DRAFT FINAL

BLACKSTONE RIVER FEASIBILITY STUDY

TASK D:

ECOLOGICAL RISK ASSESSMENT

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EXECUTIVE SUMMARY

The Army Corps of Engineers, New England District (NED) is conducting a multi-year feasibility study to identify restoration opportunities in the Blackstone River Basin in Massachusetts. The Blackstone River has historically been impacted by a wide variety of industrial and non-point sources (USACE, 1994). As part of this multi-faceted program, Battelle was contracted to conduct an Ecological Risk Assessment (ERA) of two impoundments along the Blackstone River, Fisherville Pond (Figure 1-1) and Singing Pond (Figure 1-3). Previous investigations have indicated that sediments from these areas contain elevated levels of some chemicals (USACE, 1997).

The original objective of this ERA was to evaluate the potential risks of sediment-associated contaminants to aquatic and terrestrial communities found at both Fisherville and Singing Pond. At the time the investigation was initiated in 1999, Fisherville Pond provided 69 acres of shallow open water habitat and about 21 acres of emergent, wet meadow habitat. However, in 2000, a blockage in the spillway of the Fisherville Dam failed, resulting in a substantial decrease in open water habitat and exposing mudflats that quickly became vegetated. As a result, under current conditions, both terrestrial and aquatic receptors may be exposed to the sediment-bound contaminants. As a result, Battelle has been contracted by NED to address the following questions:

- Under ponded conditions (*i.e.*, restoring the impoundment at Fisherville Pond) are there areas within Fisherville Pond or Singing Pond that require management action (*e.g.*, dredging) to reduce risks to wildlife and aquatic species?
- What is the relative magnitude of risks at Fisherville Pond under full pool versus reduced pool conditions?

To address these questions, a two-pronged approach was developed. The first focused on determining the potential risk to aquatic and piscivorous wildlife species among designated areas of Fisherville Pond and Singing Pond. Under this approach, it was assumed that the Fisherville Pond impoundment would be restored to its 1999 condition. A qualitative weight of evidence approach was used, deriving a measure of potential risk (*e.g.*, high, medium, low) for specific areas including: Fisherville Pond-North Pool, Fisherville Pond-Central Pool, Fisherville Pond-South Pool, Singing Pond-Main Channel, Singing Pond-Marsh, and the designated reference area, Lake Wildwood (Figure 1-4). The second evaluation focused on the relative risks to wildlife (*i.e.*, piscivorous or insectivorous/omnivorous) species from exposures with or without the impoundment at Fisherville Pond.

Based on the objectives of this evaluation (*i.e.*, to evaluate potential risks to the ecological community) the following assessment endpoints were identified:

- Health of the benthic invertebrate community;
- Health of the fish community;
- Sustainability of upper trophic level receptors.

A weight of evidence approach was used with multiple lines of evidence (*i.e.*, measurement endpoints) for each assessment endpoint. As outlined in the associated work plans (Battelle, 1999a,b, 2001), data considered for this evaluation included: 1) sediment chemistry; 2) porewater and surface water chemistry; 3) fish tissue chemistry; 4) fish community, 5) sediment toxicity; 6) benthic community analysis; and, 7) dose assessment for wildlife. In addition, the results of an associated Toxicity Identification Evaluation (TIE) were also considered (SAIC, 2000). These data were used to characterize potential risk to ecological resources in Fisherville Pond, Singing Pond, and Lake Wildwood.

In general, risks at Fisherville Pond-North Pool and Lake Wildwood are low. Sediment concentrations in the North Pool were relatively low, probably as a result of dredging that occurred there in 1982. In addition, the results of the bulk sediment toxicity bioassays indicated that little or minimal toxicity was associated with sediments collected from this area. Similar results were obtained for Lake Wildwood. COPC concentrations were generally very low with only one chemical (4,4'-DDE) detected at elevated concentrations. Limited toxicity was observed in the bulk sediment toxicity tests as well. All measurement endpoints for these areas were scored as low with the exception of the benthic community analysis and the metals mixtures.

Fisherville Pond-Central Pool and Singing Pond-Main Channel were both scored as medium with six of the 10 lines of evidence ranked as medium. In general, sediment concentrations throughout these areas were elevated, however, toxicity observed in the bioassays was relatively moderate. Estimated risks to wildlife species were also moderate. In Singing Pond-Marsh Area, all lines of evidence evaluated except the benthic community analysis and the metals mixtures were scored as high, therefore, this area was ranked as high. Fisherville Pond-South Pool was also scored as high, with five of the 10 lines of evidence scored as high. Station FP4, which indicated acute toxicity, is located in this area, and overall the toxicity measurement endpoint was ranked as high. The evaluation of risks to upper trophic level species also indicated high risks in this area.

Under the second assessment (*i.e.*, relative risks from full pool versus reduced pool conditions within Fisherville Pond) it was determined that risks to piscivorous species and aquatic waterfowl were generally similar under both scenarios although slightly higher under full pool conditions. However, the reduction in risk under the reduced pool scenario was also associated with a dramatic decrease in available habitat. In contrast, risks to the terrestrial songbird were greatly increased under the reduced pool conditions.

In summary, risks to ecological receptors associated with Fisherville Pond-North Pool and Lake Wildwood are negligible. However, risks associated with the remaining areas of Fisherville Pond and Singing Pond may be of concern, ranking as medium or high based on the lines of evidence evaluated. Based on this assessment, it appears that sediment remediation in these areas would be likely to reduce risks and result in an overall ecological benefit.

Regarding the relative risks associated with the presence or absence of the Fisherville Pond impoundment, the results indicate that overall risks to the wildlife species evaluated are likely to be lower under the full pool conditions. Although risks to waterfowl and piscivorous wildlife decreased slightly under the reduced pool conditions, the associated reduction in available habitat is likely to be detrimental, offsetting the potential benefit. In contrast, the available habitat increased substantially for the songbird under the reduced pool conditions, magnifying the increase in potential risks associated with that scenario. Therefore, it is concluded that restoring the former impoundment at Fisherville Pond would reduce potential risks to wildlife species.

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ACRONYMS

ANOVA	Analysis of Variance
ASTM	American Society for Testing and Materials
AVS	Acid Volatile Sulfides
AWQC	Ambient Water Quality Criteria
BRI	Blackstone River Initiative
BVA	Barry Vittor and Associates
COPC	Chemical of Potential Concern
DEQE	Department of Environmental Quality and Engineering
EPA	Environmental Protection Agency
ER-L	Effects Range - Low
ER-M	Effects Range - Median
ERA	Ecological Risk Assessment
ERED	Environmental Residue Effects Database
ESG	Equilibrium Sediment Guideline
ESGTU	Equilibrium Sediment Guideline Toxic Unit
FER-L	Fish Effects Residue - Low
FER-M	Fish Effects Residue - Median
FP	Fisherville Pond
GLEC	Great Lakes Environmental Center
HS	"Hot Spot" (refers to Southern Pool of Fisherville Pond, near the dam)
HQ	Hazard Quotient
HCL	Hydrochloric Acid
LEL	Low Effect Level
MADEP	Massachusetts Department of Environmental Protection
NED	Army Corps of Engineers, North England District
OSWER	Office of Solid Waste and Emergency Response
PAHs	Polycyclic Aromatic Hydrocarbons
PCBs	Polychlorinated biphenyl
PEC	Probable Effects Criteria
PEC-Q	Probable Effects Criteria-Quotient
RP	Reference Pond (Lake Wildwood)
SAIC	Science Application International Corp.
SEL	Severe Effect Level
SEM	Simultaneously Extractable Metals
SP	Singing Pond
SQC	Sediment Quality Criteria
TEC	Threshold Effects Criteria
TIE	Toxicity Identification Evaluation
TMDL	Total Maximum Daily Load
TOC	Total Organic Carbon
TRV	Toxicity Reference Value
USACE	United States Army Corps of Engineers
WM	Wet Meadow
WQC	Water Quality Criteria
YOY	Young of Year

1. INTRODUCTION

The Army Corps of Engineers, New England Division (NED) is conducting a multi-year feasibility study to identify restoration opportunities in the Blackstone River Basin in Massachusetts. The Blackstone River has historically been impacted by a wide variety of industrial and non-point sources (USACE, 1994). The goals of this study are to identify environmental restoration needs and opportunities in the basin, develop plans and cost estimates for restoration projects, assess benefits and costs of alternatives, select a recommended watershed restoration plan, and prepare appropriate NEPA documentation. As part of this multi-faceted program, Battelle was contracted to conduct an Ecological Risk Assessment (ERA) of two impoundments along the Blackstone River: Fisherville Pond and Singing Pond (Task D, USACE Scope of Work, July 20, 1999; August 6, 2001). Previous investigations have indicated that sediments from these areas contain elevated levels of some chemicals (USACE, 1997).

The original objective of this ERA was to evaluate the potential risks of sediment-associated contaminants to aquatic and terrestrial communities found at both Fisherville and Singing Pond. At the time the investigation was initiated in 1999, Fisherville Pond provided approximately 69 acres of shallow open water habitat and about 100 acres of emergent, wet meadow and scrub, shrub wetland habitat. However, in 2000, a blockage in the Fisherville Dam failed, resulting in a substantial decrease in open water habitat and exposing mudflats that quickly became vegetated. As a result, Battelle has been contracted by NED to address the following questions:

- Under ponded conditions (*i.e.*, restoring the impoundment at Fisherville Pond), are there areas within Fisherville Pond or Singing Pond that require management action (*e.g.*, dredging) to reduce risks to wildlife and aquatic species?
- What is the relative magnitude of these risks at Fisherville Pond under full pool versus reduced pool conditions?

The ERA provides a baseline evaluation of the nature and geographical extent of the possible ecological risks based on current knowledge of environmental conditions, chemicals of interest, and ecological receptors. The methodology used is based on risk assessment guidance developed by the United States Environmental Protection Agency (EPA, 1996a; 1998). The results of this risk assessment will be used by NED to evaluate environmental restoration alternatives at Fisherville Pond and Singing Pond.

1.1 Site Description

A detailed description of the aquatic and terrestrial environment associated with the Blackstone River is provided in the EPA's *Blackstone River Initiative* (Wright *et al.*, 2001), USACE's *Blackstone River Watershed Reconnaissance Investigation* (USACE, 1997) and the preliminary baseline ecological assessment (McLaren/Hart, 1997). The Blackstone River serves as a drainage basin for approximately 475 square miles of land in central Massachusetts and Northern Rhode Island. The source of the river is found in the southern part of Worcester, Massachusetts at the confluence of the Middle River and Mill Brook (USACE, 1997). The Blackstone River then flows south-southeasterly to the mouth of the Providence River, eventually draining into Narragansett Bay. Approximately 350 ponds, lakes, and reservoirs are present in the Blackstone River Basin, many of which are impoundments created through the construction of dams to provide water for local water supplies and hydropower needs in the 19th and early 20th centuries. Fisherville Pond and Singing Pond were both created in this manner; greater detail pertaining to these two waterbodies and the reference site is provided below.

1.1.1 Fisherville Pond

Fisherville Pond is located at the confluence of the Blackstone and Quinsigamond Rivers (Figure 1-1). The drainage area of the Blackstone River upstream of the impoundment is 134 square miles. Fisherville Dam, a 12-foot high earthen and granite block structure is located approximately 1,000 ft from the confluence; it is 650 feet long with a 200-foot long spillway. The pond covered approximately 69 acres in its full pond conditions. In 1982, the spillway was opened to drain the pond water and conduct dredging operations in the upstream portion of Fisherville Pond (*i.e.*, near locations FP1 and FP3A in Figure 1-1). Within a few years, the spillway became plugged with debris and the area flooded back to pond levels.

In its full pool state, emergent, wet meadow, and scrub-shrub wetland habitat border Fisherville Pond. Cattail was found predominantly in the wetland and woolgrass, sedges, *Bidens* sp., purple loosestrife, *Phragmites*, reed canary grass, and other grasses were found predominantly in a wet meadow region located between the south and central pools (Figure 1-1; USACE, 1997). Fisherville Pond provided habitat to many species of waterfowl (*i.e.*, black duck, mallard) and pollution-tolerant fish species (*i.e.*, white sucker, golden shiner, and carp). Just beyond the confluence of the Blackstone and the Quinsigamond, water levels were very shallow (2-3 feet). Depths in other parts of the study area were approximately 5 feet, while upstream areas of the Fisherville Pond along the Quinsigamond River were deeper as a result of dredging in the 1980's, reaching a maximum depth of approximately 15 feet (Wright *et al.*, 2001).

Fisherville Pond was used frequently for recreational purposes by local fishermen and duck hunters. There is open land to the east of the pond often used by off-road vehicles (EPA, 1997). Also in close proximity to the former pond are agricultural fields, a gravel pit, a large subdivision, and an apartment complex.

In 2000, the blockage in the outlet gate of Fisherville Dam failed, draining Fisherville Pond to a narrow stream (Figure 1-2). As a result, the pond was reduced to approximately 26 acres, including the northern pool and a narrow, shallow channel running along the eastern shoreline of the central and southern pools. Approximately 43 acres were exposed. The remaining channel is unlikely to provide sufficient habitat for fish species. The newly exposed mudflat habitat has been quickly vegetated by a variety of emergent plant species.

1.1.2 Singing Pond

Singing Pond is located in the town of Sutton, upstream of Fisherville Pond (Figure 1-3). The pond was created by a 10-foot-high, 100-foot-long dam (USACE, 1997) and is a shallow impoundment (<4" deep), extending about 2000 feet upstream. The impoundment is bordered on one side by a marshy backwater, and to the other by agricultural fields. There is a large island located near the head of the impoundment and the southern side of the island is silted in and heavily vegetated. An emergent marsh area south of the channel provides good waterfowl habitat. There is limited data available describing fishery resources, however shallow water and poor water quality may impede the development of healthy populations (USACE, 1997).

1.1.3 Lake Wildwood

Lake Wildwood (Figure 1-4) was included as a reference area. It is an impoundment with a maximum depth of approximately 10 feet, covering an estimated 38 acres. The habitat surrounding Lake Wildwood is relatively undeveloped, however, the aquatic and terrestrial species are similar to those found at Fisherville and Singing Ponds. Although no historical sediment contaminant data is available, contaminant levels were expected to be quite low given the low level of human impact. The aquatic habitat, however, is seasonally impaired by a dense growth of fanwort, an invasive aquatic weed. The Town of Upton treated the pond with SONAR, an aquatic herbicide, to reduce fanwort growth in 1998 and 2000. The EPA's Total Maximum Daily Load (TMDL) Program determined that the fanwort problem was severe enough to receive a designation as one of Massachusetts's Impaired Waters (1998) as a result of noxious aquatic plants (EPA, 2002).

1.2 Previous Investigations

The Blackstone River watershed, considered the birthplace of the Industrial Revolution, has been studied extensively over the last several decades as a result of its long history of pollution problems and its potential impacts on the downstream Narragansett Bay. The Massachusetts Department of Environmental Protection (MADEP) has produced annual reports on the river for approximately 30 years, examining water quality problems associated with the operation of hydroelectric facilities, water withdrawals, and the resuspension of contaminated sediments trapped behind impoundments. The state of Rhode Island and the Narragansett Bay Project have also lead numerous projects assessing the potential impacts of the Blackstone River on the bay.

In 1981, the Massachusetts Department of Environmental Quality Engineering (DEQE, now MADEP) completed a major state effort to address the issue of contaminated sediments at several Blackstone River sites. The report entitled "A Sediment Control Plan for the Blackstone River" (commonly known as the 1981 McGinn report) describes metal concentrations, locations of sediment accrual, sediment volumes, impacts of the sediment on river ecology, and alternatives available to eliminate or mitigate the associated adverse impacts (USACE, 1997). The data evaluated indicated elevated concentrations of both metals and polycyclic aromatic hydrocarbons (PAHs).

As part of the Blackstone River Initiative (BRI), a multi-agency, multi-phased effort initiated in 1991, extensive chemistry and toxicity testing of the river's water column and sediments was performed under both low flow (Phase I) and storm conditions (Phase II). This effort was the first water quality survey ever conducted for the entire Blackstone River from Worcester, MA to Pawtucket, RI. The sampling, assessment, and modeling work was conducted to determine necessary restoration locations in the watershed that would prevent further deterioration of the resources in both the river and the body of water at the mouth of the river: Narragansett Bay. Specifically, the BRI consisted of dry and wet weather surveys focused on analyzing the toxicity and chemistry of ambient river water, sediments and their oxygen demand, sediment pore water, significant industrial and municipal water effluents and benthic macroinvertebrate community health. In several reaches, resuspension of old materials contributed to toxicity and nutrient violations of state standards. As a result of these investigations, the EPA was better able to predict annual loading rates to the Providence River and determine that the Blackstone River is the major source of the majority of the pollutants (Wright *et al.*, 2001).

Following the BRI, the USACE continued to monitor sediment and water column toxicity and chemistry. These efforts are published as the USACE Reconnaissance Investigation (USACE, 1997). This work was performed to comprehensively examine restoring fish and wildlife habitat via flow regulation; restoring fish spawning habitat, wetland systems, and waterfowl nesting areas; constructing fish passage facilities; and remediating the contaminated sediments. The primary purpose of these analyses was to assess the extent of problems and determine the appropriate federal actions (USACE 1997). Based on the results of their investigations, the Corps determined that action was necessary and recommended proceeding to the feasibility stage of analysis. Specific areas were identified within the Blackstone River Watershed where the Corps could improve and restore fish and wildlife values. Two of the locations that they selected were Fisherville Pond and Singing Pond.

As part of the Corps Feasibility Study (USACE, 1997), McLaren/Hart Environmental Engineering Corporation also completed a Preliminary Baseline Ecological and Human Health Risk Characterization at Fisherville Pond (McLaren Hart, 1997). The report interpreted chemical and biological data including chemical analyses of surface water, porewater, sediment, and fish tissue; sediment and surface water ambient toxicity tests; and fish and benthic macroinvertebrate community surveys. Based on the results of the evaluation, McLaren/Hart (1997) concluded that the sediment contaminants at Fisherville Pond (*i.e.*, metals and PAHs) might pose a risk to the benthic community, amphibians, and muskrats, but that surface water quality did not pose significant risk to the surrounding ecological community. In their human health risk characterization, they concluded that there were potential noncarcinogenic hazards associated with consumption of fish from Fisherville Pond as a result of high PCB concentrations. McLaren/Hart (1997) also concluded that carcinogenic risks might also be associated with incidental ingestion of and dermal contact with benzo(a)pyrene and chromium in sediment.

1.3 Purpose and Scope

The purpose of this ERA is to evaluate the likelihood that adverse ecological effects may occur or are occurring as a result of exposure to contaminated sediments within Fisherville and Singing Ponds. The ERA process will be used to evaluate and organize site-specific data in a quantitative or semi-quantitative manner for the purpose of understanding or predicting the relationship between identified stressors (anthropogenic chemicals, physical conditions) and potential impacts on relevant ecological communities (EPA, 1998).

As a result of the changed conditions within Fisherville Pond, the evaluation has been expanded to address the following questions:

- Under ponded conditions (*i.e.*, restoring the impoundment at Fisherville Pond) are there areas within Fisherville Pond or Singing Pond that require management action (*e.g.*, dredging) to reduce risks to wildlife and aquatic species?
- What is the relative magnitude of these risks at Fisherville Pond under full pool versus reduced pool conditions?

To address these questions, a two-pronged approach was developed. The first focused on determining the potential risk to omnivorous and piscivorous wildlife species among designated areas of Fisherville Pond and Singing Pond. Under this approach, it was assumed that the Fisherville Pond impoundment would be restored to its 1999 condition. A qualitative weight of evidence approach was used, deriving a measure of potential risk (*e.g.*, high, medium, low) for specific areas including: Fisherville Pond-North Pool, Fisherville Pond-Central Pool, Fisherville Pond-South Pool, Singing Pond-Main Channel, Singing Pond-Marsh, and the designated reference area, Lake Wildwood (Figure 1-3). The second evaluation focused on evaluating the relative risks to wildlife (*i.e.*, piscivorous, insectivorous, or omnivorous) species from exposures with or without the impoundment at Fisherville Pond.

1.4 Report Organization

This document is organized in the following manner. Section 2 presents the problem formulation for the ERA, summarizing the assessment endpoints and the species selected for evaluation. A summary of the data collection methods is in Section 3 while Section 4 presents the results. The risk characterization for the weight of evidence is presented in Section 5, and Section 6 summarizes the results of the relative risk evaluation. Section 7 presents the summary and conclusions.

2. PROBLEM FORMULATION

The purpose of the problem formulation is to describe the risk evaluation process. The appropriate ecological receptors and endpoints are identified, and a conceptual site model is developed that depicts the pathways through which the identified receptors may be exposed.

2.1 Approach

As previously discussed, Battelle was contracted by NED to:

- Identify areas within Fisherville Pond or Singing Pond that require management action (*e.g.*, dredging) to reduce risks to wildlife and aquatic species under full pool conditions (*i.e.*, restoring the impoundment at Fisherville Pond);
- Evaluate the relative magnitude of wildlife risks at Fisherville Pond under full pool versus reduced pool conditions.

To address these questions, a two-pronged approach was developed. The first focused on determining the potential risk to ecological receptors among designated areas of Fisherville Pond and Singing Pond. It assumed that the Fisherville Pond impoundment would be restored to its 1999 condition (*i.e.*, full pool). For the purpose of this evaluation, six specific areas were identified for consideration (Figure 1-1, 1-3, 14; Appendix G):

- Fisherville Pond-North Pool: identified as the northern section of the Fisherville Pond impoundment, upstream of the confluence with the Quinsagamond River; much of this area was previously dredged;
- Fisherville Pond-Central Pool: the largest portion of Fisherville Pond, located just north of the wet meadow area;
- Fisherville Pond-South Pool: the small pool adjacent to Fisherville Dam;
- Singing Pond-Main Channel: the main channel and impoundment of Singing Pond;
- Singing Pond-Marsh Area: the marsh area to the west of the Singing Pond-Main Channel
- Lake Wildwood: the reference location

Table 2-1 provides a summary of the sampling stations grouped within each of these designated areas.

The second evaluation focused on evaluating the relative risks to selected wildlife species associated with exposures occurring with and without the presence of the impoundment at Fisherville Pond.

2.2 Conceptual Site Model

The purpose of the conceptual model is to describe the assumed sources of contaminants, routes of transport of contaminants, contaminated media, routes of exposures and ecological receptors. Previous investigations have indicated that contaminants may enter the water column or sediments from upland areas surrounding the Fisherville Pond/Blackstone River system through overland flow, stormwater runoff, mobilization and settling of sediment from upstream areas, or erosion (McLaren/Hart, 1997). Historic and current point sources (including WWTP discharges) are also a source of contaminants.

A conceptual model of exposure of the aquatic food web to contaminants in sediments in the study area is presented in Figure 2-1 (McLaren/Hart, 1997). A second conceptual model was developed to address exposures to newly exposed soil at Fisherville Pond (Figure 2-2). The conceptual models demonstrate that the receptors may be exposed to sediment or soil-associated chemicals through a variety of mechanisms including direct contact with sediment/soil, as well as indirectly through exposures to porewater concentrations and trophic transfer. The measurement endpoints described in Section 2.3 are aimed at evaluating these pathways.

2.3 Weight of Evidence Approach

To identify areas within Fisherville Pond and Singing Pond that might require remediation, a qualitative weight-of-evidence evaluation was applied. Weight-of-evidence is a process by which multiple lines of evidence, expressed as measurement endpoints, are related to assessment endpoints to evaluate whether significant risk of harm is posed to the environment (Menzie *et al.*, 1996). The approach used for this evaluation was an adaptation of that proposed by Menzie *et al.* (1996) and focused on the assessment and measurement endpoints identified below using the data described in Section 3 and 4.

As defined by EPA (1992), assessment endpoints are explicit statements of the ecological system that are to be protected. For example, species richness and abundance of the fish community or other valuable resources of the river may be evaluated as assessment endpoints. Assessment endpoints are either measured directly or are evaluated through indirect measures. Measurement endpoints represent quantifiable ecological characteristics that can be measured, interpreted, and related to the valued ecological components chosen as assessment endpoints. General considerations for selecting assessment and measurement endpoints include ecological relevance, policy goals and societal values, and susceptibility to chemical stressors (EPA, 1992; 1996a).

Based on the objectives of this evaluation (*i.e.*, to evaluate potential risks associated with exposure to sediments) and the conceptual site models developed, the specific assessment endpoints identified are:

- The health of the benthic invertebrate community;
- The health of the fish community; and,
- Sustainability of upper trophic level species.

The measurement endpoints for each of these assessment endpoints are summarized in Table 2-2 and below. Each measurement endpoint was assigned a relative weight (*i.e.*, High, Medium, or Low) reflecting the overall confidence in the measurement endpoint in terms of the strength of association with the assessment endpoint (*i.e.*, High, Medium, Low) and the quality of the data evaluated (*i.e.*, Good, Poor, Acceptable). For example, the data used to evaluate Measurement Endpoint 1a, Bulk Sediment Comparison to Sediment Quality Benchmarks was considered to be of good quality, however, these simple benchmarks do not take into account chemical and physical factors controlling bioavailability, therefore they were considered to have a low strength of association with Assessment Endpoint 1, Health of the Benthic Invertebrate Community. As a result, this measurement endpoint was given an overall weight of Medium. In contrast, the quality of the bulk sediment toxicity data was considered good and the strength of association was considered high, therefore Measurement Endpoint 1d, Results of Acute and Chronic Toxicity Tests, was given a overall weight of High. Table 2-2 also presents the relative weight of each measurement endpoint evaluated.

2.3.1 Assessment Endpoint 1: Health of the Benthic Invertebrate Community

The benthic invertebrate community includes a wide array of organisms living in close association with sediments. Due to their direct exposure to surface sediments, benthic invertebrates are a key indicator species when considering the impacts associated with contaminated sediment. There are a variety of ways, both direct and indirect to measure the potential effects of chemicals of potential concern (COPC) in sediments. For this evaluation, the health of the benthic invertebrate community was evaluated through four separate lines of evidence including bulk sediment chemistry data, the results of acute and chronic toxicity tests, a benthic invertebrate community analysis, and an evaluation of water quality.

2.3.1.1 Measurement Endpoints 1a, 1b, 1c — Bulk Sediment Chemistry

Sediment samples were collected in 1999 and 2001 from locations throughout Fisherville Pond, Singing Pond, and Lake Wildwood and analyzed for a suite of chemicals including metals, pesticides, polychlorinated biphenyls (PCBs) and polycyclic aromatic hydrocarbons (PAHs). These data were compared to freshwater sediment quality guidelines to determine if exposures to sediment concentrations have the potential to adversely impact benthic communities.

Sediment quality guidelines (SQGs) have been developed for the purpose of predicting the potential toxicity of contaminated sediments. The goal of SQGs is to provide an estimate of chemical concentrations in sediment that are not likely to be associated with an adverse effect. Most SQG have been developed for individual chemicals measured on a dry weight basis, however, there are a variety of limitations associated with this approach. For example, dry weight measurements do not account for numerous chemical and physical factors (*e.g.*, TOC, grain size, etc.) that may affect the bioavailability of contaminants in sediment. In addition, SQG for individual chemicals do not account for the potential effects of chemical mixtures.

<u>Measurement Endpoint 1a—Comparison to Sediment Quality Benchmarks</u>

For the purpose of evaluating individual chemical concentrations, Threshold Effect and Probable Effect Concentrations (TEC and PEC, respectively) developed by MacDonald *et al.* (2000) were used (Table 2-3). These values were developed based on a review of existing sediment quality guidelines using a consensus-based approach. The TEC represents the concentration below which adverse effects are not expected to occur, while exceedance of the PEC is assumed to indicate the potential for adverse effects to benthic organisms for many chemicals. Other comparable criteria were evaluated when possible in the absence of PEC values.

In addition to the evaluation of individual chemical concentrations, chemical mixtures were also addressed using the PEC quotient or PEC-Q. As described by MacDonald *et al.* (2000) and Ingersoll *et al.* (2000), the PEC-Q is derived by a three-step process developed by Long *et al.* (1998). In the first step, the concentration of each chemical in a given sample is divided by its respective sediment quality criteria, in this case defined as a Probable Effects Concentration (PEC) as derived by MacDonald *et al.* (2000). The resulting ratio is defined as a PEC quotient or PEC-Q. The PEC-Qs for each chemical are then summed and divided by the number of individual chemicals evaluated to derive a mean PEC-Q for each sample. Derivation of the mean PEC-Q facilitates comparisons between stations, particularly in situations where differing numbers of chemicals have been evaluated. Based on a sample size of 175, MacDonald *et al.* (2000) found that the incidence of toxicity in freshwater sediments could be predicted in up to 94.4 percent of sediments considered through use of the mean PEC-Q.

Ingersoll *et al.* (2000) further evaluated this relationship, exploring different methods of deriving the mean PEC-Q. They found that the best predictive relationship was associated with mean PEC-Qs calculated by equally weighting the contribution of metals, PAHs and PCBs in the evaluation of sediment chemistry and toxicity. Specifically, they calculated the geometric mean of the average PEC-Q associated with the metals, the PEC-Q with total PCBs and the PEC-Q associated with total PAHs. The geometric mean of the three PEC-Q was used in place of the arithmetic mean based on the assumption that it provides a better measure of central tendency.

<u>Measurement Endpoint 1b and 1c—Evaluation of Chemical Mixtures</u>

Mixtures were also evaluated using EPA's Draft Equilibrium Partitioning Sediment Guidelines for metals (EPA, ND) and PAH mixtures (EPA, 2000b) which evaluate whether or not the relevant PAH and metal mixtures may prove toxic to surrounding benthic organisms. PAHs are not typically believed to bioaccumulate into tissues of aquatic organisms like other persistent chemicals, however they may exhibit sublethal toxicity through narcosis effects on fish and aquatic invertebrates. Combining equilibrium partitioning theory, narcosis theory and additivity, EPA (2000b) has developed equilibrium partitioning sediment guidelines for PAH mixtures in sediments (Appendix D).

EPA's guidance for metals mixtures is based on the theory that acid volatile sulfides (AVS) control the bioavailability of metals. In the presence of AVS in sediments, certain metals, primarily copper, cadmium, lead, nickel, and zinc (Ankley, 1996; Ankley *et al.*, 1996) precipitate as their respective metal sulfides, which are not bioavailable (Di Toro *et al.*, 1991). Thus, if the molar concentration of AVS in sediments is higher than the sum of the simultaneously extracted metals (SEM; the sum of the molar concentrations of these metals in the 1 N HCl extract) these metals are assumed to be in non-bioavailable forms in the sediments. In other words, if the SEM/AVS ratio is greater than 1, the metals are believed to be bioavailable. Otherwise, it is assumed that they are bound to sulfides and not bioavailable.

2.3.1.2 Measurement Endpoint 1d—Sediment Toxicity

Acute and chronic sediment toxicity tests were performed on bulk sediment samples collected from the study area. Reductions in the survival and growth of the amphipod, *Hyalella azteca* and the insect, *Chironomid tentans*, were evaluated in 10-day bulk sediment bioassays. Chronic effects were also examined in a 42-day test with *Hyalella azteca* examining survival, growth and reproduction. Results of a toxicity identification evaluation (TIE) were also considered.

2.3.1.3 Measurement Endpoint 1e—Water Quality

To evaluate potential impacts of water quality on the benthic community, concentrations of metals were measured in porewater samples extracted from sediments collected in 1999 throughout the study area. These concentrations were compared to chronic toxicity thresholds for aquatic species, expressed as National Ambient Water Quality Criteria (AWQC; Table 2-4; EPA, 1999a). In addition, concentrations of PAHs, PCBs and selected pesticides in porewater were estimated from measured sediment concentrations in the same samples based on the theory of Equilibrium Partitioning Theory (EqP) (DiToro *et al.*, 1991). The EqP theory assumes that the primary exposure route of aquatic organisms to sediment-bound chemicals is from the partitioning of chemicals into the interstitial porewater. Under equilibrium conditions, it is assumed that this partitioning process is controlled by chemical and physical factors such as the organic carbon content of the sediment and the octanol-water partitioning coefficient (K_{ow}) of the individual contaminant. Using this theory, porewater concentrations of organic chemicals may be estimated from sediment concentrations using the K_{ow} and the fraction organic carbon. Similarly, water quality criteria can be estimated for these chemicals based on available SQGs, using the same assumptions (Table 2-5).

2.3.1.4 Measurement Endpoint 1f—Benthic Community Analysis

A quantitative evaluation of the benthic invertebrate community was conducted. Species richness and abundance were recorded for each location using a variety of diversity indices. Diversity indices are used to characterize species abundance relationships in communities. Diversity is composed of two distinct components: 1) species richness, or the total number of species and 2) evenness, or how the numbers of individuals are distributed among the species (Ludwig and Reynolds, 1988). In addition, a qualitative evaluation of mouthpart deformities was performed.

2.3.2 Assessment Endpoint 2: Health of the Fish Community

Fisherville and Singing Ponds each support a limited warmwater fish community. These communities are important ecologically as a potential food source for piscivorous birds and mammals and to local anglers as the source of a potential recreational fishery. To evaluate the potential impacts to this community, three lines of evidence were considered, surface water quality, an evaluation of fish tissue residues in comparison to literature-based effect levels and the results of a fish community assessment.

2.3.2.1 Measurement Endpoint 2a—Surface Water Quality

Concentrations of metals were measured in a limited number of samples from throughout Fisherville Pond during 2001. These concentrations were compared to the AWQC described in Table 2-4 to evaluate potential impacts to the fish community.

2.3.2.2 Measurement Endpoint 2b—Evaluation of Fish Tissue Residues

Fillet and whole body tissue concentrations were collected and analyzed for metals and PCBs. These data were evaluated based on comparisons of tissue concentrations measured in fish from Lake Wildwood, the designated reference area. In addition, measured concentrations were compared to body burden concentrations reported to be associated with adverse effects on behavior, growth, reproduction, and survival for those chemicals for which data were available. Screening effects guidelines were derived based on effects data obtained from the Environmental Residue and Effects Database (ERED; USACE, 2000; Table 2-6).

2.3.2.3 Measurement Endpoint 2c—Fish Community Analysis

The sustainability and health of the warmwater fish community was also evaluated through consideration of species diversity and productivity based on data collected from two sampling efforts conducted by NED. Fish collected were identified, counted, weighed, and inspected for external abnormalities. Length and weight data for individual fish were used to calculate the coefficient of condition, a widely used measure of fish condition or “plumpness” (Carlander, 1977). These data were used to provide a measure of the fish community’s health.

2.3.3 Assessment Endpoint 3—Wildlife Assessment

Potential risks to upper trophic level receptors were evaluated by calculating estimates of exposure (*i.e.*, dose) to sediment-associated contaminants. Doses were calculated using standard risk equations quantifying the exposures from consumption of prey items and incidental ingestion of sediment and soils. Based on the conceptual site model, three receptor types were selected:

- Piscivorous wildlife
- Omnivorous waterfowl
- Terrestrial /insectivorous songbirds

These three receptor types were represented by the river otter, the mallard duck, and the American robin, respectively. These species are all known to exist in the Blackstone River valley and were assumed to be exposed to contaminated sediments and soils through complete exposure pathways as depicted in Figures 2-1 and 2-2.

3. SUMMARY OF METHODS AND DATA COLLECTION

Data to address the identified measurement endpoints were collected by Battelle and NED in October 1999 and September 2001 (Table 3-1). A summary of the methods used to collect or derive these data is presented in this section. Additional details regarding the collection of these data and the analytical methods used are reported in the Task C Work Plan (Battelle, 1999a,b) and Final Data Report (Battelle, 2000) and Appendices A, B, and C.

3.1 Bulk Sediment

Surface sediment samples were collected from Fisherville Pond, Singing Pond, and Lake Wildwood on October 18-22, 26, and 30 in 1999 and on September 13-14 in 2001. Samples collected in 1999 were used to evaluate whole sediment (Measurement Endpoints 1a, 1b, and 1c) and porewater chemistry (Measurement Endpoint 1e), bulk sediment toxicity (Measurement Endpoint 1d), benthic infauna composition (Measurement Endpoint 1f) and to perform a Toxicity Identification Evaluation (TIE; Measurement Endpoint 1d). Samples collected in 2001 were used to further evaluate sediment chemistry, TOC, and to perform additional sediment toxicity tests. Table 3-2 provides a summary of the samples collected during these efforts and the analyses performed. Approximate sample locations are plotted in Figures 1-1, 1-3, and 1-4, and grouped by area in Table 2-1.

<u>Summary of 1999 Sampling Effort</u>

In 1999, approximately 7 gallons of sediment were collected from 12 locations in Fisherville Pond, four in Singing Pond, and two in Lake Wildwood using a 0.04 m^2 Van Veen grab sampler. Each of these samples was analyzed for ten metals, sixteen priority pollutants, seventeen pesticides and total PCBs. Sediments were collected into Teflon-lined food grade buckets and were kept on ice in coolers until homogenization. At each of those 18 stations, an additional three replicate grab samples were collected for benthic infauna analysis. Samples were all collected from an aluminum boat provided by NED with the exception of SP4. The location of SP4 was chosen in an attempt to represent a muddy sediment type within the marsh area in Singing Pond (most of the pond contained a medium to coarse sand). Because the site was so close to shore, the grab was deployed by hand, from the edge of the pond. It should be noted that this station was close to a small, gravel parking lot and could be influenced from associated runoff.

Soil and sediment samples were also collected from selected locations for more limited analyses. Specifically, six sediment grab samples were collected from the southern pool of Fisherville Pond, in the vicinity of the dam (see samples H1-H6 in Figure 1-1). The previous ecological and human health risk characterization (McLaren Hart, 1997) reported high AVS/SEM rations in this portion of the pond, thus more extensive sampling was performed to delineate the potentially heavily impacted area. Approximately 1.5 gallons of sediment were collected from each of these locations for the analysis of ten metals, acid volatile sulfide and simultaneously extracted metals (AVS/SEM), TOC, and grain size. In addition, soil samples were collected using a stainless steel shovel from three locations within the wet meadow area in Fisherville Pond (see WM-1 and WM-2 in Figure 1-1). Each of these samples was analyzed for the ten metals, TOC, and grain size.

<u>Summary of 2001 Sampling Effort</u>

In 2001, approximately 3 gallons of sediment were collected from three locations in Fisherville Pond, two in Singing Pond and one in Lake Wildwood using a 0.04 m^2 Young-modified Kynar coated Van Veen grab. The samples at Fisherville Pond were collected by hand, since the sites were in shallow water. Sampling sites FP1, FP3 and FP4 were chosen to be as close to the original sites (*i.e.*, 1999 locations) as possible, while still remaining submerged in water. The sites in Singing Pond were collected via canoe and were selected to provide more data for the marsh area.

Sediment from each sample was placed in Teflon-lined buckets and kept on ice until homogenization. The samples were transferred to Battelle Duxbury Laboratories for compositing, mixing and final distribution to the University of Connecticut Environmental Research Laboratory for chemical analysis (*i.e.*, eleven metals, 40 priority pollutants, 20 pesticides, Total PCBs; see Appendix B).

3.2 Water

Water quality was considered in the evaluation of both the benthic community (Measurement Endpoint 1e) and the fish community (Measurement Endpoint 2a). For the benthic community, concentrations of metals measured in porewater extracted from sediment samples and concentrations of organic chemicals estimated based on chemical concentrations measured in sediments were evaluated. For fish, surface water concentrations measured by the NED in 2002 were evaluated.

3.2.1 Evaluation of Porewater (Measurement Endpoint 1e)

As indicated in Table 3-2, porewater was extracted from 18 of the sediment samples collected in 1999 and analyzed for metals. In addition, porewater concentrations of PAHs, pesticides and PCBs were estimated for these samples using the EqP theory (DiToro *et al.*, 1991) described in Section 2. Specifically, porewater concentrations were estimated using the following relationship:

$$C_{\text{ porewater}} = C_{\text{ sediment}} / f_{oc} * K_{oc}$$

where:

$C_{porewater}$ = concentration of the individual PAH, pesticide or PCB in porewater

$C_{sediment}$ = concentration of the individual PAH, pesticide or PCB in sediment

f_{oc} = fraction organic carbon (f_{oc} = % total organic carbon (TOC) / 100)

K_{oc} = carbon/water partitioning coefficient log
($\log K_{oc} = 0.00028 + 0.983 * \log K_{ow}$)

Site-specific TOC and sediment concentrations for individual PAHs, pesticides or PCBs were used in the calculation of porewater PAH concentrations for each site. Log K_{oc} (*i.e.*, octanol-carbon partition coefficient) and log K_{ow} values (*i.e.*, octanol-water partition coefficient) for individual PAHs are presented in Table 2-5.

National AWQC (EPA, 1999a) for metals were evaluated (Table 2-4). Values were not available for the PAHs or pesticides; therefore, screening benchmarks were derived based on the relationship described above, using the SQG values in place of site-specific sediment concentrations and an assumed TOC content of 1 percent. Table 2-5 summarizes the estimated porewater criteria. The sediment screening benchmarks applied were the same values identified for screening the measured sediment concentrations (see Table 2-3).

3.2.2 Evaluation of Surface Water (Measurement Endpoint 2a)

Surface water samples were collected monthly from June through December 2001 at seven locations within Fisherville Pond (J. Keenan, NED, Pers. Communication, 2002; Table 3-3). At each station, samples were collected at two depths, within 1 m of the surface and within 1 m of the bottom. Metals were analyzed in each sample according to EPA Method 200.7 and Standard Methods 3120B.

3.3 Fish

Fish were collected from the study area by NED in 1996 and in 1999 (Table 3-4). In 1996, fish were collected from Fisherville Pond using four methods: gill net, hoop nets, beach seine, and backpack electrofishing (USACE, 1997). In 1999 fish were sampled from Fisherville Pond, Singing Pond, and Lake Wildwood using a 15 ft. shallow draft electroshock boat. At Fishersville, fish were sampled at five primary locations, including one location in the north pool, two in the central pool and two in the southern portion of the impoundment. Minnow traps were also set at several locations. At Singing Pond, sampling was confined to the main channel of the Blackstone River. Fish were sampled at two locations in Lake Wildwood.

3.3.1 Fish Tissue Evaluation (Measurement Endpoint 2b)

A subset of the fish collected by USACE in 1999 was retained for tissue analysis (Table 3-5). Samples were frozen and held by NED until transfer to Battelle on December 6, 1999 and were maintained frozen by Battelle until analysis. Fish samples for fillet analyses were generally a composite of two or more fish; Table 3-5 indicates the number of fish included in each composite. Fillet (skin-on) from individual fish were homogenized separately and equal amounts from each fillet composited and homogenized again. Sub-samples from this homogenate were removed for PCB and metal analyses. Fish samples for whole body analysis were homogenized whole and sub-samples removed for PCB and metal analyses.

3.3.2 Fish Community Evaluation (Measurement Endpoint 2c)

During both the 1996 and 1999 sampling events, fish collected were identified to species, counted, weighed and inspected for external abnormalities. Length and weight data for individual fish were used to calculate the coefficient of condition, a widely used measure of fish condition or “plumpness” (Carlander, 1977). The coefficient of condition, K, is calculated as follows:

$$K = \frac{W * 100}{L^3}$$

where W = weight in grams, L = length in centimeters.

Additional information about fisheries resources at Fisherville Pond is available from MADFW (see USACE 1997 and Lee McGlauglin, pers. commun. 1999). The MADFW studies were conducted using gill nets (5 nets, each set overnight for 18 hours). No fisheries studies have been conducted at Fisherville Pond since the pool level was lowered in 2000.

3.4 Bulk Sediment Toxicity Tests (Measurement Endpoint 1d)

The 10-day solid-phase static-renewal test with *Hyalella azteca* was performed by Battelle with the 1999 sediment samples according to ASTM (1994) guidelines. As described in these guidelines, eight replicates of each sediment treatment were tested with 10 amphipods in each test chamber. A more detailed discussion of the specific methods used to perform this test is in the Task C work plan (Battelle, 1999a) and Final Data Report (Battelle, 2000).

For the 2001 samples, sediment toxicity testing was performed at the Great Lakes Environmental Center (GLEC) in Traverse City, MI. GLEC analyzed both acute and chronic toxicity. The 10-day bulk sediment toxicity test with *Chironomid tentans* was performed according to the guidance outlined in EPA

(2000a) and ASI Standard Operating Procedures (SOPs). The 42-day chronic toxicity test with *Hyallela azteca* was performed according to EPA (2000a), ASTM (2000), and GLEC Standard Operating Procedures (Appendix C).

3.5 Benthic Community Analysis (Measurement Endpoint 1f)

As previously described, three replicate grab samples were collected at 18 of the sediment sampling locations for the purpose of evaluating the benthic community. These samples were sieved in the field through a 583-µm-mesh bucket sieve and fixed in buffered formalin. The samples were stored by NED until November 12, 1999, and then shipped to the benthic laboratory for processing (Appendix C).

Sediment samples collected in triplicate at Fisherville Pond, Singing Pond, and Lake Wildwood were analyzed for benthic community parameters at Barry A. Vittor and Associates, Inc. (BVA) of Mobile, Alabama. Per the direction of NED, the triplicate samples collected at each location were homogenized and an aliquot representing one third of the total sample volume was evaluated. These aliquots were sorted by hand using a Wild M-5A dissecting microscope. All benthic organisms except juveniles, damaged individuals or other forms lacking necessary taxonomic characters were identified to the lowest possible taxonomic level, typically to species. In addition, voucher specimens were prepared and examined for evidence of mouthpart deformities. A minimum of one specimen for each species encountered was evaluated in this manner.

Based on the data collected, a variety of diversity indices were calculated. The specific indices used for this evaluation include:

- The Pielou Evenness Index;
- The Equitability Index;
- The Margalef Richness Index;
- The Simpson’s Diversity Index; and,
- The Shannon-Weiner Index.

Each of these diversity indices provides a slightly different measure of species diversity. The Pielou Evenness Index and the Equitability Index both indicate how evenly the numbers of individuals are distributed among the species present (*i.e.*, diversity). These indices run from 0 to approximately 1, with increasing ‘evenness’ as the value increases. In contrast, the Margalef Richness Index is a straightforward measurement of the number of species present (*i.e.* species richness). The Simpson’s Diversity Index and the Shannon-Weiner Index examine a combination of richness and diversity indicators.

3.6 Toxicity Identification Evaluations (Measurement Endpoint 1d)

As part of the overall Feasibility Study, a TIE evaluation was performed by Science Applications International Corp. (SAIC, Newport RI) based on a subset of the data collected for the ERA. Specifically, toxicity characterizations of porewater extracted from sediments associated with ten of the 18 sediment locations were performed using a freshwater fish (*Pimephales promelus*) and an amphipod (*Hyalella azteca*). TIE manipulations were performed as described by EPA (1993) and modified in Ankley *et al.* (1991), Jop *et al.* (1991) and EPA (1991b). These results were qualitatively considered in the overall conclusions. A full summary of the TIE is provided in Battelle (2000).

3.7 Calculation of Dose Estimates for Wildlife Species (Assessment Endpoint 3)

As described in Section 2.3.3, potential impact to upper trophic level species under the scenarios evaluated were addressed through the calculation of screening-level dose estimates for three wildlife species: the river otter, the mallard duck and the American robin. It was assumed that the river otter and mallard would be exposed indirectly to sediments through incidental ingestion and consumption of prey (*i.e.*, fish or aquatic invertebrates, respectively) as depicted in the conceptual site model for sediments (Figure 2-1). Exposure to the robin, assumed to be associated with contaminated soils in the wet meadow area, was estimated indirectly through incidental ingestion and consumption of contaminated prey (*i.e.*, soil invertebrates; Figure 2-2). For each species, the exposure point concentrations were determined based on average soil or sediment concentrations for each of the designated areas.

A summary of the method used to calculate dose estimates for each species is summarized below. The calculations and dose results are presented in Appendix F. The calculated doses were compared to toxicity reference values derived based on data presented by Sample *et al.* (1996) to generate hazard quotients (HQs).

3.7.1 Piscivorous Mammals

River otters are medium-sized mammals often found near lakes, marshes, and streams with fish typically comprising from 60 to 100 percent of their diet (EPA, 1993). For the purpose of this assessment, the river otter was conservatively assumed to consume only fish and to forage exclusively in the designated areas. Whole body fish tissue concentrations for fish collected in 1999 (expressed as dry weight) for each area were averaged for use in this assessment. Tissue data were only available for PCBs and metals, therefore, concentrations of other COPC (*i.e.*, PAHs, pesticides) were estimated based on site-specific sediment concentrations. For the purpose of estimating fish tissue concentrations, it was assumed that the COPC would be transferred from the sediments to benthic invertebrates, and then to fish. The estimated concentration of each COPC in benthic invertebrates was calculated using the following equation:

$$Cb = (Cs / f_{oc}) \times BSAF \times fL$$

where:

Cb	=	Concentration of COPC in benthic invertebrates (mg/kg-wet wt)
Cs	=	Sediment concentration (mg/kg dry wt)
f_{oc}	=	Organic carbon content of sediments at the Site (reported as a fraction)
BSAF	=	Biota Sediment Accumulation Factor (mg/kg-oc/mg/kg lipid) (PAHs assigned value of 0.1; all other chemicals assumed to be 1)
fL	=	Conversion factor to convert lipid-normalized body burden to a wet-weight concentration (mg/kg-lipid/mg/kg-wet weight) (value of 0.01 assumed)

Using the resulting estimated benthic tissue concentrations, the estimated fish tissue concentrations (for those chemicals not measured in fish tissue) were calculated using the following equation:

$$Cf_s = (Cb \times IR \times AF \times FI)/(GR + ER)$$

where:

Cf_s	=	Estimated concentration of COPC in fish resulting from ingestion of contaminated invertebrates (mg/kg-wet weight)

Cb = Estimated concentration of COPC in invertebrates (mg/kg-wet weight)

IR = INGESTION RATE OF FISH (KG/KG-DAY) (5.36 G/DAY BASED ON VALUE FOR CARP REPORTED BY KEVERN, 1966)

AF = Absorption factor of COPC (PAHs assigned value of 0.1; all other chemicals assumed to be 1)

FI = Fraction of fish diet made up of invertebrates (assumed to be 100% or 1)

GR = Growth rate (equivalent to 0.01 x $(BW)^{-0.2}$ per Thomann, 1989)

ER = Excretion rate (equivalent to 0.25 x IR based on Gobas, 1993)

Using the measured (*i.e.*, metals and PCBs) and estimated (*i.e.*, other organic chemicals) COPC concentrations for fish tissue, the estimated daily intake of each COPC by river otter from ingestion of fish was calculated using the following equation:

$$DI = \{(Cf \times IRf) + (Cs \times IRs)\} \times 1/BW$$

where:

DI = Daily intake of river otter (mg/kg-d)

Cf = Measured or estimated concentration of COPC in fish (mg/kg wet-wt)

Cs = Concentration of COPC in sediment (mg/kg)

IRF = FISH INGESTION RATE OF OTTER (KG/D) (BASED ON ALLOMETRIC EQUATIONS FROM EPA,1993)

IRs = Sediment Ingestion rate of otter (kg/d) (based on information for raccoon from EPA, 1993)

BW = Body weight (kg) (EPA, 1993)

3.7.2 Insectivorous Waterfowl

Mallards are dabbling ducks feeding on aquatic invertebrates, seeds of aquatic plants, and cultivated grains (EPA, 1993). During the breeding season, aquatic invertebrates comprise 70 to 90 percent of their diet, therefore, for the purpose of this assessment, it was assumed that the primary exposures of mallards to contaminated sediments would be through the ingestion of benthic invertebrates. The estimated concentration of each COPC in benthic invertebrates was calculated using the following equation:

$$Cb = (Cs / f_{oc}) \times BSAF \times fL$$

where:

Cb = Concentration of COPC in benthic invertebrates (mg/kg-wet wt)

Cs = Sediment concentration (mg/kg dry wt)

f_{oc} = Organic carbon content of sediments at the Site (reported as a fraction)

BSAF = Biota Sediment Accumulation Factor (mg/kg-oc/mg/kg lipid) (PAHs assigned value of 0.1; all other chemicals assumed to be 1)

fL = Conversion factor to convert lipid-normalized body burden to a wet weight concentration (mg/kg-lipid/mg/kg-wet weight) (assumed value of 0.01)

Based on these estimated concentrations, the dose to the mallard was calculated using the following equation:

$$DI = \{(Ci \times IRi) + (Cs \times IRs)\} \times FD \times 1/BW$$

where:

DI	=	Daily intake of mallard (mg/kg-d)
Ci	=	Estimated concentration of COPC in benthic invertebrates (mg/kg wet-wt)
Cs	=	Concentration of COPC in sediment (mg/kg)
IRi	=	Invertebrate ingestion rate of mallard (kg/d) (based on allometric equations from EPA,1993)
IRs	=	Sediment Ingestion rate of mallard (kg/d) (from EPA, 1993)
FD	=	Fraction of diet comprised of invertebrates (75%) based on annual average for breeding female (EPA, 1993).
BW	=	Body weight (kg) (EPA, 1993)

3.7.3 Terrestrial/Insectivorous Songbird

Robins are terrestrial songbirds feeding primarily on soil invertebrates (EPA, 1993). For the purpose of this assessment, it was assumed that the primary exposures of robins to COPC would be through the ingestion of soil invertebrates (*i.e.*, earthworms) found in the wet meadow area at Fisherville Pond. The concentration of COPC in soil invertebrates was estimated using bioaccumulation factors reported by EPA (1999b)

Based on these estimated concentrations, the dose to the robin was calculated using the following equation:

$$DI = \{(Ci \times FI) + (Cs \times IRs)\} \times 1/BW$$

where:

DI	=	Daily intake of robin (mg/kg-d)
Ci	=	Estimated concentration of COPC in soil invertebrates (mg/kg wet-wt)
Cs	=	Concentration of COPC in sediment (mg/kg)
FI	=	Food (*i.e.*, invertebrate) ingestion rate of robin (kg/d) (based on allometric equations from EPA,1993)
FD	=	Fraction of diet comprised of soil invertebrates (40%) (EPA, 1993).
IRs	=	Soil Ingestion rate of robin (kg/d) (based on data for American woodcock from EPA, 1993)
BW	=	Body weight (kg) (EPA, 1993)

4. SUMMARY OF DATA EVALUATED

A summary of the data evaluated is provided in this section. The complete analytical results (*i.e.*, including quality assurance information) are provided in the Final Data Task C Report (Battelle, 2000 and in Appendices B and C).

4.1 Sediment

As previously discussed, 24 sediment samples in 1999 and 6 sediment samples in 2001 were collected and analyzed for a variety of chemicals. The results of these analyses are presented in Tables 4-1, 4-2 and 4-3 as well as in the Task C data report (Battelle, 2000) and Appendix B.

4.1.1 Fisherville Pond

As indicated in Table 4-1a, all metals analyzed were detected at each of the Fisherville Pond sampling locations. In general, concentrations were similar to those reported in previous evaluations (McLaren/Hart, 1997). The highest concentrations were typically found in the South Pool of Fisherville Pond (*i.e.*, FP4A and FP11). Arsenic, nickel, lead, and zinc concentrations exceeded minimum sediment quality guidelines *(i.e.*, the TECs) at all locations. Cadmium and copper concentrations also exceeded these sediment quality guidelines at all locations except FP1 (located in the North Pool of Fisherville Pond). Chromium concentrations exceeded the screening concentrations at all locations except FP3A (1999) and mercury exceeded screening concentrations at all locations except FP1 and FP3A (1999; located in the North Pool). Silver concentrations were below the screening values at FP1 and FP1A, FP2, FP3A (2001), FP4, and FP5. No screening concentrations were available for selenium and tin. However, selenium concentrations were low, ranging from 1.3 to 4.5 mg/kg. Measured tin concentrations in Fisherville Pond ranged from a low of 4.2 mg/kg at FP1 to a high of 707 mg/kg at FP7.

The SEM/AVS ratio was below 1 at locations FP1A, FP2, FP5, FP8, FP10, and FP11 (Table 4-1a) as well as at locations HS1, HS2, HS3, and HS6 (Table 4-2), indicating limited bioavailability of cadmium, copper, nickel, lead, and zinc. The SEM/AVS ratio at the other locations ranged from 1.05 at FP3A to 9.65 at FP1 (Table 4-1a). At location FP4A, the AVS concentration was reported as not detected, therefore, an SEM/AVS ratio was not calculated. SEM/AVS ratios reported previously (McLaren/Hart, 1997) ranged from 2.2 to 387.7 with especially high ratios reported in the samples taken close to the dam.

PAHs were detected at each of the 15 sampling locations evaluated in Fisherville Pond over the course of two sampling periods. Concentrations were typically higher than those reported in earlier investigations (McLaren/Hart, 1997) although this may be due to the use of more sensitive analytical methods. In general, concentrations of these chemicals were highest at location FP9 in the Central Pool and FP4A in the South Pool. At eleven of the locations (FP2, FP3A [2001], FP4, FP4A, FP6, FP7, FP8, FP9, FP10, FP11, and FP12) all of the PAHs were detected at concentrations exceeding the sediment quality guidelines identified. In contrast, only two of the PAHs (*i.e.* pyrene and chrysene) at site FP3A (1999) exceeded their respective sediment quality guidelines. At FP1 all PAH concentrations exceeded these guidelines except acenaphthene, acenaphthylene, anthracene, fluorene, and naphthalene, while at FP1A all exceeded these guidelines except dibenz(a,h)anthracene. At site FP5, all PAH concentrations exceeded except fluorene and naphthalene. In many cases, measured PAH concentrations were more than an order of magnitude higher than the minimum sediment quality benchmark values.

An expanded list of parent PAH and alkylated PAHs compounds was also examined for toxicity as PAH mixtures. In accordance with EPA guidance (2000b), equilibrium sediment guideline (ESG) values were derived for 23 PAH compound in 1999 sediment samples and for 34 PAH compounds in 2001 sediment samples after normalizing for organic carbon. The PAH-specific ESGs were then summed per sampling station to derive equilibrium sediment guideline toxic units (ΣSGTU). ΣSGTUs greater than 1 are considered to pose an unacceptable risk (EPA, 2000b). Appendix D describes the calculation of the ΣSGTU in detail. The ΣSGTU for Fisherville Pond exceeded one at every station except FP1 and FP3A (1999) in the north pool and FP5 in the central pool.

All of the seventeen pesticides evaluated, with the exception of aldrin, heptachlor, heptachlor epoxide, lindane, and endrin, were detected in one or more of the Fisherville Pond locations. In general, the pesticides were detected at concentrations exceeding their respective sediment quality guidelines (*i.e.*, the TEC value). Pesticide concentrations were highest in the central and south pools.

For sediment samples collected in 1999, the total PCB concentration was estimated as twice the sum of the individual PCB congeners. This value was found to exceed the minimum sediment quality guideline (*i.e.*, the TEC) for total PCBs at all locations analyzed within Fisherville Pond. At several locations within the central pool (FP6 through FP12), concentrations were more than an order of magnitude greater than the TEC concentration. For samples collected in 2001, total PCB was defined as the sum of aroclor concentrations. However, none of the aroclors analyzed were detected.

TOC at Fisherville Pond ranged from approximately three percent at FP3A (2001) to 28 percent at FP3A (1999) (Table 4-3) in 1999. All locations in Fisherville Pond were predominantly composed of fines (*i.e.*, ranging from 67 percent to 99 percent) (Table 4-3). The Fisherville Pond locations with the greatest amounts of coarse material (*i.e.*, percent sand) were FP4 (32.09 percent), FP5 (27.54 percent), and FP2 (26.98 percent). The additional Fisherville Pond sampling locations in the southern pool near the dam had similar TOC and grain size. TOC in these 6 locations ranged from 9.9 percent to 18.7 percent with percent fines ranging from 87.8 percent to 99.8 percent.

4.1.2 Singing Pond

Each of the metals evaluated were detected at all six sampling locations in Singing Pond (Table 4-1b). Concentrations were typically similar to those reported for Fisherville Pond. The highest concentrations of each metal were consistently reported in the marsh area at either station SP5 or SP6. Concentrations of arsenic, cadmium, mercury, chromium, copper, and lead exceeded the minimum sediment guidelines (*i.e.*,TEC) in most of the samples. Detected concentrations of tin ranged from 9.01 mg/kg at SP1 to 238 mg/kg at SP4. Selenium measured 4.6 mg/kg at SP5 and 4.8 mg/kg at SP6. However, no screening concentration was available for either of these chemicals.

In evaluating the measured concentrations of these metals, it is important to note that the SEM/AVS ratio was below 1 at SP2, SP4, and SP5 indicating that cadmium, copper, nickel, lead, and zinc may not be bioavailable at these locations. The SEM/AVS ratio was greater than 1 at SP1 (11.4), SP3 (24.09) and SP6 (2.24).

PAHs were also evaluated in Singing Pond sediments. Each of the 16 parent PAH compounds were detected at each of the Singing Pond sampling locations, in both the main channel and marsh area. As noted with the metals, the highest concentrations of the PAHs were found in the marsh area, associated with locations SP5 or SP6. All PAH concentrations exceeded their respective minimum sediment guidelines, and in the marsh area (*i.e.*, SP5 and SP6) all of the PAHs also exceeded the Probable Effect Concentration (PEC) sediment guideline. As described for Fisherville Pond, ESGTUs were calculated (Appendix D). All were above 1, ranging from 2.12 at SP1 in the main channel to 38.51 in SP6 (in the marsh).

All pesticides evaluated, with the exception of aldrin, heptachlor, and endrin, were detected at one or more of the six sampling locations in Singing Pond. Lindane, hexachlorobenzene, and mirex were detected but measured below the respective minimum sediment guidelines at all locations in both the marsh and main channel. Concentrations of chlordane, dieldrin, heptachor epoxide, 4.4-DDE, 4,4-DDD, and 4,4-DDT were elevated (*i.e.*, exceeding sediment guidelines) in at least three of the six stations sampled. Pesticides were not detected above method detection limits in any stations in the most recent round of sampling (2001). Similarly, total PCBs (measured as congeners) exceeded the maximum guideline concentration (PEC) at all Singing Pond locations sampled in 1999, but were not detected as aroclors in 2001. The highest concentrations of all of these chemicals were reported at either SP2 or SP4.

TOC (Table 4-3) at the 1999 Singing Pond locations ranged from 0.5 percent at SP3 to 10.2 percent at SP4. Gravel was noted in the main channel at SP1 (9.23 percent) and SP3 (25.26 percent). Sediment type at these two locations showed very little fine material (1.17 percent and 3.42 percent, respectively), whereas sediment type at SP4 was composed of mostly fines (96.4 percent). Fine material was found predominantly in the marsh area at SP5 and SP6.

4.1.3 Lake Wildwood

All metals evaluated were detected at the sampling locations in Lake Wildwood (Table 4-1c). However, concentrations were generally lower than those reported in either Fisherville Pond or Singing Pond. Only cadmium, copper, lead and mercury exceeded the minimum sediment screening concentrations at one or more of the sampling stations. The SEM/AVS ratio was below 1 at all locations except RP2A, therefore, it is assumed that the metals present have only limited bioavailability.

As noted for Fisherville and Singing Ponds, most of the individual PAH compounds were detected at all locations. However, concentrations in Lake Wildwood were substantially lower, with only pyrene and chrysene exceeding minimum sediment guidelines at two of the sampling locations. The ESGTUs calculated for these samples were all below 1.

Dieldrin, trans-nonachlor, hexachlorobenze were detected but were below the minimum sediment guidelines at both locations. 4,4’DDE, 4,4’DDD and 4,4’ DDT exceeded the minimum sediment guidelines at RP1 and 4,4’DDE, and 4,4’DDD exceeded the minimum sediment guidelines at RP1, RP2, RP2A. Total PCBs exceeded the minimum guidelines at both RP1 and RP2 but were not detected in RP2A. Aldrin, heptachlor, heptachlor epoxide, lindane, mirex, and endrin were not detected at the Lake Wildwood locations.

The three Lake Wildwood stations were similar in sediment type, ranging from 91.8(RP1) to 100 percent fines (RP2A) (Table 4-3). The percent TOC was 11.6 percent and 17.6 percent for RP1 and RP2, respectively and 16.42 percent in RP2A.

4.2 Soil

As previously discussed, three soil samples were collected from the wet meadow regions of Fisherville Pond and analyzed for the same suite of chemicals evaluated in sediments. A summary of the results obtained for these locations is presented in Table 4-4. Sampling locations are depicted in Figure 1-1 (*i.e.*, WM1, WM2, WM3)

All metals were detected in the three wet meadow soil samples. In addition, all of the detected concentrations exceeded the minimum benchmark value (*i.e.*,TECs) and at many locations, the concentrations also exceeded the PECs. No screening concentration was available for tin and concentrations ranged from a low of 98.1 mg/kg at WM2 to a high of 175 mg/kg at WM1. In general, metal concentrations were highest at WM1.

All 16 PAH compounds were detected in the three soil samples at concentrations exceeding the minimum sediment quality benchmarks, often by an order of magnitude. Concentrations tended to be highest at WM3.

Of the pesticides evaluated, aldrin, heptachlor, lindane, mirex, and 2,4'DDE were not detected in any of the soil samples. Heptachlor epoxide was below the respective screening concentrations (*i.e.*, the TEC), while chlordane, dieldrin, 4,4' DDE, 4,4'DDD and 4'4'DDT exceeded their respective minimum screening concentrations at all three locations. In general, pesticide concentrations were highest at location WM1. Total PCBs exceeded the screening concentration at all of the wet meadow locations by more than an order of magnitude.

TOC at the Wet Meadow areas within Fisherville Pond ranged from approximately 7.9 percent at WM2 to 13.4 percent at WM1 (Table 4-3). Grain size results (Table 4-3) suggest that WM1 contains more fines (96.2 percent) and less sand (3.8%) than either WM2 or WM3 (82.5 percent fines and 69.9 percent fines, respectively).

4.3 Water

Porewater concentrations for metals, PAHs, pesticides, and PCBs are discussed in Section 4.3.1, while surface water concentrations of metals measured by the USACE in 2001 are presented in Section 4.3.2.

4.3.1 Porewater

Porewater was extracted from 18 of the sediment samples collected in 1999 and analyzed for metals. In addition, concentrations of PAHs, pesticides and PCBs were estimated based on measured sediment concentrations for samples collected in both 1999 and 2001 as previously discussed (Appendix E). All measured and estimated porewater concentrations are summarized in Table 4-5.

4.3.1.1 Fisherville Pond

With the exception of silver, which was only detected at FP7 and cadmium, which was not detected at FP1, FP5 or FP6, all of the 10 metals evaluated were detected at each of the Fisherville Pond sampling locations. However, concentrations were low, with only lead exceeding the chronic AWQC value at any location (*i.e.*, location FP3A). Tin concentrations ranged from 0.0209 ug/L at FP6 to 0.78 ug/L at FP7.

Using the EqP approach described in Section 2, PAHs were estimated in porewater at all sampling locations. Many of the estimated PAH concentrations exceeded the minimum estimated AWQC, particularly in the central and south pool area. Pyrene was the only PAH that exceeded criteria in the north pool. Fluoranthene exceeded its screening concentration at FP1A, FP3A, FP4, FP4A, FP6, and FP9 only. In general, concentrations in the samples collected in 2001 were higher than in those collected in 1999.

DDE and DDD exceeded minimum sediment screening guidelines at all stations except those located in the north pool (*i.e.* FP1 and FP3) and FP5 (DDE only). Chlordane exceeded minimum screening guidelines at three of the twelve stations (FP4, FP8, FP11) and dieldrin exceeded minimum guidelines at four of the twelve stations (FP4, FP10, FP11, FP12). Total PCBs only exceeded minimum guidelines at one station (FP11). All other pesticides measured were not detected.

4.3.1.2 Singing Pond

All metals except silver were detected at all Singing Pond locations (Table 4-5b); silver was not detected at any location. Only copper and nickel exceeded their chronic AWQC screening concentrations. Measured copper and nickel porewater concentrations at SP1 and SP3 exceeded their chronic AWQC concentrations. All other metals at all other locations were below their respective chronic AWQC concentrations. No screening criteria were available for tin. Tin concentrations in porewater ranged from 0.029 ug/L at SP4 to 0.257 ug/L at SP1.

Most estimated PAH concentrations exceeded minimum AWQC at all locations (Table 4-5b). In contrast, DDE, DDD and dieldrin exceeded minimum sediment screening criteria at SP2, SP3 and SP4. The only other pesticides exceeding sediment criteria were DDT at SP2, heptachlor epoxide at SP3, and Total PCBs at SP4.

4.3.1.3 Lake Wildwood

All of the ten metals except silver and cadmium were detected in porewater at locations RP1 and RP2 (Table 4-5c). Silver was not detected at either location and cadmium was only detected at RP2. However, all of the detected concentrations were below the chronic AWQC (Table 2-4). No screening criteria were available for tin, concentrations of which ranged from 0.03 ug/L at RP1 to 0.13 ug/L at RP2. Similarly, PAH and pesticide concentrations estimated at RP1 and RP2 were all below the derived porewater screening values (Table 4-5c). However, three PAHs (acenaphthene, benzo(a)anthracene, benzo(k)fluoranthene) and total PAHs exceeded the minimum AWQC at station RP2A.

4.3.2 Surface Water

Table 4-6 summarizes the draft surface water data received from USACE, specifically, monthly and annual averages for the metals in surface water samples from the North and Central Pools of Fisherville Pond. Total and dissolved concentrations are presented. Copper and lead are the only chemicals for which the monthly average concentration of total metals exceeded their respective AWQC (Table 2-4) during any of the sampling events. Copper exceeded only in the Central Pool (End QR, FP05, FP06) for all months except July and December, while lead exceeded at most stations for one or more months. Dissolved concentrations of these chemicals were lower, exceeding the AWQC only a few times. In general, concentrations of all metals tended to be highest in August and November.

4.4 Fish

As previously described, fish were collected from Fisherville, Singing, and Wildwood Ponds by NED and analyzed for metals and total PCBs. Both whole body and fillet concentrations were determined, with the exception of Singing Pond where whole body concentrations were not obtained. Table 4-7 provides a summary of the analytical results.

4.4.1 Fisherville Pond

Fish tissue was collected from five areas within Fisherville Pond, designated as the North Pool (defined the same as Fisherville Pond-North Pool), the South Pool (part of Fisherville Pond-South Pool), the East Pool (part of Fisherville Pond-South Pool), the Central Pool, the Central Pool-South and the Central Pool-Northeast (all grouped as Fisherville Pond-Central Pool). Whole body composites were comprised of bluegill sunfish while largemouth bass were used to derive the fillet concentrations.

Of the ten metals evaluated, only silver was not detected in whole body fish tissue from any of the pools sampled. Tin was only detected in whole body fish tissue from the Central South pool. In general, detected concentrations of individual metals did not vary greatly among the pools. Typically, concentrations of most of the metals were lowest in the North or East Pool areas and highest in the South Pool or Central South Pool. Mercury was unique in that the lowest concentration was observed in whole body tissue from the South Pool and highest concentrations were seen in the North and East Pools.

In general, fillet tissue concentrations were less than whole body tissue concentrations for all metals except mercury. Mercury, because it tends to accumulate in muscle tissue, was much greater in the fillet tissue than whole body tissue. Cadmium, lead, silver and tin were not detected in fillet tissue from any of the pools sampled and nickel was not detected at the North, South, and Central Northeast Pools. Like the whole body tissue concentrations, fillet tissue concentrations for individual chemicals did not vary greatly between pools. The lowest concentrations of arsenic, copper, and zinc were seen in fillets from the North pool and the highest concentrations were seen in fillets from the Central South pool. Mercury was lowest in fillets from the Central South Pool and greatest in fillets from the Central Northeast Pool. Total PCB concentrations in fillets were less than those in whole body fish and were lowest in fillets from the North Pool and highest from fillets in the Central South Pool.

4.4.2 Singing Pond

Fish tissue was collected from two pools within the Main Channel of Singing Pond. Brown bullhead fillets were collected from the upper pool and white sucker and brown bullhead fillets were collected from the lower pool. No small forage fish (*e.g.,* bluegill sunfish) were successfully collected, therefore, no whole body analyses were conducted.

Cadmium, silver, and tin were not detected in fillet tissue from Singing Pond. The concentrations of the other metals were similar in both pools. Total PCBs in fillets from the lower pool were, however, three times greater than concentrations associated with the upper pool.

4.4.3 Lake Wildwood

Fish tissue was collected from one location within the Lake Wildwood. Both whole body (bluegill sunfish) and fillet (largemouth bass) samples were collected and analyzed. Cadmium, lead, silver, and tin were not detected in whole body or fillet tissue. Concentrations of the other chemicals tended to be slightly greater in whole body tissue except for mercury, which was greater in fillet tissue. Total PCBs in fillets and whole body tissue were low (0.0067 mg/kg and 0.011 mg/kg, respectively).

4.5 Bulk-Sediment Toxicity Tests

The results of the bulk sediment toxicity bioassays are presented in Table 4-8.

4.5.1 Acute (10-d) *Hyallela azteca* Test

The acute *H. azteca* test was performed on all of the sediment samples collected in 1999. Percent survival of *H. azteca* in the test treatments ranged from 19 percent to 93 percent (Table 4-8). Statistical analyses were performed on the data; an ANOVA was run on each sample and then the Dunnett's test was used to compare the means of the test treatments to the mean of the native control sediment. Survival in the control sample was high (86 percent) and similar to that reported for Wildwood Pond (*i.e.*, 85 to 93 percent).

In Fisherville Pond, *H. azteca* survival was highest in the three northern-most samples, ranging from 80 percent in FP1 to 85 percent in FP5. Survival in sediments from the South Pool, in the vicinity of the dam was lower, ranging from 78 percent in FP10 to 80 percent in FP4. Central Pool samples had the lowest survival, ranging from 65 percent in FP9 to 76 percent in FP7 and FP8.

Survival in sediments from Singing Pond was lower than that for sediments from Fisherville Pond. Stations SP2 and SP4 were the only two stations that had statistically significantly lower survival relative to the Native Control (*i.e.*, 86 percent) with survival values of 38 percent and 19 percent, respectively. Survival at Stations SP1 and SP3 was higher, at 64 percent and 78 percent, respectively.

The growth data in the test treatments ranged from 0.01mg/day in SP4 to 0.0346 mg/day in FP6. The statistical analysis performed on the growth data indicates that none of the test treatments were statistically significantly different from the native control sediment treatment (0.0194 mg/day).

4.5.2 Acute (10-d) *Chironomous tentans* Test

The acute *C. tentans* test was conducted on all of the sediments collected in 2001 (Appendix C). Percent survival ranged from 1.3 percent at FP4 to 90 percent in FP3A (Table 4-8). With the exception of FP4, survival was lower in the Singing Pond samples (SP5=37.5%; SP6=17.5%). Statistical analysis of the data set was conducted using a t-test to compare to reference (RP2A). Survival was significantly reduced relative to the Lake Wildwood control sample at FP4, SP5 and SP6, while growth was significantly reduced at FP1A, FP3A, SP6 and SP5.

4.5.3 Chronic (42-d) *H. azteca* Test

The chronic *H. azteca* test was conducted on all of the sediments collected in 2001 (Appendix C). Test endpoints included survival at 28 days, and growth, survival, and number of young per female at 42 days. Statistical analysis of the data set was conducted using a t-test to compare to reference. A significant reduction ($p<0.05$) in survival was observed in SP5, SP6, FP1A, and FP4 after 28 days (Table 4-8). In fact, FP4 had over 50 percent mortality within the first 24 hours of the test and severe sediment avoidance was noted at the initiation of the test, indicating acute toxicity. SP4 and SP5 were also acutely toxic, with greater than 25 percent mortality after the first 48 hours. Growth was significantly reduced relative to Lake Wildwood in FP1A and SP5, while reproduction (*i.e.*, number of young per female) was significantly reduced in all but FP3A.

4.6 Benthic Community Evaluation

A summary of the benthic species identified at each sampling location is presented in Table 4-9. Species richness (*i.e.*, total number of taxa) ranged from 5 to 19 at Fisherville Pond, 6 to 14 at Singing Pond and 5 to 10 at Lake Wildwood. Species abundance (*i.e.*, total number of individuals) ranged from 13 to 318 at Fisherville Pond, 27 to 757 at Singing Pond and 146 at RP1 to 226 at RP2 in the Lake Wildwood.

At the majority of Fisherville Ponds sites (*i.e.*, FP4, FP5, FP7, FP8, FP9, FP10, FP11, FP12) oligochaetes (particularly tubificid worms) and chironomid worms (within the order *Insecta*) were the most abundant organisms observed (*i.e.*, 53.3 percent oligochaetes at FP8 to 91.9 percent oligochaetes at FP9). At FP1, FP2, FP3A and FP6 the most abundant organisms were chironimid worms with tubificid worms second most abundant. Both tubificid and chironimid worms are general indicators of poor sediment/water quality. Similarly, Singing Pond samples also contained large proportions of tubificid worms, ranging from 68.6 percent to 92 percent of the total number of species present. In RP1 and RP2 from Lake Wildwood, chironimid worms (within the order *Insecta*) comprised 99.3 and 68.1 percent of the species identified.

Table 4-10 summaries diversity indices calculated based on the data collected. Benthic community data are associated with 1999 data only; samples collected in 2001 were archived. As a result, there is only one sample (SP4) representing the Singing Pond Marsh area. Each index calculated measures a slightly different aspect of species diversity. For example, the Pielou Evenness Index and the Equitability Index both indicate how evenly the numbers of individuals are distributed among the species present. These indices run from 0 to approximately 1, with increasing 'evenness' as the value increases. These values vary widely among the Fisherville Pond sites. Both the Equitability Index and Pielou Evenness Index suggest that FP4 and FP9 are less even (*i.e.*, have an unequal distribution of individuals among species) while FP1 and FP6 are more even (have a more equal distribution of individuals among species). Within Singing Pond, sites SP1, SP3 and SP4 are similar with respect to evenness, but SP2 is very low. The Lake Wildwood sites also are associated with relatively low values for evenness.

The Margalef Richness Index measures the number of species present (*i.e.*, species richness). Larger values indicate more species. Margalef values at Fisherville Pond sites range from a low of 1.15 at FP12 to a high of 4.00 at FP8 suggesting that FP8 contains more species and is thus more diverse than FP12. Margalef values at Singing Pond sites range from a low of 1.00 at SP3 to a high of 2.73 at SP4, while Margalef values at the Wildwood Pond sites are low (0.8 at RP1 and 1.66 at RP2).

Both Simpson's Diversity Index and Shannon-Weiner Index measure a combination of richness and diversity and are often difficult to interpret. In general larger values indicate more diversity at the site. Using the Shannon-Weiner Index for the Fisherville Pond sites, it appears FP8 and FP6 are the most diverse (Shannon-Weiner index = 2.29 and 2.28, respectively) and FP9 the least diverse (Shannon-Weiner Index = 0.54). Simpson's Diversity index suggests that FP6 is more diverse (Simpson's Index = 8.14) and FP9 is the least diverse (Simpson's Index = 1.29). Both the Shannon-Weiner Index and Simpson's Diversity Index suggest that SP2 is the least diverse among the Singing Pond Sites (index = 0.86 and 1.63, respectively) and that SP4 is the most diverse (Shannon-Weiner Index = 1.84 and Simpson's Diversity = 5.16). Of the two reference locations, both Shannon Weiner and Simpson's Diversity suggest that RP2 is more diverse than RP1.

In addition to this quantitative evaluation, a qualitative evaluation of mouthpart deformities was conducted based on the voucher specimens created for this investigation. Approximately 118 individuals, representing at least one individual from each of the identified species, were examined. No evidence of mouthpart deformities was noted.

4.7 Fish Community Evaluation

As previously discussed, fish collection efforts in the study area conducted in 1992, 1996 and 1999 were considered to evaluate the health of the warmwater fish community. Table 3-4 summarizes the fish collected during each of the three efforts. Table 4-11 summarizes the results of the most recent evaluation (*i.e.*, 1999), presenting the total biomass calculated for each area evaluated, while Table 4-12 presents the condition factors calculated for each area.

4.7.1 Fisherville Pond

Based on the available data (Table 3-4, Table 4-11, Table 4-12) Fisherville Pond supports a diverse warmwater fish community. The most common species collected by gill net in the 1992 MADFW studies were white sucker, bluegill, golden shiner, and yellow perch, while bluegill sunfish, pumpkinseed sunfish, and largemouth bass were found most frequently in 1999 (see Table 3-4). Bluegill appears to be the predominant forage species, with juvenile bluegill the predominant species found in backpack and boat electrofishing samples, beach seine samples, and minnow traps. Golden shiner and pumpkinseed sunfish were also common.

Largemouth bass and yellow perch appear to be the predominant sport fish. Several very large (> 40 cm) and robust bass were collected by USACE in 1999. Juvenile largemouth bass were also common, indicating that the pond supports a self-sustaining bass fishery. Adult white sucker was the predominant bottom fish caught in gill nets by the 1992 MADFW and 1996 USACE studies. Spawning of this species likely occurs in the Blackstone and Quinsigamond River, however, although the pond may provide habitat for juvenile white sucker, none were noted in the MADFW or USACE studies.

Given the habitat type, bullhead (*Ameiurus sp*.) appears to be underrepresented in these collection efforts. Gillnetting by the MADFW in 1992 and USACE in 1996 resulted in the collection of only 14 adult yellow bullhead and no juvenile bullhead were collected in USACE beach seines in 1996 or in minnow traps set by USACE in 1999. In addition, a single adult brown bullhead collected in 1999 (from the South Pool) was severely deformed and discolored. Similar deformities were noted by the MADFW in 1992 (McLaughlin, 1999 pers. commun.) and by anglers who frequent the pond.

The average condition factor of largemouth bass from Fishersville Pond was 1.35, with values for individual fish ranging from 1.05 to 2.5.1. Average condition factors for bluegill sunfish were about 1.75 (Table 4-12). These values are within the range of values for other New England reservoirs sampled by USACE.

4.7.2 Singing Pond

Six species of fish were collected from Singing Pond, with brown bullhead and white sucker predominating (Table 3-4, Table 4-11). Two of the 11 brown bullhead collected had deformed fins or barbells. Several other bullheads and a largemouth bass had badly eroded caudal fins. The average condition factor, based on largemouth bass was 1.54 (Table 4-12).

4.7.3 Lake Wildwood

Seven fish species were collected from Lake Wildwood, with bluegill, largemouth bass, and yellow perch comprising the majority of the sample (Table 3-4, Table 4-11). Condition factors for bluegill and largemouth bass were lower than at Fisherville Pond and at the low end of the range reported by USACE for other New England reservoirs (Table 4-12).

4.8 Toxicity Identification Evaluations (TIE)

A detailed description of the TIE results is presented in the Final Task C Data Report (Battelle, 2000). In general, good agreement was found between the baseline porewater toxicity tests and the bulk sediment toxicity bioassays. Specifically, all samples exhibiting evidence of toxicity in the porewater amphipod baseline test (SP1, SP2, SP4) were associated with at least a 20 percent reduction in survival relative to the control in the sediment bioassay. Similarly, in the fish baseline test, FP8, FP9, and SP2 were found to be slightly toxic, all of which were associated with at least some reduction in survival in the sediment bioassay. One location (FP1) was highly toxic in porewater while showing no evidence of toxicity in sediment. The toxicity in the porewater test may be attributable to high ammonia levels in the porewater sample (SAIC, 2000). In fact, subsequent TIE manipulations on all porewater samples exhibiting baseline toxicity indicates that the majority of the observed response is likely due to ammonia.

4.9 Risks Estimated for Wildlife

Hazard quotients calculated for the mallard and river otter are summarized in Table 4-13. In general, HQs for the mallard were low, all falling below 10 (Table 4-13). Hazard quotients for all chemicals were below one in Fisherville Pond-North Pool and in Lake Wildwood and hazard quotients for organic chemicals were below 1 in all areas evaluated. Chromium, lead, and mercury were associated with elevated HQs for the mallard in Fisherville Pond-South Pool, Fisherville Pond-Central Pool, and both areas of Singing Pond. For the otter, the primary risk drivers appeared to be arsenic, mercury, and PCBs (Table 4-13). Hazard quotients were highest at Singing Pond-Marsh and Fisherville Pond-South Pool.

5. RISK CHARACTERIZATION

Using the data described in Section 4 and the criteria in Table 5-1, a measure of risk (*i.e.*, high, medium, low) was derived for each area for each of the measurement endpoints, based on the results for each data type (*e.g.*, sediment concentration, percent survival, etc.) as described in Section 4. Each measurement endpoint was also assigned a relative weight, reflecting the overall confidence in the measurement endpoint in terms of the strength of association with the assessment endpoint and the quality of the data evaluated as discussed in Section 2.3.

5.1 Assessment Endpoint 1: Health of the Benthic Invertebrate Community

As described in Section 2.3.1, the health of the benthic invertebrate community was evaluated through three separate lines of evidence including bulk sediment chemistry data, the results of acute and chronic toxicity tests, and a benthic invertebrate community analysis. Three measures of sediment chemistry were evaluated: comparison of individual and combined chemical concentrations to selected SQG; consideration of effects associated with metals mixtures; and consideration of effects associated with PAH mixtures. Although the sediment data are believed to be of good quality, the relative weight assigned to each of these measures was only medium because of the uncertainties associated with predicting toxicity based on bulk sediment chemistry.

5.1.1 Measurement Endpoint 1a: Comparison to Bulk Sediment Quality Guidelines

Measurement Endpoint 1a focused on comparison of individual and combined chemical concentrations to the bulk sediment quality guidelines described in Table 2-3, specifically the PEC and PEC-Q (MacDonald *et al.*, 2000; Ingersoll *et al.*, 2000). Measurement Endpoint 1a was scored as follows:

- Low: Areas where the average PEC-Q was less than 1, no more than three COPCs exceeded their respective PEC and no COPC was more than two times greater than the PEC.
- Medium: Areas where the average PEC-Q was greater than 1 but less than 1.5, no more than four to six COPCs exceeded their respective PEC, and no COPC was more than five times greater than the PEC.
- High: Areas where the average PEC-Q was greater than 1.5, or more than six COPCs exceeded their respective PEC or at least one COPC was greater than five times the PEC.

Table 5-2 provides a summary of these comparisons. As described in Section 4.1, sediment concentrations of most chemicals, particularly the metals and PAHs were elevated throughout the study area, with the exception of Fisherville Pond-North Pool and Lake Wildwood. More than six chemicals with average concentrations exceeding their respective PECs were identified in Fisherville Pond-Central Pool, Fisherville Pond-South Pool, and Singing Pond-Marsh Area. In each of these areas, there were also several chemicals for which the average concentration was more than five times greater than the PEC and the PEC-Q was greater than 1.5. Therefore, these three areas were designated as having a high potential risk. The PEC-Q for Singing Pond-Main Channel was 1.5 and only five chemicals exceeded their respective PEC values, therefore, it was designated as medium. Fisherville Pond-North Pool and Lake Wildwood were determined to have a low probability for effect, with PEC-Q of less than 1 and no chemicals with concentrations exceeding their PEC values.

5.1.2 Measurement Endpoint 1b: Evaluation of Metals Mixtures

To evaluate the potential bioavailability of metals mixtures in sediments, Measurement Endpoint 1b considered the presence of AVS. As discussed in Section 2.3.1.1, in the presence of AVS in sediments, certain metals, primarily copper, cadmium, lead, nickel, zinc (Ankley, 1996; Ankley *et al.*, 1996) precipitate as their respective metal sulfides, which are not bioavailable (Di Toro *et al.*, 1991). Thus, if the molar concentration of AVS in sediments is higher than the sum of the molar concentration of simultaneously extracted metals (SEM; *i.e.*, the sum of the molar concentrations of these metals in the 1 N HCl extract), all of these metals are assumed to be in non-bioavailable forms in the sediments. This relationship has been used by EPA (ND) to develop sediment quality guidelines for metals mixtures. In accordance with this guidance, each area was scored based on the AVS/SEM ratio as follows:

- Low: Areas where the AVS/SEM ratio was below one;
- Medium: Areas where the AVS/SEM ratio was greater than one but below two; and
- High: Areas where the AVS/SEM ratio was greater than two.

As summarized in Table 5-3, estimated AVS/SEM ratios ranged from 0.88 in Singing Pond-Marsh Area to 11.95 in Singing Pond-Main Channel. Based on this evaluation, metals in Fisherville Pond-North Pool and Singing Pond-Main Channel were determined to be the most bioavailable, ranking high based on the criteria presented in Table 5-1. Fisherville Pond-Central Pool and Lake Wildwood were scored as medium, or moderately bioavailable, while Fisherville Pond-South Pool and Singing Pond-Marsh Area were ranked as low.

5.1.3 Measurement Endpoint 1c: Evaluation of PAH Mixtures

In addition to the metals mixtures, PAH mixtures were evaluated (Measurement Endpoint 1c). Using these data, the average ESGTU values for each area were ranked as follows:

- Low: Areas where the ESGTU was below one;
- Medium: Areas where the ESGTU was above one but less than ten; and
- High: Areas where the ESGTU was greater than ten.

Based on these scoring criteria, Fisherville Pond-North Pool and Lake Wildwood were considered to have a low potential for PAH toxicity (Table 5-3). In general, PAH concentrations in these areas were below identified sediment quality guidelines, and the ESGTU were below one, resulting in a ranking of low. In contrast, PAH concentrations in Singing Pond-Marsh Area were elevated, with most individual PAHs exceeding their respective PEC values and an ESGTU of 15.32, resulting in a rank of high for this area. The ESGTU for the remaining areas ranged from 3.18 to 6.47, resulting in a rank of medium for all three.

5.1.4 Measurement Endpoint 1d: Bulk Sediment Toxicity Tests

In contrast to predicted toxicity based on comparisons to SQG, bulk sediment toxicity tests provide a direct measure of the effects of COPC on benthic invertebrates. As a result, Measurement Endpoint 1d (*i.e.*, bulk sediment toxicity tests) was given a high relative weight, assuming a strong strength of association between toxicity test results and actual in situ toxicity of sediments. As described in Section 3.4, three bulk sediment toxicity tests were conducted using sediments from the study area, including acute (10-d) tests using the amphipod *Hyallela azteca* and the chironomid *Chironomus tentans* and a chronic (42-d) test using *H. azteca*. An average percent survival was derived for each test, for each of the designated areas. The results for each test were compared to the following criteria:

- Low: Areas where the average survival was greater than 80 percent;
- Medium: Areas where the average survival was less than 80 percent but greater than 50 percent;
- High: Areas where average survival was less than 50 percent.

The criteria for survival were based in part on the method requirements. For example, the methods for the 10-day bulk sediment bioassay (ASTM 1994) require 80 percent survival in the control sample for test acceptability. Fifty percent survival was selected as the criteria for determining high effects because the LD_{50} (*i.e.*, concentration at which 50 percent of the organisms die) is commonly used as a measure of acute effects. Based on evaluation of the results of each test, an overall score for the designated area was determined.

Table 5-4 provides a summary of the toxicity test results and the comparison to the scoring criteria. As described in Section 4.5, the results of the three toxicity test results were generally consistent. For example, average survival for Fisherville Pond-Central Pool ranged from 73.15 percent for the chronic *H. azteca* test to 86.9 percent for the acute *C. tentans* test. Survival in Lake Wildwood ranged from 83.8 percent in the chronic *H. azteca* to 89 percent for the acute *H. azteca* exposures. Average results for Fisherville Pond-South Pool and Singing Pond-Marsh Area were not as similar, however, all tests indicated some level of toxicity.

A toxicity identification evaluation (TIE) test was conducted concurrently with the 1999 acute *H. azteca* bioassay. A detailed description of the TIE results are presented in SAIC (2000). In general, good agreement was found between the baseline porewater toxicity tests and the acute *H. azteca* toxicity test. Specifically, all samples exhibiting evidence of toxicity in the porewater amphipod baseline test for the TIE (SP1, SP2, SP4) were associated with at least a 20 percent reduction in survival relative to the control in the sediment bioassay. Similarly, in the fish baseline test for the TIE, FP8, FP9 and SP2 were found to be slightly toxic, all of which were associated with at least some reduction in survival in the sediment bioassay. Porewater from one location (FP1) was highly toxic in porewater while showing no evidence of toxicity in sediment, however, the toxicity in the porewater test may be attributable to high ammonia levels in the porewater sample (SAIC, 2000). In fact, subsequent TIE manipulations on all porewater samples exhibiting baseline toxicity indicated that the majority of the observed responses were likely due to ammonia.

Based on comparison to the selected criteria, Fisherville Pond-North Pool and Lake Wildwood were scored as low while Fisherville-Central Pool and Singing Pond-Main Channel were scored as medium. Fisherville Pond-South Pool and Singing Pond-Marsh Area were both scored as high. These rankings correspond well with observations made during the testing. For example, station FP4A (located in Fisherville Pond-South Pool) had greater than 50 percent mortality within the first 24 hours of the test, and the exposed organisms exhibited sediment avoidance. Concentrations of lead and copper were very high at this station. Similarly, SP4 and SP5, both located within the Singing Pond-Marsh Area, were noted to be acutely toxic with 25 to 50 percent mortality within the first 48 hours of the test. In all of these samples, growth and reproductive success were reduced relative to Lake Wildwood.

5.1.5 Measurement Endpoint 1e: Comparison of Porewater Quality Data to AWQC

To evaluate this line of evidence (Measurement Endpoint 1e), concentrations of sediment-associated chemicals in the water column were estimated using measured and predicted concentrations of COPC in porewater, as described in Section 3.2.1 and Appendix E. Because much of the data is estimated, this measurement endpoint was given a relative weight of low.

The following criteria were used to rank the designated areas:

- Low: Areas where the average predicted concentration of no more than six COPCs exceeded their respective AWQC and no COPC was more than five times greater than the AWQC.
- Medium: Areas where the average predicted concentration of six to eight COPCs exceeded their respective AWQC and no COPC was more than ten times greater than the AWQC
- High: Areas where the average predicted concentration of more than eight COPCs exceeded their respective AWQC or at least one COPC was ten times greater than the AWQC

Table 5-5 presents a summary of the average measured and predicted porewater concentrations for each area. In general, measured metal concentrations were below the AWQC values. Copper and nickel in Singing Pond-Main Channel were the only metals that exceeded their respective AWQC. In contrast, predicted total PAH and pesticide concentrations were elevated in every area except Fisherville Pond-North Pool and Lake Wildwood. Based on comparison to the designated criteria, Fisherville Pond-North Pool and Lake Wildwood were determined to be low, while Singing Pond-Marsh Area and Singing Pond-Main Channel were ranked as high. Fisherville Pond-Central Pool and Fisherville Pond-South Pool were both scored as medium.

5.1.6 Measurement Endpoint 1f: Benthic Invertebrate Community Analyses

As described in Section 3.5, sediment samples were also collected for the purpose of characterizing the structure of the existing benthic community within the study area (Measurement Endpoint 1f). Based on these data, a variety of diversity indices were calculated (Table 4-10), each of which provides a different measure of the community (*e.g.,* richness, abundance, evenness, *etc*). It is important to note that while such community analyses provide useful information regarding the presence or absence of species, it is difficult to correlate these results with a causative factor. There are numerous physical and chemical factors that may affect the structure of the benthic community, including grain size, TOC, and vegetation. Due to these confounding factors, this measurement endpoint was given a low relative weight.

As described in Section 4.6, the most abundant species in most samples were tubificid worms and chironomids, both of which can be indicative of poor sediment/water quality. The stations varied considerably with regard to species evenness and diversity, however, in general, Fisherville Pond-North Pool appeared to be associated with the healthiest benthic community. Unexpectedly, the benthic community at Lake Wildwood was impaired, with very low diversity.

For the purpose of this assessment, the magnitude and effect criteria was based on the Shannon-Weiner Diversity Index because that is the only index for which information is available in the literature regarding the range of values associated with impacted versus unimpacted aquatic systems. Stainken (1984) reports that Shannon-Weiner Diversity Index values below 2 are considered to be indicative of pollution stress. Therefore, areas with an average Shannon-Weiner value of 2 or higher were ranked as low. Areas with an average Shannon-Weiner value between 1 and 2 were considered to be medium, while those with an average Shannon-Weiner value of less than 1 were scored as High. Based on these criteria, all the designated areas within the study area were ranked as medium, with the exception of Lake Wildwood, which was ranked as high (Table 5-6).

Unlike the toxicity test results, these data do not correspond well with the predicted toxicity of the sediments based on the measured sediment chemistry. Results for Lake Wildwood are particularly noticeable, given the low observed toxicity and bulk chemistry values. However, it is important to note that the structure of the benthic community can be highly influenced by factors other than the presence of contaminants. For example, variations in grain size and bottom substrate can have a significant impact on the presence or absence of benthic species. Observations during sample collection activities in 1999 indicated that the bottom substrate at Lake Wildwood, particularly at station RP1, was predominantly fanwort. It is likely that the results obtained for this area were highly influenced by the presence of this vegetation.

5.2 Assessment Endpoint 2: Health of the Fish Community

The health of the fish community was evaluated through three lines of evidence including surface water quality, fish tissue residues in comparison to literature-based effects levels, and the results of a fish community assessment. Section 2.3.2 describes the rationale behind each of the measurement endpoints.

5.2.1 Measurement Endpoint 2a: Comparison of Surface Water Quality Data to AWQC

As discussed in Section 4.3.2, monthly and annual average concentrations of six metals measured in surface water collected from Fisherville Pond-North Pool and Fisherville Pond-Central Pool were evaluated. For the purpose of this evaluation, annual averages of both surface and bottom samples were combined to provide an average concentration for each of the designated areas. The data used for this evaluation were considered to be of acceptable quality because they are draft, and the overall weight for this measurement endpoint was low.

The following criteria were used to rank the designated areas:

- Low: Areas where the average concentration of no more than one COPCs exceeds their respective AWQC and no COPC was more than five times greater than the AWQC.
- Medium: Areas where the average predicted concentration of two to four COPCs exceeded their respective AWQC and no COPC was more than ten times greater than the AWQC
- High: Areas where the average predicted concentration of more than four COPCs exceeded their respective AWQC or at least one COPC was ten times greater than the AWQC

Table 5-7 presents a summary of the average surface water concentrations for each area. Values in bold exceed the AWQC value for that chemical (Table 2-4). Similar to the results reported for metals in porewater, concentrations were typically below the AWQC values. In fact, lead was the only chemical for which the average concentration exceeded. Based on this evaluation, both areas were ranked as low for this measurement endpoint.

5.2.2 Measurement Endpoint 2b: Evaluation of Fish Tissue Residues

As described in Section 3.3.1, tissue residues were measured in fish collected from throughout the study area. Both whole body and fillet residues were collected, however, for the purpose of this analysis only whole body concentrations were considered. Only fillet data were collected from Singing Pond, therefore, these data were converted to estimated whole body concentrations using whole body to fillet ratios developed by Bevelhimer *et al.* (1999). This line of evidence (Measurement Endpoint 2b) assumes that there is a correlation between the body burden of contaminants resulting from bioaccumulation and the potential for adverse effects in fish.

In general, the fish sampling strategy correlated with the six designated areas. Fish designated as having been collected from the east pool of Fisherville Pond were included with data from Fisherville Pond-South Pool. For each designated area, average fish tissue concentrations were calculated and compared to a range of literature-based effects values. Specifically, concentrations of metals and PCBs in fish tissue from Fisherville Pond, Singing Pond, and Wildwood Pond were compared to effects concentrations developed from data reported in the Environmental Residue and Effects Database (ERED). The ERED

(USACE, 2000) comprises a summary of available data on tissue concentrations associated with adverse effects in various aquatic species. For the purpose of this evaluation, a low effects criteria (Fish effect range-low or FER-L) was defined as the 10^{th} percentile of all whole body concentrations reported in ERED for freshwater fish species that were associated with an adverse effect. The probable effects criteria (Fish effect range-median or FER-M) was defined as the median (50^{th} percentile) of these same data. Using these criteria, each area was scored as follows:

- Low: Areas where the average whole body tissue concentrations of all COPCs were less than the FER-L.
- Medium: Areas where the average whole body tissue concentrations of one or more COPCs exceed the FER-L, but all are below the FER-M.
- High: Areas where the average whole body tissue concentrations of at least one COPC exceeds the FER-M.

In general, concentrations within Fisherville Pond and Singing Pond were elevated relative to Lake Wildwood, with the exception of tissue collected from Fisherville Pond-North Pool. For example, lead was two orders of magnitude higher in Singing Pond than in Lake Wildwood, while at Fisherville copper and PCBs were elevated. However, as depicted in Table 5-8, when compared to the scoring criteria, total PCBs in Fisherville Pond-Central Pool and Fisherville Pond-South Pool were the only COPC that exceeded the FER-L. None of the tissue concentrations exceeded the FER-M. Based on these results, Fisherville Pond-North Pool, Singing Pond, and Lake Wildwood were all ranked as low, while Fisherville Pond-Central Pool and Fisherville Pond-South Pool were ranked as medium.

5.2.3 Measurement Endpoint 2c: Fish Community Assessment

To evaluate risk to fish based on the community assessment, three factors were considered: 1) community diversity and productivity, 2) condition of individual fish (condition factors), and 3) prevalence of external abnormalities. Ranking criteria were as follows:

- Low: Species diversity or productivity as expected for impoundments with similar physical and biological habitat. Condition factors for dominant species show no evidence of impairment that cannot be explained by physical or biological habitat quality. External abnormalities (growth deformities) absent or very rare (< 1%).
- Medium: At least one of the following apply: Community diversity or productivity less than expected. Condition factors for dominant species indicate growth impairment that cannot be explained by physical or biological habitat quality. External abnormalities rare (1 - 5 %).
- High: At least one of the following applies: Community diversity or productivity much less than expected. Condition factors for dominant species suggest severe growth impairment that cannot be explained by physical or biological habitat quality. Severe external abnormalities common (> 5 %).

5.2.3.1 Fisherville Pond

Studies conducted by the MADFW and USACE in the 1990's indicate that Fisherville Pond supports a productive warmwater fishery when water levels are at normal pool levels. For example, the data suggest that the Pond supports productive bluegill and largemouth bass populations with no sign of significant impairment. For both species, population densities were high and condition factors were within range of values reported for other New England Reservoirs. The number of fish species present (community richness) was typical of other small to medium sized eastern Massachusetts reservoirs.

However, there is evidence that the productivity of the bottom fish community is impaired. White sucker, brown or yellow bullhead, and carp are the primary bottom fish in most Massachusetts ponds and reservoirs. Although these four species occur at Fisherville Pond, the population density of bullhead appears to be reduced. For example, the 1996 and 1999 USACE studies captured fewer brown and yellow bullhead than expected given the level of effort and number of other fish captured. During more than two hours of boat electrofishing in shallow water habitat in 1999, only one yellow bullhead was caught out of a total of more than 700 fish. Although boat electrofishing is not the optimal method to capture bullhead, these data appear to suggest that Fisherville Pond is unusually unproductive for this species. Similarly, in 1996, just one yellow bullhead was caught in four gillnet sets, during which 50 other fish were caught. Numerous white sucker were in the 1992 MADFW and 1996 USACE studies, however, they are a highly mobile species, and those captured from Fisherville Pond may not have been resident fish. Bullhead are less mobile than white sucker and are, therefore, considered a much better indicator of benthic habitat quality. The assumption that the reduced population may be due to habitat quality is supported by the observation that the other bottom fish present, carp, is non-native and highly tolerant of degraded habitat

The single adult brown bullhead collected in 1999 (from Fisherville Pond-South Pool) was severely deformed and discolored. Similar deformities in bullhead from Fisherville Pond were also noted by the MADFW in 1992 (McLaughlin, 1999 pers. commun.) and by anglers who frequent the pond. One angler reported that a majority of bullhead captured from Fisherville Pond are severely deformed and discolored. Although anecdotal, this information is credible and is considered in the risk evaluation. Studies elsewhere have correlated poor sediment quality with increased incidence of tumors and other abnormalities in bottom fish.

Based on the available information, the ecological risk to the fish community at Fisherville Pond-South Pool and Fisherville Pond-Central Pool is ranked high due to possible population impacts to bullhead populations and the prevalence of external abnormalities among those species. Risk at Fisherville Pond-North Pool is ranked as low since sediment quality in this area is good, and adverse effects on the resident bullhead are less likely.

5.2.3.2 Singing Pond

USACE studies at Singing Pond-Main Channel in 1999 indicate that the fish community is moderately productive. Although impounded by a dam, the areas sampled are more typical of a riverine than a lotic habitat. Bullheads appear abundant, accounting for 50 percent of the fish caught. The fish diversity and abundance is relatively low, however, this is likely limited by physical habitat quality (*e.g.,* lack of cover). Habitat type also explains the low numbers of sunfish and largemouth bass observed. Several of the bullhead captured from the site had fin or barbell abnormalities. Based on the frequency of abnormalities in bullhead, the ecological risk in Singing Pond-Main Channel is ranked as medium. Data were not available for the Singing Pond-Marsh Area.

5.2.3.3 Lake Wildwood

Studies conducted by USACE in 1999 indicate that Lake Wildwood supports a productive warmwater fishery. The pond supports productive bluegill and largemouth bass populations. For both species condition factors were lower than at Fisherville Pond, but within the range of values reported for other New England Reservoirs. Lower than normal condition factors may reflect impairment caused by a dense growth of fanwort in the Pond (*i.e.,* reduced feeding efficiency or periodic low dissolved oxygen stress). The number of fish species present (*i.e.*, community richness) was also lower than at Fisherville Pond. This may reflect habitat impairment due to fanwort or the difficulty of sampling the pond because of the fanwort. None of the fish collected had external abnormalities.

Based on this evaluation and the fact that there is no indication of possible ecological risk due to contaminants at Lake Wildwood, the site risk is ranked as "low" for this measurement endpoint. Possible impairment of the fish community as indicated by low species richness and low condition factors for largemouth bass and bluegill sunfish is likely related to dense growth of fanwort.

5.3 Assessment Endpoint 3: Evaluation of Wildlife Exposures

To evaluate risks to upper trophic level species, doses associated with exposures to contaminated sediments were estimated for two wildlife receptor species, the river otter and the mallard duck. A summary of the methods and assumptions used are presented in Section 3.7 and in Appendix F. Briefly, the species were assumed to be exposed though consumption of contaminated prey and through incidental ingestion of sediment, as outlined in the conceptual site models (Figure 2-1 and Figure 2-2). For the purpose of calculating dose estimates and resulting HQs for each area using the dose and trophic transfer equations described in Section 3.7, average sediment and fish tissue concentrations were used.

The following criteria were used to score each area for potential risk:

- Low: Areas where HQs for all COPC were less than 1;
- Medium: Areas where HQs for no more than 3 COPC were greater than 1 and all HQs were less than 10;
- High: Areas where HQs for more than 3 COPC were greater than 1 or at least 1 HQ was greater than 10.

Based on comparison to these criteria Fisherville Pond-North Pool and Lake Wildwood were scored as medium for the otter and low for the mallard (Table 5-9), however, each of these areas was given an overall score of low, because the hazard quotients for the otter, although above one for two chemicals, were all below five. Fisherville Pond-Central Pool and Singing Pond-Main Channel were scored as medium for both the mallard and the otter and therefore ranked as medium overall. Fisherville Pond-South Pool and Singing Pond-Marsh Area were both scored as high based on rankings of high for the otter and medium for the mallard.

5.4 Summary

Table 5-10 presents a summary of the scorings by area for each line of evidence. In general, risks at Fisherville Pond-North Pool and Lake Wildwood are low. For Fisherville Pond-North Pool, the benthic community analysis was scored as medium, however, this measurement endpoint was assigned a low relative weight, indicating low confidence in this line of evidence. Measurement Endpoint 1b (metals mixtures) was scored as high, however, toxicity was low in the associated bioassays. In Lake Wildwood, all lines of evidence except the benthic community analysis and the benthic community analysis were scored as low.

Fisherville Pond-Central Pool and Singing Pond-Main Channel were both scored as medium. In each of these areas, six of the ten guidelines were scored as medium. In Fisherville Pond-Central Pool, Measurement Endpoint 1a, 1e, and 2c were each ranked as high. However, each of these lines of evidence was given a low relative weight due to the uncertainty associated with the data or the strength of association with the assessment endpoint. Results for Singing Pond-Main Channel were similar.

In Singing Pond-Marsh Area, all lines of evidence with the exception of the benthic community analysis (Measurement Endpoint 1f) and the metals mixtures (Measurement Endpoint 1b) were scored as high.

Confidence in the community assessments is low, and it is important to note that significant toxicity was observed in the bioassays, despite the apparent lack of metal bioavailability. Therefore, this area was ranked as high. Fisherville Pond-South Pool was also scored as high, with five lines of evidence scored as high. Station FP4, which indicated acute toxicity, is located in this area, and overall the toxicity measurement endpoint was ranked as high. The evaluation of risks to upper trophic level species also indicated high risks in this area.

6. RELATIVE RISKS AT FISHERVILLE POND

As described in Section 1.1.1 the outlet gate of the Fisherville Dam was cleared in 2000, causing the impoundment to be reduced to a narrow channel along the eastern shoreline. Fisherville Pond-North Pool has remained essentially unchanged, while the impoundments in the South Pool and Central Pool areas have been significantly decreased. The total acreage of the impoundment has decreased from approximately 69 acres to about 26 acres. The newly exposed mudflat area has since been vegetated by a variety of emergent species, effectively extending the wet meadow area from about 21 acres to approximately 64 acres.

The objective of this portion of the evaluation was to compare the relative risks to upper trophic level species with and without the impoundment at Fisherville Pond. Doses were calculated using the same exposure assumptions described for the area-by-area investigation (Section5). For the purpose of this evaluation two doses and HQs were calculated for each COPC for each species, one assuming the presence of the impoundment (*i.e.*, full pool conditions) and the other focusing on exposures following the reduction of the impoundment (*i.e.*, reduced pool conditions). A summary of the approach for each species is provided below.

6.1.1 River Otter

Under full pool conditions, it is assumed that the otter would consume fish from all pools within Fisherville Pond. Therefore, sediment and fish tissue concentrations from all three pools were averaged for the evaluation. Concentrations of chemicals not measured in fish (*i.e.*, PAHs, pesticides) were estimated using the same methodology described in Section 3.7.1. All other exposure parameters were the same as those described in Section 3.7.1 and Appendix F.

Following the reduction of the impoundment, the surface water area within Fisherville Pond was significantly reduced. As described above, although Fisherville Pond-North Pool remains relatively intact, the Central Pool and South Pool have been reduced to narrow, shallow channels that are unlikely to support significant populations of fish. Therefore, under reduced pool conditions, it was assumed that the otter would forage only at Fisherville Pond-North Pool. Doses and HQs were derived based on the average values for that area which consists of about 18 acres.

Table 6-1 summarizes the HQs developed under these two scenarios for the river otter. Risks were generally comparable, although slightly higher under full pool conditions. Arsenic and PCBs were the only COPC with HQs greater than 1 under the reduced pool conditions, while mercury and PAHs also exceeded under the full pool conditions. However it is important to note that the habitat available to the otter decreased from 69 acres under the full pool conditions to approximately 18 acres, or that area associated with Fisherville Pond-North Pool.

6.1.2 Mallard

As described in Section 3.2.9.2, it was assumed that the mallards primary exposure is through the consumption of aquatic invertebrates and incidental ingestion. As described for the river otter, it was assumed that under full pool conditions the mallard would forage throughout Fisherville Pond, therefore, an average sediment concentration for the entire pond was used. To evaluate reduced pool conditions, it was assumed that the mallard would forage in the north pool area and within the remaining channel in Fisherville Pond-Central Pool and Fisherville Pond-South Pool. As for the otter, all other exposure parameters were the same as those described in Section 3.7.2 and Appendix F.

Similar to the results for the river otter, the HQs associated with the two scenarios were very similar, although slightly higher under full pool conditions. Chromium, lead and mercury exceeded 1 under full pool conditions, while only chromium and lead exceeded 1 under reduced pool conditions. As noted for the otter, the available habitat for the mallard was also substantially reduced, from 69 to 26 acres.

6.1.3 Robin

The assumed exposure pathway for the robin under full pool conditions was the ingestion of soil invertebrates associated with the wet meadow area. To evaluate this scenario, the three wet meadow samples were averaged and used to derive estimated soil invertebrate concentrations as described in Section 3.7.3 and Appendix F. With the reduction in size of the impoundment, however, a large portion of the pond was exposed and has since been vegetated. The reduction in the area of the impoundment increased the wet meadow area from approximately 21 acres to 64 acres. For the purpose of this assessment, the sediment concentrations in the newly exposed sampling locations were assumed to represent soil concentrations. Therefore, to evaluate exposures to the robin under reduced pool conditions, concentrations of COPC from the exposed areas were averaged with the wet meadow data.

Hazard quotients for the robin were much higher than those calculated for the mallard or river otter. The majority of metals were associated with HQs greater than 1, as were PCBs and the DDTs (*i.e.*, DDT, DDD, and DDE). The highest HQ was 33.31 for DDE under reduced pool conditions. Although risks were elevated under both scenarios, HQs were higher for the reduced pool scenario.

7. SUMMARY AND CONCLUSIONS

For the purpose of this ERA, multiple lines of evidence were evaluated including: sediment, porewater, and fish tissue chemistry, toxicity bioassays, benthic and fish community analyses, and dose calculations for selected wildlife species. Each of these lines of evidence provides an independent estimate of the potential risks, sometimes with conflicting results. The approach used in this evaluation incorporated a qualitative weight of evidence, in an attempt to provide a more comprehensive picture of the potential risks.

As discussed in Section 2.1, the evaluation was designed to address two questions:

- Potential risks associated with designated areas within Singing Pond and Fisherville Pond under full pool conditions; and,
- The relative risks associated with exposures at Fisherville Pond before and after the draining of the impoundment.

To address the first question, six areas were defined including Fisherville Pond-North Pool, Fisherville Pond-Central Pool, Fisherville Pond-South Pool, Singing Pond-Main Channel, Singing Pond-Marsh Area, and Lake Wildwood, the reference area. Using results averaged for these areas, potential risks were qualitatively evaluated (*i.e.*, high, medium, low) for each of the assessment endpoints identified (*i.e.*, benthic community, fish community, wildlife species).

7.1 Summary of Results

Fisherville Pond-North Pool and Lake Wildwood were determined to have low potential risk. Sediment concentrations in the North Pool were relatively low, probably as a result of dredging that occurred there in 1982. In addition, the results of the bulk sediment toxicity bioassays indicated that little or minimal toxicity was associated with sediments collected from this area. Similar results were obtained for Lake Wildwood. COPC concentrations were generally very low with only one chemical (4,4'-DDE) detected at elevated concentrations. Limited toxicity was observed in the bulk sediment toxicity tests as well.

Fisherville Pond-Central Pool and Singing Pond-Main Channel were both scored as medium. In general, sediment concentrations throughout these areas were elevated, however, toxicity observed in the bioassays was relatively moderate. Estimated risks to wildlife species were also moderate. In contrast, high toxicity was observed in sediment samples from the Singing Pond-Marsh Area and Fisherville Pond-South Pool. Sediment concentrations in these areas were also high, resulting in an overall rating of high risk.

Under the second assessment (*i.e.*, relative risks from full pool versus reduced pool conditions within Fisherville Pond) it was determined that risks to piscivorous species and aquatic waterfowl were generally similar under both scenarios although slightly higher under full pool conditions. However, the reduction in risk under the reduced pool scenario was also associated with a dramatic decrease in available habitat. In contrast, risks to the terrestrial songbird were greatly increased under the reduced pool conditions.

7.2 Uncertainty Evaluation

There are a number of uncertainties associated with this assessment (Table 7-1). For example, the scoring criteria developed, while derived based on best professional judgment and applied systematically across all areas, is somewhat subjective. In addition, it is impossible to account for all factors that might influence the observed results. For example, physical factors such as grain size, habitat availability, etc. are difficult to factor in to the scoring process and can only be addressed qualitatively. Also, there is uncertainty inherent in the selection of measurement endpoints.

The assessment relies on comparison to a number of screening benchmarks; it is important to recognize that the predictive ability of such benchmarks is limited. In addition, concentrations of several chemicals were estimated in porewater, invertebrate tissue and fish tissue; actual concentrations may vary.

7.3 Conclusions

This evaluation indicates that risks to ecological receptors associated with Fisherville Pond-North Pool and Lake Wildwood are negligible. However, risks associated with the remaining areas of Fisherville Pond and Singing Pond may be of concern, ranking as medium or high based on the lines of evidence evaluated. Based on this assessment, it appears that sediment remediation in these areas would be likely to reduce risks and result in an overall ecological benefit.

Regarding the relative risks associated with the presence or absence of the Fisherville Pond impoundment, the results indicate that overall risks to the wildlife species evaluated are likely to be lower under the full pool conditions. Although risks to waterfowl and piscivorous wildlife decreased slightly under the reduced pool conditions, the associated reduction in available habitat is likely to be detrimental, offsetting the potential benefit. In contrast, the available habitat increased substantially for the songbird under the reduced pool conditions, magnifying the increase in potential risks associated with that scenario. Therefore, it is concluded that restoring the former impoundment at Fisherville Pond would reduce potential risks to wildlife species.

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TABLES

FIGURES