to COPCs from sediment and surface water during critical life stages including embryonic development of eggs and fry. American eel generally dwell in the sediment and feed on small fish, insect larvae,

and crustaceans (Scarola, 1973). Sunfish (including largemouth bass, black crappie, pumpkinseed, bluegill, tessellated darter and other perch species) prefer to live in or near vegetation; they tend to feed on mollusks, crustaceans, insects, worms and small fish and are preyed on by larger fish and turtles (Etnier and Starnes, 1993; Page and Burr, 1991; Scott and Crossman, 1973). Minnows and carp (including blacknose dace, fallfish, and shiner species) feed on aquatic and terrestrial insects and algae (Etnier and Starnes, 1993). Trout are cold-water species whose distribution in Massachusetts is maintained by hatcheries; their diet mainly consists of aquatic insect larvae and adults (Scarola, 1973). Suckers and catfish (including brown bullhead, white sucker, creek chubsucker, and yellow bullhead) are omnivorous bottom feeders and are prey items for birds, larger fish and wetland mammals (Scott and Crossman, 1973; Scarola, 1973). Pickerel are predatory fish that consume insects and crustaceans when young but fish and amphibians as adults (Etnier and Starnes, 1993).

Wetland Birds

Aquatic and wetland exposure areas provide habitat which may be used by bird species for feeding or nesting. Mallards (*Anas platyrhynchos*), great blue heron (*Ardea herodias*), and osprey (*Pandion haliaetus*) were selected as representative species of waterfowl and aquatic predatory birds. These species, to varying degrees, may be exposed to COPCs through ingestion of plants, animals, detritus, sediment, and surface water.

Mallards require dense grassy vegetation at least 1.6 ft in height for nesting (Bellrose, 1976 cited in USEPA, 1993a). They forage for food in ponds and wetlands by dabbling and filtering through sediments (USFWS, 1991 cited in USEPA, 1993a). Mallards may be food items for larger mammals such as foxes (Johnson *et al.*, 1988 cited in USEPA, 1993a). Their diet consists mainly of aquatic plants, seeds, and aquatic invertebrates.

Great blue herons tend to nest in dense colonies located close to foraging grounds. Fish are their preferred prey, but great blue herons also eat amphibians, crustaceans, insects, birds, and mammals (USEPA, 1993a). When fishing, they generally employ one of two foraging techniques: standing still and waiting for fish to swim to within striking distance, or slow wading to catch sedentary prey. Great blue herons migrate from summer breeding grounds during mid-autumn and generally return by mid-spring. Great blue herons have not specifically been observed at the Site; however, they are representative of the wildlife receptors that may use the Site, and habitat present at the Site suggests that they could potentially use portions of the Site.

Osprey are top predators that have a diet consisting almost entirely of fish. They are adapted for hunting, hovering over the water and diving feet-first to seize fish using their talons (Robbins *et al.*, 1983 cited in USEPA, 1993a). They nest in close proximity to open, shallow water at the tops of isolated and often dead trees (Poole, 1989 cited in USEPA, 1993a). Osprey migrate away from the northern temperate region in early autumn and return mid-March (Poole, 1989; Poole and Gill, 1992; Reese, 1977 cited in USEPA, 1993a). Osprey have not specifically been observed at the Site; however, they are representative of the wildlife receptors that may inhabit the Site.

Wetland Mammals

Aquatic and wetland mammalian receptors may be exposed to and accumulate COPCs in sediment, surface water, and food items. The northern short-tailed shrew (*Blarina brevicauda*) and raccoon (*Procyon lotor*) were selected as representative wetland mammals.

In particular, the northern short-tailed shrew may receive substantial exposure to and accumulate COPCs due to diet and high ingestion rate (Morrison *et al.*, 1957 *cited in* USEPA, 1993a). Shrews have high metabolic rates and can eat their approximate weight in food each day, with a diet that consists of insects, worms, snails, and other invertebrates and at times, mice, voles, frogs, and other vertebrates (Robinson and Brodie, 1982 *cited in* USEPA, 1993a). They are important food items for owls and are also prey for other raptors, fox, weasels and other carnivorous mammals (Palmer and Fowler, 1975; Burt and Grossenheider, 1980; Buckner, 1966, cited in USEPA, 1993a). Shrews are most common in areas with abundant vegetative cover (Miller and Getz, 1977 *cited in* USEPA, 1993a).

They are active all year and do not hibernate. Shrews build underground nests and maintain underground runways, usually in the top 4 inches of soil but sometimes as deep as 20 inches. Habitat ranges from 0.03 hectares (0.07 acres) when prey is abundant to 2.19 hectares (5.4 acres) when food is scarce (USEPA, 1993a).

Raccoons are opportunistic omnivores (USEPA, 1993a) that forage along marsh edges (Weller, 1981). Raccoons may ingest COPCs through the consumption of prey, ingestion of drinking water, and incidental ingestion of sediment. The raccoon feeds on a variety of animal and vegetable matter, though plants are usually a more important component of the diet. The home range of an adult raccoon is approximately 156 hectares (385 acres) (Stuewer 1943, *cited in* USEPA, 1993a).

2.5.2 Terrestrial Receptors

<u>Terrestrial Plants</u>
Terrestrial plant species may be exposed to and accumulate COPCs from soil pore water, where ions are freely available for absorption by plant roots (Brady and Weil, 1999). Plants may accumulate COPCs in roots, stems, leaves, or fruits which get transferred to herbivores when consumed. Plant 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.

<u>Terrestrial Invertebrate Communities</u>
Terrestrial invertebrates, such as earthworms and soil-dwelling insects, may be exposed to and accumulate COPCs. Exposure could result from direct contact between soil and outer membranes and respiratory surfaces, from the direct ingestion of soil during feeding activities, and from the consumption of affected prey or detritus, depending upon specific feeding habits. Consumers, including amphibians, reptiles, birds, and mammals may be exposed to COPCS accumulated in the tissues of terrestrial invertebrate prey.

<u>Songbirds</u>
Terrestrial habitat at and around the Site provides feeding and nesting sites attractive to songbirds. Birds can be exposed to COPCs through ingestion of plant forage or animal prey, as well as through incidental ingestion of detritus, soil, and surface water. Northern cardinal (*Cardinalis cardinalis*) and American robin (*Turdus migratorius*) were selected as representative omnivorous and invertivorous songbirds, respectively.

Northern cardinals live in thickets and brushy areas, edges and clearings, riparian woodlands, and residential areas. They are year-round residents of the Northeast (Cornell Lab of Ornithology, 2009; Dewey *et al.*, 2001) and have a home range of ~21 hectares (Halkin and

Linville, 1999). Northern cardinals eat mainly seeds, leaf buds, flowers, berries, and fruit, supplementing these with soil-dwelling insects such as boll weevils, cutworms, and caterpillars. They hop on low branches and forage on or near the ground.

Robins eat largely soil invertebrates which may have significant direct contact with soil and may bioaccumulate COPCs (USEPA, 1993a). Robins forage by hopping along the ground in search of ground-dwelling invertebrates, such as earthworms, and by searching for fruit in shrubs and low tree branches. Robins forage on the ground in open areas, along habitat edges, or near edges of water. Nests are constructed with mud or vegetation and are located on or near the ground (USEPA, 1993a). During breeding season, foraging is generally confined to a territory approximately 0.48 hectares (1.2 acres) in size. During non-breeding roosting periods, robins are likely to return to the same foraging sites for many weeks (USEPA, 1993a). Robins migrate, leaving breeding grounds in late summer and returning by early spring. Adult robins often return to the same territory in succeeding years.

Terrestrial Mammalian Community

Mammals may also be exposed to COPCs in terrestrial soils. Organisms including mice, voles, woodchucks, skunks, and foxes may be exposed to COPCs through the consumption of contaminated vegetation and prey, incidental ingestion of soil, and consumption of dietary water. The meadow vole (*Microtus pennsylvanicus*) and red fox (*Vulpes vulpes*) were selected as representative terrestrial mammals.

The meadow vole burrows along surface runways in grasses or other herbaceous vegetation and therefore has a high exposure to COPCs in surface soils (USEPA, 1993a). The vole is mainly an herbivore, with a diet consisting primarily of green succulent vegetation, sedges, seeds, roots, bark and fungi (USEPA, 1993a). They do not hibernate but are active year round (Didow and Hayward, 1969; Johnson and Johnson, 1982 cited in USEPA, 1993a). Voles maintain a small home range, approximately 0.06 hectares (0.15 acres).

The red fox is an omnivore that predominantly feeds on small mammals such as the meadow vole, but may also eat insects, fruits, berries, seeds, and nuts (USEPA, 1993a). They are exposed to COPCs through surface soils that they may ingest accidentally while scavenging for food and through bioaccumulation of COPCs in prey items. Foxes do not undergo hibernation or torpor and are active year-round (USEPA, 1993a). The home range of a fox is approximately 699 hectares, which is equivalent to 1,727 acres (Sargeant, 1972).

2.6 Selection of 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 occurred or may occur. Assessment endpoints most often relate to attributes of biological populations or communities. They contain an entity (e.g., invertebrate populations) and an attribute of that entity (e.g., survival). At hazardous waste sites, the entity in the assessment endpoint is typically an individual species, population, or community, often referred to as a receptor species or receptor community, respectively. In the case of specially protected species, the assessment endpoint frequently focuses on individuals. Measurement endpoints are related to the assessment endpoint, and the effects that can be measured or observed (e.g., survival or apparent toxicity in invertebrate bioassays). Measurement endpoints are most often used to characterize assessment endpoints since in most cases the assessment endpoint itself expresses a value that cannot be readily measured or observed (Suter, 1993).

As part of the problem formulation, measurement endpoints are assigned a relative weight for each attribute that accounts for strength of association between the assessment and the measurement endpoint (which is also a function of data quality, study design and execution). The relative importance is indicated by an attribute weighting factor (inference weight) following guidance provided by Menzie *et al.* (1996):

- The level of confidence, or weight, given to the various measures;
- Whether the result of the measurement indicates there is an effect;
- The strength of the result (e.g. no toxicity in reference area vs. high site-related toxicity); and
- Concurrence among the various measures.

Measurement endpoints were each assigned an inference weight, based upon how closely they represented the assessment endpoint. Conclusions regarding risks to that assessment endpoint were reached by considering the inference weight for each measurement endpoint, i.e., the overall weight of evidence. In other words, conclusions from higher weighted endpoints were given a stronger consideration than the lower weighted endpoints.

The following assessment endpoints, measurement endpoints, and inference weights are presented below and summarized in Table 2-1. These assessment and measurement endpoints were established in the Study Design (*de maximis*, 2006b) which was approved by USEPA.

2.6.1 Assessment Endpoint #1: Benthic Invertebrate Community Structure

Measurement Endpoint 1A: Compare Site surface water concentrations to published chronic surface water benchmarks and to background. *Low/Medium* Inference Weight.

Measurement Endpoint 1B: Compare Site sediment concentrations to published sediment benchmarks and to background. *Low/Medium* Inference Weight.

Measurement Endpoint 1C: Perform laboratory toxicity tests to measure survival and growth of a freshwater benthic invertebrate (*Chironomus dilutus*) exposed to sediments collected from the Site and compare to background. *Medium/High* Inference Weight.

Measurement Endpoint 1D: Compare the community structure of benthic invertebrates in Site sediments to background. *Low/Medium* Inference Weight.

2.6.2 Assessment Endpoint #2: Growth, Survival, and Reproduction of Amphibian Populations

Measurement Endpoint 2A: Compare Site surface water concentrations to published chronic surface water benchmarks and to background. *Low/Medium* Inference Weight.

Measurement Endpoint 2B: Compare Site sediment concentrations to published sediment benchmarks and to background. *Low/Medium* Inference Weight.

Measurement Endpoint 2C: Compare average tissue COPC concentrations from amphibians in the bog to average concentrations measured in amphibians caught from the background (reference) bog. Additionally, if Critical Body Burden concentrations are available, observe if bog amphibian COPCs concentrations exceed those benchmarks. *Medium/High* Inference Weight.

2.6.3 Assessment Endpoint #3: Growth, Survival, and Reproduction of Fish Populations

Measurement Endpoint 3A: Compare Site surface water concentrations to published chronic surface water benchmarks and to background. *Low/Medium* Inference Weight.

2.6.4 Assessment Endpoint #4: Growth, Survival, and Reproduction of Wetland Bird Populations

Measurement Endpoint 4A: Compare estimated daily dose for omnivorous waterfowl (mallard duck) based on ingestion of prey in Site exposure areas to published avian toxicity reference values (TRVs) and to background. *Medium* Inference Weight.

Measurement Endpoint 4B: Compare estimated daily dose for predatory wading birds (great blue heron) based on ingestion of prey in Site exposure areas to published avian TRVs and to background. *Medium* Inference Weight.

Measurement Endpoint 4C: Compare estimated daily dose for piscivorous birds (osprey) based on ingestion of prey in Site exposure areas to published avian TRVs and to background. *Medium* Inference Weight.

2.6.5 Assessment Endpoint #5: Growth, Survival, and Reproduction of Wetland Mammal Populations

Measurement Endpoint 5A: Compare estimated daily dose for omnivorous small mammals (shrew) based on ingestion of prey in Site exposure areas to published mammalian TRVs and to background. *Medium* Inference Weight.

Measurement Endpoint 5B: Compare estimated daily dose for omnivorous larger mammals (raccoon) based on ingestion of prey in Site exposure areas to published mammalian TRVs and to background. *Medium* Inference Weight.

2.6.6 Assessment Endpoint #6: Growth, Survival, and Reproduction of Terrestrial Plants

Measurement Endpoint 6A: Compare Site soil concentrations to published soil benchmarks and to background. *Low/Medium* Inference Weight.

2.6.7 Assessment Endpoint #7: Growth, Survival, and Reproduction of Terrestrial Invertebrate Populations

Measurement Endpoint 7A: Compare Site soil concentrations to published soil benchmarks and to background. *Low/Medium* Inference Weight.

2.6.8 Assessment Endpoint #8: Growth, Survival, and Reproduction of Terrestrial Songbird Populations

Measurement Endpoint 8A: Compare estimated daily dose for omnivorous songbirds (cardinal) based on ingestion of prey in Site exposure areas to published avian TRVs and to background. *Medium* Inference Weight.

Measurement Endpoint 8B: Compare estimated daily dose for invertivorous songbirds (American robin) based on ingestion of prey in Site exposure areas to published avian TRVs and to background. *Medium* Inference Weight.

2.6.9 Assessment Endpoint #9: Growth, Survival, and Reproduction of Terrestrial Mammal Populations

Measurement Endpoint 9A: Compare estimated daily dose for herbivorous small mammals (meadow vole) based on ingestion of prey in Site exposure areas to published mammalian TRVs and to background. *Medium* Inference Weight.

Measurement Endpoint 9B: Compare estimated daily dose for omnivorous large mammals (red fox) based on ingestion of prey in Site exposure areas to published mammalian TRVs and to background. *Medium* Inference Weight.

2.6.10 Assessment Endpoint #10: Growth, Survival, and Reproduction of Rare, Threatened, or Endangered Species (Eastern Pondmussel, Triangle Floater, and Creeper)

Measurement Endpoint 10A: Compare Site sediment and surface water concentrations to published sediment and chronic surface water benchmarks and to background. *Low/Medium* Inference Weight.

2.7 Ecological Conceptual Site Model

Figure 2-1 presents a CSM that depicts the potential migration pathways through which contaminants may have been transported from source areas to other environmental media where possible ecological exposure may occur.

3. BASELINE ECOLOGICAL EXPOSURE AND EFFECTS EVALUATION

The Exposure Assessment identifies the exposure areas and receptors evaluated in the BERA, and calculates exposure point concentrations (EPCs) (Section 3.1). The Exposure Assessment also describes the methods that are used to quantify exposure (Section 3.2). In the Effects Assessment, available evidence from the literature for existing and potential adverse effects on the assessment endpoints is tabulated and analyzed (Section 3.3). Interpretation of results is generally reserved until the Section 4.0 Risk Characterization.

3.1 Exposure Pathways and Exposure Point Concentrations

The first part of the exposure assessment identifies the exposure areas, receptors that are evaluated, and the EPCs that are used to quantify exposures to those receptors.

3.1.1 Exposure Areas and Receptors

The following exposure areas (also referred to as study areas) were evaluated in the BERA:

- AOI 6 – Sphagnum Bog
- AOI 10 – Northeast Wetland
- AOI 18A – Assabet River Main Channel
- AOI 18B – Assabet River Embayment Area
- Site-Wide Soils

In accordance with USEPA policy, a SLERA can be sufficient to document risk in areas where a known remedy will be implemented. Based on the results of the SLERA and as documented in the RI Report (*de maximis*, 2010), the need for a presumptive remedy at AOI 4 (Cooling Water Recharge Pond) has already been identified. As a result, additional evaluation of ecological risk at AOI 4 is no longer necessary because risk associated with potential exposure to ecological receptors will be addressed by the presumptive remedy.

The above exposure areas correspond to the habitats described in Section 2.1.2 of the Problem Formulation. Receptors at the Site were identified in Section 2.5 and were selected based on the types of habitat present at the Site. Aquatic and wetland receptors include aquatic invertebrates, amphibians, fish, and wetland wildlife such as the mallard duck (omnivorous waterfowl), great blue heron (predatory wading bird), osprey (piscivorous birds), short-tailed shrew (omnivorous small mammal), and raccoon (omnivorous larger mammal). Terrestrial receptors include terrestrial plants, terrestrial invertebrates such as earthworms, and terrestrial wildlife receptors such as cardinal (omnivorous songbird), American robin (invertivorous songbird), meadow vole (a small mammal), and red fox (a larger mammal). Consideration of rare, threatened, or endangered species includes three species of freshwater mussels listed as species of special concern in Massachusetts: the eastern pondmussel, triangle floater, and creeper.

3.1.2 Calculation of Exposure Point Concentrations

EPCs represent concentrations to which ecological receptors may be exposed. EPCs were calculated by medium (surface water, sediment, sphagnum moss, peat, surface soil, and tissue) and exposure area. As described later in Section 3.2, EPCs were compared to screening and effects benchmarks, or to chronic surface water benchmarks, and were also used as input values in food chain models.

3.1.2.1 Data Used to Calculate EPCs

The BERA evaluated data collected during the Phase 1A, 1B, and 1C field events (Table 1-4 and Table 1-5).

Sample data were validated according to USEPA Region 1 procedures and guidelines, as described in the QAPP (*de maximis* 2004b). As outlined in the QAPP, the first laboratory deliverable package for a given sampling event was validated to a Tier III validation level. All subsequent data packages were validated to a Tier II validation level. The qualification and validation of the analytical data included a comparison of the Site data to corresponding laboratory-, field-, equipment-, and trip-blank concentrations data. Data rejected by validation ("R" qualified) were not used. Estimated values (ex. "J" qualified) were used in the risk assessment without modification. Validated data were also reviewed to determine if quantitation limits were adequate to ensure that any concentrations of concern from a risk or regulatory perspective were detected and quantified. Detailed discussions of data validation observations and qualification actions are presented in the RI Report. In cases when data for a given sample consisted of a primary sample and a duplicate sample, the risk assessment used the higher of the two concentrations. Validated data used in the BERA are presented in Appendix D.

Soil EPCs were calculated from samples collected from the 0 to 2 foot interval as this depth is considered to be the most biologically active horizon (Brady & Weil, 1999); however in accordance with the USEPA-approved Risk Assessment Work Plan, most samples were collected from the top foot. The skin of burrowing mammals is generally covered with fur, minimizing potential exposures in deeper soils. For sediments, EPCs were calculated from samples collected from the 0 to 6 inch interval because this interval is considered the most biologically active zone (USEPA, 2001d).

As documented in the QAPP, USEPA Method 8310 was used to set and meet the Project Action Levels (PALs) for sediment, soil, and surface water because it is specifically designed for the detection of trace level PAHs. Later, Method 8270C was used to analyze site media because this method offers more certainty and less false positive results across a broader range of concentrations. The two methods sometimes provide conflicting results stemming from false positives in the high range of Method 8310 data. Based on conclusions from a detailed data quality review, only PAH data from Method 8310 are used to select and evaluate surface water COPCs. PAH data analyzed by Method 8270C are used to select and evaluate sediment, peat, and soil COPCs; Method 8310 sediment, peat, and soil data are used only when Method 8270C data are not available.

3.1.2.2 Exposure Point Concentrations

Both reasonable maximum exposure (RME) EPCs and central tendency exposure (CTE) EPCs were calculated by exposure area/background area and medium (surface water, sediment, sphagnum moss, peat, surface soil, and tissue) as applicable. Evaluations based on the RME EPC are considered much more conservative than evaluations based on the CTE EPC. This is because the use of the RME EPCs implies that receptors are exposed to a high (i.e. maximum or 95% UCL) contaminant concentration. Animals are mobile, however, and populations foraging within an AOI are exposed to a wide range of concentrations, as represented by the CTE EPCs. Therefore, evaluations based on CTE EPCs concentrations represent a less conservative but more realistic exposure scenario. Although plants are not mobile and benthic organisms will move locally but generally exhibit a "clustered" distribution, these communities cover a large area, not a single point, so exposures based on CTE EPCs are also appropriate with regard to plants and benthic organisms. The methods used to derive RME and CTE EPCs for study area and background area samples are explained below.

Study Areas

Surface water, sediment (including sphagnum moss and peat), soil, and tissue (consisting of aquatic invertebrate and frog) EPCs for Site samples are presented in Table 3-1 through Table 3-4. In accordance with USEPA guidance (USEPA, 2002), RME EPCs for study area surface water, sediment, peat, sphagnum moss, soil, and tissue media were identified as being the lesser of the 95% upper confidence limit (95% UCL) on the arithmetic mean and the maximum detected concentration in the data set. CTE EPCs for all study area media used in the BERA are based on the lesser of either the arithmetic mean concentration or the maximum detected concentration of the data set, using one-half the Sample Quantitation Limit (SQL) for non-detects.

The 95% UCL values were calculated by exposure area and medium using the ProUCL software (V. 4.00.04; USEPA, 2009). The ProUCL software performs a goodness-of-fit test that determines the distribution of the data set for which the EPC is being derived (e.g., normal, lognormal, gamma, or non-discernible). Then, the software calculates a conservative and stable 95% UCL value in accordance with the framework described in *Calculating Upper Confidence Limits for Exposure Point Concentrations at Hazardous Waste Sites* (USEPA, 2002). The software includes 5 parametric and 10 non-parametric methods for calculating 95% UCL values, and provides recommendations for the appropriate UCL value to use as the EPC, based on the statistical characteristics of the data set. Ninety-fifth percentile UCL calculations are documented in Appendix E.

Background Areas

Surface water, sediment, soil, and tissue EPCs for background samples are presented in Table 3-5 through Table 3-8. RME EPCs for the background pond, Sphagnum bog, wetland, and upstream Assabet River area were based on at least one of the following: 95^{th} percentile on the data, the 95% coverage, the 95% confidence upper tolerance limit (95/95 UTL), the sample mean plus three standard deviations, and the maximum value of the data set. The background data and statistical descriptors of the data sets were presented in *Draft Statistical Evaluation of Background and Site-Specific Data, Fall 2004 and Spring 2005 Phase 1A Remedial Investigation Data* (*de maximis,* 2006c). CTE EPCs for background areas were based on the average (arithmetic mean) by location and medium, using one-half the SQL for non-detects.

RME and CTE EPCs for background embayment area surface water and sediment were calculated using ProUCL software (V. 4.00.04; USEPA, 2009) following the same method as the study areas. The RME for the background embayment is identified as the lesser of either the 95% UCL on the arithmetic mean or the maximum detected concentration in the data set. The CTE for the background embayment is identified as the lesser of the maximum detected concentration and the arithmetic mean.

3.2 Methods Used to Evaluate Exposures

The BERA used several methods to assess risks to receptors. Each of these methods are described below and include: screening benchmark and effects benchmark evaluations and risk calculation, ΣPAH modeling, SEM AVS sediment chemistry, surface water and sediment toxicity tests, benthic macroinvertebrate community surveys, frog and invertebrate body burden testing, and food chain models.

3.2.1 Benchmark Comparisons and Risk Calculations

For some measurement endpoints, RME and CTE EPCs were compared to screening and effects benchmarks, and to chronic surface water benchmarks, to calculate a hazard quotient (HQ). This section first describes the sources used to develop surface water, sediment, and soil benchmarks. Then, the method for calculating an HQ is explained.

3.2.1.1 Benchmarks

Benchmarks are based on toxicity tests and experimental observations published in the scientific literature. Except for surface water, two general types of benchmarks were used in the BERA to characterize risk: screening benchmarks and effects benchmarks. Screening benchmarks concentrations represent levels of COPCs in environmental media at or below which adverse effects are unlikely to occur. If EPCs are below screening benchmarks, then risk is assumed to be unlikely. Screening benchmarks were initially used to select COPCs in the SLERA (Appendix A).

Effects benchmarks represent concentrations at or above which adverse effects are likely to occur. Effects benchmarks are often reported based on the degree of measured response observed at a particular site (e.g., EC_{50}, a concentration affecting 50% of a test population). If EPCs are greater than effects benchmarks, adverse effects may be possible.

The surface water benchmarks consisted only of chronic benchmarks, because exceeding such benchmarks can result in long-term harm to aquatic community-level receptors.

The regulatory agencies overseeing the NMI RI (and associated BERA) provided project-specific guidance for selecting screening benchmarks. Since the sources recommended by the regulatory agencies did not include screening benchmarks for some important Site COPCs (e.g. cobalt, molybdenum, selenium, uranium, and vanadium in sediment), additional sources were added as described in subsequent paragraphs. The hierarchy of sources used for selecting screening benchmarks was also followed for selecting effects benchmarks when those sources included effects benchmarks. Benchmarks are presented below by medium (surface water, sediment, and soil) and summarized in Table 3-9 to Table 3-14.

The degree of certainty that an effect will occur at the Site if a COPC concentration is above a benchmark depends on a number of factors including bioavailability, similarity of species used to measure the effect, and similarity between experimental conditions and Site conditions. Screening and effects benchmarks often conservatively assume 100% bioavailability. Bioavailability is defined as the individual physical, chemical, and biological interactions that determine the exposure of plants and animals to chemicals associated with soils and sediments. When available, studies with species and habitats similar to those within the Site were used to establish effects benchmarks.

Sources used for screening and effects benchmarks are discussed below, by medium.

Surface Water Benchmarks

The following sources were used to select the chronic surface water benchmarks in the order presented (Table 3-9):

1. USEPA Freshwater Chronic Ambient Water Quality Criteria (AWQC) (USEPA, 2006a);
2. USEPA Ecotox Thresholds (ET) for Surface Water (USEPA 1996a); and
3. Oak Ridge National Laboratory (ORNL) Secondary Chronic Values (SCVs) for Aquatic Biota (Suter & Tsao, 1996).

If chronic surface water benchmarks were not available from any of the above sources, then the lowest value from the following three sources was selected:

- USEPA Region IV Screening Values (USEPA, 2001c);
- Chronic Aquatic Information Retrieval (AQUIRE) (USEPA, 2007; MassDEP, 2000); or
- USEPA Ecological Structure Activity Relationships (ECOSAR) benchmarks (USEPA, 2000a).

Where applicable, the available chronic surface water benchmarks were compared to the dissolved fractions of metals because EPA policy stipulates that only the dissolved fraction is considered bioavailable to aquatic receptors. Data on total metals measured in surface water are presented in the tables but were not included in the analysis.

Where applicable, the chronic surface water benchmarks were also adjusted based on average hardness values (calculated as calcium carbonate per liter, or $CaCO_3$/L) specific to each individual AOI, as follows:

- AOI 4 - Cooling Water Recharge Pond, 47.4 milligrams (mg) $CaCO_3$/L;
- AOI 6 - Sphagnum Bog, 14.9 mg $CaCO_3$/L;
- AOI 10 - Northeast Wetland, 14.5 mg $CaCO_3$/L;
- AOI 18A - Assabet River Main Channel, 65.8 mg $CaCO_3$/L; and
- AOI 18B - Assabet River Embayment Area, 47.4 mg $CaCO_3$/L.

Background Area:

- Background Pond, 3.1 mg $CaCO_3$/L;
- Background Bog, 7.3 mg $CaCO_3$/L;
- Background Wetland, 22.3 mg $CaCO_3$/L;
- Upstream River Channel, 65.4 mg$CaCO_3$/L; and

- Background Embayment Area, 65.4 mg $CaCO_3$/L.

Levels of hardness <60 mg $CaCO_3$/L are generally categorized as "soft" water, while levels between 61 – 120 mg $CaCO_3$/L fall into the range of "moderately hard" (Hem, 1989). This distinction is important from the standpoint the toxicity of dissolved metals to aquatic invertebrates, amphibians and fish as an increase in calcium (hardness) generally affords more protection to the organism and is reflected in the development of National Recommended Ambient Water Quality Criteria. The soft waters encountered on Site will therefore exhibit lower water quality criteria for metals. Lower water quality criteria translate into benchmarks that are, therefore, more protective of sensitive aquatic receptors.

<u>Sediment Benchmarks</u>
Screening Benchmarks: The lowest value of the following sources was selected as the sediment screening benchmark (Table 3-11):

- Consensus Value (CV) Threshold Effect Concentrations (TECs) (MacDonald *et al.,* 2000);
- USEPA ETs Sediment Quality Benchmarks (SQBs) (USEPA, 1993a,b,c,d; 1996);
- National Oceanic and Atmospheric Administration (NOAA) Effects Range-Low (ER-L) (Long & Morgan, 1991, Long *et al.,* 1995, *cited in* Jones *et al.,* 1997); and
- Ontario Ministry of the Environment (OMOE) Lowest Effects Levels (LELs) (Persaud *et al.*, 1993).

If benchmarks were not available from any of the above sources, then the lowest value from the following sources was selected:

- Washington State Freshwater Sediment Quality Values (SQVs) – Probable Apparent Effects Thresholds (PAETs) (Cubbage *et al.,* 1997);
- Assessment and Remediation of Contaminated Sediments (ARCS) Program TECs (USEPA, 1996b *cited in* Jones *et al.,* 1997);
- USEPA Equilibrium-partitioning Sediment Guidelines (ESGs)/Sediment Quality Criteria (SQCs) (USEPA, 1998b, 1993b,c,d);
- ORNL SCVs and Lowest Chronic Values (LCVs) for Sediment (Jones *et al.*, 1997).

If benchmarks were still unavailable, an equilibrium-partitioning (EqP)-based benchmark was derived using the source below:

- EqP-based Benchmarks (Di Toro, 1985).

After reviewing sources recommended by the project regulators, several important COPCs (including cobalt, molybdenum, selenium, uranium, and vanadium) still lacked screening benchmarks. The following sources were consulted to identify screening level benchmarks for those COPCs:

- Thompson's LEL Sediment Quality Guidelines (SQGs) for metals and radionuclides (Thompson *et al.*, 2005); and

- USEPA Region III Biological Technical Assistance Group (BTAG) Screening Benchmarks for Freshwater Sediment (USEPA, 2006b).

Benchmarks presented in Table 3-11 are based on 1% TOC. During the COPC selection process (Appendix A) and benchmark evaluations (below), ETs, OMOE LELs, Washington State SQVs, USEPA ESG/SQC, and ORNL SCVs were adjusted to AOI-specific sediment TOC. Benchmark adjustment based on sediment TOC is consistent with the intended use as described in the applicable source documents. Site-specific arithmetic mean TOC values are as follows:

- AOI 4 - Cooling Water Recharge Pond, 0.88%
- AOI 6 - Sphagnum Bog Mineral Sediment, 14.6%
- AOI 6 - Sphagnum Bog Peat, 35.6%
- AOI 6 - Sphagnum Bog Moss, 37.67%
- AOI 10 - Northeast Wetland, 32.2%
- AOI 18A - Assabet River Main Channel, 1.0%
- AOI 18B - Assabet River Embayment Area, 13.2%

Background Area:

- Background Pond, 2.1%
- Background Bog, 8.5%
- Background Wetland: 16.8%
- Upstream River Channel: 0.87%

Effects benchmarks: The lowest value of the following sources was selected as the sediment effects benchmark (Table 3-12):

- CV Probable Effects Concentrations (PECs) (MacDonald *et al.*, 2000);
- NOAA Effects Range-Median (ER-M) (Long & Morgan, 1991, Long *et al.,* 1995 *cited in* Jones *et al.,* 1997); and
- OMOE Severe Effect Levels (SELs) (Persaud *et al.,* 1993).

If benchmarks were not available from any of the above sources, then the lowest value from the following sources was selected:

- Washington State Freshwater SQVs – Apparent Effects Thresholds (AETs) (Cubbage *et al.,* 1997);
- ARCS Program PECs (USEPA, 1996b *cited in* Jones *et al.,* 1997); and
- ORNL LCVs for Sediment (Jones et al., 1997).

For chemicals still lacking sediment effects benchmarks, literature based benchmarks were derived from the following source:

- Thompson's SEL SQGs for metals and radionuclides (Thompson *et al.*, 2005)

USEPA ETs, and USEPA ESG/equilibrium partitioning sediment benchmark (ESB) values were not considered because these sources do not provide effects benchmarks. Where applicable, sediment benchmarks were also adjusted based on TOC values specific to each AOI as presented previously.

Surface Soil Benchmarks

Screening Benchmarks: The following sources were used to select surface soil screening benchmarks in the order presented (Table 3-13):

1. Ecological Soil Screening Levels (EcoSSLs) (USEPA, 2003-2007);
2. USEPA Region IV Soil Benchmarks (USEPA, 1998c, *cited in* Friday, 1998);
3. ORNL Toxicological Benchmarks for Screening Potential Effects on Terrestrial Plants (Efroymson *et al.,* 1997a);
4. ORNL Toxicological Benchmarks for Screening Potential Contaminants of Concern for Effects on Soil and Litter Invertebrates and Heterotrophic Process (Efroymson *et al.,* 1997b).

If benchmarks were not available from any of the above sources, then the lower value from the following two sources was selected:

- USEPA ECOSAR Soil Invertebrate Benchmarks (USEPA, 2000a); and
- EqP-Based Benchmarks (Di Toro, 1985).

Effects Benchmarks: Site-wide soil effects benchmarks (Table 3-14) are based on the same hierarchy as the soil screening benchmarks; however, the effects benchmark is based on taxon-specific values. For example, effects benchmarks for terrestrial plants are based on benchmarks with phytotoxic endpoints, and effects benchmarks for terrestrial invertebrates are based on benchmarks with terrestrial invertebrate endpoints. The following sources were used to select surface soil effects benchmarks in the order presented:

1. USEPA Eco-SSLs for plants and soil invertebrates (USEPA, 2003-2007);
2. ORNL Toxicological Benchmarks for Screening Potential Effects on Terrestrial Plants and Soil and Litter Invertebrates and Heterotrophic Processes (Efroymson *et al.*, 1997a; Efroymson *et al.*, 1997b).
3. USEPA ECOSAR Soil Invertebrate Benchmarks (USEPA, 2000a).

USEPA Region IV and EqP values were not considered because these sources do not provide effects benchmarks.

3.2.1.2 Benchmark Evaluation Risk Calculation

An HQ method was used to characterize the magnitude of risks associated with COPC exposure. RME and CTE EPCs of individual COPCs were compared to screening and effects benchmarks in order to calculate an HQ:

$$HQ = \frac{EPC}{Benchmark} \qquad \text{(Equation 1)}$$

Where:
EPC = RME or CTE
Benchmark = Screening or Effects Benchmark (or chronic surface water benchmark)

A RME EPC coupled with a screening benchmark describes the most conservative risk scenario, while a CTE EPC coupled with an effects benchmark describes a more realistic risk scenario. Therefore, an HQ less than or equal to 1 based on an RME and screening benchmark indicates that the contaminant alone is very unlikely to cause adverse ecological effects, while an HQ above 1 based on a CTE and effects benchmark suggests that a COPC is present at a concentration that may have an adverse effect on an exposed population.

This analysis is more simplified when dealing with surface water COPCs. An HQ less than or equal to 1 based on an RME and a chronic surface water benchmark indicates that the contaminant alone is unlikely to cause adverse ecological effects, while an HQ above 1 based on a CTE and the same chronic surface water benchmark suggests that a COPC is present at a concentration that may have an adverse effect on an exposed population.

The risk characterization also includes an evaluation of incremental risks, which take into account the contribution of background concentrations to the overall Site risks:

$$\text{Incremental Risk HQ} = \text{Site HQ} - \text{Background HQ} \qquad \text{(Equation 2)}$$

Incremental risk is used to distinguish between risks that may be associated with the Site and those that are considered attributable to naturally occurring (background) conditions. Risk managers typically do not require remediation for risks that are consistent with background levels, and therefore incremental risks are important in supporting risk management decisions.

3.2.2 Evaluation of PAHs in Sediment using the $\sum$PAH Model

The bioavailability of PAHs in sediment can significantly affect their potential toxicity to benthic organisms. Bioavailability of PAHs is influenced by the amount of TOC within the substrate and the chemical properties of the individual PAH constituents, particularly the organic carbon partition coefficient (Koc). The bioavailability of PAHs in sediment was evaluated following *Procedures for the Derivation of Equilibrium Partitioning Sediment Benchmarks (ESBs) for the Protection of Benthic Organisms: PAH Mixtures, November, 2003* (USEPA, 2003), herein referred to as the Sum-PAH ($\sum$PAH) method. This model calculates ESBs using individual toxicity quotients based on measured sediment concentrations of thirteen individual PAHs and site-specific TOC concentrations. The individual toxicity quotients are summed to calculate the equilibrium sediment benchmark toxic unit ($\sum ESBTU_{fcv}$). Freshwater sediments with a $\sum ESBTU_{fcv} \leq 1.0$ are considered protective of benthic organisms. If the $\sum ESBTU_{fcv} > 1.0$, sensitive benthic organisms may be potentially affected. Thus, the $\sum$PAH model is useful for predicting lack of toxicity, i.e., if $\sum ESBTU_{fcv} < 1$ at all given locations within an exposure area, it may be concluded that PAHs would not pose an adverse effect to the benthic community because tissue concentrations are unlikely to exceed a critical body burden (i.e. a toxic threshold). Conclusions from the $\sum$PAH model are considered more accurate than conventional sediment screening benchmarks. The detection limit was used where concentrations were reported as non-detect. The $\sum ESBTU_{fcvs}$ were calculated to achieve a confidence percentile of 95%, thus the certainty in risk conclusions is high. $\sum$PAH calculations are presented in Appendix F.

3.2.3 Simultaneously Extracted Metals/Acid Volatile Sulfide

The bioavailability of metals in sediment can significantly affect their potential toxicity to benthic organisms. Bioavailability of certain divalent metals (cadmium, copper, lead, mercury, nickel, silver, and zinc) is 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 anticipated to be bound by divalent sulfide and therefore considered to be unavailable for leaching from the substrate into pore water or the overlying water column (USEPA, 2005).

Sediment samples for SEM AVS analysis were collected in the study area from AOI 4 - Cooling Water Recharge Pond (five samples), AOI 6 - Sphagnum Bog (five samples), and AOI 18A Assabet River (five samples), in order to assess the bioavailability of divalent metals (USEPA, 2005). SEM AVS samples were also collected from the background pond (three samples) and the background upstream river channel (three samples). SEM AVS data were used in the risk characterization when the screening and effects benchmark evaluations identified possible risk from divalent metals.

SEM AVS samples were obtained from hand auger grab samples by inserting a syringe into the upper 2 centimeters of substrate prior to homogenization. Samples were collected from the Cooling Water Recharge Pond and the Sphagnum Bog during the months of April and May in 2005 and were collected from AOI 18A Assabet River, the background pond, and the background upstream river channel in November of 2004. Detailed discussions of sampling approaches and the quality assurance and control activities implemented during data collection are provided in the RI report.

As documented within the guidance (USEPA, 2005), the comparison between AVS and SEM consists of calculating the amount of AVS and SEM in units of micromoles per gram (μmol/g), subtracting the AVS value from the SEM value, and then normalizing this difference by dividing by the amount of organic carbon (expressed as the fraction organic carbon) in the sediment:

$$\text{Normalized Value} = \frac{(SEM - AVS)}{f_{oc}} \qquad \text{(Equation 3)}$$

Where:
Normalized Value = μmol/g_{OC}
SEM = measured concentration of SEM metals (μmol/g sediment)
AVS = measured concentrations of AVS (μmol/g sediment)
f_{oc} = fraction of organic carbon in sediment (g_{OC} /g sediment)

SEM AVS calculations are presented in Appendix G. Per USEPA Guidance (USEPA, 2005), if the normalized value is less than 130 μmol/ g_{OC} (including negative values), then divalent metals in the sample are unlikely to be toxic to benthic macroinvertebrates within the sediment substrate. If the normalized value is between 130 μmol/ g_{OC} and 3,000 μmol/ g_{OC}, then the toxicity of the sample is uncertain. If the normalized value exceeds 3,000 μmol/ g_{OC}, then divalent metals are likely to be toxic. A negative value indicates that AVS exceeds SEM, and therefore the divalent metals are likely unavailable for leaching into porewater or the overlying water column.

Because AVS is generated by microbial reduction of sediment sulfate, AVS values may vary seasonally, with AVS concentrations typically higher in the warmer months and lower in the colder months, implying that metals tend to be less bioavailable in the summer (USEPA, 2005). Due to seasonal differences between the sample collection dates, bioavailability in the Cooling Water Recharge Pond and in the Sphagnum Bog (sampled in the spring) may be overestimated while bioavailability in the Assabet River, background pond, and background upstream river channel (sampled in the fall) may be underestimated. The confidence of risk conclusions based on SEM AVS data is therefore moderate.

3.2.4 Toxicity Tests

Laboratory toxicity tests are an important tool for assessing the impact of chemicals on aquatic ecosystems because they integrate the effects of complex chemical mixtures as well as physicochemical variables (e.g. grain size, percent TOC). In toxicity tests, groups of selected organisms are exposed to test media under controlled laboratory conditions to determine potential adverse ecological effects. Sediment toxicity at the NMI Site were evaluated using the methods described below.

3.2.4.1 Sediment Toxicity Tests

Sediment samples were collected between September 20 and October 4, 2007 at twelve Site Bog locations (SD-RI-0600100R, SD-RI-0602100R, SD-RI-0600500R, SD-RI-0605100, SD-RI-0602500R, SD-RI-0603600R, SD-RI-0603800R, SD-RI-0605200, SD-RI-0605300, SD-RI-0605000, SD-RI-0600900R, and SD-RI-0601700R) and five background bog locations (SD-RI-175300, SD-RI-175400, SD-RI-175500, SD-RI-175600, SD-RI-175700). Investigative sample SDRl0602900R was determined to be unsuitable for whole sediment toxicity testing due to the physical characteristics (peat moss) of the sediment sample. Following sample preparation (sieving and equilibration), each sample was subjected to chronic 10-day *Chironomus dilutus* tests with survival and growth as endpoints (Appendix H). *Chironomus dilutus* is the aquatic [larval] stage of a small freshwater fly (midge).

The sediment samples were collected in the field from the top six inches using a hand auger and preserved at 4°C for shipment to the Great Lakes Environmental Center's (GLEC) Traverse City, Michigan laboratory. Toxicity testing was conducted following EPA/600/R-99/064 *Methods for Measuring the Toxicity and Bioaccumulation of Sediment-Associated Contaminants with Freshwater Invertebrates, Second Edition* (USEPA, 2000b), and American Society for Testing and Materials (ASTM) 1705-95B *Standard Test Methods for Measuring the Toxicity of Sediment Associated Contaminants with Freshwater Invertebrates* (ASTM, 2000).

Each sample consisted of eight replicates, ten test organisms per replicate. At the end of the 10-day test period, survival frequency was determined by counting the number of surviving embryos and dividing by 10 (the total number of test organisms per replicate). Growth was measured as the ash free dry weight (mg) for each replicate.

In addition to the twelve study area and the five background area samples, the laboratory also ran a control sample to determine the health of the test organisms and validity of the tests.

Upon test completion, results of the laboratory control were reviewed to evaluate test validity. Results of the laboratory controls indicated that sediment toxicity test was performed within acceptable method parameters.

Results from the study area samples were then compared to the background area results to determine if any observed differences in effects were statistically significant. Statistical significance was calculated in accordance with the requirements and recommendations published with the testing method protocols (USEPA, 2000b; ASTM, 2000). Because study area samples were not paired to specific background samples, background data were first evaluated to determine if they satisfied assumptions of parametric statistical tests for normality and homogeneity of variance. If those assumptions were met, a one tailed Analysis of Variance (ANOVA) was used to determine if any site samples exhibited adverse effects when compared with background. If those assumptions were not met, a nonparametric (distribution-free) test was used. Statistical significance was defined as $p=0.05$.

All five background (reference) sediment sample results were analyzed individually to determine significant differences from one another using the Kruskal Wallis test for similarity (a non-parametric test). During the statistical review process it was determined that results of one background sample (SD-RI-175600) was significantly different from the other four background samples (SD-RI-175300 through SD-RI-175500 and SD-RI-75700). Consequently, sample SD-RI-175600 was removed from the background results.

If an effect on survival was found to be statistically significant at a particular location, then the effects on growth were not measured. This is a standard method in hypothesis tests for the growth endpoint for two reasons. First, an adverse effect has already been demonstrated on survival, so it is not necessary to demonstrate an effect on the non-lethal growth endpoint. Second, growth may increase in replicates with low survival because the remaining organisms have less competition for critical resources (such as food), thus the system is altered and not comparable to replicates with more survivors competing for resources.

If a sample demonstrated a statistically significant effect, then the test results were evaluated for ecological significance. Statistically significant differences between study area and background are not necessarily indicative of ecologically significant results, i.e., an association that is important mathematically may not be important biologically, especially at a population level scale. Because ecological risk assessments evaluate risk to receptors at the population level (USEPA, 1997), an ecologically "significant" risk should be interpreted relative to the results of any effects seen in the background (reference) area(s).

Sediment samples for concurrent chemical analysis were also collected at each sediment toxicity testing location. Sample-specific chemical concentrations can sometimes be used to identify COPCs that correlate with observed effects (Appendix L).

3.2.4.2 Frog Embryo Toxicity Assay with Xenopus (FETAX)

Surface water samples collected between September 20 and October 3, 2007 at five Site bog locations (SW-RI-06001R, SW-RI-06006R, SW-RI-06007R, SW-RI-06050, SW-

RI-06003R) and five background bog location (SW-RI-1753 through SW-RI-1757) were tested on frog embryos using 96-hour FETAX tests for survival, malformation, and growth (Appendix I). Toxicity tests were conducted following *Standard Guide for Conducting the Frog Embryo Teratogenesis Assay - Xenopus, ASTM E1439-98, Reapproved 2004* (ASTM, 2004).

The acidic water of the bog, however, was outside of the range of acceptable pHs for the standard assay (i.e.. pH between 6.5 and 9.0). This endpoint was therefore dropped as a measurement endpoint and replaced with an evaluation of frog tissue body burdens (see Section 3.2.6.2 and Measurement Endpoint 2C). The results of the FETAX tests are presented in Appendix I.

3.2.5 Benthic Macroinvertebrate Community Study

A quantitative survey was used to characterize the benthic macroinvertebrate community. Sediment samples were collected between September 19 and September 27, 2007, at thirteen locations Site bog locations (SD-RI-0600100R, SD-RI-0600500R, SD-RI-0600900R, SD-RI-0601700R, SD-RI-0602100R, SD-RI-0602500R, SD-RI-0602900R, SD-RI-0603600R, SD-RI-0603800R, and SD-RI-0605000, SD-RI-0605100, SD-RI-0605200 and SD-RI-06053) and five background bog locations (SD-RI-175300, SD-RI-175400, SD-RI-175500, SD-RI-175600, SD-RI-175700).

Sediment samples were collected in the field from the top six inches of substrate using either a hand auger or shovel, placed in jars, preserved in ethanol, and sealed. Samples were then transported to the Normandeau Associates, Inc. laboratory in Bedford, NH for enumeration and taxonomic identification. In the lab, samples were rinsed and transferred into a pan partially filled with water. Specimens were removed under magnification using forceps and subsequently transferred to glass vials. Specimens were identified to the lowest possible taxon, usually genus or species, using dissection and compound microscopes.

Six invertebrate metrics (wet weight biomass, taxa richness, percent dominant taxon, Hilsenhoff biotic index (HBI), Shannon diversity index, and community loss index) were calculated to evaluate community composition. The selected metrics are standard for evaluating benthic macroinvertebrate communities (USEPA, 1989a, 1999; Hilsenhoff, 1988). Individually, each metric does not provide conclusive information, but when considered collectively, metric values can highlight overall trends in the relationship between study area and background conditions. The collective results were quantified by the Scoring Index which is discussed after each of the individual metrics. Each metric is described below:

Wet Weight Biomass: Biomass is the total mass of living benthic macroinvertebrates in a given sample. Wet weight biomass is calculated by first removing excess water from the sieved macroinvertebrates, then measuring the mass of the total invertebrate sample in milligrams. High biomass generally correlates well with an increased abundance of organisms in the sample.

Taxa Richness: Taxa richness is the number of unique taxa present in a sample. Taxa richness generally increases with improved water quality, habitat diversity, and habitat suitability.

Percent Dominance: Percent dominance is a measure of the distribution of dominant individuals among all of the species present. Percent dominance is calculated as follows:

$$\% \text{ Dominance} = \frac{\text{Number of Individuals in the Most Dominant Taxon}}{\text{Total Abundance}} \qquad \text{(Equation 3)}$$

A lower percent dominance generally indicates a more robust ecosystem.

Hilsenhoff Biotic Index (HBI): The HBI produces a numerical value to indicate the level of organic pollution (Hilsenhoff, 1988). Each individual taxon is assigned a value ranging from zero to ten based on their tolerance of organic pollution as listed in USEPA, 1999. A low tolerance value (zero) indicates that a taxon is pollution intolerant and is likely to be found only in a pristine or pollution-free habitat. A high value (ten) indicates a taxon is pollution tolerant and is likely to dominate polluted habitat. The tolerance value for each taxon identified in a sample is multiplied by the number of specimens represented by that taxon. These products are summed and divided by the total number of specimens in the sample:

$$HBI = \Sigma \frac{V_1 N_1}{N} \qquad \text{(Equation 4)}$$

Where:
HBI = Hilsenhoff Biotic Index value
V_1 = Tolerance value for a given taxon
N_1 = Number of specimens for a given taxon
N = Total number of specimens in the given sample

HBI values range between 0 and 10 and are interpreted as follows (USEPA, 1999):

Index Value	Habitat Quality
0.00 – 3.50	Excellent
3.51 – 4.50	Very Good
4.51 – 5.50	Good
5.51 – 6.50	Fair
6.51 – 7.50	Fairly Poor
7.51 – 8.50	Poor
8.51 – 10.00	Very Poor

Shannon-Weiner Diversity Index: Diversity (H) is a measure of both total abundance and number of taxa. A community dominated by one species is considered less diverse than a community with a similar number of species that are all equally abundant. Chemical, physical, or biological disturbances are often thought to reduce diversity by eliminating taxa not capable of withstanding them. A diversity value will range from 0 for a community with a single species (no diversity) to over 7 (high diversity).

The Shannon-Weiner Index is calculated as follows:

$$H = - \Sigma (P_i * \log [P_i]) \qquad \text{(Equation 5)}$$

Where:

H = Diversity Index Value
P_i = relative abundance (n_i/N)

and:

n_i = number of individuals in species i
N = total number of individuals in all species

Community Loss Index: The Community Loss Index measures the relative difference in benthic species between study sites and background sites. Values range from zero to infinity, with an increasing value indicating a larger degree of dissimilarity between study area and background increases (Plafkin *et al.*, 1989). The Community Loss Index (CLI) is calculated using the following formula:

$$\text{Community Loss} = \frac{d - a}{e} \qquad \text{(Equation 6)}$$

Where:
d = Number of taxa present in background samples
a = Number of genera in common between the background and study area samples
e = Number of taxa present in study area sample

CLI values typically range from less than 0.50 (study area is similar to background) to greater than 4.0 (study area is different from background).

Scoring Index: A Scoring Index was calculated as a way to interpret the benthic community metrics to determine potential benthic community impairment (USEPA, 1989a). The Scoring Index quantifies impacts to the benthic community in study area samples relative to the background condition, expressed as a percentage. The Scoring Index was then assigned a narrative descriptor (non-impaired, slightly impaired, moderately impaired, or severely impaired) as shown in Table 3-15.

In order to quantify the background condition, functional metrics for each of the background benthic macroinvertebrate samples was calculated. Then, for each study area sample, calculated metrics (except wet weight biomass) were assigned a score ranging from 0 to 6 based on impairment relative to background (Table 3-16). A zero value signifies impairment relative to background. A value of six signifies non-impairment relative to background. Wet weight biomass was not included in the scoring index as per the procedures identified by USEPA (1989a).

Next, the five individual metric scores from each study area sample were totaled to yield a total score, with a maximum possible score of 30 (five metrics per sample times a maximum score of 6 per metric).

The Scoring Index for each study area sample was calculated by comparing the total score for each study area sample to the total score for the background condition:

$$\text{Scoring Index} = \frac{\text{Total Score For Study Area Sample}}{\text{Total Background Score}} \qquad \text{(Equation 7)}$$

Where:

Scoring Index = impairment of study area samples expressed as a percentage of the background condition.

Following the USEPA methodology (USEPA, 1989a), a total background score of 26 was calculated by using a score of 6 to represent the background condition of richness, HBI, diversity, and community loss index, and a value of 2 for the Percent Dominant Taxa. Scoring methodology is described in more detail in the Benthic Community Survey Report (Appendix J).

Sediment samples for concurrent chemical analysis were also collected at each benthic macroinvertebrate community survey location. Sample-specific chemical concentrations can sometimes be used to identify COPCs that correlate with observed community survey results.

3.2.6 Biological Tissue Testing

Biological tissue samples from representative prey species such as frogs and aquatic invertebrates were collected from the Site and background bogs to provide site-specific tissue data to be incorporated into the food chain models (discussed in Section 3.2.7).

Frogs and invertebrates were captured using wire mesh minnow traps measuring approximately 18” x 12” x 6.” Traps were deployed during the late afternoon, baited with cat food, and checked the following morning. Hand nets were also used to collect aquatic invertebrates and frog specimens. Captured specimens were identified in the field, measured, grouped into 50 g to 100 g samples, and frozen until laboratory analysis.

3.2.6.1 Invertebrate Body Burdens

Between September 24 and September 27, 2007, four aquatic invertebrate tissue samples were collected from the Site bog (IN-RI-06009, IN-RI-06025, IN-RI-06036, IN-RI-06050) and on October 9 one sample was collected from the background bog (IN-RI-1757). Captured specimens included dragonfly larvae, cranefly larvae, water boatman, and other unidentified beetles. Invertebrate tissue samples were analyzed for PAHs (Method 8270C), PCBs (Method 8082), TAL Metals (Method 6010B/7471), specialty metals (Method 6020), and percent lipids; however one study area sample (IN-RI-06036) was analyzed only for TAL metals, specialty metals, and percent lipids due to limited tissue quantity.

3.2.6.2 Frog Body Burdens

Between September 24 and September 27, 2007, eleven frog tissue samples were collected from the Site bog (AM-RI-06001, AM-RI-06005, AM-RI-06009, AM-RI-06017, AM-RI-06021, AM-RI-06025, AM-RI-06029, AM-RI-06036, AM-RI-06050, AM-RI-06051, AM-RI-06053), and between September 28 and October 11 five were collected from the background bog (AM-RI-1753, AM-RI-1754, AM-RI-1755, AM-RI-1756, AM-RI-1757). Captured specimens included tadpole and adult bullfrogs (*Rana catesbeiana*), green frogs (*Rana clamitans)*, and wood frogs (*Rana sylvatica*). Amphibian tissue samples were analyzed for PAHs (Method 8270C), PCBs (Method 8082), TAL Metals (Method 6010B/7471), specialty metals (Method 6020), and percent lipids; however one study area sample (AM-RI-06-029) and three background samples (AM-RI-1754, AM-RI-1755, and AM-RI-1756) were analyzed only for TAL metals, specialty metals, and percent lipids due to limited tissue quantity.

3.2.7 Food Chain Models

Exposure of terrestrial and wetland wildlife (i.e., birds and mammals) to site COPCs was estimated using food chain models. Surface water, sediment, soil, and tissue EPCs were entered into the food chain model to calculate an estimated daily intake (EDI) to which the receptor may be exposed. EPCs for prey items were either directly measured in tissue samples, estimated using biota-sediment accumulation factors (BSAFs) or bioaccumulation factors (BAFs) derived from site-specific data, or estimated using literature based BSAFs; literature based BSAFs were used only when measured tissue concentrations or site-specific BSAFs were not available.

Food chain models incorporate a site foraging frequency (SFF), which accounts for the proportion of a receptor's diet assumed to be obtained from an exposure area. If the exposure area is larger than the receptor's foraging (home) range, it is assumed that the receptor obtained all of its food from within that exposure area. If the exposure area is smaller than the receptor's foraging (home) range, it is assumed to obtain a fraction of its food from the exposure area. That fraction is calculated by dividing the exposure area by the foraging range.

EDIs for individual COPCs were compared to wildlife TRVs, expressed as milligrams per kilogram body weight day (mg/k BW-day), to evaluate the effect of exposure on representative species. The comparison was quantified using the HQ approach, as follows:

$$HQ = \frac{EDI}{TRV} \qquad \text{(Equation 8)}$$

Where:
EDI = Total body dose estimated from the food chain model (mg/kg BW-day)
TRV = Toxicity Reference Value (mg/kg BW-day)

TRVs were obtained from studies published in primary literature resources or review articles that reported No-Observable-Adverse-Effects Levels (NOAELs) and Lowest-Observed-Adverse-Effect Level (LOAELs) with survival, growth, or reproductive endpoints. Chronic studies were generally selected over acute or subchronic studies. USEPA-derived TRVs established to calculate Eco-SSLs were used preferentially when available. NOAEL and LOAEL TRVs are roughly analogous to screening and effects benchmarks used for other media, except that they represent screening and effects doses rather than concentrations. Wildlife TRVs used in the food chain model are presented in Appendix K.

There are uncertainties associated with these literature-based NOAEL and LOAEL values. Studies upon which these TRVs are based are typically performed in a laboratory setting, and are often based on common laboratory test species that may or may not be closely related to receptors being evaluated in the BERA. To reduce study variability, most strains of laboratory rats and mice are inbred and therefore are typically more sensitive to various toxicological effects. Thus, effects that may be observed in laboratory species may not be observed in wild populations of the same species or genus.

The details of the food chain model, including exposure assumptions, BAFs, BSAFs, TRVs, and receptor summaries are provided in Appendix K along with the food chain calculation spreadsheets. Table 3-17 presents a summary of receptors, exposure areas, and the sources of data used in the food chain model. Food chain models for AOI 6 – Sphagnum Bog were separately performed for the three sediment fractions (mineral, peat, and moss).

3.3 Effects Assessment

This section presents the results of the data evaluations described in Section 3.2. Interpretation of these results in generally reserved for the Risk Characterization (Section 4.0).

3.3.1 Assessment Endpoint #1 (Benthic Invertebrates)

Measurement Endpoint 1A: Compare Site surface water concentrations to published chronic surface water benchmarks and to background.

Both total and incremental risk HQs calculated using study area and background surface water RME and CTE EPCs, as part of the chronic surface water benchmark evaluations for benthic invertebrates, are presented in Table 3-18 (AOI 6 - Sphagnum Bog), Table 3-19 (AOI 10 - Northeast Wetland), Table 3-20 (AOI 18A - Assabet River), and Table 3-21 (AOI-18B Assabet River Embayment). This measurement endpoint was not evaluated in AOI 4 - Cooling Water Recharge Pond because the need for a presumptive remedy of the sediment in this pond has already been identified. As shown on Table 3-18 though Table 3-21, HQs could not be calculated for those COPCs which lacked chronic surface water benchmarks, and incremental risk HQs could not be calculated for those COPCs which were not analyzed in background samples. HQs above 1.0 are presented in boldface type on each table.

Measurement Endpoint 1B: Compare Site sediment concentrations to published sediment benchmarks and to background.

Both total and incremental risk HQs were calculated using study area and background sediment RME and CTE EPCs as part of the screening and the effects benchmark evaluation for benthic invertebrates were evaluated for three categories of sediment in AOI 6: mineral, moss and peat. Table 3-22.1 presents HQs for the Southwest Corner of AOI 6 (mineral sediment) while Table 3-22.2 presents HQs for mineral sediment in the Remaining Area. Table 3-23 and 3-24 presents HQs for, respectively, moss and peat in AOI 6.

An initial analysis of sediment data for AOI 6 Sphagnum Bog suggested that COPC concentrations were elevated with respect to screening benchmarks for PCBs and several metals. Consequently, the nature and extent of contamination within the Bog was reviewed to determine if the elevated concentrations were associated with a particular area of the Bog. The evaluation suggested that concentrations of PCBs, uranium, and several other metals were higher in the mineral sediment within the Southwest Corner of the Bog than in the remainder of the bog. Therefore, the mineral sediment data for the Sphagnum Bog were segregated into two exposure units: "Southwest Corner" and "Remaining Area". Figure 3-1 shows the demarcation of these areas and the sediment samples included in the areas. This approach was used only to evaluate Measurement Endpoint 1B, under the rationale that benthic organisms are primarily sedentary and hence effects to the benthic community could occur over a portion of the bog. In contrast, higher trophic level receptors (amphibians, birds, mammals) would be expected to forage throughout the bog so potential exposure would be lower than for benthic organisms. The entirety of the bog was therefore evaluated as one exposure unit to evaluate effects following food chain exposures to ecological receptors.

Benchmark HQs for sediment are also presented in Table 3-25 (AOI-10 Northeast Wetland), Table 3-26 (AOI 18A - Assabet River), and Table 3-27 (AOI 18B - Assabet River Embayment). This measurement endpoint was not evaluated in AOI 4 (Cooling Water Recharge Pond) because the need for a presumptive has already been identified. As shown on Table 3-22 though Table 3-27, HQs could not be calculated for those COPCs which lacked available benchmarks, and incremental risk HQs could not be calculated for those COPCs which were not analyzed in background samples. HQs above 1.0 are presented in boldface type on each table.

For those exposure areas where incremental risk HQs (see Section 4.7) for sediment PAHs were above 1.0, risk from PAHs was further evaluated using the ΣPAH model (USEPA, 2003). Results of the ΣPAH model are shown in Table 3-28. With the exception of one location in AOI 6 – Sphagnum Bog (SD-RI-0600500R) where the ΣPAH value is 1.38, all ΣPAH values are below 1.0 which indicates that PAHs would be bound to total organic carbon and therefore not bioavailable to benthic organisms. The calculation for the elevated result used full detection limits in the calculation for non-detects which produced over-conservative results, and when the one-half detection limits are used instead, the ΣPAH value at SD-RI-0600500R drops to 0.79.

SEM and AVS data (Table 3-29) were used to further evaluate those exposure areas where, prior to segregating out the data from the Southwest Corner of AOI 6 (Sphagnum Bog), incremental risk HQs for divalent metals were above 1. Total organic carbon adjusted SEM/AVS values from ranged from 0.5 micromole per gram of organic carbon (μmole/gOC) to 34 μmole/gOC. These values were all below the brightline value of 130 μmole/gOC which, according to the USEPA guidance, renders divalent metals completely bound up to naturally occurring sulfides (therefore not bioavailable to benthic organisms).

Measurement Endpoint 1C: Perform laboratory toxicity tests to measure survival and growth of a freshwater benthic invertebrate (*Chironomus dilutus*) exposed to sediments collected from the Site and compare to background.

Results from AIO 6 – Sphagnum Bog *Chironomus dilutus* toxicity test samples were compared to pooled background (reference sample) data (Table 3-30). The reference data set excluded sample SD-RI-175600 because survival was significantly different compared to the other four reference samples (Appendix H). Results of the sediment toxicity tests suggest that there were statistically significant differences in observed survival and growth effects in the study bog compared to data pooled for the background bog. Compared to pooled reference samples, *C. dilutus* demonstrated statistically significant differences for decreased survival at SD-RI-0600100R and SD-RI-0600900. Compared to pooled reference samples, statistically significant differences in growth were seen at SD-RI-0602100R and SD-RI-0605300.

Compared to the clean laboratory control sediment samples, *C. dilutus* demonstrated statistically significant differences for decreased survival at SD-RI-0600100R, SD-RI-

0602100R, SD-RI-0600500R, SD-RI-0603600R, SD-RI-0603800R, and SD-RI-0600900R. Compared to the clean laboratory control samples, statistically significant differences in growth were seen at SD-RI-0605300 and SD-RI-0600700R.

Measurement Endpoint 1D: Compare the community structure of benthic invertebrates in Site sediments to background.

Benthic community survey samples were collected from 13 locations in AOI 6 - Sphagnum Bog and from five locations in the background bog (Table 3-31). Biomass in the study area ranged from 1.4 mg wet weight to 473.1 mg wet weight, including elevated values at SD-RI-0603800R (223.8 mg wet weight) and SD-RI-0602500R (473.1 mg). The mean for biomass calculated in the study area was 60.2 mg wet weight while the median was 7.2 mg wet weight. Biomass in the background area, ranging from 1.4 mg wet weight to 11.7 mg wet weight with an average of 4.6 mg wet weight, was generally lower than the study area biomass. Richness in the study area, which ranged from 2 to 20 with a mean of 11, was comparable to a background richness mean of 11 (range from 7 to 20). Percent dominance, which ranged from 20% to 80%, was generally higher (more impaired) than reference area values that ranged from 17.1% to 43.0%. The most dominant species (chironomid larvae from the genera *Polypedilum* and *Bezzia*) were the most dominant organisms in both the study area and background samples; in two samples (SD-RI-0600500R and SD-RI-0603600R), earthworms (*Lumbriculus variegates*) were identified as the dominant taxon. The HBI values, a general index of organic pollution, ranged from 5.09 (good) to 7.63 (poor) with a mean of 6.86 (fairly poor) in the study area. HBI values calculated from the background bog were slightly higher (worse) and ranged from 6.4 (fair) to 7.6 (poor) with an average of 7.2 (fairly poor). Diversity in the study area ranged from 0.5 to 2.4 with an average of 1.7, and was slightly lower than background diversity, which ranged from 1.6 to 2.5 with an average of 1.8. CLI values ranged from 0.5 to 7.5. Ten of the thirteen study area locations received a CLI value of less than 2, indicating little overall impairment compared to background.

Scoring Index values in the study area ranged from 8 to 26. Eight of the thirteen samples (SD-RI-0600100R, SD-RI-0600900R, SD-RI-0602100R, SD-RI-0602500R, SD-RI-0603600R, SD-RI-0603800R, SD-RI-060500, and SD-RI-0605100) were identified as non-impaired. Two samples (SD-RI-0600500R and SD-RI-0605300) were identified as slightly impaired, one sample (SD-RI-0602900R) was identified as slightly/moderately impaired, and two samples (SD-RI-061700R and SD-RI-0605100) were identified as moderately impaired relative to background. No samples were identified as severely impaired relative to background.

3.3.2 Assessment Endpoint #2 (Amphibians)

Measurement Endpoint 2A: Compare Site surface water concentrations to published chronic surface water benchmarks and to background.

Site surface water concentrations were compared to chronic surface water benchmarks and background as described in Section 3.1.1 under Measurement Endpoint 1A.

Measurement Endpoint 2B: Compare Site sediment concentrations to published sediment benchmarks and to background.

Site sediment concentrations were compared to sediment benchmarks and background as described in Section 3.1.1 under Measurement Endpoint 1B.

Measurement Endpoint 2C: Compare amphibian tissue COPC concentrations to both concentrations in background bog organisms as well as Critical Body Burdens (tissue concentration thresholds thought to be adverse to the organism).

Because the degree of an adverse effect to a COPC is proportional to the exposure and/or dose of the receptor, it is instructive to compare the *average* concentrations of COPCs in amphibians (frogs) that were caught in both AOI 6 (Sphagnum Bog, N = 11) to the background (reference, N = 5) bog. Table 3-32 presents the data for both PCBs and metals in frog tissue (mg/kg). The data show that aluminum, barium, cadmium, mercury, and zinc were slightly lower in AOI 6 amphibians than the reference bog. Arsenic, beryllium, cobalt, copper, lead, nickel and titanium were slightly higher in bog tissue than the reference samples. PCBs and uranium were markedly increased in AOI 6 frogs as compared to organisms captured in the reference bog, although it is important to mention that not every sample was above the detection limit for the average calculation.

Only four Critical Body Burden values could be found in the literature (NAVFAC, 2004): cadmium (20 mg/kg), copper (16 mg/kg), lead (620 mg/kg) and zinc (170 mg/kg). Table 3-32 shows that all of the concentrations measured in AOI 6 frog tissue were well below these values, suggesting that, for these four metals, Sphagnum Bog surface water and/or sediment would not result in significant adverse effects to survival or growth when compared to background.

The fate and transport of COPCs can, of course, be affected by aquatic chemistry, particularly for metals in low pH environments as increased acidity generally increases the concentration of dissolved metals. Field pH measurements of the study area and background bogs ranged largely between 3.55 and 4.69 standard units (Table 3-33), conditions common for New England Sphagnum bogs. There was also no apparent difference between temperature, specific conductivity and dissolved oxygen values for both bogs. It can therefore be concluded that the aquatic chemistry of the bogs cannot explain any observed differences in tissue concentrations of site-related COPCs.

3.3.3 Assessment Endpoint #3 (Fish)

Measurement Endpoint 3A: Compare Site surface water concentrations to published chronic surface water benchmarks and to background.

Site surface water concentrations were compared to surface water benchmarks and background as described in Section 3.1.1 under Measurement Endpoint 1A.

3.3.4 Assessment Endpoint #4 (Waterfowl)

Measurement Endpoint 4A: Compare estimated daily dose for omnivorous waterfowl (mallard duck) based on ingestion of prey in Site exposure areas to published avian TRVs and to background.

Both total and incremental risk HQs calculated using the mallard duck food chain model are presented in Table 3-34 (AOI 6 - Sphagnum Bog – mineral sediment), Table 3-35 (AOI 6 - Sphagnum Bog – peat), Table 3-36 (AOI 6 - Sphagnum Bog – moss), Table 3-37 (AOI 10 - Northeast Wetland), Table 3-38 (AOI 18A - Assabet River Main Channel), and Table 3-39 (AOI 18B - Assabet River Embayment).

Measurement Endpoint 4B: Compare estimated daily dose for predatory wading birds (great blue heron) based on ingestion of prey in Site exposure areas to published avian TRVs and to background.

Both total and incremental risk HQs calculated using the great blue heron food chain model are presented in Table 3-34 (AOI 6 - Sphagnum Bog – mineral sediment), Table 3-35 (AOI 6 - Sphagnum Bog – peat), Table 3-36 (AOI 6 - Sphagnum Bog – moss), Table 3-37 (AOI 10 - Northeast Wetland), Table 3-38 (AOI 18A - Assabet River Main Channel), and Table 3-39 (AOI 18B - Assabet River Embayment).

Measurement Endpoint 4C: Compare estimated daily dose for piscivorous birds (osprey) based on ingestion of prey in Site exposure areas to published avian TRVs and to background.

Both total and incremental risk HQs calculated using osprey food chain model are presented in Table 3-34 (AOI 6 - Sphagnum Bog – mineral sediment), Table 3-35 (AOI 6 - Sphagnum Bog – peat), Table 3-36 (AOI 6 - Sphagnum Bog – moss), Table 3-37 (AOI 10 - Northeast Wetland), Table 3-38 (AOI 18A - Assabet River Main Channel), and Table 3-39 (AOI 18B - Assabet River Embayment).

3.3.5 Assessment Endpoint #5 (Semi-Aquatic Mammals)

Measurement Endpoint 5A: Compare estimated daily dose for omnivorous small mammals (shrew) based on ingestion of prey in Site exposure areas to published mammalian TRVs and to background.

Both total and incremental risk HQs calculated using the shrew food chain model are presented in Table 3-34 (AOI 6 - Sphagnum Bog – mineral sediment), Table 3-35 (AOI 6 - Sphagnum Bog – peat), Table 3-36 (AOI 6 - Sphagnum Bog – moss), Table 3-37 (AOI 10 - Northeast Wetland), Table 3-38 (AOI 18A - Assabet River Main Channel), and Table 3-39 (AOI 18B - Assabet River Embayment).

Measurement Endpoint 5B: Compare estimated daily dose for omnivorous larger mammals (raccoon) based on ingestion of prey in Site exposure areas to published mammalian TRVs and to background.

Both total and incremental risk HQs calculated using the raccoon food chain model are presented in Table 3-34 (AOI 6 - Sphagnum Bog – mineral sediment), Table 3-35 (AOI 6 - Sphagnum Bog – peat), Table 3-36 (AOI 6 - Sphagnum Bog – moss), Table 3-37 (AOI 10 - Northeast Wetland), Table 3-38 (AOI 18A - Assabet River Main Channel), and Table 3-39 (AOI 18B - Assabet River Embayment).

3.3.6 Assessment Endpoint #6 (Terrestrial Plants)

Both total and incremental risk HQs calculated using study area and background soil RME and CTE EPCs as part of the screening benchmarks evaluation and effects benchmark evaluation for terrestrial plants are presented in Table 3-40. HQs could not be calculated for those COPCs which lacked available benchmarks, and incremental risk HQs could not be calculated for those COPCs which were not analyzed in background samples.

3.3.7 Assessment Endpoint #7 (Terrestrial Invertebrates)

Measurement Endpoint 7A: Compare Site soil concentrations to published soil benchmarks and to background.

Both total and incremental risk HQs calculated using study area and background soil RME and CTE EPCs as part of the screening benchmarks evaluation and effects benchmark evaluation for terrestrial invertebrates are presented in Table 3-41. HQs could not be calculated for those COPCs which lacked available benchmarks, and incremental risk HQs could not be calculated for those COPCs which were not analyzed in background samples.

3.3.8 Assessment Endpoint #8 (Songbirds)

Measurement Endpoint 8A: Compare estimated daily dose for omnivorous songbirds (cardinal) based on ingestion of prey in Site exposure areas to published avian TRVs and to background.

Both total and incremental risk HQs calculated using the cardinal food chain model for site-wide soils are presented in Table 3-42.

Measurement Endpoint 8B: Compare estimated daily dose for invertivorous songbirds (American robin) based on ingestion of prey in Site exposure areas to published avian TRVs and to background.

Both total and incremental risk HQs calculated using the robin food chain model for site-wide soils are presented in Table 3-42.

3.3.9 Assessment Endpoint #9 (Terrestrial Mammals)

Measurement Endpoint 9A: Compare estimated daily dose for herbivorous small mammals (meadow vole) based on ingestion of prey in Site exposure areas to published mammalian TRVs and to background.

Both total and incremental risk HQs calculated using the meadow vole food chain model for site-wide soils are presented in Table 3-42.

Measurement Endpoint 9B: Compare estimated daily dose for omnivorous large mammals (red fox) based on ingestion of prey in Site exposure areas to published mammalian TRVs and to background.

Both total and incremental risk HQs calculated using the red fox food chain model for site-wide soils are presented in Table 3-42.

3.3.10 Assessment Endpoint #10 (Rare, Threatened, & Endangered Species)

Measurement Endpoint 10A: Compare Site sediment and surface water concentrations to published sediment and chronic surface water benchmarks and to background.

Both total and incremental risk HQs calculated using study area and background RME and CTE EPCs as part of the sediment (screening and effects benchmark evaluation) and surface water (chronic benchmark evaluation) for benthic invertebrates are presented in Table 3-20 (AOI 18A

- Assabet River Main Channel surface water), and Table 3-26 (AOI 18A - Assabet River Main Channel sediment) as described in Section 3.3.1 and 3.3.2.

4. RISK CHARACTERIZATION

Risk characterization includes two major components: risk estimation and risk description. Risk estimation consists of integrating the exposure profiles with the effects information. Risk description provides information important for interpreting the risk results. Several species and communities were used to evaluate risk to ecological receptors in the aquatic/wetland exposure areas (AOI 6 - Sphagnum Bog, AOI 10 - Northeast Wetland, AOI 18A - Assabet River, and AOI 18B - Assabet River Embayment) and the site-wide soil exposure area. The risk for AOI 4 - Cooling Water Recharge Pond is not characterized because the need for a presumptive remedy has already been identified.

In this section, each of the assessment endpoints are reviewed, results for measurement endpoints are analyzed, and the relationship between assessment and measurement endpoints is discussed, including the confidence in the relationship relative to accurately predicting risk. A weight of evidence approach is used to make conclusions regarding risk of harm for assessment endpoints with more than one measurement endpoint. Measurement endpoints were each assigned an inference weight, based upon how closely they represented the assessment endpoint (Table 2-1). Conclusions regarding risks to that assessment endpoint were reached by considering the inference weight for each measurement endpoint, i.e., the overall weight of evidence.

Ideally, risk characterization would be based on a dose-response curve for each COPC and receptor combination. However, for most ecological receptors, sufficient information to establish dose-response curves is not available in the published scientific literature. Instead, the likelihood of an adverse population level effect was based largely on whether the risk HQs was greater than or less than unity. In the case of food chain models (measurement endpoints 4A, 4B, 4C, 5A, 5B, 8A, 8B, 9A, and 9B) where four sets of HQs were calculated using the various combination of NOAEL and LOAEL TRVs with the RME and CTE EPCs, risk was initially characterized using the Four-Way Interpretative Risk Matrix (Pauwels, 2008) that incorporates all four EPC/TRV combinations to derive a risk conclusion and associated confidence level:

Four-Way Interpretative Risk Matrix					
RME/NOAEL HQ	**RME/LOAEL HQ**	**CTE/NOAEL HQ**	**CTE/LOAEL HQ**	**Risk Conclusion:** Adverse population-level effects:	**Confidence Level**
≤ 1	≤ 1	≤ 1	≤ 1	Unlikely	High
> 1	≤1	≤ 1	≤ 1	Unlikely	High
> 1	> 1	≤ 1	≤ 1	Unlikely	Moderate
> 1	≤ 1	> 1	≤ 1	Possible	Low
> 1	> 1	≥ 1	≤ 1	Possible	Moderate
> 1	> 1	> 1	> 1	Possible	High (increases with higher HQs)

In the case of benchmark evaluations (measurement endpoints 1B, 2B, 6A, and 7A) where HQs were calculated using toxicity benchmarks and not TRVs, the screening benchmark was used in place of the

NOAEL and the effects benchmark was used in place of the LOAEL. Where incremental risk HQs were not available because a COPC was not analyzed in background samples, then the risk conclusion is based only on study area HQs.

It was not possible to characterize risk using the Four-Way interpretative matrix when HQs were calculated using only screening benchmarks where effects benchmarks were unavailable. In such cases, the Two-Way Interpretative Risk Matrix shown below was used as a guide to address the range of risk conclusions and confidence levels.

Two-Way Interpretative Risk Matrix for screening benchmarks only			
RME/Screening Benchmark HQ	**CTE/Screening Benchmark HQ**	**Risk Conclusion:** Adverse population-level effects:	**Confidence Level**
≤ 1	≤ 1	Unlikely	High
> 1	≤ 1	Unlikely	Moderate
> 1	> 1	Possible	Low (increases with higher HQs)

It was also not possible to characterize risk using the Four-Way interpretative matrix for HQs calculated using the chronic surface water benchmarks. In such cases, the Two-Way Interpretative Risk Matrix shown below was used as a guide to address the range of risk conclusions and confidence levels.

Two-Way Interpretative Risk Matrix for chronic surface water benchmarks			
RME/Chronic Benchmark HQ	**CTE/Chronic Benchmark HQ**	**Risk Conclusion:** Adverse population-level effects:	**Confidence Level**
≤ 1	≤ 1	Unlikely	High
> 1	≤ 1	Unlikely	Moderate
> 1	> 1	Possible	high

Note that the reason for the separate two-way interpretative risk matrices is that chronic surface water benchmarks are "effects" thresholds which, when exceeded, can be expected to result in risk with a high level of confidence. On the other hand, exceeding a screening (i.e., no effect) benchmark which lacks an effect threshold may result in risk but only at a low level of confidence.

If the HQ based on the RME EPC and screening benchmark is greater than 1 but the HQ based on the CTE and effects benchmark is less than 1, then risk is interpreted to be unlikely with low confidence as long as the HQ is also less than 10. This interpretation tacitly extrapolates an effects benchmark by adjusting the screening benchmark upward 10-fold, an assumption which is commonly used by risk assessors to extrapolate LOAELs from NOAELS; an HQ based on the RME and screening benchmark that is less than 1 would also correspond to an HQ less than 10 following the 10-fold upward benchmark adjustment. Likewise, an HQ based on the RME and screening benchmark which is greater than 1 would correspond to an HQ greater than 10 after the 10-fold upward benchmark adjustment.

Risk could not be characterized if neither a screening level nor an effects benchmark, or a chronic surface water benchmark, were available for a given COPC. Incremental risk HQs below zero indicate that site related risk is less than background risk. Confidence level increases with higher HQs because the higher the HQ, the more likely adverse effects will occur.

Because a SLERA was performed and because most of the screening benchmarks used in the SLERA were carried over to the BERA, the summaries of each of the exposure pathways will primarily focus on the "Effects Benchmark Evaluation" HQs and the chronic surface water benchmark evaluation. Incremental risks based on Screening Benchmarks will be discussed, but it is still important to consider that the Effects Benchmarks will reflect a more realistic receptor exposure. This approach is valid as most, if not all, of the Effects HQs are lower than the Screening HQs because the benchmarks for the latter are more conservative in nature.

4.1 AOI 4 – Cooling Water Recharge Pond

Measurement endpoints were not evaluated in AOI 4 - Cooling Water Recharge Pond because the need for a presumptive remedy, primarily based on the level of COPCs in sediment as a result of receiving discharge of water from the facility in the past, has already been identified.

4.2 AOI 6 – Sphagnum Bog

This section characterizes risk from COPCs identified in AOI 6 – Sphagnum Bog. Surface water in the bog, as is typical of all bog ecosystems, has an acid pH but also has a high concentration of dissolved organic matter which helps buffer the 'activity' of metals in surface water. Sediment in the Sphagnum bog was divided into three fractions: mineral sediment, peat, and moss.

4.2.1 Assessment Endpoint #1 (Benthic Invertebrates)

Measurement Endpoint 1A: Compare Site surface water concentrations to published chronic surface water benchmarks and to background.

Chronic Benchmark HQs: Risk from the elements titanium and tungsten, and from organic compounds categorized as Extractable Petroleum Hydrocarbon (EPH) fractions, could not be characterized because these COPCs lacked chronic surface water benchmarks. The Study area RME and CTE EPCs for titanium are below background EPCs (Table 3-18), implying that risk – if it were present - would also fall below background and would therefore not be actionable for this metal. Irwin (1997) reports that the primary hazards from petroleum hydrocarbon mixtures are more strongly related to PAHs, which were evaluated individually.

Risk from VOCs and PCBs in Sphagnum Bog surface water is negligible because they were screened out during the initial COPC selection process.

Table 3-18 shows that RME HQs above unity consisted of chrysene (2.2) and ten dissolved metals, with the highest exceedances shown by lead (150), copper (72), cadmium (34), silver (5.6) and uranium (5.9).

Incremental Risks: The surface water benchmark risk characterizations in AOI 6 - Sphagnum Bog based on incremental risk suggest that risk from 4-methylphenol is possible with a high confidence level (Table 4-1).

Risk from dissolved chromium, iron, manganese, mercury, nickel, selenium, vanadium, and zinc in Sphagnum Bog surface water is unlikely (Table 4-1).

Risk from dissolved aluminum, barium, beryllium, and silver was characterized as unlikely but only with a moderate level of confidence because the RME IR (but not the CTE IR) exceeded 1.0. Finally, risk from dissolved cadmium, copper, lead, and uranium was characterized as possible with a high confidence level based on RME IRs and CTE IRs above 1.0 (Table 4-1). Lead (150 and 17), copper (72 and 19) and uranium (5.9 and 4.0) had the highest RME IRs and CTE IRs.

Therefore, incremental risk to benthic invertebrates exposed to surface water at AOI 6 – Sphagnum Bog is possible primarily from exposure to dissolved lead, copper, and uranium (high confidence) but also from exposure to dissolved silver, barium, and aluminum (moderate confidence). Bogs are known to bind divalent metals generated from anthropogenic sources, and this issue is further discussed in the Uncertainty Analysis (Section 4.7).

<u>Measurement Endpoint 1B</u>: Compare Site sediment concentrations to published sediment benchmarks and to background.

Effects Benchmark HQs: This evaluation addresses HQs for both the Southwest Corner (Table 3-22.1) and Remaining Area (Table 3-22.2) of the Sphagnum Bog.

Risk to the benthic community from m/p-methylphenol, barium, beryllium, thallium, thorium, titanium, tungsten, and zirconium could not be evaluated in bog sediment because sediment benchmarks were not available for these COPCs. Study area RME and CTE EPCs for barium, thorium, and titanium are also below background EPCs (Table 3-22.1 and 22.2), implying that risk would also be below background and therefore would be unlikely.

Risk from EPH fractions could not be characterized using benchmark comparisons because benchmarks could not be identified in the literature. However, Irwin (1997) reports that the primary hazards from petroleum hydrocarbon mixtures are more strongly related to PAHs, which were evaluated individually using the ΣPAH model.

Sediment benchmark risk characterizations based on ΣPAH models (Table 3-28), SEM/AVS calculations (Table 3-29) and incremental risk HQs (Table 4-2 through Table 4-4) for both the Southwest Corner and the Remaining Area of the Sphagnum Bog all suggest that risks to the benthic community from PAHs, benzoic acid, divalent cations (i.e., cadmium, copper, lead, mercury, nickel, silver, and zinc), aluminum, antimony, arsenic, and chromium are unlikely across all three mineral, peat, and moss sediment fractions. PAH concentrations are protective of the benthic community because all ΣPAH values were below 1.0. Divalent cations are not likely to be bioavailable and therefore not toxic based on the fact that the SEM/AVS values were all below the toxic threshold of 130 μmole/gOC (USEPA, 2005). Risk from COPCs which could not be eliminated are further discussed below.

Acetone was the only VOC identified as a COPC in AOI 6 – Sphagnum Bog for mineral sediment in both the Southwest Corner and the Remaining Area. Based on screening benchmarks, acetone was characterized as posing a possible risk to the benthic community in the mineral fraction, but was screened out as a COPC in peat and moss.

For the Southwest Corner, Site RME HQs for Aroclor 1254, copper and mercury were 3, 8.2 and 4.5, respectively; Site CTE HQs for the same respective compounds were 1.1, 3.4 and 1.0. For the Remaining Area, HQs for both the Site RME and CTE were all below unity. These results reflect the higher concentrations of these constituents in the Southwest Corner as compared to the Remainder of the bog.

Incremental Risk HQs: Acetone incremental risk HQs for both the Southwest Corner and the Remaining Area were based only on screening benchmarks (RME HQ = 6.1 and CTE HQ = 3.4; effects benchmarks could not be identified), so there is some uncertainty as to whether detected concentrations are above concentrations at which adverse effects might occur. Also, given the conservative nature of screening benchmarks, risk from acetone is likely overestimated. It is therefore concluded that risk to aquatic receptors from acetone in Sphagnum Bog mineral sediment is unlikely.

The ΣPAH model indicated that risk to the benthic community from sediment PAHs for the whole bog was unlikely.

Sediment benchmark evaluations based on incremental risk HQs (screening benchmarks) show possible effects in the Southwest Corner of Sphagnum Bog mineral sediment from phenol, PCBs, cadmium, chromium, copper, lead, mercury, molybdenum, nickel, silver and uranium. However, effects benchmark incremental risks, based on RME/CTE incremental HQs, only show risk from Aroclor 1254 (3.0/1.1), copper (8.2/3.4) and mercury (4.5/1.0). Although incremental risks based on screening benchmarks for the Remaining Area appear feasible, incremental risks based on effects HQs were all below 1.0.

Risk from 2-methylnaphthalene was screened out of mineral and moss fractions, but risk to the benthic community was characterized as possible in peat because screening HQs for the RME EPC (1.3) and CTE EPC (1.2) were above 1.0. Considering that screening benchmarks tend to overestimate risk, and given the low HQs based on those screening levels, risk from 2-methylnaphthalene is likely overestimated and can be characterized as unlikely. Incremental risk was estimated in peat for acenaphthylene for the RME scenario (1.5) but not for the CTE.

Risk from 2,4-dinitrophenol was screened out of the peat and moss fractions, but risk to the benthic community was characterized as possible in the mineral fraction of the Remaining Area. However, this conclusion is based only on screening benchmarks because effects benchmarks could not be identified. Furthermore, this conclusion is based on only one detection; 2,4-dinitrophenol was detected in only one of nineteen mineral samples and was not detected in any of 17 peat samples and 4 moss samples. Although the screening RME EPC and CTE EPC HQs (both are 160) are above 1, there is a significant amount of uncertainty in concluding risk based on only screening benchmarks and one detection out of forty bog sediment samples.

Risk from 4–methylphenol to the benthic community was screened out of peat and moss, but was characterized as possible in the mineral fraction for the Remaining Area. This conclusion is based on only the screening benchmark (HQ = 1.5) because effects benchmarks could not be identified. Given the conservative nature of screening benchmarks and the lack of effects benchmarks, it is likely that risk is overestimated. Risk from 4-methylphenol in Sphagnum Bog sediment is therefore unlikely.

In the Southwest Corner, risk to the benthic community from phenol was screened out of the moss factions, but was characterized as possible in the mineral fraction (screening benchmark, low confidence) and peat (effects benchmark, high confidence). The HQs for phenol in the Southwest Corner and Remaining Area were 1.9 and 2.4, respectively; for peat, the screening and effects HQs for phenol were 9.2 and 7.7, respectively.

The sediment benchmark evaluations for Aroclor-1254 and Aroclor-1260 suggest that risk to the benthic community from these two COPCs is possible in the Southwest Corner mineral fraction (moderate and low confidence, respectively), unlikely (high confidence) in peat, and were screened out altogether as COPCs in moss (Tables 4-1 through 4-4).

The sediment benchmark evaluations identified molybdenum, mercury and uranium as metals for which risk to the benthic community was characterized as possible, and HQs are generally more elevated in the Southwest Corner; the SEM/AVS evaluation indicated that divalent metals would not be bioavailable. Confidence is low for both molybdenum and uranium across all three sediment fractions. Molybdenum HQs range from 0.056 (moss CTE/ Effects) to 21 (mineral and peat RME/Screening). Uranium HQs range from 0.029 to 5.2 in the mineral fraction, 0.01 to 1.7 in peat, and 0.019 to 6.4 in moss.

Risk to benthic invertebrates from sediment at AOI 6 – Sphagnum Bog therefore is possible from phenol (low to high confidence), Aroclor-1254 (moderate confidence), Aroclor-1260 (low confidence), mercury (high confidence), molybdenum (low confidence), and uranium (low confidence). In general, the more elevated concentrations are seen in the Southwest Corner (particularly PCBs and mercury) and the risk estimates are improved by segregating data from that location away from the bog as a whole.

<u>Measurement Endpoint 1C</u>: Perform laboratory toxicity tests to measure survival and growth of a freshwater benthic invertebrate (*Chironomus dilutus*) exposed to sediments collected from the Site and compare to background.

Statistically significant effects on *C. dilutus* survival were observed at SD-RI-0600100R (56.3%), SD-RI-0600900R (52.5%), SD-RI-0602100R (62.5%) and SD-RI-0603600R (62.5%). Statistically significant effects on *C. dilutus* growth were observed at SD-RI-0600900R, SD-RI-0602100R, SD-RI-06053000.

Correlation coefficients were calculated (Table 4-5) and concentrations responses were plotted (Appendix K) to better relate observed effects with measured sediment PCB and metal concentrations. Sediment toxicity did not appear to be strongly correlated with any single COPC (Table 4-5); correlation coefficients ranged from -0.673 to 0.273 for survival, and from -0.525 to -0.017 for growth. However, the maximum concentration for Aroclor-1254 and nineteen of the twenty-five metals occurred at SD-RI-0600900, which could reflect the observed 52% survival rate at that location. Location SD-RI-0600100R, which exhibited a 51% survival rate, was associated with the second-highest detected concentrations for eleven metals. In these two cases the sediment chemistry appears to reflect the toxicity data.

The lack of strong correlation coefficient does not necessarily mean that a particular COPC is not responsible for any adverse effects to laboratory organisms. The reason for this is that sediments contain a mixture of chemical constituents, some of them COPCs but some not related to the Site (e.g. naturally occurring metals; humic and fulvic acids). Also, it only takes two or three data points to bias the dose/response (X vs. Y axis) regression line. For example, the percent survival vs. sediment concentration for nickel has a correlation coefficient of -0.673 (Appendix K), but the graph shows considerable scatter in the data, with three data points biasing the regression trend line.

Curiously, station SD-RI-0605300, which resulted in ecologically significant effects on growth and which were associated with an impaired biological condition (see Measurement Endpoint 1D below) were consistent with COPC concentrations at other AOI 6 locations that did not exhibit adverse effects (Table 4-5). Thus, characterizing risk as a result of site-related contamination at SD-RI-0605300 is uncertain.

Measurement Endpoint 1D: Compare the community structure of benthic invertebrates in Site sediments to background.

Of the thirteen benthic community samples, eight were characterized as non-impaired (Table 4-6), making this the prevailing condition of the Sphagnum Bog with respect to the benthic community. Five samples were characterized as having either slight or moderate impairment. No samples were characterized as having severe impairment. One important caveat is that only one sample per location was obtained during the benthic assessment, whereas routine procedure typically requires at least three. Thus, these results are uncertain as the lack of replicates limits the statistical power of the results.

Summary of Measurement Endpoint 1:

Table 4-6 compares multiple lines of evidence pertaining to the benthic macroinvertebrate community within AOI 6 – Sphagnum Bog. In total, thirteen individual sample locations were evaluated using at least one leg of the sediment triad approach (benchmark comparisons, benthic community surveys, and sediment toxicity tests). Co-located (mineral) sediment toxicity test data and benthic community survey data were collected from twelve locations; SEM/AVS data were available at five locations, and the ΣPAH model was applied at all thirteen sample locations. One location was evaluated only by the benthic community survey and not with toxicity tests.

Investigation results from SD-RI-0600100R, SD-RI-0602500R, SD-RI-0605200, SD-RI-0605300, SD-RI-0603800R, SD-RI-060500, and SD-RI-060510 demonstrate that these locations have not been adversely affected relative to the reference locations. Toxicity test results and benthic community survey at all seven of these locations indicate there were no adverse effects observed relative to background. The only SEM/AVS sample in this sub-group, collected at SD-RI-0602500R, also suggests that divalent metals are not likely to be bioavailable there. PAH concentrations are predicted to be protective of the benthic community at all six locations.

Investigation results also suggest that SD-RI-0600500R, SD-RI-0601700R, and SD-RI-0605200 have not been affected by historical releases. While the medium weight benthic community survey indicates a slight to moderate level of impairment, the low weighted SEM/AVS and ΣPAH results and the highest weighted sediment toxicity tests all suggest no adverse ecological effects.

At SD-RI-0600100R and SD-RI-0600900R, SEM/AVS data suggest divalent cations are unlikely to be bioavailable and the ΣPAH values corroborate the benthic community data that suggest there is no community impairment relative to background. However, the sediment toxicity test results, which are weighted higher than the SEM AVS data, ΣPAH values, and the benthic community survey, demonstrated adverse ecologically significant effects. The highest and second-highest concentrations of metals and PCBs in sediment toxicity samples frequently occurred at SD-RI-0600900 and SD-RI-0600100R, respectively.

At SD-RI-0605300, sediment toxicity tests identified significant growth effects and the benthic community was scored as moderately impaired relative to background; SEM/AVS data were not available for this location, and ΣPAH values predict that PAH concentrations are protective of the benthic community. However, there was no apparent chemical explanation for the observed effects since concentrations at SD-RI-0605300 were consistent with concentrations at locations that did not exhibit adverse effects, suggesting that risk there has been overestimated. Therefore, there is a fair degree of uncertainty in concluding risk at SD-RI-06-05300.

Overall, eight of the thirteen macroinvertebrate samples obtained directly from AOI 6 - Sphagnum Bog demonstrate that there are no adverse impacts to the benthic community. Adverse effects seen in the toxicity tests conducted on samples SD-RI-0600100R and SD-RI-0600900 appear to reflect the elevated sediment metal and PCB concentrations. While risk at SD-RI-0605300 was characterized as possible using the available lines of evidence, chemical concentrations at SD-RI-0605300 were consistent with concentrations at other locations which did not exhibit adverse effects.

Benchmark evaluations were the only line of evidence used to evaluate risk in moss and peat. Molybdenum and uranium were the only two metals that were characterized as causing possible risk in peat and moss, but confidence is low.

4.2.2 Assessment Endpoint 2 (Amphibians)

Measurement Endpoint 2A: Compare Site surface water concentrations to published chronic surface water benchmarks and to background.

Incremental risk to amphibians from surface water at AOI 6 – Sphagnum Bog is possible primarily from exposure to dissolved lead, copper, and uranium (high confidence) but also from exposure to dissolved silver, barium, and aluminum (moderate confidence). Potential risk to amphibians and pelagic aquatic receptors is indicated based on comparison of Site surface water concentrations with chronic surface water benchmarks. Interpretation of, and response, to this potential risk is further discussed with regard to reaching a scientific management decision point (SDMP) on Sphagnum Bog surface water in this section. This single line of evidence carries significant uncertainty in a bog setting, where contaminants are uniquely subject to binding with naturally-occurring organic substances such as tannins that may greatly reduce bioavailability and/or toxicity.

This uncertainty prompted EPA to re-evaluate whether useful site-specific information on aquatic toxicity could be gleaned from the FETAX toxicity test, in spite of the limitations imposed by sample pH values below test acceptability criteria. Appendix O evaluates the available evidence from this test. Alternative site-specific aquatic toxicity values were developed from the FETAX test which suggest that comparison with generic chronic surface water benchmarks over-estimates risk in a bog setting. This issue is further discussed in the Uncertainty Analysis (Section 4.7).

Measurement Endpoint 2B: Compare Site sediment concentrations to published sediment benchmarks and to background.

Risk from sediment COPCs in AOI 6 – Sphagnum Bog was characterized as described in Section 4.2.1. Risk to amphibians from sediment at AOI 6 – Sphagnum Bog therefore is possible from phenol (low to high confidence), Aroclor-1254 (moderate confidence), Aroclor-1260 (low confidence), molybdenum (low confidence), and uranium (low confidence).

Measurement Endpoint 2C: Compare amphibian tissue COPC concentrations to both concentrations in background bog organisms as well as Critical Body Burdens (tissue concentration thresholds thought to be adverse to the organism).

Section 3.3.2 discussed the data for both PCBs and metals in AOI 6 and background bog frog tissue (mg/kg). The data showed that most metals were near or below background. PCBs and uranium were increased in AOI 6 frogs as compared to organisms captured in the reference bog, although not every sample was above the detection limit for the average calculation. Cadmium, copper, lead and zinc were well below Critical Body Burden values identified in the literature.

Summary of Measurement Endpoint 2:

Since the direct measurement of COPCs in amphibian tissue would have a higher weight than the benchmark evaluations, it could be concluded that risk to amphibians at AOI 6 – Sphagnum Bog is low as some organisms had elevated levels of uranium and PCBs while others were below the detection limit. A walkover of the bog showed that frogs and their offspring appear to be the most plentiful in the Southwest Corner. Since the latter tended to have the higher concentration of COPCs, relative to the remainder of the bog, and since PCBs can bioaccumulate, the potential for an increased opportunity for bioaccumulation of PCBs into amphibians should be weighted into the risk assessment conclusions.

4.2.3 Assessment Endpoint #4 (Wetland Birds)

Measurement Endpoint 4A: Compare estimated daily dose for omnivorous waterfowl (mallard duck), based on ingestion of prey in Site exposure areas, to published avian TRVs and to background.

Risk RME and CTE HQs based on food chain models for mallard ducks in AOI 6 – Sphagnum Bog were characterized for mineral (Table 4-7), moss (Table 4-8), and peat fractions (Table 4-9). In the mineral, peat, and moss fractions, food chain models suggest that risk to mallard duck from COPCs is unlikely (high confidence). Risk from iron, titanium, tungsten, and EPHs could not be characterized because TRVs could not be identified.

Measurement Endpoint 4B: Compare estimated daily dose for predatory wading birds (great blue heron) based on ingestion of prey in Site exposure areas to published avian TRVs and to background.

Risk RME and CTE HQs based on food chain models for great blue heron in AOI 6 – Sphagnum Bog were characterized for mineral (Table 4-7), moss (Table 4-8), and peat (Table 4-9) fractions. In the mineral, peat, and moss fractions, food chain models suggest that risk to great blue heron is possible from beryllium (high confidence). Risk from iron, titanium, tungsten, and EPHs could not be characterized because TRVs could not be identified. Ingestion of substrate accounts for approximately 89% to 94% of risk and ingestion of amphibians accounts for approximately 5% to 11%of risk (Appendix K).

4.2.4 Assessment Endpoint #5 (Wetland Mammals)

Measurement Endpoint 5A: Compare estimated daily dose for omnivorous small mammals (shrew), based on ingestion of prey in Site exposure areas, to published mammalian TRVs and to background.

Risk RME and CTE HQs based on food chain models for short-tailed shrew in AOI 6 – Sphagnum Bog were calculated for mineral (Table 4-7), moss (Table 4-8), and peat (Table 4-9) fractions. In the mineral, peat, and moss fractions, food chain models suggest that risk from molybdenum (low to moderate confidence) is possible. In the mineral sediment, the HQs are above unity (although risk is unlikely) from Aroclor 1254 (RME).

Risk from vanadium in the peat fraction was identified as possible (low confidence). However, incremental risk HQs based on screening benchmarks were near unity (1.6 and 1.4). Given the conservative nature of screening benchmarks, it is likely that these HQs overestimate risk from vanadium. Risk from vanadium to shrew at AOI 6 Sphagnum bog is therefore unlikely.

Risk from iron, titanium, tungsten, and EPHs could not be characterized because TRVs could not be identified. Ingestion of substrate (mineral, peat, or moss) accounts for approximately 59% to 84% of risk, ingestion of plants accounts for approximately 3% to 5% of risk, ingestion of invertebrates accounts for approximately 13% to 35% of risk, and ingestion of mammal prey accounts for approximately 1% to 4% of risk (Appendix K).

Measurement Endpoint 5B: Compare estimated daily dose for omnivorous larger mammals (raccoon), based on ingestion of prey in Site exposure areas, to published mammalian TRVs and to background.

Risk RME and CTE HQs based on food chain models for raccoon in AOI 6 – Sphagnum Bog were calculated for mineral (Table 4-7), moss (Table 4-8), and peat (Table 4-9) fractions. In the mineral, peat, and moss fractions, food chain models suggest that risk to raccoon from all COPCs is unlikely (high confidence). Risk from iron, titanium, tungsten, and EPHs could not be characterized because TRVs could not be identified.

4.2.5 Summary for AOI – 6 Sphagnum Bog

Several dissolved metals showed RME and CTE incremental risks above unity, indicating possible risk from exposure to surface water with a high level of confidence. The sediment benchmark evaluations for Aroclor-1254 and Aroclor-1260 suggest that risk to the benthic community from these two COPCs is possible in the Southwest Corner mineral fraction. The sediment benchmark evaluations identified molybdenum, mercury and uranium as metals for which risk to the benthic community was characterized as possible, and HQs are generally more elevated in the Southwest Corner; the SEM/AVS evaluation indicated that divalent metals would not be bioavailable.

Based on the weight of evidence, eight of the thirteen mineral sediment samples from AOI 6 - Sphagnum Bog demonstrate that there is no adverse impact to the benthic macroinvertebrate community, making this the predominant condition in the bog (although this result was based on a single sample obtained at each location during a drought conditions). Adverse effects in *C. dilutus* sediment toxicity tests at SD-RI-0600900 and SD-RI-0600100R appear to reflect the elevated sediment metal and PCB concentrations at these stations. While risk at SD-RI-0605300 was characterized as possible using the available lines of evidence, chemical concentrations at this location were consistent with concentrations at other locations which did not exhibit adverse effects, suggesting risk there may be overestimated. Sediment toxicity did not appear to be strongly correlated with any single COPC. Benchmark evaluations were the only line of evidence used to evaluate risk in moss and peat; molybdenum and uranium were the only two metals that were characterized as causing possible risk in peat and moss, but confidence is low.

Comparison of COPCs in amphibian tissue from the bog vs. the background bog showed that only uranium and PCBs were elevated in amphibians from the former; there is some uncertainty in this result as concentrations in some of the frogs from AOI 6 were below the detection limit, so the average result is slightly misleading. Tissue metal concentrations were well below Critical Body Burdens identified from the literature for amphibians (cadmium, copper, lead and zinc).

Medium weighted food chain models characterized risk to mallard duck and raccoon as unlikely. Food chain models suggest that risk to great blue heron is possible from beryllium (high confidence) in mineral, peat, and moss fractions. Food chain models suggest that risk to shrew is possible from molybdenum (low to moderate confidence) in all three sediment fractions. The highest concentrations of beryllium and molybdenum were detected in sediment samples SD-RI-06001 and SD-RI-06009, which were associated with toxicity to the benthic community, and sample SD-RI-06032 which is located near SD-RI-06009.

Segregation of the Southwest Corner data from the remainder of the bog sample data showed that most of the risk can be attributed to generally higher levels of COPCs (PCBs, Cu, Hg) in that portion of the wetland and lower concentrations in the Remaining Area. This appears to be logical given the historical operations that occurred at the former Nuclear Metals facility.

4.3 AOI 10 – Northeast Wetland

This section characterizes risks from COPCs identified in AOI 10 – Northeast Wetland.

4.3.1 Assessment Endpoint #1 (Benthic Invertebrates)

<u>Measurement Endpoint 1A:</u> Compare Site surface water concentrations to published chronic surface water benchmarks and to background.

Chronic Benchmark HQs: For the RME dissolved metals, the HQs above unity ranged from 1.0 (zinc) to 15 (lead), whereas for the CTE dissolved metals, the HQs above unity ranged from 1.1 (zinc) to 7.5 (lead) (Table 3-19).

Risk to the benthic community from exposure to titanium in surface water could not be evaluated because this COPC lacked a chronic surface water benchmark.

Incremental Risks: Surface water incremental risks (Table 4-10) suggest that risk from dissolved aluminum, barium, iron, and zinc are unlikely at AOI 10 – Northeast Wetland.

Risk to the benthic community from exposure to copper and lead (and to a minor degree, manganese) in the Northeast Wetland surface water was characterized as possible based on the chronic benchmark evaluation. Confidence in the conclusion that copper and lead pose risk to the benthic community is high for the both metals (Table 4-10).

Even though the confidence in the conclusion for zinc was moderate, the RME incremental risk HQ only equaled 1.0. Because RME exposure is local, it appears likely that the risk for this metal is overestimated. Thus, risk from zinc in Northeast Wetland surface water is unlikely. Divalent metals are also bound strongly to natural organic compounds in bog environs, which is further discussed in the Uncertainty Analysis.

Risk to benthic invertebrates exposed to surface water in AOI 10 - Northeast Wetland is therefore possible for copper and lead (high confidence) but unlikely for all other COPCs.

Measurement Endpoint 1B. Compare Site sediment concentrations to published sediment benchmarks and to background.

Effects Benchmark HQs: Risk to the benthic community from sediment 1,4-dioxane, barium, beryllium, thallium, thorium, titanium, tungsten, and zirconium could not be evaluated because these COPCs lacked both screening and effects benchmarks. However, study area RME and CTE EPCs for beryllium, thallium, thorium, titanium, and zirconium are below background EPCs (Table 3-25), implying that any risk would be below background and therefore non-actionable.

EPHs also lacked benchmarks. Irwin (1997) reports that the primary hazards from petroleum hydrocarbon mixtures are more strongly related to PAHs, which were evaluated individually and using the ΣPAH model.

For VOCs, the screening benchmark RME and CTE for acetone were 1.1 and 0.85, respectively. For SVOCs, RME HQs ranged from 1.1 to 71, with acenaphthylene contributing the apparent bulk of the hazard; this elevated HQ drops to near unity for the effects benchmark HQ. The screening HQ for the CTE ranged from 1.1 to 8.1, with acenaphthylene again contributing most of the apparent hazard. It can be observed that acenaphthylene was detected in only 3 of 12 samples, so the HQs for this COPEC are biased high. Screening HQs for both RME and CTE rarely exceeded 3.0 and most exceedances were for PAHs, levels of which have already been shown to be below the threshold for narcotic effects (see Table 3-28). Effects HQs only exceeded unity for acenaphthylene which, as mentioned, would most likely not pose a risk based on the ΣPAH narcosis model.

Incremental Risk HQs: Sediment benchmark risk characterizations based on incremental risk HQs (Table 4-11) suggest that risk from VOCs and metals (aluminum, cadmium, copper, lead, mercury, selenium) at AOI 10 – Northeast Wetland is unlikely. While risk from some PAHs (2-methylnaphthalene, acenaphthylene, benzo(a)anthracene, and fluorene) was characterized as possible using the screening and effects benchmarks, the ΣPAH values for Northeast Wetland sediment (Table 3-28) indicate that PAH concentrations are protective of the benthic community since all ΣPAH $ESBTU_{FCV}$ values are below 1.Risk to the benthic community in the Northeast Wetland from sediment COPCs is therefore unlikely.

In summary, risk to benthic invertebrates from surface water at AOI 10 - Northeast Wetland is possible for total and dissolved lead (low to moderate confidence). Recognizing that AOI 10 is adjacent to a busy road, the lead may not be related to historical activities at the site. Risk to the benthic community in the Northeast Wetland from sediment COPCs appears to be unlikely.

4.3.2 Assessment Endpoint #2 (Amphibians)

Measurement Endpoint 2A: Compare Site surface water concentrations to published chronic surface water benchmarks and to background.

Risk from surface water COPCs in AOI 10 – Northeast Wetland were characterized as described in Section 4.3.1. Hence, risk to amphibians exposed to surface water in AOI 10 - Northeast Wetland is possible for copper, lead and manganese (high confidence) but unlikely for all other COPCs.

Measurement Endpoint 2B: Compare Site sediment concentrations to published sediment benchmarks and to background.

Risk from sediment COPCs in AOI 10 – Northeast Wetland were characterized to benthic invertebrates as described in Section 4.3.1. Amphibians have less intimate contact with the benthos than macroinvertebrates. Risk to the amphibians in the Northeast Wetland from sediment COPCs is, therefore, unlikely.

4.3.3 Assessment Endpoint #4 (Wetland Birds)

Measurement Endpoint 4A: Compare estimated daily dose for omnivorous waterfowl (mallard duck) based on ingestion of prey in Site exposure areas to published avian TRVs and to background.

Risk RME and CTE HQs based on food chain models for mallard ducks in AOI 10 – Northeast Wetland are all below unity, which suggest that risk from COPCs is unlikely (high confidence) (Table 4-12). Risk from 1,4-dioxane, iron, titanium, tungsten, and EPHs could not be characterized because TRVs could not be identified.

Measurement Endpoint 4B: Compare estimated daily dose for predatory wading birds (great blue heron), based on ingestion of prey in Site exposure areas, to published avian TRVs and to background.

Risk RME and CTE HQs based on food chain models for great blue heron in AOI 10 – Northeast Wetland are all below unity, (Table 4-12) which suggest that risk from COPCs is unlikely (high confidence). Risk from 1,4-dioxane, iron, titanium, tungsten, and EPHs could not be characterized because TRVs could not be identified.

4.3.4 Assessment Endpoint #5 (Wetland Mammals)

Measurement Endpoint 5A: Compare estimated daily dose for omnivorous small mammals, (shrew) based on ingestion of prey in Site exposure areas, to published mammalian TRVs and to background.

Risk RME and CTE HQs based on food chain models for shrew in AOI 10 – Northeast Wetland (Table 4-12) suggest that risk from COPCs is unlikely (high confidence). HQs for 1,4-dioxane, iron, titanium, tungsten, and EPHs could not be characterized because TRVs could not be identified.

Measurement Endpoint 5B: Compare estimated daily dose for omnivorous larger mammals, (raccoon) based on ingestion of prey in Site exposure areas, to published mammalian TRVs and to background.

Risk RME and CTE HQs based on food chain models for raccoon in AOI 10 – Northeast Wetland (Table 4-12) suggest that risk from COPCs is unlikely (high confidence). HQs for 1,4-dioxane, iron, titanium, tungsten, and EPHs could not be characterized because TRVs could not be identified.

4.3.5 Summary for AOI 10 – Northeast Wetland

Low/medium weighted benchmark comparisons characterized risk to benthic invertebrates and amphibians at AOI 10 - Northeast Wetland as possible in surface water for lead, copper, and manganese (high confidence), and characterized risk as unlikely in sediment. Medium-weighted food chain models characterized risk to wetland birds and mammals as unlikely.

The Northeast Wetland is a seasonally-flooded isolated habitat, with standing water observed in May 2004, and dry substrate observed in June 2004 (*de maximus*, 2006). Sediment/soil represents the primary exposure pathway for community-level receptors since surface water is only present for a short period in early spring. The BERA risk evaluation for this habitat showed no actionable risk from sediment exposure, but potential risk from surface water exposure.

The surface water exposure concentrations were calculated from five samples (four locations) collected from the wetland in late April and early May 2004 (BERA Table 1-4). The risk conclusion for surface water was based on comparing the maximum measured concentrations to chronic surface water criteria. The dissolved metals levels, particularly for copper and lead, resulting in HQs above 1.0 (Table 3-19). The measured surface water hardness for the Northeast Wetland was very low (14.5 mg/L $CaCO_3$), and nearly identical to that measured in the Sphagnum Bog (i.e., 14.9 mg/L $CaCO_3$, BERA Table 3-9, footnote [d]). Therefore, the chronic surface water benchmarks were adjusted for the low hardness (Table 3-9).

These low benchmarks were developed for use in the BERA, even though true water-column species such as fish are absent from the Northeast Wetland. Benthic invertebrates exposed to wetland sediment best represent the most likely exposure conditions in this habitat. Even though the BERA found exceedances of chronic surface water benchmarks, the risk associated with this exposure pathway is considered low with high uncertainty because of the seasonal nature of the inundation and the small areal extent of the flooded areas in the wetland. Based on these considerations, it is recommended not to carry the surface water exposure pathway from the Northeast Wetland into the FS for remedial consideration since risk to the benthic invertebrate community in that same habitat is considered unlikely.

Ecological risk at AOI 10 – Northeast Wetland is considered to be unlikely based on the above discussion and the weight of evidence and confidence/uncertainties in the data (see Section 4.7, Uncertainty Analysis).

4.4 AOI 18A – Assabet River Main Channel

This section characterizes risk from COPCs identified in AOI 18A - Assabet River Main Channel.

4.4.1 Assessment Endpoint #1 (Benthic Invertebrates)

Measurement Endpoint 1A. Compare Site surface water concentrations to published chronic surface water benchmarks and to background.

Chronic Benchmark HQs: Barium was the only metal with a Site RME and CTE above unity (4.5 and 4.4, respectively) (Table 3-20). However, both the background RME and CTE exceeded these values, resulting in no site risk.

Incremental Risks: Surface water benchmark evaluations based on incremental risk (Table 4-13) suggests that risk to the benthic community exposed to dissolved metals at AOI 18A - Assabet River Main Channel is unlikely.

Measurement Endpoint 1B. Compare Site sediment concentrations to published sediment benchmarks and to background.

Effects Benchmark HQs: Risk from barium, beryllium, thallium, thorium, titanium, and tungsten could not be evaluated because these COPCs lacked both screening and effects benchmarks (Table 3-26). However, study area RME and CTE EPCs for these COPCs are below background EPCs, implying that risk would be below background levels.

Acetone was the only VOC characterized as causing possible risk in AOI 18A – Assabet River Main Channel sediment using the benchmark evaluation. Acetone HQs were based only on screening benchmarks (RME HQ = 7.3 and CTE HQ = 3.6), so there is some uncertainty as to whether detected concentrations are above concentrations at which adverse effects might occur. Also, given the conservative nature of screening benchmarks, risk from acetone is likely overestimated. Risk from VOCs in Assabet River Main Channel sediment is therefore unlikely.

Screening HQs ranged from 0.37 (carbon disulfide) to 8.0 (arsenic), but none of the RME or CTE screening or effects benchmark HQs exceeded the latter value, and many of the background HQs were equivocal.

Incremental Risk HQs: Sediment benchmark evaluations based on incremental risk HQs (Table 4-14) suggest that risk to the benthic community from carbon disulfide, aluminum, arsenic, cadmium, copper, iron, lead, manganese, mercury, nickel, vanadium and zinc at AOI 18A – Assabet River Main Channel is unlikely.

Chromium was the only metal characterized as causing possible risk (moderate confidence) to the benthic community using the benchmark evaluation. Incremental risk HQs ranged from 0.60 (CTE/Effects) to 6.0 (RME/Screening), with the CTE/Screening HQ (1.6) being just above 1.0. Given the conservative nature of screening benchmarks and the fact that bioavailability of chromium is sediment is generally low in reducing sediments, risk to the benthic community from the CTE/Screening HQ is likely overestimated, suggesting that risk from chromium in sediment is probably unlikely.

The risk from sediment COPCs to the benthic community at AOI 18A – Assabet River Main Channel is unlikely.

Based on the above analysis, it can be concluded that risk from surface water and sediment COPCs to the benthic community at AOI 18A – Assabet River Main Channel is unlikely.

4.4.2 Assessment Endpoint #2 (Amphibians)

Measurement Endpoint 2A: Compare Site surface water concentrations to published chronic surface water benchmarks and to background.

Risk from surface water COPCs in AOI 18A – Assabet River Main Channel was characterized as described in Section 4.4.1. Risk to amphibians from exposure to surface water at AOI 18A – Assabet River Main Channel is considered unlikely.

Measurement Endpoint 2B: Compare Site sediment concentrations to published sediment benchmarks and to background.

Risk from sediment COPCs in AOI 18A – Assabet River Main Channel was characterized as described in Section 4.4.1. Amphibians have less intimate contact with the benthos than macroinvertebrates. Risk to amphibians from sediment at AOI 18A – Assabet River Main Channel is, therefore, unlikely.

Based on the concentrations of COPCs in both surface water and sediment, risk from site-related media to amphibians at AOI 18A – Assabet River Main Channel is unlikely.

4.4.3 Assessment Endpoint #3 (Fish)

Measurement Endpoint 3A: Compare Site surface water concentrations to published chronic surface water benchmarks and to background.

Risk from surface water COPCs in AOI 18A – Assabet River Main Channel was characterized as described in Section 4.4.1. Risk to fish from surface water at AOI 18A – Assabet River Main Channel is considered unlikely.

4.4.4 Assessment Endpoint #4 (Wetland Birds)

Measurement Endpoint 4A: Compare estimated daily dose for omnivorous waterfowl (mallard duck) based on ingestion of prey in Site exposure areas to published avian TRVs and to background.

Risk RME and CTE HQs based on food chain models for the mallard duck in AOI 18A – Assabet River Main Channel (Table 4-15) suggest that risk from COPCs is unlikely (high confidence). HQs for iron, titanium, and tungsten could not be characterized because TRVs could not be identified.

Measurement Endpoint 4B: Compare estimated daily dose for predatory wading birds (great blue heron), based on ingestion of prey in Site exposure areas, to published avian TRVs and to background.

Risk RME and CTE HQs based on food chain models for great blue heron in AOI 18A – Assabet River Main Channel (Table 4-15) suggest that risk from chromium and zinc is possible (moderate confidence). However, risk HQs for chromium and zinc that are based on the

CTE/NOAEL are both 2.1, and the CTE/LOAEL HQ is <1.0. Given the conservative assumptions associated with NOAEL benchmarks, it is likely that risk to great blue heron is overestimated and that risk to great blue heron from these metals is unlikely (these metals also do not bioaccumulate or biomagnify). HQs for iron, titanium, and tungsten could not be characterized because TRVs could not be identified. Ingestion of sediment accounts for approximately 9% to 12% of risk, ingestion of amphibians account for approximately 4% of risk, and ingestion of fish accounts for approximately 84% to 88% of risk (Appendix K). It should also be noted that statistical analysis of the Assabet River Main Channel sediment data showed that there we no significant differences between chromium and zinc concentrations in the portion of the channel adjacent to the Site and the portion of the channel up-gradient of the Site. In addition, a migration pathway for these metals to migrate from the Site to the Main Channel does not exist. Collectively, this indicates that the HQs are not attributable to any Site-related contribution.

<u>Measurement Endpoint 4C</u>: Compare estimated daily dose for piscivorous birds (osprey), based on ingestion of prey in Site exposure areas, to published avian TRVs and to background.

Risk RME and CTE HQs based on food chain models for osprey in AOI 18A – Assabet River Main Channel (Table 4-15) suggest that risk from COPCs is unlikely (high confidence). HQs for iron, titanium, and tungsten could not be characterized because TRVs could not be identified.

4.4.5 Assessment Endpoint #5 (Wetland Mammals)

<u>Measurement Endpoint 5B</u>: Compare estimated daily dose for omnivorous larger mammals (raccoon), based on ingestion of prey in Site exposure areas, to published mammalian TRVs and to background.

Risk RME and CTE HQs based on food chain models for raccoon in AOI 18A – Assabet River Main Channel (Table 4-15) suggest that risk from COPCs is unlikely (moderate to high confidence). HQs for iron, titanium, and tungsten could not be characterized because TRVs could not be identified.

4.4.6 Assessment Endpoint #10 (Rare, Threatened, & Endangered Species)

<u>Measurement Endpoint 10A:</u> Compared Site sediment and surface water concentrations to published sediment and chronic surface water benchmarks and to background.

AOI 18B - Assabet River Main Channel also included freshwater mussels as an assessment endpoint population because the MANHESP indicated the presence of three species of concern, namely the eastern pondmussel (*Ligumia nasuta*), triangle floater (*Alasmidonta undulata*), and creeper (*Strophitus undulates*).

Risk from surface water and sediment COPCs in AOI 18A – Assabet River Main Channel was characterized as unlikely as described in Section 4.4.1. Therefore, risk to these species of special concern from COPCs is also considered unlikely.

Developmental success of freshwater mussels depends on a parasitic phase when larvae (called glochidia) must attach to the gills or fins of a vertebrate host to develop into juveniles (MANHESP, 2009). Studies have identified many vertebrate hosts, including a suite of species common in cool to warm-water streams in Massachusetts such as largemouth bass, fallfish, longnose dace, blacknose dace, common shiner, golden shiner, slimy sculpin, bluegill, and rock bass. This parasitic phase is also the only period during which mussels can disperse long distances, especially upstream. Any impacts to fish will also impact mussels. However, Assessment Endpoint 3 demonstrated that risk to fish from Site COPCs is unlikely. Distribution of these mussels is likely more attributable to the many impoundments that lie on the Assabet River that impede or prevent movement of fish (and therefore mussels) upstream.

4.4.7 Summary for AOI 18A – Assabet River Main Channel

Low/medium weighted benchmark comparisons characterized risk to benthic invertebrates, amphibians, and fish at AOI 18A – Assabet River Main Channel as unlikely for surface water and sediment. Medium weighted food chain models characterized risk to wetland birds and mammals as unlikely. Distribution of rare mussels is likely more attributable to the many impoundments that lie on the Assabet River that impede or prevent movement of fish (and therefore mussels) upstream. Based on the weight of evidence and confidence/ uncertainties in the data, ecological risk at AOI 18A – Assabet River Main Channel is unlikely.

4.5 AOI 18B – Assabet River Embayment

This section characterizes risk from COPCs identified in AOI 18B- Assabet River Embayment.

4.5.1 Assessment Endpoint #1 (Benthic Invertebrates)

<u>Measurement Endpoint 1A</u>. Compare Site surface water concentrations to published chronic surface water benchmarks and to background.

Chronic Benchmark HQs: Table 3-21 presents HQs which ranged from < 1.0 (zinc) to 3.6 (RME barium).

Incremental Risks: Chronic surface water benchmark evaluations based on incremental risk (Table 4-16) suggest that risk to the benthic community from metals at AOI 18B – Assabet River Embayment is unlikely. HQs for barium and lead were below background HQs. Risk from surface water COPCs to the benthic community in AOI 18B – Assabet River Embayment is, therefore, unlikely.

<u>Measurement Endpoint 1B</u>. Compare Site sediment concentrations to published sediment benchmarks and to background.

Effects Benchmark HQs: Metals were the only COPCs evaluated in AOI 18B – Assabet River Embayment sediment. Table 3-27 presents screening benchmark HQs for metals which ranged between 1.1 (CTE nickel) and 21 (RME chromium), although the majority of the HQs were below 10. Effects benchmark HQs only exceeded unity for chromium, lead, mercury and silver, although background effects HQs were also similar for these metals, which is indicative of a low risk based on total HQs.

Risk from barium, beryllium, thallium, thorium, titanium, tungsten, and zirconium could not be evaluated because these COPCs lacked both screening and effects benchmarks. However, study area RME and CTE EPCs for barium and thallium are below background EPCs (Table 3-27), implying that risk would be below background and therefore would be unlikely.

Incremental Risk HQs: Sediment benchmark evaluations based on incremental risk HQs (Table 4-17) suggest that risk to the benthic community from metals (aluminum, arsenic, cadmium, chromium, copper, iron, lead, manganese, mercury, nickel, selenium, silver, vanadium, and zinc) is unlikely. Incremental risk HQs for the majority of these metals were below zero, signifying that study area concentrations were below background conditions. Risk from surface water and sediment COPCs to the benthic community at AOI 18B – Assabet River Embayment is, therefore, unlikely.

4.5.2 Assessment Endpoint #2 (Amphibians)

<u>Measurement Endpoint 2A</u>: Compare Site surface water concentrations to published chronic surface water benchmarks and to background.

Risk from surface water COPCs in AOI 18B – Assabet River Embayment was characterized as described in Section 4.5.1. Risk to amphibians from exposure to surface water at AOI 18B – Assabet River Embayment is considered unlikely.

<u>Measurement Endpoint 2B</u>: Compare Site sediment concentrations to published sediment benchmarks and to background.

Risk from sediment COPCs in AOI 18B – Assabet River Embayment was characterized as described in Section 4.5.1. Amphibians have less intimate contact with the benthos than macroinvertebrates. Risk to amphibians from sediment at AOI 18B – Assabet River Embayment is, therefore, unlikely.

It can be concluded with a fair degree of confidence that risk from surface water and sediment COPCs to amphibians at AOI 18B – Assabet River Embayment is unlikely.

4.5.3 Assessment Endpoint #3 (Fish)

<u>Measurement Endpoint 3A</u>: Compare Site surface water concentrations to published chronic surface water benchmarks and to background.

Risk from surface water COPCs in AOI 18B – Assabet River Embayment was characterized as described in Section 4.5.1. Risk to fish from exposure to surface water at AOI 18B – Assabet River Embayment is considered unlikely.

4.5.4 Assessment Endpoint #4 (Wetland Birds)

<u>Measurement Endpoint 4A</u>: Compare estimated daily dose for omnivorous waterfowl (mallard duck), based on ingestion of prey in Site exposure areas, to published avian TRVs and to background.

Risk HQs based on food chain models for the mallard duck in AOI 18B – Assabet River Embayment (Table 4-18) suggest that risk from COPCs is unlikely (high confidence). HQs for iron, titanium, and tungsten could not be characterized because TRVs could not be identified.

Measurement Endpoint 4B: Compare estimated daily dose for predatory wading birds (great blue heron), based on ingestion of prey in Site exposure areas, to published avian TRVs and to background.

Risk HQs based on food chain models for great blue heron in AOI 18B – Assabet River Embayment (Table 4-18) suggest that risk from COPCs is unlikely (high confidence). HQs for iron, titanium, and tungsten could not be characterized because TRVs could not be identified.

Measurement Endpoint 4C: Compare estimated daily dose for piscivorous birds (osprey), based on ingestion of prey in Site exposure areas, to published avian TRVs and to background.

Risk HQs based on food chain models for piscivorous birds (osprey) in AOI 18B – Assabet River Embayment (Table 4-18) suggest that risk from COPCs is unlikely (high confidence). HQs for iron, titanium, and tungsten could not be characterized because TRVs could not be identified.

4.5.5 Assessment Endpoint #5 (Wetland Mammals)

Measurement Endpoint 5B: Compare estimated daily dose for omnivorous larger mammals (raccoon), based on ingestion of prey in Site exposure areas, to published mammalian TRVs and to background.

Risk HQs based on food chain models for the raccoon in AOI 18B – Assabet River Embayment (Table 4-18) suggest that risk from COPCs is unlikely (high confidence). HQs for iron, titanium, and tungsten could not be characterized because TRVs could not be identified.

4.5.6 Summary for AOI 18B – Assabet River Embayment

Low/medium weighted benchmark comparisons characterized risk to benthic invertebrates, amphibians, and fish at AOI 18B – Assabet River Embayment as unlikely for surface water and sediment. Medium weighted food chain models characterized risk to wetland birds and mammals as unlikely. Based on the weight of evidence and confidence/uncertainties in the data, ecological risk at AOI 18B – Assabet River Embayment is unlikely. Moreover, as documented in Appendix N, comparison of metals concentrations in the Assabet River Embayment to concentrations in upstream impoundments and a similar upstream embayment area indicated that metals concentrations in AOI 18B sediment are consistent with background conditions.

4.6 Site-Wide Soils

This section characterizes risk from COPCs identified in site-wide soils.

4.6.1 Assessment Endpoint #6 (Terrestrial Plants)

Effects Benchmark HQs: Risk to plants from VOCs, PCBs, PAHs and EPHs were characterized using only screening benchmarks because effects benchmarks for plants were not available. Screening benchmarks ranged between 0.054 (chrysene CTE) to 7.8 (pyrene RME). HQs for effects benchmarks only exceeded unity for chromium, mercury, uranium and vanadium; background HQs were similar for all of these COPCs except uranium.

Risk to terrestrial plants from thorium and zirconium could not be evaluated because these COPCs lacked both screening and effects benchmarks. Risk from aluminum and iron could not be evaluated in soil because toxicity is associated with pH (i.e., toxic at pH <5.5), and only one soil pH measurement was collected (pH = 5.5). Site soil RME and CTE EPCs for aluminum, iron, thorium, and zirconium are below respective background EPCs (Table 3-53) implying that Site risk would be lower than background risk.

Risk from EPH fractions could not be characterized using benchmark comparisons because benchmarks could not be identified. However, Irwin (1997) reports that the primary hazards from petroleum hydrocarbon mixtures are more strongly related to PAHs, which were evaluated individually.

Incremental Risk HQs: Trichloroethene was the only VOC characterized as causing possible risk to plants in soil using the benchmark evaluation (Table 4-19). Trichloroethene HQs were based only on screening benchmarks (RME HQ = 4.2 and CTE HQ = 1.4), so there is some uncertainty as to whether detected concentrations are above concentrations at which adverse effects might occur. Also, given the conservative nature of screening benchmarks, risk from trichloroethene is likely overestimated. Risk to plants from VOCs in soil is therefore unlikely.

Sediment benchmark evaluations based on incremental risk HQs (Table 4-19) characterized risk from anthracene and chrysene as unlikely. Risk from benzo(a)pyrene, fluoranthene, phenanthrene, and pyrene were characterized as possible with low confidence; since HQs (1.6 to 6.1) were based only on screening benchmarks, there is considerable uncertainty as to whether detected PAH concentrations are above concentrations at which adverse effects might occur. This is because PAH benchmarks do not take into account site-specific effects of TOC in soil or dissolved organic carbon in porewater which can bind PAHs and reduce their toxicity (USEPA, 2006c). Risk to plants from benzo(a)pyrene, fluoranthene, phenanthrene, and pyrene in soil is, therefore, likely to be overestimated and should be characterized as unlikely.

Soil benchmark evaluations based on incremental risk HQs (Table 4-19) suggest that risk from PCBs to plants was characterized as unlikely in site-wide soil.

Table 4-19 also presents screening risk to plants from antimony, arsenic, cadmium, chromium, cobalt, lead, manganese, mercury, molybdenum, nickel, selenium, titanium, vanadium, and zinc as unlikely. Risk from copper was characterized as possible, but the only incremental risk HQs that were above 1.0 were based on screening benchmarks (CTE/Screening HQ of 1.5 and RME/Screening HQ of 2.0). Given the conservative nature of screening benchmarks, the actual risk from soil copper to plants is likely to be overestimated. Risk from uranium was characterized as possible with high confidence as incremental HQs ranged from 3.0 to 5.1.

Based on this analysis, uranium is the only compound in soil that may present a 'possible' risk to terrestrial plants. As the incremental HQs are very close to unity, the conservatism present in the indirect exposure benchmarks would, more likely than not, favor a condition no adverse effect.

4.6.2 Assessment Endpoint #7 (Terrestrial Invertebrates)

Effects Benchmark HQs: Risk to terrestrial invertebrates from thorium and zirconium could not be evaluated because these COPCs lacked both screening and effects benchmarks. Only screening benchmarks were available for evaluating risks from tetrachloroethene, trichloroethene, aluminum, cobalt, iron, molybdenum, thorium, titanium, uranium and vanadium. Risk from aluminum and iron could not be evaluated in soil because toxicity is associated with pH (i.e., toxic at pH <5.5), and only one soil pH measurement was collected (pH = 5.5). Site soil RME and CTE EPCs for aluminum, iron, thorium, and zirconium are below respective background EPCs (Table 3-41) implying that Site risk would be lower than background risk.

Risk from EPH fractions could not be characterized using benchmark comparisons because benchmarks could not be identified. Irwin (1997) reports that the primary hazards from petroleum hydrocarbon mixtures are more strongly related to PAHs, which were evaluated individually.

Most screening benchmark HQs were below unity, and those that were above an HQ of 1.0 did not exceed 7.8 (RME, pyrene). Effects benchmark HQs were only greater than unity for chromium and the Site RME for mercury; background HQs for chromium were higher than the Site HQs.

Incremental Risk HQs: Trichloroethene was the only VOC characterized as causing possible risk to invertebrates in soil using the benchmark evaluation (Table 4-20). Trichloroethene HQs were based only on screening benchmarks (RME HQ = 4.2 and CTE HQ = 1.4), so there is some uncertainty as to whether detected concentrations are above concentrations at which adverse effects might occur. Also, given the conservative nature of screening benchmarks, risk from trichloroethene is likely overestimated. Risk to invertebrates from VOCs in soil is therefore unlikely.

Soil benchmark evaluations based on incremental risk HQs (Table 4-20) suggest that risk to soil invertebrates from anthracene and chrysene is unlikely. Risk from benzo(a)pyrene, fluoranthene, phenanthrene, and pyrene were characterized as possible with low confidence; however, incremental risk HQs based on screening benchmarks were above one but low (1.6 to 6.1), while incremental risk HQs based on effects benchmarks were two to three orders of magnitude below respective screening benchmark HQs. Furthermore, PAH benchmarks do not take into account site-specific effects of TOC in soil or dissolved organic carbon in porewater which can bind soil PAHs and reduce their toxicity (USEPA, 2006c). Risk from benzo(a)pyrene, fluoranthene, phenanthrene, and pyrene to invertebrates in soil is probably overestimated and should be characterized as unlikely.

Soil benchmark evaluations based on incremental risk HQs (Table 4-20) suggest that risk from PCBs to invertebrates was characterized as unlikely in site-wide soil.

Soil benchmark evaluations based on incremental risk HQs (Table 4-20) suggest that risk to invertebrates from antimony, arsenic, cadmium, chromium, cobalt, lead, manganese, mercury, molybdenum, nickel, selenium, titanium, vanadium, and zinc is unlikely. Risk from copper was characterized as possible, but the only incremental risk HQs that were above 1 were based on screening benchmarks, and ranged from 1.5 (CTE/Screening) to 2.0 (RME/Screening). Given the conservative nature of screening benchmarks, risk from soil copper to invertebrates is most likely overestimated and therefore unlikely.

Risk from uranium was characterized as possible with high confidence as screening RME and CTE incremental HQs ranged from 3.0 to 5.1, respectively. However, less conservative effects benchmarks were not available, so there is some uncertainty as to whether detected concentrations are above concentrations at which adverse effects might occur.

4.6.3 Assessment Endpoint #8 (Songbirds)

Measurement Endpoint 8A: Compare estimated daily dose for omnivorous songbirds (cardinal), based on ingestion of prey in Site exposure areas, to published avian TRVs and to background.

Risk HQs based on food chain models for cardinal in the site-wide soil exposure area (Table 4-21) suggest that risk from most COPCs is unlikely (low to high confidence) but that risk is possible for Aroclor-1254 (low confidence), Aroclor-1260 (low confidence) and zinc (low confidence). Risk HQs based on LOAELs for Aroclor-1254, Aroclor-1260, zinc were below 1, and incremental risk HQs based on NOAELs ranged from 2.2 to 5.3 for Aroclor-1254, from 1.4 to 5.3 for Aroclor-1260; HQs for zinc ranged from 1.7 to 1.8, suggesting that there is a low risk from these COPCs. HQs for iron, titanium, and EPHs could not be characterized because TRVs could not be identified. Ingestion of soil accounts for approximately 59% to 70% of risk, ingestion of plants accounts for approximately 4% to 11% of risk and ingestion of invertebrates accounts for approximately 21% to 37% of risk (Appendix K).

Measurement Endpoint 8B: Compare estimated daily dose for invertivorous songbirds (American robin), based on ingestion of prey in Site exposure areas, to published avian TRVs and to background.
Risk HQs based on food chain models for the American robin in the site-wide soil exposure area suggest that risk from most COPCs is unlikely (low to high confidence). The risk to American robin was characterized as possible for Aroclor-1254 (moderate confidence), Aroclor-1260 (moderate confidence) and zinc (low confidence) (Table 4-21). HQs for iron, titanium, and EPHs could not be characterized because TRVs could not be identified. Ingestion of soil accounts for approximately 24% to 39% of risk, and ingestion of invertebrates accounts for approximately 61% to 76% of risk (Appendix K).

4.6.4 Assessment Endpoint #9 (Terrestrial Mammals)

Measurement Endpoint 9A: Compare estimated daily dose for herbivorous small mammals (meadow vole), based on ingestion of prey in Site exposure areas, to published mammalian TRVs and to background.

Risk HQs based on food chain models for meadow vole in the site-wide soil exposure area suggests that risk from COPCs is unlikely (high confidence) (Table 4-21). HQs for iron, titanium, and EPHs could not be characterized because TRVs could not be identified.

Measurement Endpoint 9B: Compare estimated daily dose for omnivorous large mammals (red fox), based on ingestion of prey in Site exposure areas, to published mammalian TRVs and to background.

Risk HQs based on food chain models for fox in the site-wide soil exposure area suggest that risk from COPCs is unlikely (high confidence) (Table 4-21). HQs for iron, titanium, and EPHs could not be characterized because TRVs could not be identified.

Risk of PCBs to fox and other burrowing animals to deeper soil is expected to be unlikely since COPC concentrations in deeper soils are lower than surface soils, as presented in the RI Report (*de maximis*, 2010).

4.6.5 Summary for Site-Wide Soil

Low/medium weighted benchmark comparisons characterized risk to plant and terrestrial invertebrates in site-wide soil as highly unlikely for all COPCs except uranium which was characterized as possible (high confidence). Medium weighted food chain models characterized risk from COPCs to mammals as unlikely; food chain models for cardinal and American robin characterized risk as possible for Aroclor-1254 (low to moderate confidence), Aroclor-1260 (low to moderate confidence) and zinc (low confidence). Based on the weight of evidence and confidence/ uncertainties in the data, ecological risk in site-wide soil is possible from uranium (plants and soil invertebrates), Arcolor-1254 (birds) and Arolor-1260 (birds).

4.7 Uncertainty analysis

There is uncertainty associated with estimates of risk in any BERA because the risk estimates are based on a number of assumptions regarding exposure and toxicity. More specifically, there is inherent variability and uncertainty associated with the exposure and modeling parameters (e.g. dietary intake, body weight, age), toxicological data (e.g. TRVs), and risk characterization (USEPA, 1997). A thorough understanding of the uncertainties associated with risk estimates is critical to understanding predicted risks and placing them into proper perspective.

Uncertainties related to the exposure assessment step of the BERA affect the interpretation and the significance of the HQs. The main uncertainties are associated with the EPCs, exposure parameters, estimation of BSAFs and BAFs for various environmental media, assumptions of bioavailability of COPCs, and ingestion exposure pathways. These uncertainties may result in an over- or under-estimation of risk. Uncertainty was minimized by using and citing information published in peer-reviewed scientific journals whenever possible. Major uncertainties and assumptions are summarized below.

4.7.1 Data Summary/Exposure Point Concentrations

Risks were assessed only for analytes that were detected in at least one sample in a given medium and exposure area; analytes that were not detected in a given medium were assumed to pose no significant risk. This assumption is supported by the fact that DQOs, including required detection limits, were established early in the RI process to ensure that proposed analytical detection limits were sensitive enough to measure concentrations of analytes that might pose a risk to ecological receptors. This was accomplished by comparing proposed detection limits to Project Action Levels (PALs) that included ecological screening benchmarks, and requesting lower detection limits for those analytes with very low benchmarks (e.g., some metals in water).

COPCs were selected based upon larger site-wide data sets for surface water, sediment, and soil (Appendix A). Analytes eliminated as COPCs were not evaluated further in the BERA; since concentrations of these analytes were below conservative screening values, their overall contribution to risk is negligible and their exclusion is expected to have an insignificant impact on the risk characterization.

As documented in the QAPP, USEPA Method 8310 was used to establish and meet the (PALs) for sediment, soil, and surface water because it is specifically designed for the detection of trace level PAHs. Later, Method 8270C was used to analyze site media because this method offers more certainty and less false positive results across a broader range of low and high concentration ranges. The two methods sometimes provided conflicting results stemming from false positives in the high range of Method 8310 data. Based on conclusions from a detailed data quality review, only PAH data from Method 8310 are used to select and evaluate surface water COPCs. Data analyzed by Method 8270C are used to select and evaluate sediment, peat, and soil PAHs; Method 8310 sediment, peat, and soil data are used only when Method 8270C data are not available.

4.7.2 Receptor Exposure Parameters

The relationship between receptor size and dietary intake is a critical factor in estimating exposure. In addition, dietary composition affects exposure because different food sources contain varying levels of COPCs. Although data for dose calculation inputs such as body weight, ingestion rate, and dietary composition for the measurement endpoint receptors evaluated at the Site are found in the scientific literature, there is a natural level of variability in these parameters within a population of organisms (often differing by age, gender and body weight). Uncertainty is inherent in the use of these values because they were generated from literature sources which may rely on non-regional data, rather than being measured at the Site or surrounding areas. Therefore, the use of literature-derived exposure parameters increases uncertainty that may over- or underestimate actual exposures encountered by receptors at the Site.

4.7.3 Bioaccumulation Factors

Site-specific prey tissue data were available for benthic macroinvertebrates and amphibians at AOI 6 – Sphagnum Bog. Site-specific tissue data reduce uncertainty and result in high confidence in the risk estimation because they are direct measures of potential exposures to receptors.

Literature-based BSAFs or BAFs were used to estimate uptake into other food chain components and for COPCs that were not directly measured. These literature-based BSAFs and BAFs are generally considered to be more conservative than Site-specific factors, and therefore their use for some COPCs may overestimate risks associated with those COPCs.

4.7.4 Bioavailability of COPCs

Assumptions about bioavailability are made in several steps of the food chain model: from medium to prey, from prey to predator, and from medium directly to predator via incidental ingestion. Bioavailability of COPCs in the each step of the food chain depends on a number of factors including particle size, matrix/tissue type, tissue chemistry (e.g. stomach pH), and an organisms' ability to detoxify and eliminate a COPC. For many species, this information is not available, thus the food chain models assumes 100% bioavailability so as not to underestimate risk. In most cases, this assumption leads to a risk conclusion which is clearly overly conservative given the uncertainties involved. For example, bioavailability of lead or arsenic following the ingestion of soil or sediment soil is generally much lower than an "assumed" value of 100%.

In order to reduce uncertainty associated with exposure of aquatic organisms to metals in surface water, AWQC were adjusted based on average hardness values (calculated as mg $CaCO_3$/L) specific to each individual AOI, where applicable. Sediment benchmarks (ETs, OMOE LELs, Washington State SQVs, USEPA ESG/SQC, and ORNL SCVs) were adjusted to

AOI-specific sediment TOC; benchmark adjustment based on sediment TOC is consistent with the intended use as described in the applicable source documents and helps to reduce uncertainty. The ΣPAH method was also used to limit uncertainty associated with bioavailability of PAHs.

SEM/AVS values may vary seasonally, with AVS concentrations typically higher in the warmer months and lower in the colder months, implying that metals tend to be less bioavailable in the summer (USEPA, 2005). Due to seasonal differences between the sample collection dates, bioavailability of divalent metals in the Sphagnum Bog may be overestimated while bioavailability in the Assabet River and background upstream river channel may be underestimated.

It is also known that peat bogs bind metals strongly, as the vertical metal profiles in bogs are often used to estimate air deposition of lead, copper, zinc and other metallic species over long periods of time (Novak *et al.*, 2003). An undated USGS report (Cameron and Wright) reports relatively high concentrations of trace metals (including copper, lead, nickel and zinc) in the upper horizons of 39 bogs from Washington County, Maine. Bioavailability of copper has also been reported to be low in bogs that have excessive amounts of this metal (which, in turn, would explain a decrease in bioavailability (Brewina *et al.*, 2007; MacDonald, 2010). These observations are important because the Sphagnum Bog may have much lower bioavailability of copper and other COPCs that were identified at a fairly high frequency, even though the pH is acidic. Humic and tannic acids are negatively charged compounds that can sequester metals, which would render the metals "dissolved" but still unavailable for uptake through the gills of aquatic organisms.

4.7.5 Inhalation and Dermal Exposure Pathways

Dermal exposures to wildlife were not evaluated because there are few data relating dermal exposure to toxic responses in wildlife. An assumption was made that fur, feathers, or chitinous exoskeleton limit dermal exposures of most wildlife. Inhalation exposures are assumed to be important only for VOCs, which were infrequently detected at fairly low concentrations and therefore not assessed. Risks to wildlife may therefore have been underestimated, but the overall effect on risk estimates is likely negligible.

4.7.6 Toxicological Uncertainties

Uncertainties associated with effects assessment steps for the BERA also affect the interpretation of the significance of the HQs. The main uncertainties are associated with the literature-derived toxicity data and interpretation of Site-specific toxicity tests. Literature-based toxicity endpoints always err on the conservative side (through the use of multiple safety factors) so the estimation of risk using, for example, TRVs is overestimated.

4.7.7 Literature-Derived TRVs and Benchmarks

TRVs used in the BERA are based on an extensive search of both primary peer reviewed literature and secondary literature, such as government reports and technical conference proceedings. The number and types of information sources reviewed is adequate to capture the majority of sources of ecotoxicological information.

However, there is also uncertainty associated with the extrapolation of literature-derived toxicity endpoints to measurement endpoint receptors at the Site because of differences in exposure conditions. The majority of the literature-based toxicity data evaluated in the BERA were derived from laboratory studies which also use the most soluble, and therefore the most bioavailable, form of a toxicant. The most soluble form of a metal is also generally the most toxic form (e.g. barium chloride vs. barium sulfate). Laboratory settings do not necessarily represent field conditions and exposures, and typically are designed to control various factors in order to isolate one parameter in particular. Strains of laboratory rats and mice have also been inbred to reduce variability and are generally more sensitive to toxic agents, so effects that may be observed in laboratory species may not be observed in wild populations of the same genus or species. Several measures were instituted to minimize these uncertainties. Chronic studies with NOAEL endpoints were selected preferentially in developing TRVs. This makes for a more conservative risk assessment as NOAEL's are not "bounded" on the lower end of the dose range and therefore represent "safe" exposure levels. This practice will have a tendency to overestimate risk.

Available toxicological data are not always associated with chronic exposure duration, in which case other test durations (for example, subchronic NOAELs or LC50 values) were sometimes selected to identify TRVs. When an endpoint other than a chronic NOAEL was selected as a TRV, an uncertainty factor was applied to the reported value to provide an additional level of conservatism in the risk estimation process. Application of conservative safety factors may result in risks being overestimated.

TRVs are limited for some specific receptors (e.g., omnivorous birds, herbivorous mammals, soil invertebrates) in the toxicological literature. Applying TRVs to broad classes of receptors may over or underestimate risks, depending on the sensitivity of the receptor relative to that of the test organism, which often has not been determined.

Sediment and surface soil screening benchmarks used in comparison tables were generally the lowest available screening value for a particular analyte. Since screening benchmark values typically varied by up to several orders of magnitude, selection of the lowest available value likely resulted in an overestimation of risk.

Due to the variability in media within the bog to which ecological receptors could potentially be exposed, sediment, peat, and sphagnum moss were sampled for analysis. There are no screening benchmarks available for peat or sphagnum, and therefore these two data sets are treated as sediment-like media in the BERA and are compared to sediment benchmarks. Both peat and sphagnum likely support some type of invertebrate community, and therefore the sediment benchmarks are the most appropriate benchmarks currently available for these media.

Some COPCs had very little or no toxicological data (TRVs, benchmarks) available. When appropriate, toxicity data for similar chemicals were used as surrogates for these COPCs. Use of such surrogate toxicity data may underestimate or overestimate risks from the COPCs. In some cases, no appropriate toxicity or surrogate data were identified in the toxicological literature. The lack of toxicological data means that risk from some COPCs could not be quantified. This may underestimate ecological risks at the Site. However, where possible, COPCs lacking benchmarks or TRVs were compared to reference conditions in order to estimate any incremental difference in risk.

Toxicity of COPCs to Sphagnum Species

It is unknown if COPCs are toxic to Sphagnum spp. itself since Sphagnum-specific toxicity data were not readily available from the scientific literature, although no "readily apparent harm" was visible upon direct observation in the bog. Most studies on sphagnum bogs have focused on the effects of acidification and nitrification rather than chemical effects.

Toxicity Tests

In risk assessments, generally speaking, laboratory toxicity tests typically receive a relatively high weight of inference, though there is still some uncertainty associated with their use. For example, it is assumed that the test species - in this case *Chironomus dilutus* - are representative of the benthic invertebrate community at the site. In the case of the NMI site, *C. dilutus* were commonly identified in benthic community surveys, so there is less uncertainty in extrapolating the laboratory results to field conditions. However, laboratory reared test organisms may overestimate toxic effects since they have not had the opportunity to acclimate to ambient sediment conditions while natural populations of *C. dilutus* would have had hundreds of generations to adapt to site conditions. There is documentation in the scientific literature to indicate that Hyallela *spp*. are more sensitive to the effects of divalent metals than Chironomus *spp*. so employing this organism may have resulted in toxicity.

If a sample demonstrated a statistically significant effect, then the test results were evaluated for ecological significance. Statistically significant differences between study area and background are not necessarily indicative of ecologically significant results (i.e., an association that is important mathematically may not be important biologically, especially at a population level scale). Because ecological risk assessments evaluate risk to receptors at the population level (USEPA, 1997), an ecologically "significant" risk should be interpreted relative to the results of any effects seen in the background (reference) area(s). Evaluating toxicity test results relative to background levels helps to reduce uncertainty.

As described in the *Chironomus* toxicity test reports (Appendix H), a predator was found in one replicate chamber of four study area samples (SD-RI-0600500R, SD-RI-0602500R, SD-RI-0603800R, and SD-RI-0650500) and one background sample (SD-RI-1745400). Results from test chambers where a predator was identified were excluded from the evaluation. The loss of a replicate does not bias the results either high or low, but it reduces power in hypothesis tests and may make it difficult to detect an ecologically meaningful effect if the p value is close to 0.05, which did not occur.

4.7.8 Uncertainty in Risk Characterization

Interpretation of Hazard Quotients

The largest source of uncertainty associated with risk characterization involves the interpretation of HQs. The Four-Way and Two-Way risk matrix, developed by Stan Pauwels at TechLaw, Inc.,, were set up to provide a logical decision process for interpreting sometimes complex scenarios based on various permutations of benchmarks, TRVs, and EPCs. However, even with this systematic approach, professional judgment is still required when weighing uncertainties.

Incremental risk could not be characterized if a COPC in a given AOI was not analyzed in its respective background data set; in this case, it was assumed that the background concentration would have been below the instrument detection limit. This may result in an overestimation of risk.

The results of the effect of metals in surface water of the bog were also, by inference, extended to the results of the NE Wetland. Metals in sediment in the NE Wetland did not show a risk to ecological receptors and, given that the wetland is small and that it is inundated with standing water for only a short time in the spring, it was concluded that metals would not pose a risk to this habitat. As a result, there is some degree of uncertainty with regard to the impact of metals to this ecosystem, but the level of uncertainty is low.

Benthic Community Surveys

There is some uncertainty regarding the interpretation of the benthic community survey results in AOI 6 - Sphagnum Bog where the aquatic habitat is poorly suited to standard metrics for benthic community structure. Most benthic sampling also employs a minimum of three replicates because macroinvertebrates typically exhibit a clustered distribution in sediment. For the NMI study, only one sample was obtained per location, which reduces the statistical power when interpreting the results. Therefore, the benthic community survey was given a "low" weight in the interpretation scheme. The toxicity testing, which provides a more direct line of evidence, was given a higher weight in the evaluation of risk to the benthic community.

Impacts of Soil pH

The risk from aluminum and iron in soil depends on pH. Since only one pH sample (5.5 standard pH units) was evaluated in soil, risk from aluminum and iron could not be properly screened in Appendix A. Aluminum and iron were therefore characterized in the BERA using additional measurement endpoints, including food chain models.

Risk from EPH fractions could not be characterized using benchmark comparisons because benchmarks could not be identified. Irwin (1997) reports that the primary hazards from petroleum hydrocarbon mixtures are more strongly related to PAHs, which were evaluated individually and using the ΣPAH model and benchmark comparisons.

5. SUMMARY AND CONCLUSIONS

The BERA evaluated the potential for COPCs in surface water, sediment, and soil to impact ecological receptor populations present within or adjacent to areas affected by historical operations at the NMI facility in Concord, MA.

Five aquatic/wetland exposure areas (AOI 4 - Cooling Water Recharge Pond, AOI 6 - Sphagnum Bog, AOI 10 – Northeast Westland, AOI 18A – Assabet River Main Channel, and AOI18B – Assabet River Embayment) and one terrestrial exposure area (site-wide soils) were identified based on habitat assessment and site investigations conducted in 2005 through 2009. Although a SLERA was previously performed using Phase 1A and Phase 1B data, COPCs were re-selected using the pooled Phase 1A, 1B, and 1C data sets (Appendix A) per agreement with all parties. Based on the RI data, the need for a presumptive remedy at AOI 4 - Cooling Water Recharge Pond has been identified. In accordance with USEPA policy, a SLERA can be sufficient to document risk in areas where a known remedy will be implemented. As a result, additional evaluation of ecological risk within the AOI 4 - Cooling Water Recharge Pond was not necessary since risk associated with potential exposure to ecological receptors will be addressed by the presumptive remedy.

Site-specific data consisted of surface water, sediment (including SEM/AVS), and soil chemistry data, sediment toxicity test data, benthic community survey data, and amphibian and benthic macroinvertebrate tissue concentration data. In some locations, sediment chemical data, sediment toxicity tests, and benthic community surveys were co-located, an approach commonly referred to as a sediment "triad."

The BERA considered up to six assessment endpoints to evaluate risk in aquatic/wetland exposure areas (depending on habitat in each exposure area), and four assessment endpoints to evaluate risk in the terrestrial area:

<u>Aquatic/Wetland Endpoints</u>

- Aquatic benthic invertebrate community structure
- Growth, survival, and reproduction of amphibian populations
- Growth, survival, and reproduction of fish populations
- Growth, survival, and reproduction of semi-aquatic bird populations
- Growth, survival, and reproduction of semi-aquatic mammal populations
- Growth, survival and reproduction of rare, threatened, or endangered species (Eastern Pondmussel, Triangle Floater, and Creeper)

<u>Terrestrial Assessment Endpoints</u>

- Growth, survival, and reproduction of terrestrial plants
- Growth, survival, and reproduction of terrestrial plants
- Growth, survival, and reproduction of terrestrial songbird populations
- Growth, survival, and reproduction of terrestrial mammal populations

Measurement endpoints used to evaluate these assessment endpoints included:

- Comparison of COPC EPCs in surface water, sediment, and surface soil to chronic surface water benchmarks, plus screening and/or effects benchmarks, derived from the scientific literature.
- Evaluation of SEM/AVS data and ΣPAH models.
- Sediment toxicity tests with statistical analysis of co-located sediment data to try to relate a dose response relationship to observed effects.
- Comparison of COPCs in site and background amphibian tissue and available Critical Body Burden data.
- Quantitative benthic macroinvertebrate surveys.
- Food chain modeling for terrestrial and wetland birds and mammals, by which modeled doses were compared to both NOAEL- and LOAEL-based TRVs.

Assessment populations evaluated in wetland and aquatic AOI food chain models included:

- Mallard ducks, representing omnivorous birds,
- Great blue heron, representing piscivorous birds,
- Osprey, representing piscivorous birds,
- Short-tailed shrew, representing carnivorous small mammals; and
- Raccoon, representing omnivorous mammals.

The AOI 18B - Assabet River Main Channel also included freshwater mussels as an assessment population because the MANHESP indicated the presence of three species of concern including the eastern pondmussel (*Ligumia nasuta*), triangle floater (*Alasmidonta undulata*), and creeper (*Strophitus undulates*) (Appendix B).

Assessment populations evaluated in Terrestrial Soil food chain models included:

- American robin, representing invertivorous songbirds,
- Cardinal, representing omnivorous songbirds,
- Meadow vole, representing a small herbaceous mammal; and
- Red Fox, representing omnivorous mammals.

Reference locations were identified and paired with Site study areas based on important habitat characteristics including hydrology, plant community characteristics, and surrounding land use.

Both "reasonable maximum exposure" (RME) and the "central tendency exposure" (CTE) were considered when assessing and characterizing risk. CTE represents the most likely concentration to which a population of receptors would be exposed. CTE EPCs were calculated as the lower of either the maximum concentration or the arithmetic mean. RME EPCs were calculated as either the lower of the 95 percent UCL or the maximum concentration.

A weight of evidence approach was used to make conclusions regarding risk of harm of each assessment endpoint with more than one measurement endpoint. Measurement endpoints were each assigned an inference weight, based upon how closely they represented the assessment endpoint. Conclusions regarding risks to that assessment endpoint were reached by considering the inference weight for each measurement endpoint, i.e., the overall weight of evidence.

Measurement endpoints involving food chain models and comparison of media concentrations to benchmarks were assessed using a HQ approach. When HQs were calculated as part of the Effects Assessment (comparison to benchmarks, food chain models), the likelihood of adverse population effects was determined using a four-way matrix that incorporated all four combinations using RME and CTE EPCs and NOAEL and LOAEL TRVs (or screening and effects benchmark) combinations. Where only screening benchmarks were used to assess effects, a two-way table based on the four-way table was used to characterize risk. When possible, incremental risk HQs which accounted for the background contribution to Site risk were used as the basis for the risk characterization.

Other assessment endpoints which were not based on HQs, such as quantitative benthic community surveys, sediment toxicity tests, and amphibian tissue comparisons, were compared to conditions in reference areas. SEM/AVS data were compared to threshold concentrations established by USEPA (USEPA, 2005).

AOI 6 – Sphagnum Bog

Low/medium weighted chronic surface water benchmark comparisons characterized risk to benthic invertebrates and amphibians at AOI 6 – Sphagnum bog as possible in surface water for eight dissolved metals. Copper, lead, and cadmium represented the biggest risk drivers in this matrix (Table 6-1). Based on the weight of evidence (Table 6-1), eight of the thirteen mineral sediment samples from AOI 6 - Sphagnum Bog demonstrate that there is no adverse impact to the benthic community, making this the predominant condition in the bog. Adverse effects in *C. dilutus* sediment toxicity tests at SD-RI-0600900 and SD-RI-0600100R appear to reflect elevated sediment metal and PCB concentrations at those specific locations; most of the samples with elevated levels of PCBs and mercury were located in the Southwest Corner of the bog. Benchmark evaluations were the only line of evidence used to evaluate risk to benthic invertebrates in moss and peat, and identified possible risk from molybdenum and uranium (low confidence). These evaluations also showed that risks to the benthic community for mineral sediment were elevated in the Southwest Corner due to PCBs, copper, and mercury, but were low in the remainder of the bog. Comparison of amphibian tissue residues from AOI 6 – Sphagnum Bog showed, relative to background bog tissue, elevated levels of PCBs and uranium; this observation is not uniform as the result is based on a bog-wide average (i.e. some AOI 6 tissues were below the limit of detection). Amphibian tissues were all well below Critical Body Burden values for metals. Medium weighted food chain models characterized risk to mallard duck and raccoon at AOI 6 – Sphagnum Bog as unlikely. Food chain models suggest that risk to great blue heron is possible from beryllium (high confidence) in mineral, peat and moss sediment fractions. Food chain models also suggest that in the mineral and peat fractions, risk to shrew is possible from molybdenum (low to moderate confidence) in all three sediment fractions. The highest concentrations of beryllium and molybdenum were detected in sediment samples SD-RI-06001 and SD-RI-06009, which were associated with toxicity to the benthic community, and sample SD-RI-06032 which is located near SD-RI-06009. These locations are in or near the Southwest Corner of the bog, which received historic discharges of water from the former NMI facility.

AOI 10 – Northeast Wetland

Low/medium weighted benchmark comparisons characterized risk to benthic invertebrates and amphibians at AOI 10 - Northeast Wetland as possible in surface water for dissolved copper, lead, and manganese, and characterized risk as unlikely in sediment (Table 6-1). Medium weighted food chain models characterized risk to wetland birds and mammals as unlikely. Based on the weight of evidence and confidence/ uncertainties in the data, ecological risk at AOI 10 – Northeast Wetland is unlikely.

Assabet River – Main Channel

Low/medium weighted benchmark comparisons characterized risk to benthic invertebrates, amphibians, and fish at AOI 18A – Assabet River Main Channel as unlikely for surface water and sediment (Table 6-1). Medium weighted food chain models characterized risk to wetland birds and mammals as unlikely. Distribution of rare mussels is likely more attributable to the many impoundments that lie on the Assabet River that impede or prevent movement of fish upstream. Based on the weight of evidence and confidence/ uncertainties in the data, ecological risk at AOI 18A – Assabet River Embayment is unlikely.

Assabet River – Embayment Area

Low/medium weighted benchmark comparisons characterized risk to benthic invertebrates, amphibians, and fish at AOI 18B – Assabet River Embayment as unlikely for surface water and sediment (Table 6-1). Medium weighted food chain models characterized risk to wetland birds and mammals as unlikely. Based on the weight of evidence and confidence/ uncertainties in the data, ecological risk at AOI 18B – Assabet River Embayment is unlikely.

Site-Wide Surface Soil

Low/medium weighted benchmark comparisons characterized risk to plant and terrestrial invertebrates in site-wide soils as unlikely for all COPCs except uranium which was characterized as possible (high confidence) (Table 6-1). Medium weighted food chain models characterized risk to mammals as unlikely; food chain models for cardinal and American robin characterized risk as possible for Aroclor-1254 (low to moderate confidence), Aroclor-1260 (low to moderate confidence), and zinc (low confidence). Based on the weight of evidence and confidence/ uncertainties in the data, ecological risk in site-wide soils is possible from uranium (plants and soil invertebrates), Arcolor-1254 (birds), and Aroclor-1260 (birds).

Taken as a whole, both surface water and sediment, as well as the results of the food chain models, reveal no significant ecological risk for the following operational units:

- Northeast Wetland (AOI 10)
- Assabet River (AOI 18A)
- Assabet River Embayment Area (AOI 18B)
- Site-Wide Surface Soil (overlaps several AOI's)

Remedial action will therefore not be required at any of the above locations. A previous agreement to declare a Presumptive Remedy for the Cooling Water Recharge Pond (AOI 4) obviates any need to evaluate ecological risk or harm to this waterbody.

Surface water in AOI 6 (Sphagnum Bog) showed that incremental risk to benthic invertebrates and amphibians was possible due to dissolved cadmium, copper, lead, and uranium.

With regard to sediment, two of the sediment bioassays that showed toxicity also were a) located in the Southwest Corner and b) had relatively elevated concentrations of PCBs, mercury and other trace metals. Comparing the Southwest Corner data to the results from the Remaining Area (Table 3-22.1 and 3-22.2), it is clear that there is no risk to COPCs within the Remaining Area, yet there is some incremental risk from sediment in the Southwest Corner (PCBs, copper and mercury).

Although some of the lines of evidence are equivocal (e.g. some apparently uncontaminated samples showed toxicity), the weight of evidence leans toward the Southwest Corner as a feasible 'source' term for copper, mercury and PCBs in the bog (and, to a lesser extent, trace metals such as Mo, Ag and U). However, because of the sensitive nature of the bog mat and the nature and extent of persistent, bioaccumulative and/or toxic COPCs, it is recommended to confine any remedial activity, if warranted, to the lag zone [open water] segment of the Southwest Corner. Any remedial activity based on ecological risk should focus on a) removal of any of any contaminated sediment from the Southwest Corner lag zone and b) restoration and mitigation of the shoreline to enhance amphibian habitat.

A recent communication from USEPA (USEPA, 2012) provides recommendations for remedial goals for Sphagnum Bog (Appendix M). The sediment PRGs developed for PCBs, copper, lead and mercury appear to be a logical assessment of the results of the measurement endpoints. Other metals that exceed risk standard thresholds, such as mercury, would appear to be removed along with PCBs and copper.

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