REMEDIAL INVESTIGATION REPORT
MAYBROOK LAGOON SITE

TOWN OF HAMPTONBURGH
ORANGE COUNTY, NEW YORK

VOLUME I OF IV - TEXT, FIGURES AND TABLES

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TABLE OF CONTENTS

Page

TABLE OF CONTENTS

Page

TABLE OF CONTENTS

TABLE OF CONTENTS

Page

TABLE OF CONTENTS

Page

TABLE OF CONTENTS

Page

TABLE OF CONTENTS

Page

TABLE OF CONTENTS

Page

TABLE OF CONTENTS

Page

TABLE OF CONTENTS

Page

TABLE OF CONTENTS

Page

LIST OF FIGURES
(Following Text)

LIST OF FIGURES
(Following Text)

LIST OF FIGURES
(Following Text)

LIST OF TABLES
(Following Text)

LIST OF TABLES
(Following Text)

LIST OF TABLES
(Following Text)

LIST OF TABLES
(Following Text)

LIST OF TABLES
(Following Text)

LIST OF TABLES
(Following Text)

LIST OF APPENDICES

LIST OF APPENDICES

LIST OF APPENDICES

LIST OF ACRONYMS

AMSL	Above Mean Sea Level
ANOVA	Analysis of Variance between groups
APE	Area of Potential Effect
ARARs	Appropriate, Relevant and Applicable Regulations
ARCS	Assessment and Remediation of Contaminated Sediments
ASTM	American Society for Testing and Materials
AWQC	Ambient Water Quality Criteria, U.S. EPA
AWQS	Ambient Water Quality Standards, New York State
BCC	Bioaccumulative Chemicals of Concern
BCF	Bioconcentration Factors
BERA	Baseline Ecological Risk Assessment
BGS	Below Ground Surface
BnC	Bath-Nassau shaley silt loam, 8 to 15 percent slopes
BTAG	Biological Technical Assisstance Group
BTEX	Benzene, toluene, ethylbenzene, xylene(s)
CCME	Canadian Council of Ministers of the Environment
CDI	Chronic Daily Intake
CERCLA	Comprehensive Environmental Response, Compensation and Liability Act
COC	Chemicals of Concern
COEC	Constituents of Ecological Concern
COPCs	Chemicals of Potential Concern
CRA	Conestoga Rovers & Associates
CSF	Cancer Slope Factor
CT	Central Tendancy
CY	Cubic Yards
DMP	Data Management Plan, CRA, 1990
EA	Preliminary Ecological Assessment
EDQL	Environmental Data Quality Limit
EF	Exposure Frequency
EqP	Equilibrium Partitioning
ERA	Ecological Risk Assessment
ER-L	Effects Range - Low

LIST OF ACRONYMS

ER-M	Effects Range – Medium
ERRD	Emergency and Remedial Response Division
ESV	Ecological Screening Values
ET	Ecotox Threshold
FAC	Facultative Wetland Plants
FACU	Facultative Wetland Plants
FACW	Facultative-wet Wetland Plants
FS	Feasibility Study
FWIA	Fish and Wildlife Impact Analysis
GPS	Global Positioning System
HASP	Health and Safety Plan, CRA, 1990
HEAST	Health Effects Assessment Summary Table
HELP	Hydrological Evaluation of Landfill Performance Computer Model
HHRA	Human Health Risk Assessment
HI	Hazard Index
HoB	Hoosic gravelly sandy loam, 3 to 8 percent slopes
HoC	Hoosic gravelly sandy loam, 8 to 15 percent slopes
HQ	Hazard Quotient
HSA	hollow stem augers
HWSB	Hazardous Waste Support Branch
ID	inner diameter
IEUBK	Integrated Exposure Uptake Biokenetic
IRIS	Intefrated Risk Information System
LADD	Lifetime Average Daily Dose
LCV	Lowest Chronic Values
LEL	Lowest Effect Level
LMS	Linearized Multistage
LOAEL	Lowest Observed Adverse Effect Level
MdB	Mardin gravelly silt loam, 3 to 8 percent slope
MCLs	Maximum Contaminant Levels
MF	Matrix Factor
MLEs	Maximum Likelihood Estimates
MOE	Ontario Ministry of the Environment

LIST OF ACRONYMS

NAPL	Non-Aqueous Phase Liquids
NEL	No Effect Level
Nepera	Nepera, Inc.
NOAA	National Oceanic and Atmospheric Administration
NOAEL	No Observed Adverse Effect Level
NOEC	No Observed Effect Concentration
NYDOH	New York Department of Health
NYS	New York State
NYSDEC	New York State Department of Environmental Conservation
OBL	Obligate Wetland Plants
ORNL	Oak Ridge National Library
OSWER	Office of Solid Water and Emergency Response
OU-1	Operable Unit 1
OU-2	Operable Unit 2
PAHs	Polyaromatic Hydrocarbons
PCBs	Polychlorinated biphenyls
PEC	Probable Effects Concentration
Pg	Pits, Gravel
PID	Photoionization Detector
PFO	Palustrine-forested Wetland
PRGs	Preliminary Remediation Goals
QAPP	Quality Assurance Project Plan, CRA, 1990
Qu	Quarries
RA	Risk Assessment
RAGS	Risk Assessment Guidance for Superfund
RBC	Risk Based Criteria
RD	Remedial Design
RDA	Recommended Daily Allowance
REDOX	Reduction and Oxidization
RfD	Reference Dose
RI	Remedial Investigation
RME	Reasonable Maximum Exposure
RQD	Rock Quality Designation

LIST OF ACRONYMS

SA	Stipulation Agreement Index No. W3-006-8102
SERA	Screening Ecological Risk Assessment
SCL	Screening Level Concentration
SCS	Soil Conservation Service
Site	Former Lagoon Site
SOP	Site Operations Plan, CRA, 1990
SOW	Scope of Work
SP	Self Potential
SQB	Sediment Quality Benchmark
SQC	Sediment Quality Criteria
SVE	Soil Vapor Extraction
SVOCs	Semivolatile Organic Compounds
TAL	Target Analyte List
TCL	Target Compound List
TCR	Target Cumulative Risk
TEC	Threshold Effects Concentration
TICs	Tentatively Identified Compounds
TOC	Total Organic Carbon
TPH	Total Petroleum Hydrocarbons
TRW	Technical Review Workshop
TWA	Time Weighted Average
UCL	Upper Confidence Limit
UF	Uptake Factor
USCS	Unified Soil Classification System
USDA	United States Department of Agriculture
U.S.EPA	United States Environmental Protection Agency
USFWS	United States Fish and Wildlife Services
USGS	United States Geological Survery
VHG	Vertical Hydraulic Gradient
VOCs	Volatile Organic Compounds
WHO	World Health Organization
WLC	Warner-Lambert Company
Work Plan	RI/FS Work Plan, Dames and Moore, December, 1989

1.0 INTRODUCTION

This Remedial Investigation (RI) Report has been prepared by Conestoga-Rovers & Associates (CRA) on behalf of the Maybrook and Harriman Environmental Trust (the Trust) for the Former Lagoon site (Site) located in Hamptonburgh, New York. This report incorporates all additional field activities, results, and changes made into the previously submitted RI Report (RI-95) to further and more distinctly characterize the Site.

1.1 STATEMENT OF PURPOSE

The RI/FS program for the Site is being completed in accordance with Stipulation Agreement Index No. W3-0006-8102 entered into by Nepera, Inc. (Nepera) and Warner-Lambert Company (WLC) with the New York State Department of Environmental Conservation (NYSDEC). The Stipulation Agreement, signed by the NYSDEC on March 21, 1988, included the following RI/FS objectives:

1. to conduct an RI, including a Risk Assessment (RA), to determine the nature and extent of the release or threatened release of hazardous substances, pollutants or contaminants from the Site; and
2. to perform a Feasibility Study (FS) to identify and evaluate alternatives for the appropriate extent of remedial action, if any, to prevent or mitigate the migration, or the release, or threatened release of hazardous substances, pollutants, or contaminants from the Site.

Following the Stipulation Agreement, a Consent Decree between the State of New York and the Estate of William Lasdon, Nepera, Inc., and Warner-Lambert Company was filed in US District Court, Southern District of New York on May 5, 1998. In order for the Estate to be absolved of its liability at the Maybrook Site, the Estate placed funds into a Trust established for the Harriman and Maybrook, New York Sites. The Trust was established by an Indenture of Trust dated March 25, 1998 and is defined in the following sections of the RI as "The Maybrook & Harriman Environmental Trust" or simply the Trust. As indicated in the Consent Decree, following approval of the FS by NYSDEC and United States Environmental Protection Agency (U.S. EPA), the State may develop and make available for public comment, a Preliminary Remedial Action Plan (PRAP) for the Maybrook Site.

In order to effectively meet the stated objectives for the RI/FS Study Program, an RI/FS Work Plan (Work Plan) (Dames and Moore, 1989) was prepared for the review and approval of NYSDEC. The approved Work Plan culminated in the development, review, and finalization of the following Project Plans which were prepared to aid in the effective implementation of the Work Plan:

1. Site Operations Plan (SOP);
2. Quality Assurance Project Plan (QAPP);
3. Health and Safety Plan (HASP);
4. Data Management Plan (DMP); and
5. Appropriate, Relevant and Applicable Regulations (ARARs) Report.

All Project Plans were approved by NYSDEC prior to the initiation of Site investigative activities.

The Scope of Work (SOW) for the RI/FS is a compilation of all the work tasks to be performed during the study. A summary of the Tasks identified in the SOW is presented in Table 1.1.

In addition to the SOW, an Additional Soil Sampling Work Plan was developed by the Trust during 2002 to 2003. The Work Plan outlined additional investigative tasks related to soil and defined a process to evaluate inorganics for the purposes of the FS. The Additional Soil Work Plan was approved by U.S. EPA in February 2003 and incorporated agreed methodology based on extensive discussions with U.S. EPA and NYSDEC. This RI report reflects the additional work outlined in the Work Plan and is discussed in subsequent sections.

1.2 RI OBJECTIVES

The specific objectives of the RI are summarized as follows:

- to determine the nature of the remaining contents buried, if any, in the former lagoons;
- to assess Site hydrogeologic and environmental conditions;
- to determine the nature and extent of contaminants on Site;
- to characterize the subsurface geologic conditions;
- to determine the nature and extent of contaminants in the groundwater;
- to determine the nature and extent of contaminants in the soil;

- to determine the nature and extent of contaminants in surface water and sediments;
- to determine the potential contaminant migration pathways;
- to determine the nature and extent, if any, of contaminant migration;
- to assess the extent to which detected contamination poses a threat to public health, welfare, or the environment;
- to develop an RA, including an Ecological Risk Assessment (ERA) and Human Health Risk Assessment (HHRA);
- to gather data for the development and evaluation of remedial action alternatives (Feasibility Study), if required; and
- to summarize the conclusions of the Site investigation and develop Site-specific chemicals of concern and remedial action objectives which will form the basis for the FS.

1.3 SITE DESCRIPTION

As indicated on Figure 1.1, the Site is located approximately 1.5 miles southwest of the Town of Maybrook, New York on the southern side of Orange County Highway 4. The 29.3-acre Site includes a portion of Beaverdam Brook and a portion of an abandoned railroad bed. The Site is surrounded mainly by farmland and is bounded on the north by Orange County Highway 4. Three residences exist in the immediate vicinity of the Site, one just west of the southwest marsh area, and two to the north/northeast of the Site on the opposite side of Orange County Highway 4. In addition, a residential subdivision has recently been developed to the south of the Site. A Site plan is presented as Figure 1.2.

Approximately five acres of the Site were affected by the historical lagoon operations. Access from Orange County Highway 4 is via a gravel road leading past the former lagoon area to an abandoned railway bed. Beaverdam Brook traverses the western edge of the Site flowing south to the Otter Kill just beyond the southern edge of the Site. The historical lagoon operations of the Site is currently fenced around its perimeter (see Figure 1.2).

The Site is in an area of rolling hill topography. Two hills, and a portion of a third one, with an approximate elevation of 400 feet above mean sea level (amsl), occupy the Site. The maximum local relief is on the order of 40 feet. Most of the Site is forested. The former lagoon area was originally stripped of vegetation but is now covered with grasses, wild flowers, and mixed brush.

1.4 SITE HISTORY

Nepera is currently a producer of bulk pharmaceutical chemicals, hydrogels and pyridine-based industrial chemical intermediates. The plant facility located approximately 25 miles away at Harriman, New York was originally operated by the Pyridium Corporation beginning in 1942. Since that time, several other companies have owned and operated Nepera.

The Maybrook Site was purchased by the Pyridium Corporation in 1952. The Site was used for wastewater disposal in six constructed lagoons between 1953 and 1967. These lagoons were used to dispose of wastewater generated at the plant site located in Harriman, New York. No wastewater disposal has taken place at the Site since December 1967.

1.5 REPORT ORGANIZATION

This report has been prepared consistent with the United States Environmental Protection Agency (U.S. EPA) "Guidance for Conducting Remedial Investigations and Feasibility Studies Under CERCLA" (October 1988), and as such, is organized in the following manner:

Section 1.0	Introduction;
Section 2.0	Previous Investigations (before 1991);
Section 3.0	Recent Investigations;
Section 4.0	Regional Setting;
Section 5.0	Site Setting;
Section 6.0	Chemical Distribution;
Section 7.0	Chemical Fate and Transport;
Section 8.0	Ecological Risk Assessment;
Section 9.0	Human Health Risk Assessment;
Section 10.0	Summary and Conclusions; and
Section 11.0	References.

2.0 PREVIOUS INVESTIGATIONS

Prior to the RI, a number of investigations were conducted that are relevant to the RI/FS. Leggette, Brashears and Graham conducted an initial investigation of the site in 1967, Groundwater Technology performed a site investigation in 1983, and C.A. Rich completed a site investigation in 1985. The results of these investigations have been used in this RI Report, where appropriate. The following sections briefly summarize the pre-RI work.

2.1 INITIAL INVESTIGATION BY LEGGETTE, BRASHEARS & GRAHAM, 1967

The initial investigation of the Former Lagoon Site conducted by Leggette, Brashears and Graham was designed to determine if there was pollution on the site, and if so, how extensive.

Leggette, Brashears & Graham installed test wells T-1, T-2, and T-3 at the locations presented on Figure 2.1 in May 1967 as part of the groundwater study program conducted for Nepera. The well locations were selected based on the most probable groundwater flow directions. Well T-1 was located in the direction of northerly groundwater flow and well T-2 was completed in the direction of northwesterly groundwater flow. Well T-3 was to have been completed in the direction of southerly flow. However, do to accessibility problems, T-3 was relocated in the direction of westerly flow.

As these wells were drilled, temporary perforated pipe was used in each hole to collect groundwater samples from the permeable overburden. Following overburden groundwater sampling, the holes were drilled approximately 10 feet into bedrock. The overburden was isolated from the bedrock with 3-inch diameter steel pipe. The logs for wells T-1, T-2 and T-3, provided in Appendix D.1, indicate that an open bedrock corehole was completed to a depth of approximately 25 feet below ground surface (bgs) for wells T-2 and T-3 and 9.5 feet bgs for well T-1.

The results of the groundwater analyses were not located during our review of archive data and are therefore not included in the RI.

2.2 GROUNDWATER TECHNOLOGY INVESTIGATION

Groundwater/Technology installed 7 overburden wells (MW-1 to MW-7) and 3 piezometers (PZ-1, PZ-2, and PZ-3), as shown on Figure 2.1, at the Site in April 1983. Wells MW-1, MW-2, MW-3, and MW-6 and piezometer PZ-3 were screened approximately 0.5 to 1.0 foot into the fractured bedrock. Wells MW-4, MW-5, and MW-7 and piezometers PZ-1 and PZ-2 were completed only within the overburden. Well logs can be found in Appendix D.2.

Wells MW-1 and MW-2 were located downgradient of the former lagoon area to the north/northeast. Wells MW-5, MW-6, and MW-7 were located downgradient of the former lagoon area to the south/southwest. Wells MW-3 and MW-4 were completed within the area of the former lagoons.

Any analyses that Groundwater/Technology may have conducted were not located during our review of archive data and therefore is not included in the RI.

2.3 C.A. RICH INVESTIGATION

The work activity conducted on site by C.A. Rich consisted of:

- a seismic profiling survey;
- lagoon sampling;
- test drilling and logging; and
- sampling and analyses.

The seismic refraction profile survey was conducted by C.A. Rich in May, 1985. The survey was conducted for the purpose of determining subsurface configuration and distribution of geologic media beneath the site.

Methodology, location of the seismic profiles and interpretation of the collected data can be found in the C.A. Rich Summary Report (C.A. Rich, 1986) along with cross-sections derived from the survey in Appendix III of that report.

Lagoon sampling activities were conducted to characterize the vertical extent of waste materials, depth to bedrock, types of non-water materials encountered and sample analyses for the parameters approved by Nepera Inc. and the NYSDEC. Locations of the samples that were collected are presented on Figure 3.1. Characterization of the lagoon material is presented on Table 2.1 and results of the laboratory analyses are included in the tables for soil, surface water and sediment in Section 6.0.

In August 1985, C.A. Rich installed 10 shallow monitoring wells (SW-1 to SW-10) screened at the bedrock/overburden interface and 2 deep bedrock wells (DW-1 and DW-2). The locations were chosen in conjunction with the NYSDEC. The well locations were selected to provide more complete coverage of the Site in the vicinity of the former lagoons.

All overburden wells were screened at the overburden/fractured bedrock interface with the exception of well SW-9 which was completed only in the overburden.

Deep bedrock well DW-1 was nested with shallow well SW-9 and deep bedrock well DW-2 was nested with shallow well SW-2 to determine the hydraulic connection between the overburden and the bedrock groundwater flow regimes. Bedrock well DW-2 was located midway between the former lagoon area and the Maybrook wellfield. Bedrock well DW-1 was located in the southwest portion of the Site in the direction of southwesterly groundwater flow downgradient from the former lagoons. Wells DW-1 and DW-2 were installed as open coreholes in the bedrock from 24 to 152 feet bgs and from 15 to 175 feet bgs, respectively. The bedrock wells were, reportedly, properly cased and sealed off from the overburden/fractured bedrock to prevent the downward migration of chemicals from the overburden to the deep bedrock.

A total of 16 groundwater samples, 4 surface water samples, 6 lagoon soil samples and 4 stream sediment samples were taken from the site and surrounding area. Figure 3.1 and Figure 3.3 presents the locations of the soil samples, and surface water and sediment samples respectively. The samples were analyzed for select VOCs, SVOCs and metals, and the results are included in Appendix I.1 (soil), Appendix I.2 (groundwater), Appendix I.3 (surface water) and Appendix I.4 (sediment). The results are considered and discussed where relevant in the chemical distribution analysis in Section 6.0 of this report.

Water elevations measured by C.A. Rich Consultants, Inc. in 1985 are presented in Table 3.9 along with CRA's water level monitoring.

3.0 RI- RELATED INVESTIGATIONS

The characterization of the nature and extent of contamination at the Site was achieved by conducting a variety of investigative activities. The purpose of this section is to provide a summary of all the investigative activities which have been completed at the Site during the period of 1991 to 2003. A detailed chronology of events for this period is provided in Table 3.1.

These investigations include the following:

- Phase I Field Investigations – 1991;
- Phase II Field Investigations – 1995;
- Treatability Study Investigations – 1996;
- Interim Groundwater Monitoring Program – 1997 to present;
- Additional Groundwater Sampling – 2001/2002;
- Natural Attenuation Study – 2001/2002;
- Additional Soil Investigations – 2003;
- Wetland Delineation – 2003; and
- Regional Water Wells - 1986 and 1995.

3.1 PHASE I FIELD INVESTIGATIONS

The Phase I RI field investigation conducted in 1991 included:

- detailed Site inspection to evaluate and assess present conditions;
- Site survey to establish Site boundaries and Site features;
- field reconnaissance of pre-RI existing monitoring wells;
- installation of four (4) boreholes;
- installation of four (4) bedrock monitoring wells and one (1) overburden monitoring well;
- single well response testing of newly installed monitoring wells;
- pump testing at one (1) newly installed bedrock monitoring well and one (1) existing bedrock monitoring well, and monitoring response of surrounding overburden wells;
- excavation of test pits to determine the boundaries of the former lagoons;

- collection of subsurface soil, groundwater, surface water and sediment samples for laboratory analysis;
- a magnetometer survey and subsequent investigation of magnetic anomalies;
- a downhole geophysical survey at two (2) existing and four (4) newly installed bedrock wells;
- water level measurements in all monitoring wells;
- geologic mapping; and
- on-Site ecological assessment.

The field investigative activities for the Phase I RI were reported to NYSDEC as a series of eleven Technical Memoranda. The eleven Technical Memoranda were presented to NYSDEC on an ongoing basis throughout the completion of the Phase I Site investigation. Table 3.2 summarizes the twelve Technical Memoranda (one from the Phase II activities) previously submitted.

The investigative activities for the Phase I RI are further discussed in the following sections.

3.1.1 DETAILED SITE INSPECTION

A Site inspection was conducted during the Phase I RI to evaluate and assess the conditions at the Site and to confirm the locations for all sampling activities and the accessibility of monitoring well locations. Based upon this inspection the following changes to the SOP were implemented:

- proposed locations for well cluster MW-2U-91 and MW-2D-91 were revised as the original locations were at the top of a steep bedrock outcrop which was both inaccessible and not suitable for the completion of an overburden well; and
- proposed locations for monitoring wells MW-1U-91 and MW-3D-91 were revised to complete the two wells as a nested pair, no overburden groundwater was encountered and MW-1U-91 was subsequently used to replace pre-RI monitoring well MW-5.

3.1.2 SITE SURVEY AND BASE MAP

Total station surveying was completed by Loch Surveyors and Engineers (New York State licensed surveyor) under the direct supervision of CRA during May 1991. Additional surveying was completed by Hayward & Pakin (on an as-required basis).

The Site survey information including the former lagoon locations are presented on Figure 2.1. The results of this Task were presented to NYSDEC in Technical Memorandum No. 1 and are utilized throughout this report.

3.1.3 SURFACE GEOLOGICAL MAPPING

Geologic mapping of outcrops at the Site was conducted during the well installation program in July 1991 to identify:

1. lithostratigraphic units,
2. contacts,
3. bedding,
4. foliation,
5. parting,
6. cleavage,
7. jointing,
8. fracturing,
9. faults,
10. fault zones,
11. folds,
12. aerial planes, and
13. strike and dip of bedrock.

The results of the surface geological mapping were presented to NYSDEC in Technical Memorandum No. 4 and are reproduced in Section 5.0 of this Report.

3.1.4 PRELIMINARY ECOLOGICAL ASSESSMENT

A preliminary ecological assessment (EA) was conducted during the Phase I RI to assess the possible impact of potential releases, if any, discovered during the RI. The EA was based on an on-Site reconnaissance and existing data sources such as aerial photographs, NYSDEC Wildlife Resources Center, NYSDEC Division of Regulatory Affairs, and Town of Hamptonburgh Master Plan. The on-Site EA field reconnaissance was completed on October 14, 1991.

The results of the EA were presented to NYSDEC in Technical Memorandum No. 8 and were reproduced in Section 8.0 of the original submission of the RI Report (RI-92).

The EA was expanded as part of the Phase II RI to include a more detailed EA in accordance with the NYSDEC guidance document entitled "Fish and Wildlife Impact Analysis for Inactive Hazardous Waste Sites", dated October 1994 (FWIA). The FWIA is discussed in Section 3.2.1.

3.1.5 TEST PITS

Test pits were excavated during the Phase I RI to assist in defining the lagoon boundaries and to obtain samples of the sludge material for chemical analysis. Test pits were initially excavated (Phase I) during the period from June 25 to July 12, 1991. Two drums were encountered during excavation of the Phase I test pits and subsequently, a magnetometer survey was requested by the NYSDEC to determine the extent of potential additional drums and other magnetic anomalies. The test pit locations are presented on Figure 3.1. Test pit logs and photographs are presented in Appendix B.1.

Test pits were excavated and backfilled using a Case 780 rubber tired backhoe. Excavation and backfilling was conducted under Level B or Level C health and safety protection, based upon air monitoring results. Air in the breathing zone and in the test pits was monitored continuously using an HNU photoionization device and a Jerome 411 Mercury vapor analyzer. Readings of organic vapors and mercury vapors were measured, and were recorded on the test pit logs (see Appendix B.1).

Test pits were excavated in approximately 12-inch passes. The top 12 inches of excavated materials was stockpiled separately from the remaining excavated material for each test pit. All excavated materials were stockpiled on 6-mil polyethylene adjacent to the test pit. Test pits were not excavated below the water table, if encountered.

After logging, photographing and sampling (if necessary) each test pit was backfilled. Excavated materials were placed roughly in the same location of the pit from which they were removed. The backhoe bucket was used to compact material into the pit. Equipment was steam cleaned at the end of each day and after completing test pit activities associated with each lagoon.

Two steel 55-gallon drums (D001 and D002) were excavated from test pit TP-17 during Phase I activities. The backhoe bucket was used to remove the drums and to place each drum directly into 85-gallon overpack drums under the direction of the NYSDEC field representative. The overpacks were then staged on pallets (pending disposal). One drum (D001) contained greasy soil and the other drum (D002) contained liquid (which was solidified in the field).

Thirty-four test pits were completed during Phase I and a total of 12 samples were submitted for chemical analysis. Sampling locations were selected from the interval which indicated the highest potential for chemical presence based upon field screening (both visual and HNU/Jerome 411) and were determined in consultation with the NYSDEC field representative.

A summary of the test pit and drum samples collected for laboratory analysis is presented in Table 3.3. Results of the chemical analyses are discussed in Section 6.2.

Soil/sludge samples were taken directly from the backhoe bucket utilizing stainless steel spoons. The samples were then placed directly into laboratory provided sample containers in a manner consistent with the approved QAPP. The samples were stored in a cooler on ice and maintained at or below approximately 4°C. At the end of each day, samples were shipped to H2M Labs, Inc. under Chain of Custody procedures (via Federal Express).

For the majority of the test pits, the horizontal edge and vertical extent of the former lagoons could not be determined. Hence, test pits were terminated at a point agreed upon in the field by both the CRA and the NYSDEC representatives. In absence of the confirmatory data from the test pits, the areal extent of the lagoons are as defined previously based upon the 1963 aerial photograph, shown on Figure 3.12.

The results of the Phase I test pit activities were presented to NYSDEC in Technical Memoranda No. 2 and 6.

3.1.6 BOREHOLE SOIL SAMPLING

In accordance with the SOP, soil samples were collected during the Phase I RI from both soil borings and from boreholes drilled for the purpose of monitoring well installation. Borehole and monitoring well locations are presented on Figure 3.1.

Four soil borings (BH-1-91, BH-3-91, BH-4-91 and BH-5-91) were drilled using a Canterra CT 150 truck-mounted drill rig. Borehole BH-2-91 was not installed separately since stratigraphic information and soil samples for chemical analysis could be collected from monitoring well MW-1D-91, which was installed immediately adjacent to the proposed location of borehole BH-2-91. The drill rig advanced the boreholes using 4 1/4-inch inside diameter (ID) hollow stem augers (HSA). Borehole stratigraphic logs are presented in Appendix C.

Additional soil samples were collected from boreholes installed for the completion of monitoring wells MW-1D-91, MW-2D-91, MW-4D-91 and MW-1U-91. Soil samples were retrieved in accordance with ASTM Method D1586 using 2-inch and 3-inch diameter split spoon samplers. Soil samples were collected continuously to bedrock and the samples were described and recorded in accordance with the Unified Soil Classification System (USCS).

Stainless steel spatulas were used to place soil samples into glass sampling jars for classification and chemical analysis. Soil headspace readings were measured and recorded using an HNU photoionization detector (PID). Efforts were made to collect soil samples for chemical analysis from depth intervals 1 to 2 feet, 4 to 5 bgs, and 1 foot above bedrock, as stipulated in the SOP. However, this protocol could not be met due to actual borehole depths and/or insufficient soil recovery. Chemical analyses included Target Compound List (TCL) Volatile Organic Compounds (VOCs), Semi-volatile Organic Compounds (SVOCs), Pesticides/Polychlorinated biphenyls (PCBs), Target Analyte List (TAL) Metals, Cyanide, total petroleum hydrocarbons (TPH) and Site-specific parameters. A summary of soil samples submitted for chemical analysis is presented in Table 3.4.

Upon completion of the soil borings, the boreholes were tremie backfilled with a neat cement/bentonite grout to ground surface. Monitoring wells were installed in the boreholes for MW-1D-91, MW-2D-91, MW-4D-91 and MW-1U-91 as described in Section 3.1.8. Auger cuttings from the soil borings were distributed evenly on the ground in the location of each borehole in accordance with the SOP.

The results of the borehole soil sampling were presented to NYSDEC in Technical Memoranda No. 4 and No. 6.

3.1.7 HYDROGEOLOGIC EVALUATION

The hydrogeologic evaluation during the Phase I RI included the following:

- installation of four (4) bedrock monitoring wells (MW-1D-91, MW-2D-91, MW-3D-91, and MW-4D-91) and one (1) overburden monitoring well (MW-1U-91);

- well development and groundwater sampling of the newly-installed wells and selected existing monitoring wells;
- in situ hydraulic conductivity testing of selected monitoring wells;
- downhole geophysical logging at six (6) monitoring wells; and
- surveying of all newly installed and existing monitoring wells for vertical and horizontal control.

The results of the Phase I hydrogeologic investigation were presented to NYSDEC in Technical Memoranda No. 4, 6, and 7.

Details of the hydrogeologic evaluation are presented in the following subsections. Details of the in situ hydraulic conductivity testing of selected monitoring wells are combined with the Phase II RI testing and is presented in Section 3.2.4.6.

3.1.7.1 NEW MONITORING WELL INSTALLATION

The Phase I RI included drilling and installation of four bedrock and one overburden monitoring wells. The monitoring wells were installed by Marcor of New York, Inc., of Rochester, New York between the periods of June and July 1991 under the supervision of CRA personnel. The locations of the boreholes and monitoring wells are presented on Figure 2.1.

During the Phase I RI several deviations from the SOP were necessary in order to accommodate rig accessibility and existing Site conditions. Monitoring wells MW-2U-91 and MW-2D-91 were originally proposed to be installed at the top of a steep bedrock outcrop which was both inaccessible and not suitable for the completion of an overburden well. It was agreed between NYSDEC, Nepera, WLC, and CRA that the MW-2 monitoring well cluster should be relocated adjacent to the existing monitoring well SW-4. It was also determined that SW-4 would be sufficient for overburden groundwater monitoring and that monitoring well MW-2U-91 was not required.

The installation of proposed monitoring wells MW-1U-91 and MW-3D-91 were originally to be completed in heavily wooded areas. An agreement was reached between NYSDEC, Nepera, WLC, and CRA to centralize and incorporate both wells into a single monitoring well cluster. However, overburden groundwater was not encountered at this location so ultimately only MW-3D-91 was installed.

Existing monitoring well MW-5 was damaged while clearing access to the proposed monitoring well location MW-4D-91. Monitoring well MW-5 was subsequently abandoned and monitoring well MW-1U-91 was relocated and installed as its replacement.

Overburden Monitoring Well

A Canterra CT 150 truck mounted drill rig was used to the install overburden monitoring well (MW-1U-91). The overburden monitoring well completion details are presented in Table 3.5, and the stratigraphic and instrumentation log is presented in Appendix D.3.

Soil samples were retrieved in accordance with ASTM Method D1586 using 2-inch and 3-inch diameter split spoon samplers. Soil sample headspace readings were measured and recorded.

The monitoring well was installed using the hollow-stem auger method. The borehole was completed with a monitoring well using a 5-foot length of 2-inch diameter stainless steel, 0.010-inch continuous slot well screen, attached to 2-inch diameter stainless steel riser pipe and end cap. The deviation of using 2-inch diameter instead of 4-inch diameter riser pipe and screen, as stipulated in the SOP, was approved by the NYSDEC. Quartzite sand (#4) was placed in the annular space to a height of at least one foot above the top of the well screen. A 2-foot bentonite seal was placed above the sandpack. The annulus above the bentonite seal was tremie backfilled with neat cement/bentonite grout to ground surface. A steel locking protective casing was placed over the riser pipe and was embedded in the cement/bentonite grout. A padlock was placed on the protective casings for security. A minimum of three protective posts were installed around the well to complete the installation.

With NYSDEC approval, the well screen was installed in the saturated zone where the PID detected the highest soil headspace readings.

Auger cuttings from the boring was distributed evenly on the ground in the general vicinity of the monitoring well, in accordance with the SOP.

Bedrock Monitoring Wells

Four bedrock monitoring wells (MW-1D-91, MW-2D-91, MW-3D-91, and MW-4D-91) were installed using a Canterra CT 350 truck-mounted drill rig. A summary of bedrock well completion details is presented in Table 3.5, and stratigraphic and instrumentation logs are presented in Appendix D.3.

Soil samples within the overburden were retrieved in accordance with ASTM Method D1586 using 2-inch and 3-inch diameter split spoon samplers. Soil sample headspace readings were measured and recorded for the Phase I bedrock monitoring wells.

The bedrock was cored using NX-size coring equipment during the Phase I RI. The core was logged in the field for its composition, formation name, bedding, orientation, fractures, joints, folding and faulting, core length, percent recovery and Rock Quality Designation (RQD). The cores were placed into labeled core boxes which were subsequently sent for storage to the Nepera, Inc. facility in Harriman, New York.

With the concurrence of the NYSDEC, the bedrock wells were completed as 3 7/8-inch diameter open hole wells using wet-rotary methods, instead of 6-inch diameter open hole wells using air-rotary methods, as proposed in the SOP. Water was utilized as the drilling fluid. The bedrock wells were installed using the following procedures:

1. Hollow-stem augers (4 1/4-inch ID) were advanced through the overburden to bedrock and were removed.
2. Temporary 6-inch diameter flush threaded steel casing was installed in the borehole and was seated firmly onto the bedrock surface.
3. NX coring was conducted until five feet of competent bedrock was recovered.
4. The core hole was reamed to 5 7/8-inch diameter using a tricone bit and wet-rotary methods. Water was used as the drilling fluid.
5. A 4-inch diameter flush threaded stainless steel surface casing was installed to the bottom of the reamed hole.
6. The surface casing was tremie grouted and was allowed to set for a minimum 24-hour period.
7. NX coring was again implemented until a water producing zone was intersected.
8. The core hole was reamed using a 3 7/8-inch diameter tricone wet rotary bit to the final depth.

Steel locking protective casings were placed over the surface casing and padlocks were placed on the protective casings for security. A minimum of three protective posts were installed around the monitoring wells to complete the installation.

3.1.7.2 WELL DEVELOPMENT

Monitoring wells were developed utilizing surging, pumping and bailing techniques. Development was accomplished using stainless steel and steel bailers, and a 2-inch diameter stainless steel submersible pump.

A minimum of four well volumes were removed from each monitoring well. The purged water was tested for pH, conductivity, temperature and turbidity to determine chemical stability and clarity. The wells were considered to be stabilized when three consecutive and consistent readings of the stabilization parameters were achieved. Development records for the RI monitoring wells are presented in Table 3.6.

3.1.7.3 GROUNDWATER SAMPLING

Groundwater sampling was conducted during the period from August 23 to August 27, 1991 for the Phase I RI. The Monitoring Well Network for the sampling event is presented in Table 3.7. The summary of groundwater sample collection is presented in Table 3.8.

During the Phase I RI, monitoring wells SW-6 and SW-8, could not be sampled for the following reasons:

- SW-6 Inner well cap seized and could not be opened with a pipe wrench; and
- SW-8 During purging of the well, the sand pack collapsed into the riser pipe.

Maybrook Town wells #1 and #3 were sampled on October 16, 1991. Sampling of the town wells was not concurrent with the August sampling of the on-Site wells since it was not possible to obtain access to the Town wells at that time.

A minimum of three well volumes were either pumped or bailed from each well prior to sampling for all parameters. Well purging was completed utilizing a 2-inch stainless steel submersible pump or stainless steel bailers. Purge records are presented in Appendix E.

Samples were collected from each well to determine pH, conductivity, temperature and turbidity.

Samples for chemical analyses were removed directly from the wells and poured into sample jars (with the appropriate preservatives) utilizing pre-cleaned stainless steel bailers.

After sampling each monitoring well, the sample jars were placed directly into the shipping containers with ice or freezer packs to maintain samples at approximately 4°C. Samples were shipped via Federal Express and submitted for analysis of TCL VOCs, SVOCs, Pesticides/PCB, TAL Metals, Cyanide, TPH and Site-specific parameters.

All groundwater samples were collected in accordance with the protocols in the QAPP.

3.1.7.4 WATER LEVEL MONITORING

Groundwater elevations were measured to define the groundwater flow directions. Water level measurements were taken between August and October 1991 during the Phase I RI. Table 3.9 presents the groundwater elevations measured.

Water levels were used to determine the effects of seasonal fluctuations, and horizontal and vertical hydraulic gradients, which are discussed in Section 5.2. Water level versus time graphs (hydrographs) were plotted for selected monitoring wells in the Shallow and Bedrock Aquifer to illustrate the effects of seasonal fluctuations. These hydrographs are presented in Appendix F.

3.1.7.5 DOWNHOLE GEOPHYSICAL LOGGING

A downhole geophysical logging survey was conducted by GeEx during the period from October 16 to October 18, 1991 for the Phase I RI. The suite of geophysical logs used for this study were: Caliper, Self-Potential (SP), Natural Gamma-Ray, Electrical Resistance, and Temperature. The survey included bedrock monitoring wells DW-1, DW-2, MW-1D-91, MW-2D-91, MW-3D-91, and MW-4D-91. The investigation did not include the Town of Maybrook's production wells, as proposed in the SOP, since it would disrupt the Town's water supply.

The equipment used for the survey was a Century Geophysical Corporation's Compu-Log Portable Logging System with a combination SP, Natural Gamma-Ray, Single Point Resistance, and Temperature tool, and a separate 3-arm Caliper tool. At each borehole/well, the first log conducted was the Temperature log on the downward pass. These readings represented temperatures when the water column was least disturbed. The SP, Natural Gamma-Ray, Single Point Resistance logs, and a second Temperature log were conducted simultaneously in each bedrock well starting from the bottom. A Caliper log was conducted in each well using a separate tool, starting from the bottom. All logging tools and cables were rinsed with distilled water prior to being used in the next bedrock well in order to prevent cross-contamination. Results of the downhole geophysical logging are presented in the report prepared by GeEx "Report on the Geophysical Well Logging Study At The Former Lagoon Site For Nepera, Inc.", as presented in Appendix H.1.

3.1.8 SURFACE WATER AND SEDIMENT SAMPLING

Two rounds of surface water samples and one round of sediment samples were collected during the Phase I RI. The first round of surface water samples was collected during low flow conditions (August 13 and 14, 1991), and the second round of surface water samples was collected during high flow conditions (August 20 and 21, 1991). The high flow sampling round was collected within 48 hours of a rainfall event that produced runoff from the Site. One round of sediment samples was collected on August 13, 14 and 21, 1991. No surface water samples were collected from locations 1, 2 and 5 since these locations did not have sufficient quantities of water. Phase I surface water and sediment samples were analyzed for TCL VOCs, SVOCs, Pesticides/PCBs, TAL metals, cyanide, Site-specific parameters, and TPH.

Surface water and sediment sampling results are summarized in Tables 3.10 and 3.11, respectively. Surface water and sediment sampling locations are presented on Figure 3.3, and were chosen in consultation with the NYSDEC field representative.

All surface water and sediment samples were collected upstream of the sampler to minimize the potential for disturbing the sediments prior to sample collection. Sampling started at a downstream location and proceeded upstream. Stream characteristics (depth, width, flow direction, odor, color, etc.) and field conditions were noted in the sampling record.

Surface water samples were collected in pre-cleaned glass bottles provided by the laboratory. Samples were collected as close to mid-depth and mid-stream as possible at a point of sufficient depth to allow complete immersion of the collection bottles without the collection of bottom sediments. Surface water which was collected was subsequently transferred to sample bottles which contained the appropriate preservatives.

Conductivity, pH, and temperature measurements were obtained and recorded for each surface water sampling point. Turbidity was also measured for the Phase I RI surface water samples. Probes were rinsed with deionized water between tests.

Sediment samples were collected using a stainless steel piston sampler, or stainless steel ladles if the water was shallow. Care was exercised to avoid the loss of fine materials. Rocks and vegetative materials were removed from the samples. Sample jars were completely filled, capped and labeled.

Surface water and sediment samples were placed in the shipping container with ice or freezer packs to maintain samples at or below 4°C. Samples were subsequently shipped via Federal Express.

The results of the Phase I RI surface water and sediment sampling were presented to NYSDEC in Technical Memoranda No. 5 and 6.

3.1.9 MAGNETOMETER SURVEY

Although not originally included in the Phase I RI Work Plan, a magnetometer survey was requested by the NYSDEC after the completion of the Phase I test pit excavations.

The purpose of the magnetometer survey was to determine the location and extent of any additional drums and other magnetic anomalies in the area. The magnetometer survey was conducted at the Site during the period from October 14 to October 19, 1991 by GeEx of Slinger, Wisconsin under the direct supervision of CRA.

The magnetometer survey was performed in the area as presented on Figure 3.4, in accordance with the SOP Addendum No. 1. The SOP Addendum No. 1 was approved by the NYSDEC on September 18, 1991.

Grid positions were flagged every 20 feet with the coordinates labeled every 100 feet. The Site was surveyed using a GEMS GSM-19G Proton Precession Gradiometer Memory Magnetometer, with a spacing between stations and survey lines of 10 feet. A total of 2,235 stations were measured.

A base station was established along the abandoned railroad bed as indicated on Figure 3.4. Readings were taken at the base station approximately every 30 minutes to one hour during the survey period. These measurements were used to correct the survey data for drift of the magnetic field during the survey period.

The results of the magnetometer survey are presented in the GeEx report presented in Appendix H.2. Several anomalies are apparent, which may be caused by metal objects on the surface. However, the results of the magnetometer survey indicate one significant anomaly in the area of Lagoon 1. This area was further investigated during the test pit excavations and one buried drum was encountered, removed and overpacked.

The results of the magnetometer survey were presented to NYSDEC in Technical Memorandum No. 3.

3.1.10 EQUIPMENT CLEANING

All sampling equipment, drilling equipment and excavating equipment were cleaned as outlined in the approved QAPP. All decontamination water generated during the Phase I RI was stored at the decontamination pad in a separate 2,500-gallon compartment of a 5,000-gallon tanker. A sample of the decontamination water was collected and analyzed for TCL VOCs and no compounds were detected. This decontamination water was subsequently disposed.

3.2 PHASE II FIELD INVESTIGATIONS

The Phase II RI field investigation conducted at the Site included the following tasks to supplement the information obtained during the Phase I RI:

- evaluation of existing wells to determine which wells would be included in a revised Monitoring Well Network;
- installation of three (3) overburden wells and two (2) bedrock wells;
- conversion of existing monitoring wells DW-1 and DW-2 to piezometers to provide a definitive depth interval for determination of potentiometric water level elevations at these locations;
- collection of additional water level measurements for all monitoring wells;
- conducting single well response tests and short-duration pumping tests on selected wells;
- conducting water injection tests on the bedrock coreholes to determine water bearing fracture zones;

- groundwater sampling of all wells in the Monitoring Well Network including the newly installed bedrock and overburden wells;
- determination of surface water elevations at selected locations in Beaverdam Brook, Otter Kill, and the northeast swamp to determine probable groundwater flow patterns;
- additional surface water sampling during low, medium, and high flow conditions;
- additional sediment sampling in the northeast swamp and southeast marsh area;
- surficial soil sampling at each of the six (6) former lagoons and along the Site access road;
- evaluation of the "curtain drain";
- further evaluation of residential water supplies and wells in the vicinity of the Site;
- sampling of the Town of Maybrook potable water supply wells; and
- a Fish and Wildlife Impact Analysis (FWIA).

The investigative activities for the Phase II RI are further discussed in the following sections.

3.2.1 FISH AND WILDLIFE IMPACT ANALYSIS

An FWIA was conducted during the Phase II RI in accordance with the NYSDEC guidance document entitled "Fish and Wildlife Impact Analysis for Inactive Hazardous Waste Sites" (FWIA Guidance) dated October 1994.

The FWIA included in this RI Report consists of Steps I and II of the FWIA Guidance.

Step I includes the Site description including topographic, cover type, and drainage maps. Fish and wildlife resources within a one-half mile radius of the Site were identified and described. Step I also includes the identification of plant and animal species considered to be rare, endangered, or threatened, if present. The results of the Step I FWIA are presented in Appendix J.

Step II of the FWIA was conducted to determine the impact, if any, of Site-specific contaminants on fish and wildlife based on the following factors:

- contaminants of concern;
- contaminant concentrations in the various media;

- exposure of biota to contaminants; and
- toxicological effects of the exposures.

Step II followed a stepwise approach consisting of the following steps:

- pathway analysis;
- criteria-specific analysis; and
- toxic effect analysis.

The results of Step II are presented in Section 8.3 of this report.

The remaining steps of the FWIA; including Step III (Ecological Effects of Remedial Alternatives), Step IV (Fish and Wildlife Requirements for Implementation of Remedial Action), and Step V (Monitoring Program) will be completed, as required, during the FS and/or the remedial design/remedial action stages.

3.2.2 TEST PITS

Additional test pits (Phase II) were excavated on December 4 and 5, 1991, after the magnetic anomalies had been located as described in Section 3.1.9. The test pit locations are presented on Figure 3.1. Test pit logs and photographs are presented in Appendix B.1.

Five test pits were excavated during the Phase II RI and eleven samples were submitted for chemical analysis. Sampling locations were selected from the interval which indicated the highest potential for chemical presence based upon field screening (both visual and HNU/Jerome 411) and were determined in consultation with the NYSDEC field representative.

Test pit activities for Phase II were completed similarly to Phase I activities as described in Section 3.1.5.

An additional 55-gallon steel drum (D003) was removed from TP-47 during the Phase II excavations.

Samples from drums D001, D002 and D003 were collected in December 1991. All of these samples were submitted for chemical analyses which included TCL VOCs, SVOCs, Pesticides/PCB, TAL Metals, Cyanide, TPH and Site-specific parameters (pyridine, 2-amino pyridine, and alpha picoline).

A summary of the test pit and drum samples collected for laboratory analysis is presented in Table 3.3. Results of the chemical analyses are discussed in Section 6.0.

The results of the Phase II test pit activities were presented to NYSDEC in Technical Memoranda No. 6.

3.2.3 SURFACE SOIL SAMPLING

Eight surface soil samples were collected and analyzed during the Phase II RI from the six former lagoons and along the Site access road at the locations presented on Figure 3.1. The surface soil samples were collected to further delineate surface soil conditions on and in the vicinity of the lagoons.

Surface soil samples were collected from the top three inches of soil in accordance with the protocols and procedures presented in the SOP and QAPP. A summary of surface soil sample collection is presented in Table 3.12.

Surface soil samples were analyzed for TCL VOCs, SVOCs, Pesticides, PCBs, TAL inorganics, TOC, and Site-specific compounds. Results of the analyses are discussed in Section 6.1.

3.2.4 HYDROGEOLOGIC EVALUATION

The Phase II RI included a supplemental hydrogeologic investigation to further characterize the Site hydrogeologic setting. The Phase II RI evaluation included the following:

- completion of a Monitoring Well Reconnaissance;
- installation of two (2) bedrock monitoring wells (MW-5D-95 and MW-6D-95) and three (3) overburden monitoring wells (MW-5U-95, MW-7U-95, and MW-8U-95);
- conversion of two (2) existing monitoring wells (DW-1 and DW-2) to piezometers (DW-1-95 and DW-2-95);
- water injection tests (packer test) on the bedrock coreholes to determine water bearing fracture zones;

- well development and groundwater sampling of newly-installed monitoring wells and selected existing monitoring wells;
- water level measurements in all newly-installed and existing monitoring wells and piezometers;
- single-well response tests and short-duration pumping tests on selected monitoring wells; and
- survey of all newly-installed Phase II RI monitoring wells for vertical and horizontal control.

Details of the hydrogeologic evaluation conducted during the Phase II RI are presented in the following subsections.

3.2.4.1 NEW MONITORING WELL INSTALLATION

The Phase II RI included drilling and installation of two bedrock and three overburden monitoring wells. In addition, two existing monitoring wells (DW-1 and DW-2) were converted to bedrock piezometers in accordance with the Phase II RI Work Plan. The monitoring wells were installed by Marcor of New York, Inc., of Rochester, New York between the period of May and June 1995 under the supervision of CRA personnel. The locations of the boreholes and monitoring wells are presented on Figure 2.1.

Overburden Monitoring Wells

For the Phase II RI, a Canterra CT 150 truck mounted drill rig was used to install the overburden monitoring wells (MW-5U-95, MW-7U-95, and MW-8U-95). The overburden monitoring well completion details are presented in Table 3.5, and the stratigraphic and instrumentation logs are presented in Appendix D.4.

Soil samples were retrieved in accordance with ASTM Method D1586 using 2-inch and 3-inch diameter split spoon samplers. Soil sample headspace readings were measured and recorded.

The monitoring wells were installed using the hollow-stem auger method. Each borehole was completed with a monitoring well using a 5-foot length of 2-inch diameter stainless steel, 0.010-inch continuous slot well screen, attached to 2-inch diameter stainless steel riser pipe and end cap, except at MW-5U-95. A ten-foot stainless steel screen with the same slot size and diameter specifications was installed in MW-5U-95. The deviation of using 2-inch diameter instead of 4-inch diameter riser pipe and screen, as stipulated in the SOP, was approved by the NYSDEC. Quartzite sand (#4) was placed in the annular space to a height of at least one foot above the top of each well screen. A 2-foot bentonite seal was placed above the sandpack. The annulus above the bentonite seal was tremie backfilled with neat cement/bentonite grout to ground surface. A steel locking protective casing was placed over each riser pipe and was embedded in the cement/bentonite grout. Padlocks were placed on the protective casings for security. A minimum of three protective posts were installed around the wells to complete the installation.

With NYSDEC approval, the well screens were installed in the saturated zone where the PID detected the highest soil headspace readings.

Auger cuttings from the borings were distributed evenly on the ground in the general vicinity of each monitoring well, in accordance with the SOP.

Bedrock Monitoring Wells

Two bedrock monitoring wells (MW-5D-95 and MW-6D-95) were installed using a Canterra CT 350 truck-mounted drill rig. A summary of bedrock well completion details is presented in Table 3.5, and stratigraphic and instrumentation logs are presented in Appendix D.4.

Soil samples within the overburden were retrieved in accordance with ASTM Method D1586 using 2-inch and 3-inch diameter split spoon samplers.

The bedrock was cored using HQ-size coring equipment during the Phase II RI. The core was logged in the field for its composition, formation name, bedding, orientation, fractures, joints, folding and faulting, core length, percent recovery and RQD. The cores were placed into labeled core boxes which were subsequently sent for storage to the Nepera, Inc. facility in Harriman, New York.

With the concurrence of the NYSDEC, the bedrock wells were completed as 3 7/8-inch diameter open hole wells using wet-rotary methods, instead of 6-inch diameter open hole wells using air-rotary methods, as proposed in the SOP. Water was utilized as the drilling fluid. The bedrock wells were installed using the same procedures as the Phase I installations.

In May 1995, existing monitoring wells DW-1 and DW-2 were converted to piezometers at depth intervals of 115 to 120 feet bgs and 100 to 105 feet bgs, respectively. These bedrock wells were originally open cored holes monitoring as much as 160 feet (DW-2) of the bedrock aquifer. Each piezometer was completed using a 5-foot length of 2-inch diameter stainless steel, 0.010-inch continuous slot well screen, attached to 2-inch diameter stainless steel riser pipe and end cap. Quartzite sand (#4) was placed in the annular space to a height of at least 3 feet above the top of each well screen. A 2-foot bentonite seal was placed above the sandpack. The annulus above the bentonite seal was tremie backfilled with neat cement/bentonite grout (to ground surface). A steel locking protective casing was placed over the riser pipe and was embedded in the cement/bentonite grout. Padlocks were placed on the protective casing for security. A minimum of three protective posts were installed around the piezometers to complete the installation.

All bedrock monitoring wells were installed in accordance with the SOP and Phase II RI Work Plan with the exception of MW-5D-95. With the concurrence of NYSDEC, the inner 4-inch diameter surface casing was set into 17 feet of competent bedrock (36 feet bgs). No major water bearing fracture zones were encountered at MW-5D-95 until 88.5 feet bgs. Therefore, MW-5D-95 was completed as an open hole with a monitoring interval between 37 feet and 97 feet bgs.

3.2.4.2 WELL DEVELOPMENT

Monitoring wells were developed utilizing surging, pumping and bailing techniques. Development was accomplished using stainless steel and steel bailers, and a 2-inch diameter stainless steel submersible pump.

A minimum of four well volumes were removed from each monitoring well, with the exception of MW-6D-95, MW-7U-95, and DW-1-95. Monitoring well MW-7U-95 was dry at the time of development. Only one well volume could be removed from MW-6D-95 and DW-1-95 before they went dry. MW-6D-95 was pumped and DW-1-95 was bailed. To ensure that MW-6D-95 and DW-1-95 were stabilized, a minimum of three well volumes were removed (purged) from the well prior to sampling (see Table E.1, Appendix E). The purged water was tested for pH, conductivity, temperature and turbidity to determine chemical stability and clarity. The wells were considered to be stabilized when three consecutive and consistent readings of the stabilization parameters were achieved. Development records for the RI monitoring wells are presented in Table 3.6.

3.2.4.3 MONITORING WELL RECONNAISSANCE

An initial inventory of all monitoring wells, including the wells installed during the Phase I RI, was conducted by CRA on July 12, 1991. Four years after this initial inventory, the monitoring wells were re-evaluated by CRA on January 11, 1995 to assess which wells would be used for either hydraulic or chemical monitoring during the Phase II RI. The results of this reconnaissance are presented in Table 3.13. The monitoring well evaluation was utilized to determine which monitoring wells would be included in the revised Monitoring Well Network. The revised Monitoring Well Network consists of existing on-Site monitoring wells and new on-Site and off-Site monitoring wells installed as part of the Phase II RI. The revised monitoring well network was agreed upon by the NYSDEC and CRA (representing Nepera/WLC). The Monitoring Well Reconnaissance completed on January 11, 1995 was submitted as a Technical Memorandum to NYSDEC on January 26, 1995.

3.2.4.4 GROUNDWATER SAMPLING

Groundwater sampling was conducted during the period from June 5 to June 9, 1995 for the Phase II RI. The Monitoring Well Network for the sampling event is presented in Table 3.7. The summary of groundwater sample collection is presented in Table 3.8.

Monitoring wells MW-6, SW-1, and MW-7U-95 were not sampled due to insufficient volume of water in the monitoring wells. The Town of Maybrook supply wells were also sampled in June 1995. A composite sample was submitted for analysis for Town wells #1, 2 and 3. A dug well ("A") located approximately 20 feet north of the Site on County Route 4 (see Figure 2.3) was not sampled, as the well appeared to be dry and contained metal and wood debris.

A minimum of three well volumes were either pumped or bailed from each well prior to sampling for all parameters. Well purging was completed utilizing a 2-inch stainless steel submersible pump or stainless steel bailers. Purged water was discharged directly to the ground surface in accordance with the SOP. Purge records are presented in Appendix E.

Samples were collected from each well to determine pH, conductivity, temperature and turbidity.

Samples for chemical analyses were removed directly from the wells and poured into sample jars (with the appropriate preservatives) utilizing pre-cleaned stainless steel bailers.

After sampling each monitoring well, the sample jars were placed directly into the shipping containers with ice or freezer packs to maintain samples at approximately 4°C. Samples were shipped via Federal Express and submitted for analysis of TCL VOCs, SVOCs, Pesticides/PCB, TAL Metals, Cyanide, TPH and Site-specific parameters. Groundwater samples collected for the Phase II RI were also analyzed for chloride and dissolved metals.

All groundwater samples were collected in accordance with the protocols in the QAPP. Equipment cleaning protocols are presented in Section 3.1.10.

3.2.4.5 WATER LEVEL MONITORING

Water level measurements were recorded six times during the Phase II RI period to address the comments from the NYSDEC on the Phase I RI. Measurements were taken from May 12, 1993 to June 17, 1993, January 12, 1995, June 1 and June 20, 1995 and July 24, 1995. These groundwater elevations are discussed in Section 5.2 and summarized in Table 3.9. Historical water elevations measured by C.A. Rich Consultants, Inc. are also presented in Table 3.9.

It is important to note that CRA's and C.A. Rich's survey datum differ. The historical water level elevations measured by C.A. Rich presented in Table 3.9 have not been corrected or recalculated. Monitoring wells MW-2 and MW-7 were re-surveyed in June 1995; therefore, the 1995 water elevations are based on the new survey results. Water levels were measured from the top of the well casings using a Solinst electronic water level meter. The AMSL elevation of the water was determined for each well with reference to the top of the well casing elevations.

Water levels were used to determine the effects of seasonal fluctuations, and horizontal and vertical hydraulic gradients, which are discussed in Section 5.2. Water level versus time graphs (hydrographs) were plotted for selected monitoring wells in the Shallow and Bedrock Aquifer to illustrate the effects of seasonal fluctuations. These hydrographs are presented in Appendix F.

3.2.4.6 IN SITU HYDRAULIC CONDUCTIVITY

Single-Well Response Test

During the Phase I RI, single-well response tests were conducted on the newly installed bedrock monitoring wells (MW-1D-91, MW-2D-91, MW-3D-91, and MW-4D-91) on July 13, 1991, and on the overburden monitoring well MW-1U-91 on October 18, 1991. On May 12, 1993, monitoring well MW-1U-91 was retested to verify the 1991 results.

As recommended by NYSDEC (comments dated April 12, 1993), additional in situ hydraulic conductivity tests were conducted during the Phase II RI in accordance with the Phase II RI Work Plan. Single-well response tests were performed on the new monitoring wells installed in May and June 1995 and selected existing wells (including the monitoring wells on which tests were performed in 1991) to further refine the hydrogeologic parameters. The 1995 single-well response tests (including the test conducted in 1993 for MW-1U-91) are provided in Appendix G.1.

Monitoring wells where single-well response tests were conducted in June and July 1995 may be divided into three sections: north, south, and former lagoon area, as follows:

i) North - MW-2, MW-3, SW-2, SW-3, SW-10, MW-5U-95, and MW-8U-95;

ii) South - MW-7, SW-8, SW-9, and MW-1U-91 (note: single-well response test conducted on MW-1U-91 was performed on May 12, 1993); and

iii) Former Lagoon Area - SW-4.

Single-well response tests involve changing the water level within a monitoring well and measuring the rate of recovery of the water level to static position. Prior to each test, the static water level measurement was recorded. A slug of known dimensions was introduced into the water column to displace the water level. Measurements of depth to water and the corresponding time were recorded until the water level reached its static level (falling head test). After the water level reached its static level, the slug was removed and depth to water and the corresponding time were recorded again until the water level rose to its static level (rising head test). Water levels in the tested monitoring well were measured at timed intervals using a Solinst™ electric water level indicator and/or Telog™ datalogger and pressure transducer.

A falling head test is not valid in the case where the static water level in the monitoring well is below the top of the well screen or open hole section. If a known volume of water is added to the well, the subsequent flow of water into the aquifer due to the falling level will occur through the well screen (or open section) below the original water table as well as through the vadose (unsaturated) zone above the original water table. This result increases the rate of fall of the water level in the well and therefore, leads to an overestimation of the hydraulic conductivity. A falling head test was not conducted on SW-3, SW-4, SW-8, SW-9, SW-10 as the static water levels in these wells were below the top of the well screen.

Rising head tests were only conducted on DW-1-95 and DW-2-95. Approximately 0.875 and 1.44 gallons were removed by means of a 2-inch diameter stainless steel bailer from DW-1-95 and DW-2-95, respectively. The water level in the piezometers were measured (at timed intervals) subsequent to the withdrawal of water from the well.

Subsequent to the initial falling/rising head test conducted on SW-2, a second rising head test was performed. Approximately 0.875 gallons were removed by means of a 2-inch diameter stainless steel bailer from SW-2. The water level at SW-2 was measured (at timed intervals) subsequent to the withdrawal of water from the well.

The Bouwer and Rice Method (Bouwer, 1976) was used to evaluate the single-well response test results and determine the hydraulic conductivity of the Shallow and Bedrock Aquifers in the immediate vicinity of each well. The results of the single-well response tests are summarized in Table 3.14. The Bouwer and Rice Method was selected because of the unconfined conditions in both the Shallow and Bedrock Aquifers.

The recovery data for the monitoring wells were plotted and analyzed using the AQTESOLV™ computer software package Version 2.0 (Geraghty and Miller, 1988 to 1994) to determine the hydraulic conductivity. The diagnostic plots and water level data are found in Appendix G.

Based on the results of the single-well response tests conducted in June and July 1995 and May 1993 (MW-1U-91), the geometric mean of the hydraulic conductivity for the Shallow Aquifer and Bedrock Aquifer was determined. In the northern, southern and former lagoon area of the Shallow Aquifer, the hydraulic conductivities were calculated as 9.85×10^{-4} cm/s, 8.75×10^{-4} cm/s and 1.07×10^{-4} cm/s, respectively. In the northern, southern, and former lagoon area of the Bedrock Aquifer, the hydraulic conductivities were calculated as 3.19×10^{-6} cm/s, 1.18×10^{-4} cm/s and 4.29×10^{-4} cm/s, respectively. Data is tabulated on Table 3.14.

It should be noted that the hydraulic conductivity calculated from DW-2-95 (5.22×10^{-7} cm/s) is lower than one would expect based on the results of the water injection tests (included in this section) and the caliper logs (C.A. Rich, 1986 and Appendix H.1).

Short-Duration Pumping Tests

Short-duration pumping tests were conducted during the Phase I RI at bedrock monitoring wells MW-2D-91 and DW-2 on October 14, 1991 in order to determine the potential interconnection between the overburden and bedrock. These bedrock wells were chosen since they were in close proximity to overburden wells which could be monitored during the test. The pumping test monitoring network for bedrock well MW-2D-91 included overburden wells SW-3, SW-4, PZ-2, MW-3, and bedrock well MW-1D-91. The pumping test monitoring network for bedrock well DW-2 included overburden wells SW-2 and MW-2. Each pumping test was conducted for a period of approximately 2 to 4 hours and water levels were measured in the selected wells. These activities and results are described in Hydrogeologic Investigation Technical Memorandum No. 4.

Bedrock monitoring well MW-2D-91 was pumped at a rate of 0.3 gpm for 90 minutes followed continuously by 1.0 gpm for 150 minutes. The total drawdown resulting from pumping at 0.3 gpm and 1.0 gpm was 8 feet and 27 feet, respectively. No drawdown was observed in the monitored observations wells, most notably at adjacent shallow monitoring well SW-4. Monitoring well MW-2D-91 was retested in 1995 and these later results were used during subsequent analysis.

Bedrock monitoring well DW-2 was pumped at 3.6 gpm and 5.6 gpm for a total period of 2.0 hours. The total drawdown resulting from pumping at 3.6 gpm and 5.6 gpm was 11 and 15 feet, respectively. Adjacent shallow monitoring well SW-2 responded quickly to the pumping of DW-2. However, inspection of DW-2 identified flowing water entering the DW-2 casing at a depth of approximately 10 feet. It was concluded that the water was either flowing into DW-2 from a hole in the casing or at the contact of the casing and the bedrock. Review of C.A. Rich's Summary Report, (C.A. Rich, 1986) indicates that the top of bedrock was encountered at 9 feet bgs at this location. Monitoring well SW-2 is screened in the overburden and bedrock from 7 to 17 feet bgs. DW-2 was cased to a 15 feet bgs and completed as an open bedrock hole to a depth of 175 feet. This indicates that the effective monitoring intervals of these two wells overlap the interval between 15 and 17 feet bgs (6 to 8 feet below the top of bedrock). As such, this test was not suitable to evaluate the interconnectivity between the overburden and bedrock.

To further define hydrogeologic properties in the bedrock aquifer, short duration pumping tests were conducted on MW-1D-91, MW-2D-91, MW-3D-91, and MW-4D-91 during June and July 1995. The duration of pumping tests conducted on MW-1D-91, MW-2D-91, MW-3D-91, and MW-4D-91 were 4.4 hours, 2 hours, 1 hour, and 2.53 hours, respectively. The pumping rate for MW-1D-91, MW-2D-91, MW-3D-91, and MW-4D-91 were 1.08 gpm, 0.85 gpm, 1.0 gpm, and 1.46 gpm, respectively. The water was pumped from each well using a 2-inch diameter submersible Grundfos™ pump. The evacuated water was discharged directly to the ground surface in accordance with the SOP.

Water level measurements were obtained using a Solinst™ electronic water level indicator. The water levels were measured only at the pumped well to determine in situ hydraulic conductivity values.

Following the completion of the pumping portion of each short duration test, the pump was shutdown and the water level was measured in the pumped well until 70 percent of the measured drawdown in the pumped well recovered.

The Theis Method (Theis, 1935) was used to evaluate the test results to determine the hydraulic conductivity of the bedrock in the immediate vicinity of the well. Although the bedrock aquifer is unconfined, the Theis Method was appropriate in evaluating the test results as the time-drawdown data conformed to the Theis-type curves.

The drawdown and recovery data for the monitoring wells were plotted and analyzed using the AQTESOLV™ computer software package Version 2.0 to determine the transmissivity. The drawdown and recovery data for the pumped wells during the short-term tests are presented in Appendix G.2.

A summary of the estimated transmissivity values obtained using the Theis Method is presented in Table 3.15. The transmissivity in the former lagoon area ranges between 4.38 ft^2/day and 28.02 ft^2/day, with a mean transmissivity of 13.8 ft^2/day. The transmissivity in the southern portion of the Site ranges between 19.12 ft^2/day and 75.31 ft^2/day, with a mean transmissivity of 54.3 ft^2/day.

Transmissivity, which is a rate at which water is transmitted through a unit width of aquifer under a unit hydraulic gradient, can be used to calculate the hydraulic conductivity using the following relationship (Driscoll, 1986):

$$T = Kb$$

where:

T = transmissivity (ft^2/day)
K = hydraulic conductivity (feet/day)
b = aquifer thickness (feet)

The saturated thickness in the immediate vicinity of each pumped well is also summarized in Table 3.15. Using the above equation, the hydraulic conductivities of the Bedrock Aquifer in the former lagoon area and southern portion of the Site were calculated as 2.54×10^{-4} cm/s and 1.38×10^{-3} cm/s, respectively.

The hydraulic conductivities calculated from the single-well response tests for monitoring wells: MW-1D-91, MW-2D-91, MW-3D-91, and MW-4D-91 (see Table 3.14) are similar to the hydraulic conductivities calculated from the corresponding short-duration pumping tests (see Table 3.15).

Water Injection Tests

Water injection tests were conducted on the 3 7/8-inch diameter cored holes at monitoring well locations MW-5D-95, MW-6D-95, DW-1-95, and DW-2-95. The purpose of the water injection tests was to determine the water-bearing fracture zones. The hydraulic conductivity at various depth intervals within the coreholes being tested was also determined.

Water injection tests were conducted using the packer test method, which consisted of the injection of potable water at a constant pressure and flow rate into sealed intervals of the coreholes. All intervals tested were 5.5 feet in length. The injection rate was controlled using a gate valve in the injection line and the water pressure was measured using an in-line pressure gauge.

At each tested interval, the packers were inflated to seal off the selected interval of the corehole. Potable water was injected into the interval at constant pressure for five minutes. For each interval tested, the test was repeated for five settings. The flow rates and injection pressure were monitored once a minute during each 5-minute test. The elapsed time, injection pressures and flow rates were recorded.

A deviation from the above procedure was necessary after several attempts were made to perform water injection tests on DW-2-95. The packer assembly could not be securely seated in the open core interval of this well. The packer assembly was removed and fully inflated above ground to ascertain that the packer was inflating properly. The packer was reinserted and several attempts were made to seal it at different intervals between 100 and 120 feet bgs. The Caliper log for DW-2 (C.A. Rich, 1986) indicated that, due to probable fracturing in the open-core interval, the borehole diameter was inconsistent and often in excess of six inches in diameter, which is likely due to washouts or zones of caving in the fractured area.

Based on the information in the Caliper log (C.A. Rich, 1986 and Appendix H) and the attempted injection tests, concurrence between NYSDEC/Nepera/WLC/CRA was reached to have the monitoring interval at DW-2-95 at 100 to 105 feet bgs.

No problems were encountered during the water injection tests at DW-1-95.

After the water bearing fracture zones for DW-1-95 and DW-2-95 were determined, cement/bentonite grout was used to fill the open core interval below the interval to be screened, the grout was allowed to cure for twenty-four hours or longer. The piezometers were then completed in accordance with the SOPs.

Water injection tests were attempted on MW-5D-95 and MW-6D-95. Less than 0.20 gallons of water was injected at MW-5D-95 between the interval of 71 feet to 97 feet bgs. No visible change in pressure was noted while the packer was inflated which indicated that there were limited fractures within this tested interval. No major water bearing zones were encountered at MW-5D-95 until 88.5 feet bgs. Therefore, MW-5D-95 was completed as an open hole with a monitoring interval between 37 feet and 97 feet bgs.

The packer assembly could not be securely seated in the open corehole interval (62.5 to 72 feet bgs) at MW-6D-95. The packer assembly was removed and fully inflated aboveground to ascertain that the packer was inflating properly. The packer was re-inserted in attempt to test the same interval. Again the packer assembly could not be securely seated due to the highly fractured nature of this open interval. It was concluded that the interval between 62.5 to 72 feet bgs was highly fractured. Well MW-6D-95 was set at this interval.

The hydraulic conductivity for each tested interval was calculated using the following formula:

$$K = \frac{Q}{2pLHo} \ln \left\{ \frac{L}{2r_o} + \sqrt{1 + (L/2r_o)^2} \right\} \qquad \text{(Hvorslev, 1951)}$$

where:

K = hydraulic conductivity (centimeters per second);
Ho = injection pressure (centimeters of water);
L = length of interval (centimeters);
ro = radius of borehole (centimeters), and
Q = flow rate (cubic centimeters per second).

The hydraulic conductivity values calculated for DW-1-95 from the water injection tests are presented in Table 3.16. The results for DW-1-95 indicate lower hydraulic conductivities at depth.

3.2.4.7 WATER WELL INVENTORY

A historical water well inventory was previously completed by C.A. Rich (Rich, 1986) within a radius of approximately two miles from the Site. In June 1995, CRA updated the water well inventory to include residents with water wells located within the immediate vicinity of the Site. A summary of the water wells which were identified is presented in Table 3.17. The locations of the water wells are presented on Figure 3.2.

3.2.4.8 INFILTRATION ESTIMATION

Surface infiltration was estimated at areas both on and off the surface of the former lagoons using a percolation test. Additionally, surficial soils were collected for grain size analysis so that the estimates of saturated permeability could be made using the Hazen method. Subsequently, the surface layer of overburden was modelled utilizing the Hydrological Evaluation of Landfill Performance computer model (HELP, Version 2.0). The results of the infiltration rate assessment were presented in Technical Memoranda No. 9 and are further evaluated in Section 5.3.

3.2.5 SURFACE WATER AND SEDIMENT SAMPLING

During the Phase II RI, three rounds of surface water samples were collected at the locations presented on Figure 3.3. One round was collected during a period of low flow (June 5, 1995). The second round was collected during a period of relatively high water table elevations (June 14, 1995). The third round was collected within 48 hours following a significant rainfall event which resulted in runoff from the Site (June 19, 1995).

Two sediment samples were collected during the Phase II RI. Sample SDII-2 was collected from the northeast swamp in accordance with the Phase II RI Work Plan. A second sediment sample, SDII-12, was added to the program at the request of the NYSDEC (NYSDEC letter dated April 13, 1995 to Nepera) to address concerns about the possibility of contaminated sediments existing in a ponded area of Beaverdam Brook southwest of the Site.

Phase II surface water and sediment samples were analyzed for TCL VOCs, SVOCs, pesticides/PCBs, TAL metals, cyanide, and Site-specific parameters. Phase II surface water samples were also analyzed for hardness while Phase II sediment samples were also analyzed for total organic carbon.

Surface water and sediment sampling results are summarized on Tables 3.10 and 3.11, respectively. Surface water and sediment sampling locations are presented on Figure 3.3, and were chosen in consultation with the NYSDEC field representative.

All surface water and sediment samples were collected with similar procedures to the Phase I sampling outlined in Section 3.1.8.

3.2.6 "CURTAIN DRAIN" INVESTIGATION

Following the completion of the Phase I RI, the NYSDEC requested that the "curtain drain" described in a New York Department of Health (NYDOH) Memorandum dated June 1, 1967 be investigated.

The NYDOH memorandum presents the following information pertaining to the construction of the "Curtain Drain":

- on May 11, 1967 a contractor was observed in the vicinity of a previously identified wastewater breakout north of the lagoons. The contractor was in the process of installing the "Curtain Drain";
- the bottom of the "Curtain Drain" was to contain gravel and perforated orangeburg pipe;
- the ditch was completed and backfilled to a point adjacent to the driveway where an existing culvert pipe was located;
- a ditch on the west side of the driveway was excavated but no gravel was put in this trench;
- the orangeburg pipe extended from the "Curtain Drain", under the driveway, and into the open ditch on the west side of the driveway;
- the "Curtain Drain" pipe runs several feet under the culvert; and
- a dark hued wastewater with an odor described as chemical in nature was observed running from the "Curtain Drain" pipe.

The "curtain drain" is believed to have been installed as a typical perforated french drain to convey groundwater away from the lagoon area.

In accordance with the NYSDEC request, an assessment of the "Curtain Drain" was made during the Phase II RI to determine its operational status and impact of Site-related chemicals, if any, in the area of the "Curtain Drain".

Field work for the "Curtain Drain" investigation was conducted on June 29, 1995 in accordance with the "Curtain Drain" Investigation Work Plan outlined in letters to the NYSDEC from CRA dated June 5, 1995 and June 13, 1995.

A New York State-licensed asbestos abatement contractor, Marcor of New York Inc., was retained to excavate the test pits as the "Curtain Drain" was constructed of "orangeburg" pipe which was identified by the NYSDEC as a potential asbestos-containing pipe.

Five test pits (TP-48, TP-49, TP-50, TP-51, and TP-52) were excavated to determine the alignment and limits of the "Curtain Drain". The locations of the test pits are presented on Figure 3.1. Test pit logs are presented in Appendix B.2.

The test pits were excavated with a backhoe in approximately 12-inch passes. All excavated material was temporarily stockpiled on polyethylene sheeting adjacent to each test pit. Following logging and sampling, each test pit was backfilled. Excavated materials were placed roughly in the same location of the test pit from which they were removed. The backhoe bucket was used to compact the material returned to each test pit. The backhoe used for excavation, or any portions of it coming into contact with potentially contaminated soil, was steam cleaned prior to demobilization from the Site.

In accordance with the "Curtain Drain" Investigation Work Plan, soil samples were to be collected from the test pits that exhibited a potential for contamination based on visual and PID screening. However, based on field observations, no evidence of contamination was encountered during the excavation of the test pits. Soil samples were, therefore, collected at locations determined in the field with the concurrence of the NYSDEC field representative. Soil samples were analyzed for TCL VOCs, SVOCs, Pesticides/PCBs, TAL metals, cyanide, TOC, and Site-specific parameters in accordance with the protocols and procedures presented in the SOP and QAPP. Soil samples were collected from the backhoe bucket. A summary of soil samples collected during the "Curtain Drain" investigation is presented in Table 3.18.

Groundwater was not encountered during the investigation of the "Curtain Drain" (i.e., the drain was not operating).

The "Curtain Drain" was investigated by the excavation of test pits beginning with TP-48 located approximately 10 feet west of the culvert. The "Curtain Drain" pipe was encountered at a depth of approximately 7 feet bgs running in an east-west orientation. The 4-inch diameter pipe was not perforated. During the excavation of TP-48, the backhoe damaged the pipe. The broken pieces of the asbestos-containing pipe were immediately removed from the excavation, bagged, handled, and disposed of in accordance with applicable federal and New York State regulations including 12 NYCRR Part 56. The interior of the pipe was partially filled with sediment. No groundwater was encountered within the pipe or the test pit. No evidence of chemical contamination was encountered at TP-48 based on field observations.

TP-49 was excavated approximately 80 feet west of TP-48. The perforated "Curtain Drain" pipe was encountered at a depth of approximately 4 feet bgs running in an east-west orientation. With the concurrence of the NYSDEC field representative, a soil sample was collected from just below the pipe bedding. No evidence of chemical contamination was encountered at TP-49 based on field observations.

TP-50 was excavated approximately 65 feet west of TP-49. The "Curtain Drain" pipe was not encountered at TP-50 which was excavated to a depth of approximately 5 feet bgs. Although the western end of the "Curtain Drain" was not encountered, the NYSDEC field representative indicated that the general extent of the "Curtain Drain" had been adequately defined to the west of the Site access road. No evidence of chemical contamination was encountered at TP-50 based on field observations.

The investigation of the "Curtain Drain" to the east of the Site access road was initiated at TP-51 located approximately 90 feet east of the access road. The "Curtain Drain" pipe was encountered at a depth of approximately 4 feet bgs running in an east-west orientation. With the concurrence of the NYSDEC field representative, investigative and duplicate soil samples were collected from silty clay immediately beneath the pipe. No evidence of chemical contamination was encountered at TP-51 based on field observations.

TP-52 was excavated approximately 73 feet east of TP-51. The "Curtain Drain" pipe was not encountered at TP-52 which was excavated to a depth of approximately 6 feet bgs. As directed by the NYSDEC field representative, a 3-point composite sample was collected from the silty material within the bedding which consisted primarily of 2-inch diameter stones at approximately 2 to 3.5 feet bgs. Although the eastern end of the "Curtain Drain" was not encountered at TP-52, the NYSDEC field representative indicated that the general extent of "Curtain Drain" had been adequately defined to the east of the Site access road. No evidence of chemical contamination was encountered at TP-52 based on field observations.

The test pits were surveyed for horizontal and vertical control as presented on Figure 3.1. The test pit logs are presented in Appendix B.2.

The analytical results for the "Curtain Drain" investigation soil samples are discussed in Section 6.3.

3.3 TREATABILITY STUDY INVESTIGATIONS

CRA conducted a Treatability Study to determine the effectiveness of Soil Vapor Extraction (SVE) and bioremediation technologies as potential remedies for the Site. The Treatability Study was conducted in accordance with the Soil Vapor Extraction/Bioremediation Bench Scale Treatability Study Work Plan (Treatability Study Work Plan) dated September 16, 1996.

Test pitting activities were conducted in accordance with the Treatability Study Work Plan during the period November 21 to 26, 1996. Test pit locations were established in the field by the NYSDEC field representative and CRA's field supervisor. Sample selection activities were coordinated with both the U.S. EPA and the NYSDEC field representatives.

A total of 23 investigative samples were collected from the following 15 test pits:

Lagoon	*Test Pit*	*Number of Investigative Samples*	*Samples/Lagoon*
1	1	2	4
	2	0	
	3	2	
2	1	2	3
	2	1	
3	1	1	7
	2	3	
	3	3	
4	1	2	5
	2	3	
5	1	1	3
	2	2	
6	1	0	1
	2	0	
	3	1	

A test pit sampling summary is presented in Table 3.19. Test pit locations are presented on Figure 3.1. Test pit logs are provided in Appendix B.3. A photographic log of the material excavated from each of the test pits is provided in Appendix K.3.

Investigative and duplicate samples were analyzed for the following parameters to determine levels of the Target Compounds, chemical and physical characteristics, and baseline nutrient levels:

- total organic carbon (TOC) by EPA Modified Method 160.4;
- pH by U.S. EPA Method 9045;

- orthophosphate phosphorus by U.S. EPA Modified Method 365.3;
- ammonia nitrogen by U.S. EPA Modified Method 350.2;
- soil particle size by ASTM D22;
- TCL VOCs;
- TCL SVOCs, including Site-specific parameters;
- TCL PCB/pesticides; and
- TAL inorganics.

The validated analytical results for these parameters along with the New York State Soil Cleanup Objectives are summarized in Section 6.0. The analytical results are included in the Appendix I.1 tables and the validation report is presented in Appendix L.2.

The test pit samples were also analyzed for grain size distribution. The results of the grain size distribution analyses are summarized in Table 3.20. Consistent with the findings of the RI, the grain size distribution results indicate that the lagoon material is generally sand with some gravel and varying amounts of silt and clay.

3.4 INTERIM GROUNDWATER MONITORING PROGRAM

Following the completion of the Phase II RI groundwater sampling, selected sentry wells were monitored semi-annually for the Interim Groundwater Monitoring Program beginning in 1997. The Interim Groundwater Monitoring Program included wells MW-1, SW-9, T-2, MW-5D-95, and MW-5U-95 in 1997 and 1998, and well MW-8U-95 in May and August 1998 as well as April and August 1999. These samples were analyzed for benzene, toluene, ethylbenzene, xylenes (BTEX) and Site-related pyridine based compounds as identified by the NYSDEC.

At the October 21, 1998 meeting with U.S. EPA and the NYSDEC, U.S. EPA expressed some potential concerns regarding the adequacy of the current groundwater monitoring well network and groundwater quality database for the Site. It was suggested that the respective project hydrogeologists from the U.S. EPA, NYSDEC, and CRA review the groundwater monitoring well network to determine if any additional wells are required, and to evaluate the current condition of the existing older wells.

Given the hydrogeologic conditions (groundwater flow paths) and groundwater quality beneath the Site, and a review of the existing monitoring well network, an additional eight new groundwater monitoring wells were installed during the Natural Attenuation

Study in June 2001 to further investigate the groundwater contaminant plume in the overburden and bedrock at the Site. The locations of the monitoring wells were selected in the field in conjunction with the U.S. EPA and the NYSDEC during a Site Reconnaissance visit on March 15, 2000. The installation of these wells were to provide further definition of the groundwater plume, particularly along property boundaries, and would also serve as perimeter sentry wells during and following remediation. The well installation and development details are defined in Section 3.6.

Aside from the pre-mentioned wells, beginning July 2001, the Interim Groundwater Monitoring Program included the newly installed overburden wells: MW-9U-01, MW-10-01, MW-11-01; the newly installed bedrock wells: MW-9D-01, MW-10D-01, MW-11D-01, MW-12D-01, MW-13D-01; and the converted well MW-5D-95 which was changed from an open corehole to a screened interval. (Well conversion details can be found in Section 3.6). These samples were also analyzed for the contaminants of concern.

Table 3.21 summarizes the wells included, and contaminants analyzed in each groundwater sampling event to date. A summary of the samples is presented on Table 3.22. Analytical results from the interim groundwater monitoring program can be found in Section 6.0.

3.5 CULTURAL RESOURCE SURVEY

Hunter Research conducted a Phase I Archeological Survey on the former Maybrook Lagoon site in 2001 to establish whether cultural resources existed on or within the Area of Potential Effect (APE) of the site activities. This section summarizes the Cultural Resource Survey prepared for CRA by Hunter Research in January 2002. The full report can be found in Appendix M. The APE adopted corresponds to the project boundary. This Phase I survey was authorized as part of remedial operations on this Superfund site, under CERCLA.

No previous archaeological, architectural, or historic research had been reported on the Maybrook Lagoon site. The initial scope of work consisted of a Phase 1A evaluation of the prehistoric, environmental and historic background of the site and surround area followed by a day of field inspection. A Phase 1B evaluation of surface and subsurface testing was also conducted.

No National Register of Historic Places or National Register of Historic Places-eligible properties were identified within the Site boundaries as a result of the Phase 1A and 1B surveys. The historical research undertaken for the project and the subsequent field investigations did not find any locations where such resources might be anticipated. No architectural resources of landscape features in the immediate vicinity of the Site are considered for the National Register.

The prehistoric sensitivity of the Site has also proved to be limited. Shovel testing of the archaeological sensitive areas impacted by the proposed remediation gave no evidence of prehistoric or historic work. Stratigraphy was consistent throughout the project area with sandy slit and cobbles overlying decaying shale bedrock. Based on these results, no additional archaeological investigations are considered necessary on the Site. The regulatory requirements relating to the identification and protection of historic properties are therefore considered to be complete.

3.6 ADDITIONAL GROUNDWATER SAMPLING

3.6.1 NEW GROUNDWATER WELL INSTALLATIONS

Eight new groundwater monitoring wells were installed during the Natural Attenuation Study to further investigate the groundwater contaminant plume in the overburden and bedrock at the Site. Well installations and strategies were in accordance with the Additional Investigation Work Plan for OU-2 (Additional Investigation Work Plan). The locations of the monitoring wells were selected in the field in conjunction with the U.S. EPA and the NYSDEC during a Site Reconnaissance visit on March 15, 2000.

The following monitoring wells, shown on Figure 2.1, were installed in the area of the Site in June 2001:

- MW-9U-01/MW-9D-01 - overburden/bedrock well cluster located northeast of the Site, approximately 400 feet southwest of the Village of Maybrook pumping station on County Road 4. The purpose of this well cluster is to define the northeast extent of the groundwater contaminant plume and serves to further investigate the chemistry recently detected at upgradient overburden well MW-8U-95.
- MW-10U-01/MW-10D-01 – overburden/bedrock well cluster located north of County Road 4 and across from the entrance to the Site from County Road 4. The purpose of this well cluster is to define the northern extent of the contaminant plume in the overburden and bedrock groundwater.
- MW-11U-01/MW-11D-01 - overburden/bedrock well cluster located southwest of the Site and adjacent to Beaverdam Brook. The purpose of this well cluster is to

define the southwest extent of the contaminant plume and to investigate the potential for contaminant migration to Beaverdam Brook.

- MW-12D-01 - bedrock well located approximately 300 feet east of the eastern Site property line. The purpose of this well is to verify the direction of groundwater flow and to confirm the absence of Site-related contaminants east of the Site.
- MW-13D-01 - bedrock well located approximately 500 feet east of the eastern Site property line and approximately 600 feet north of well MW-12D-01. The purpose of this well is to verify the direction of groundwater flow and to confirm the absence of Site-related contaminants east of the Site.

Overburden well MW-10U-01 was not originally proposed in the Work Plan. However, CRA, the Trust, NYSDEC, and the U.S. EPA agreed that the installation of an overburden well at this location of bedrock well MW-10D-01 would be beneficial given the thickness of the overburden encountered (25 feet), the abundance of groundwater in the overburden, and the fact that the upgradient overburden wells (MW-7U-95 and MW1) are sometimes dry.

Overburden monitoring wells MW-12U-01 and MW-13U-01 proposed in the Work Plan were not installed due an insufficient thickness of overburden and the absence of groundwater at these locations. The NYSDEC and U.S. EPA concurred with the decision that overburden wells not be installed at these locations.

The screened interval for each monitoring well was approved by the U.S. EPA and NYSDEC prior to construction. The installation details for each well are summarized in Table 3.23. Monitoring well completion logs are presented in Appendix D.5.

3.6.2 BEDROCK WELL CONVERSIONS

In accordance with the Work Plan, a number of existing open corehole bedrock wells were converted to screened interval wells. Four open coreholes, MW-1D-91, MW-2D-91, MW-3D-91, and MW-5D-95 were retrofitted with 10-foot long screens to monitor a discrete interval at the most transmissive depth. The well conversion details for each well are summarized in Table 3.24. Well logs for each of the converted wells are presented in Appendix D.6.

Packer testing was used to select the monitoring zone for placement of the screen at each well at each location. The results of the "constant rate packer flow" testing analyses are presented in Appendix G.3.

In accordance with the Work Plan, once the "optimum bedrock groundwater monitoring zone" was determined at each location, concurrence was obtained from NYSDEC and U.S. EPA prior to the installation of the screen. Based on this concurrence, the four wells were screened as follows:

Well Designation	*Screened Interval (ft bgs)*
MW-1D-91	21.5-31.5
MW-2D-91	42.0-52.0
MW-3D-91	43.0-53.0
MW-5D-95	87.0-97.0

The following provides the basis for selecting the screen interval for the conversion of each of the four open corehole bedrock wells:

<u>MW-1D-91</u>

The bedrock open corehole extends from approximately 13 feet to 32 feet below ground surface (bgs) at MW-1D-91. Results of the packer testing for the first interval (10 feet to 21 feet bgs) indicated a hydraulic conductivity of approximately 0.0004 cm/sec based on a constant rate test at 1 gallon per minute (gpm) for 64 minutes. Results of the packer testing for the second interval (21 feet to 32 feet bgs) indicated a hydraulic conductivity of approximately 0.002 cm/sec based on a constant rate test at 2 gpm for 55 minutes. Consequently, MW-1D-91 was completed with a 10-foot screen set at 21.5 feet to 31.5 feet bgs.

<u>MW-2D-91</u>

The bedrock open corehole extends from approximately 24 feet to 52 feet bgs at MW-2D-91. There was no recovery after pumping during the step test for the first two intervals, which extended from 22 feet to 33 feet bgs and 30 feet to 41 feet bgs, respectively. A constant rate test was conducted for 65 minutes at a flow rate of 1 gpm for the third interval extending from 41 feet to 52 feet bgs. The results indicated a hydraulic conductivity of approximately 0.0008 cm/sec. Consequently, MW-2D-91 was completed with a 10-foot screen set at 42 feet to 52 feet bgs.

MW-3D-91

The bedrock open corehole extends from approximately 24 feet to 53 feet bgs at MW-3D-91. There was no recovery after pumping during the step test for the first two intervals, which extended from 20 feet to 31 feet bgs and 31 feet to 42 feet bgs, respectively. A constant rate test was conducted for 60 minutes at a flow rate of 1 gpm for the third interval extending from 42 feet to 52 feet bgs. The results indicated a hydraulic conductivity of approximately 0.0006 cm/sec. Consequently, MW-3D-91 was completed with a 10-foot screen set at 43 feet to 53 feet bgs.

MW-5D-95

The bedrock open corehole extends from approximately 36 feet to 97 feet bgs at MW-5D-95. A total of seven intervals were selected for testing (34 feet to 45 feet bgs, 42 feet to 53 feet bgs, 53 feet to 64 feet bgs, 64 feet to 75 feet bgs, 75 feet to 86 feet bgs, and 86 feet to 97 feet bgs). The results were similar for each interval with hydraulic conductivities ranging from 0.0002 to 0.0007 cm/sec at a pumping rate of 1 gpm for approximately 1 hour. However, during the packer tests, significant drawdown was observed above the top packer during pumping for the middle five intervals indicating interconnectivity with the overlying interval(s) most likely due to vertical fracturing. Therefore, the results of these five intervals were considered questionable. Consequently, MW-5D-95 was completed with a 10-foot screen at 87 feet to 97 feet bgs. This depth was chosen over the shallower interval (34 feet to 45 feet bgs) in order to monitor the deeper bedrock groundwater and the potential for vertical migration in the bedrock aquifer. MW-5D-95 could then have the potential to be used as a sentinel well for the Maybrook supply well, which is completed in the deeper bedrock downgradient of MW-5D-95.

3.6.3 GROUNDWATER SAMPLING

In accordance with the Additional Investigation Work Plan, one comprehensive round of groundwater sampling was conducted for the Monitoring Well Network consisting of 38 wells as outlined in Table 3.25. The monitoring well locations are presented on Figure 2.1. Groundwater sampling for the first round of the Natural Attenuation Study was also completed concurrently with the Additional Round of Groundwater Sampling at 24 monitoring locations as presented in Table 3.25.

Groundwater samples were collected from 32 of the 38 wells in July 2001 for the Additional Round of Groundwater Sampling using low flow sampling techniques and analyzed for TCL VOCs, SVOCs, TAL inorganics, cyanide, and Site-specific pyridines (pyridine, 2-aminopyridine, and alpha-picoline) in accordance with the QAPP. Samples were not collected from overburden wells MW-6, MW-7U-95, SW-1, SW-6, and SW-10 as these wells were dry at the time of sample collection. In addition, well T-1 was not sampled as it could not be located. The sample key is presented in Table 3.26.

Groundwater elevations were measured and can be found in Table 3.9. Purging and sampling information is presented in Appendix E.2 for overburden and bedrock wells.

The analytical results for the Additional Round of Groundwater Sampling for the overburden and bedrock wells are presented and discussed in Section 6.0. The laboratory analysis data sheets for the Volatile and Semi-Volatile Tentatively Identified Compounds (TICs) are included in the data assessment and validation which is presented in Appendix L.3.

3.7 NATURAL ATTENUATION STUDY

Samples were collected for geochemical parameters and Site-related Chemicals of Potential Concern (COPCs) to demonstrate the type(s) of natural attenuation processes that are occurring in the overburden and bedrock groundwater. Two rounds of groundwater sampling were conducted for natural attenuation parameters. The first round (July 2001) was conducted during the dry season and the second round (June 2002) was conducted during the wet season. The second round was conducted to determine if there are any seasonal variations in the COPCs and geochemical concentrations in the overburden or bedrock.

The Natural Attenuation Study for Operable Unit 2 (OU-2) was submitted as an initial draft to the agencies in November 2003 and was subsequently resubmitted in June 2004 in response to comments dated January 20, 2004. Additional comments were provided by the U.S. EPA dated September 9, 2004. The Natural Attenuation Study for OU-2 has not been finalized.

The Groundwater sample key for the Natural Attenuation Study samples can be found on Table 3.27. Purging and sampling information is presented in Appendix E.2 for overburden and bedrock wells. The analytical results for the overburden and bedrock wells are presented and discussed in Section 6.0.

3.8 2003 SOIL INVESTIGATIONS

At the request of the U.S. EPA and the NYSDEC, additional soil sampling related to Operable Unit 1 (OU-1) was completed as per the Additional Soil Sampling Work Plan (CRA, March 2003). The purpose of the additional soil sampling was to characterize the presence of TAL inorganics and total cyanide as well as to collect additional background samples.

Soil samples were collected and analyzed at the Maybrook Lagoon Site for TAL inorganic parameters and cyanide as per the Additional Soil Sampling Work Plan. A total of 30 background soil samples were collected from 15 soil borings conducted using direct-push methods. The locations of background samples are presented on Figure 3.5. Sample locations were agreed upon in the field by U.S. EPA, NYSDEC, CRA, and Trust personnel. Two soil samples were collected from each boring: one from the 0- to 2-foot interval, and another from an intermediate depth between the surficial interval and bedrock using direct-push methods.

In the former lagoons area, 120 additional soil samples were collected and added to the existing inorganic database to result in a total of 168 soil samples in the area of the former lagoons. The locations of the additional soil samples are shown on Figure 3.6. Forty-seven historic soil samples collected in the area of the former lagoons were added to the data. The historic samples include:

- 17 subsurface soil samples collected from boreholes in 1991;
- 7 surface soil samples collected in 1995; and
- 23 subsurface soil samples collected during the test pit program in November 1996.

Historic samples outside the lagoon areas were not used in this analysis.

As per the work plan, a uniform grid was developed to determine the locations of the additional borings within the lagoons. The actual sample location within each grid square was randomly positioned to maintain a random component to the sample location selection process. The locations of borings for lagoons 4 and 5 were chosen using an in-field random placement. The remaining lagoon boring locations (Lagoons 1, 2, 3, and 6), were determined using the random number generator within Excel and paced off in the field. The samples were collected from the black-stained material within the area of the former lagoons and a second sample was collected from an intermediate depth between this interval and the top of the bedrock. Geological conditions occasionally prevented the collection of the second sample due to the shallow depth of the bedrock. This was especially prevalent in lagoon 6, where the bedrock was typically between 2- to 5-feet deep; the deeper sample was not obtained for three out of seven samples.

All soil samples were analyzed for the following 24 inorganic parameters:

- Aluminum
- Antimony
- Arsenic
- Barium
- Beryllium
- Cadmium
- Calcium
- Chromium
- Cobalt
- Copper
- Iron
- Lead
- Magnesium
- Manganese
- Mercury
- Nickel
- Potassium
- Selenium
- Silver
- Sodium
- Thallium
- Vanadium
- Zinc
- Cyanide (total)

The analytical results for the background and the former lagoon areas are summarized and discussed in Section 6.0.

Mercury speciation sampling was performed in accordance with the Additional Soil Sampling Work Plan for soil samples collected from the area of the former lagoons to determine the proportion of methylmercury present in the soils. Initially, seven samples were selected for mercury speciation as they exceeded the most stringent residential health-based criterion for methylmercury of 8.34 mg/kg. These seven samples were collected from Lagoons 1, 2 and 5. There were no exceedences of the residential health-based criterion for samples collected from Lagoons 3, 4, and 6. The U.S. EPA requested that 8 additional samples be included for speciation analysis from lagoons 3, 4, and 6 to provide coverage across the Site. The number of samples submitted for speciation analysis from each lagoon is as follows:

Sampling Area	*Number of Samples*
Lagoon 1	2
Lagoon 2	4
Lagoon 3	2
Lagoon 4	2
Lagoon 5	4
Lagoon 6	1
Sub-total:	*15*

The analytical results for the mercury speciation are summarized and discussed in Section 6.0.

3.9 WETLAND DELINEATION

On May 15 and 16, 2003, CRA conducted a field investigation to delineate the extent of jurisdictional wetlands on the Site. The methodologies and results of CRA's delineation was presented in the Wetland Delineation Report (CRA, 2003) and is reiterated in this section of the RI report.

The delineation of the wetlands was conducted in support of the remedial action that will be completed at the Site. Comments provided by the U.S. EPA were incorporated in the Wetland Delineation Report.

Section 3.9.1 of this section describes the methodology used for the delineation. In Section 3.9.2, information obtained from the U.S. Geological Survey (USGS) topographic quadrangle, the United States Department of Agriculture (USDA) Orange County soil survey, and the New York State freshwater wetland map is discussed. Section 3.9.3 presents the location and extent of the wetlands delineated by CRA and a detailed description of the dominant vegetation, soils, and hydrology of the wetlands delineated by CRA. On-site watercourses are also described in Section 3.9.4. A description of the upland area of the Site is provided in Section 3.9.5. Summary and conclusions are discussed in Section 3.9.6.

Completed Field Data Forms for each sample point are provided in Appendix O. Color photographs of wetlands and uplands on the Site taken during the wetland delineation are provided in Appendix K.6.

3.9.1 METHODOLOGY

Wetland scientists Scott Bush and Donald Knorr from CRA conducted the wetland delineation on May 15, 2003 and May 16, 2003. The delineation was conducted in accordance with the 1987 Corps of Engineers *Wetland Delineation Manual*. The method for the routine on-site delineation was used. This method is based on the three-parameter approach, in which vegetation, soils, and hydrology are used to identify the boundary between jurisdictional wetlands and uplands. During the delineation, CRA established sample points in wetlands and adjacent uplands that existed on the Site. At each sample point, the soil horizons to a minimum depth of 18 inches below ground surface were evaluated for texture and color using a Munsell chart. Hydrological indicators, such as inundation and saturation, were also noted. When present, the depth of standing water, depth to saturation, and depth to standing water in boreholes were recorded. At each sample point, the dominant vegetation within each vegetative strata (i.e., tree, shrub/sapling, herbaceous) was identified and assigned an indicator status (i.e., Obligate, Facultative Wetland, Facultative, Facultative Upland, Upland) in accordance with the U.S. Fish and Wildlife Service's (USFWS) regional list of plant species (Reed 1988). A summary of observed on-Site plant taxa and indicator status is included in Table 3.28. Each of these wetland parameters is further discussed below.

Plants that have successfully adapted to saturated or inundated soil conditions, and characteristically dominate wetland areas are collectively referred to as hydrophytes or hydrophytic vegetation. To help identify wetland areas, the USFWS has assigned a wetland indicator status category to plants that have been found in wetland areas. The wetland indicator status category defines the estimated probability of a plant species occurring in wetlands. The categories are as follows:

Category	*Estimated Frequency of Occurrence in Wetlands*
Obligate Wetland Plants (OBL)	> 99%
Facultative-Wet Wetland Plants (FACW)	> 67 to 99%
Facultative Wetland Plants (FAC)	> 33 to 67%
Facultative Upland Plants (FACU)	> 1 to 33%
Obligate Upland Plants (U)	< 1%

Modifiers such as "+" and "-" are sometimes used to indicate that a plant species has a greater or lesser probability of occurring in a wetland than a plant species that has the general indicator status. For example, a species designated as FACW+ is more likely to be found in a wetland than FACW but not as likely as an obligate wetland species. If a plant is not listed then it is assumed not to occur in wetlands.

Consistent with the 1987 manual, CRA determined the presence of hydrophytic vegetation by comparing the plants identified in the study area with the USFWS's list. If an area had 50 percent or more of the dominant plant species having a USFWS wetland indicator status of OBL, FACW, or FAC) the area was designated as positive for hydrophytic vegetation.

Wetland soils are collectively referred to as hydric soils and are saturated, flooded, or ponded long enough during the growing season to develop anaerobic conditions in the upper part of the soil horizon. Water saturation can result from shallow or perched groundwater conditions or surface inundation. These soils usually support hydrophytic vegetation. Hydric soils are determined visually by soil color: brightly colored soils (red, orange and brown) are indicative of well drained conditions; subdued colors such as grays are typical of soils subject to constant saturation or a fluctuating water table.

The presence of hydric soils was initially assessed by reviewing the Soil Conservation Service's (SCS) Soil Survey of Orange County, New York and a list of hydric soils provided by the USDA.

In accordance with the 1987 manual, CRA also identified hydric soils by field indicators of saturated soil conditions. Constantly saturated soils typically are low chroma or gray in color. These soils are often referred to as gleyed soils. The presence of gleyed soils is typically indicative of a wetland area. Therefore, examination of soil color often reveals the most visible indicator of wet soils. Soil color was determined by comparing soil samples with Munsell Soil Color Charts. Munsell colors for soils are designated in a symbolic notation that identifies the hue, value, and chroma. An example of soil color notation is 5 YR 7/1; where 5 YR is the hue, 7 is the value, and 1 is the chroma. Soils within 18 inches which exhibited either of the following characteristics were designated as hydric: (1) a matrix chroma of one or less; or, (2) a chroma of two or less with mottles.

Other field indicators of hydric soils include the presence of any of the following: sulfidic materials (hydrogen sulfide odor), mottles which are generally iron and manganese concretions in the form of black or brown aggregates in the soils, accumulated organic materials in the surface horizon, and oxidized root-rhizome channels associated with living roots and rhizomes in reduced soils. CRA soil borings were collected in accordance with the 1987 manual by using a hand auger to extract 18-inch deep soil samples. The entire profile was then characterized using the Munsell color chart.

CRA determined the presence of wetland hydrology by observing field indicators such as saturated or inundated soils or other signs of previous prolonged inundation (discussed below) during the growing season. Permanent or periodic inundations, or soil saturation to the surface, at least seasonally, are the primary forces associated with wetland formation. Hydrologic characteristics indicative of a wetland would include saturation by ground or surface water for prolonged periods of time during the growing season. Hydrology may also be indicated by morphological adaptations of plants in response to inundation or soil saturation such as buttressed tree trunks, multiple trunks, and shallow roots. Other positive indicators of hydrology include drainage patterns, sediment deposits, water-stained leaves, surface scoured areas and drift lines. CRA applied these collective criteria for wetland determination at the property.

CRA walked the entire Site and recorded wetland characteristics as determined by observation of vegetation, soils and hydrology using the criteria outlined above. Each wetland area was marked using alpha-numeric symbols on pink flagging tape (tied to vegetation). The delineated areas were surveyed by CRA field technicians using GPS survey equipment.

3.9.2 REVIEW OF SECONDARY DATA

Prior to conducting the delineation, CRA reviewed secondary sources of information that may indicate the presence of wetlands on the Site. Secondary sources reviewed included the USGS topographic map (Maybrook and Goshen, NY quadrangles), USDA Orange County soil survey report and map, and the New York State freshwater wetlands map (Maybrook quadrangle). The USGS map (Figure 1.1) identifies Beaverdam Brook entering the Site near the northwestern property boundary, flowing north to south, and exiting the Site at the west-central property boundary. Beaverdam Brook continues to flow south after exiting the property, eventually flowing into Otter Kill. No indications of wetlands are identified on the USGS map (Figure 1.1).

The 1981 USDA Orange County soil survey map (Figure 3.7) indicates several soil series as occurring on the Site (Sheet No. 40). In the central portion of the Site, the map indicates surface mining activity historically. The soils are mapped as Quarries (Qu) and Pits, Gravel (Pg). Nepera bought the land in 1952 and began using the property in 1953 through 1967. There is no known usage of this property for quarrying or gravel pits, and these references may be a misclassification on the 1981 USDA Orange County soil survey maps. Other soil types mapped for the central portion of the Site are Bath-Nassau shaley silt loam, 8 to 15 percent slopes (BnC), Mardin gravelly silt loam, 3 to 8 percent slope (MdB), Hoosic gravelly sandy loam, 3 to 8 percent slopes (HoB), and Hoosic gravelly sandy loam, 8 to 15 percent slopes (HoC). None of these soils are listed as hydric on either the National Hydric Soils List or the New York Hydric Soils List.

The USDA Orange County soil survey map indicates Alden silt loam and Halsey silt loam as occurring on the Site. Both soils are listed as hydric on both the National Hydric Soils List and the New York Hydric Soils List. Alden silt loam is mapped in the southeastern portion of the Site. Halsey silt loam is mapped in the northwestern portion of the Site, corresponding with the location of Beaverdam Brook.

A copy of the New York State freshwater wetlands map is provided as Figure 3.8. The New York State freshwater wetlands map (Maybrook quadrangle) indicates a state regulated freshwater wetland occurring in the northwestern portion of the Site. This regulated wetland area is designated by the state as MB-1 and corresponds to CRA's Wetland Area DD.

3.9.3 WATERCOURSES AND WETLANDS

3.9.3.1 OVERVIEW

One watercourse (Beaverdam Brook) and three wetland areas are present on the Maybrook Lagoon Site. The location of Beaverdam Brook and wetlands delineated by CRA are identified on the topographic site plan provided as Figures 3.9 and 3.10. The locations of the sample points and photographs are also identified on Figures 3.9 and 3.10, respectively. Completed Field Data Forms for each sample point are provided in Appendix O. Color photographs are provided in Appendix K.6. Descriptions of each watercourse and wetland are provided in the following sections. A total of 2.0 acres of wetlands were delineated on the Site.

3.9.3.2 WATERCOURSES

Beaverdam Brook is a perennial watercourse that enters the Site near the northwestern property corner, flows south, and exits the Site at the west-central property boundary. Beaverdam Brook flows south, and eventually flows into Otter Kill.

3.9.3.3 WETLANDS

3.9.3.3.1 WETLAND AA

Wetland AA is a palustrine-forested wetland (PFO) located off-site and east of the southeastern Site boundary. Dominant species in the tree stratum are pin oak (*Quercus palustris*) and red maple (*Acer rubrum*). Dominant species in the sapling/shrub stratum are red maple, southern arrowwood (*Viburnum dentatum*), green ash (*Fraxinus pennsylvanicus*), silky dogwood (*Cornus amomum*), red chokeberry (*Aronia arbutifolia*), black cherry (*Prunus serotina*), black gum (*Nyssa sylvatica*), and poison ivy (*Toxicodendron radicans*). Both the tree and sapling/shrub cover in this wetland is fairly dense. Dominant taxa in the herbaceous stratum are marsh blue violet (*Viola cucullata*), enchanters nightshade (*Circaea quadrisulata*), jewelweed (*Impatiens capensis*), sedges

(*Carex sp.*), and ferns. Eighty-five (85) percent of the dominant species are facultative-wet or facultative.

Soils in Wetland AA are mapped as Alden silt loam, which is listed as a hydric soil on the National Hydric Soils List and the New York Hydric Soils List. At a depth of 0 to 3 inches, the matrix color is 10 YR 3/2 monochromatic. At a depth of 3 to 18 inches, soil is stony silt loam with a matrix color of 10 YR 4/1, with mottles of 10 YR 4/6. The soil profile observed during the field investigation is consistent with the USDA description for Alden silt loam.

At the time of the field investigation, the soil within Wetland AA was saturated to the surface. Depth of free water in the borehole was 2 inches below ground surface. Evidence of vernal ponds within the forested wetland was also observed.

Wetland AA is delineated with flags AA-1 through AA-9 and is located outside the Site boundaries. Sample Point 1 and Photographs 1 and 3 document conditions in Wetland AA.

3.9.3.3.2 WETLAND BB

Wetland BB is a PFO in the southeastern portion of the Site, south of the right of way of the abandoned railroad tracks. The limits of Wetland BB extend beyond the Maybrook Lagoon Site boundaries. Approximately 1.1 acres of Wetland BB occurs on the Site. Dominant species in the tree stratum are red maple, pin oak, swamp white oak (*Quercus bicolor*), and slippery elm (*Ulmus rubra*). Dominant species in the sapling/shrub stratum are green ash, red maple, silky dogwood, southern arrowwood, witch hazel (*Hamamelis virginiana*), multiflora rose (*Rosa multiflora*), and poison ivy. Dominant species in the herbaceous stratum are jack-in-the-pulpit (*Arisaema atrorubens*), wild geranium (*Gerrnium maculatum*), sensitive fern (*Onoclea sensilibis*), marsh blue violet, skunk cabbage (*Symplocarpus foetidus*), and trout lily (*Erythronium americanum*). Seventy-eight (78) percent of the dominant species are obligate wetland, facultative-wet, or facultative.

Soils in Wetland BB are mapped as Alden silt loam, which is listed as a hydric soil on the National Hydric Soils List and the New York Hydric Soils List. At a depth of 0 to 8 inches, soil is silt loam with a matrix color of 10 YR 3/1 and mottles of 10 YR 4/4. At a depth of 8 to 15 inches, soil is stony loam with a matrix color of 10 YR 4/1 and mottles of 7.5 YR 4/6. The soil profile observed during the field investigation is consistent with the USDA description for Alden silt loam.

At the time of the field investigation, the soil within Wetland BB was saturated to the surface. Depth of free water in the borehole was 2 inches below ground surface. Several vernal pools, supporting tussock sedge, (*Carex stricta*) were observed in the vicinity of the sample point.

Wetland BB is delineated with flags BB-1 through BB-13. Sample Point 3 and Photographs 4 and 5 document conditions in Wetland BB.

3.9.3.3.3 WETLAND CC

Wetland CC is a 0.5-acre PFO associated with Beaverdam Brook, which flows along the western boundary of the Site. The limits of Wetland CC extend beyond the Maybrook Lagoon Site boundaries. Approximately 0.3-acres of Wetland CC occurs on the Site. Wetland CC is south of the concrete bridge in the west-central portion of the Site. Dominant species in the tree stratum are swamp white oak and green ash. Dominant species in the sapling/shrub stratum are wild black current (*Ribes americanum*), choke cherry (*Prunus virginiana*), southern arrowwood, and poison ivy. Dominant taxa in the herbaceous stratum are skunk cabbage, sensitive fern, tussock sedge, arrow arum (*Peltandra virginica*), and sedges (*Carex sp.*). Ninety percent of the dominant species are obligate wetland, facultative-wet, or facultative.

Soils in Wetland CC are mapped as Halsey silt loam, which is listed as a hydric soil on the National Hydric Soils List and the New York Hydric Soils List. From 0 to 12 inches, the soil is silt loam with a matrix color of 7.5 YR 3/1 with mottles of 7.5 YR 3/3. The soil profile observed during the field investigation is consistent with the USDA description for Halsey silt loam.

At the time of the field investigation, the soil within Wetland CC was saturated to the surface. Free water in the borehole was at the ground surface. Stained leaves, indicating the presence of inundation, were also observed.

Wetland CC is delineated by flags CC-1 through CC-27. Sample Point 6 and Photograph 6 document conditions in Wetland CC.

3.9.3.3.4 WETLAND DD

Wetland DD is a 0.85-acre PFO) associated with Beaverdam Brook in the northwestern portion of the Site. The limits of Wetland DD extend beyond the Maybrook Lagoon Site boundaries. Approximately 0.6-acres of Wetland DD occurs on the Site. It is separated from Wetland CC by the concrete bridge in the west-central portion of the Site. Dominant species in the tree stratum are swamp white oak and pin oak. Dominant species in the sapling/shrub stratum are swamp white oak, pin oak, red maple, silky dogwood, multiflora rose, and tartarian honeysuckle. Dominant taxa in the herbaceous stratum are sensitive fern, purple loosestrife (*Lythrum salicaria*), grass-leaved goldenrod (*Solidago graminifolia*), steeplebush (*Spiraea tomentosa*), Kentucky bluegrass (*Poa pratensis*), fragrant bedstraw, iris (*Iris sp.*), sedge (*Carex sp.*), and grass (Gramineae). Fifty (50) to 100 percent of the dominant taxa in Wetland DD are facultative wetland or facultative.

Soils in Wetland DD are mapped as Halsey silt loam, which is listed as a hydric soil on the National Hydric Soils List and the New York Hydric Soils List. Soil borings evaluated during the field investigation identified an A horizon of silt loam ranging for 3 to 8 inches below ground surface. The matrix color of this horizon is 10 YR 3/1 or 10 YR 4/1 with mottles of 10 YR 4/4 or 10 YR 6/6. The B horizon extends to a depth of 15 to 16 inches. Soil in this horizon is clay with a matrix color of 10 YR 5/1 and mottles of 10 YR 6/8 or 7.5 YR 4/6. The soil profiles observed during the field investigation are consistent with the description for Halsey silt loam.

At the time of the field investigation, the soil within Wetland DD was saturated to the surface. Free water in one of the boreholes was 2 inches below ground surface. Stained leaves, indicating the presence of inundation, were also observed.

Wetland DD is delineated by flags DD-1 through DD-13 and EE-1 through EE-4. Sample Points 8, 10, and 11 and Photographs 7, 8 and 9 document conditions in Wetland DD.

3.9.3.3.5 WETLAND FF

Wetland FF is a PFO associated with Otter Kill and is located off-site, south of Wetland CC. Wetland FF was not delineated by CRA because the limits of Wetland FF occur outside the Maybrook Lagoon Site boundaries. However, CRA did perform a global positioning system (GPS) survey of the limits of the wetland. Photograph 11 documents conditions in Wetland FF.

3.9.4 UPLANDS

Uplands adjacent to Wetland AA, Wetland BB, and Wetland CC are forested. However, the upland forest adjacent to Wetland AA is younger than the forested Wetland AA.

Dominant tree species in the forested uplands consist of red maple, white ash (*Fraxinus americana*), and shagbark hickory (*Carya ovata*), sugar maple, pin oak, black cherry, black locust (*Robinia pseudoacacia*), hickory (*Carya sp.*). Dominant species in the sapling/shrub stratum are blackhaw (*Viburnum prunifolium*), red maple, shagbark hickory, sugar maple (*Acer saccarum*), pin oak, witch hazel, wild raspberry (*Rubus sp.*), black cherry, green ash, tartarian honeysuckle (*Lonicera tatarica*), poison ivy, and Virginia creeper (*Parthenocissus quinquefolia*). Dominant species in the herbaceous stratum are garlic mustard (*Alliaria petiolata*), marsh blue violet, wild strawberry (*Fragaria virginiana*), fragrant bedstraw (*Galium triflorum*), trout lily, jewelweed, avens (*Geum sp.*), jumpseed (*Polygonum virginianum*), and lily of the valley (*Maianthemum canadense*).

The USDA maps the soils in portions of the forested uplands as Alden silt loam and Halsey silt loam. These map types were not confirmed during the field investigation. Soil in the forested uplands range from a 10 YR 3/3 monochromatic very shaley silt loam to a 10 YR 5/3 monochromatic silt loam. No indicators of wetland hydrology were observed in the uplands during the field investigation.

Sample Points 2, 4, 5, and 7 and Photographs 2, 12, 13, and 14 document conditions in the upland forest adjacent to Wetland AA, Wetland BB, and Wetland CC.

The uplands adjacent to Wetland DD consist of a scrub-shrub and forested vegetative community. Dominant species in the sapling/shrub stratum are red maple, white ash, and poison ivy. Trout lily is the dominant species in the herbaceous stratum. Seventy-five (75) percent of the dominant species are facultative. The right-of-way for County Highway No. 4 is located north of Wetland DD.

The USDA maps the soil in the scrub-shrub uplands adjacent to Wetland DD as Halsey silt loam. This map type was not confirmed during the field investigation. At a depth from 0 to 15 inches below ground surface, soil is silt loam with a matrix color of 10 YR 4/4 monochromatic. No indicators of wetland hydrology were observed during the field investigation.

Sample Point 9 and Photograph 10 document conditions in the scrub-shrub uplands adjacent to Wetland DD and along County Highway No. 4.

3.9.5 SUMMARY AND CONCLUSIONS

Three (3) wetland areas (Wetlands BB, CC, and DD) were delineated within the property boundaries of the Maybrook Lagoon Site. The State of New York also regulates freshwater wetlands. In general, only those wetlands in excess of 12.4-acres or those wetlands specifically listed on state freshwater wetland maps are regulated by the state. The New York State freshwater wetlands map (Figure 3.8) indicates a state regulated freshwater wetland occurring in the northwestern portion of the Site. This regulated wetland area is designated by the state as MB-1 and corresponds to CRA's Wetland Area DD. According to the regulations, CRA's Wetland Area CC would also be regulated by the State of New York as a freshwater wetland because it is within 100 feet of Wetland DD. Wetland Area BB is likely not regulated by the State of New York.

In general, the wetlands on-site appear to have good potential to serve as areas of groundwater recharge, floodwater storage areas, and wildlife habitat. The open water and emergent wetlands found on-site may also provide sediment stabilization, toxicant retention, and nutrient removal and transformation functions. All of the wetlands on-site are found within a deciduous forest complex providing wintering, breeding and migratory rest areas for migratory birds and other wildlife. Upland areas adjacent to the wetlands are dominated primarily by mesic deciduous forest typical for this region. Several smaller areas of upland herbaceous field are located in the central portion of the Site and are associated with disturbed areas and former lagoons. Plant diversity on the Site is fairly good. Areas of stressed vegetation were not observed.

3.10 EPA SEDIMENT SAMPLING

Twenty-seven sediment samples were collected by the U.S. EPA from Beaverdam Brook and analyzed for TCL VOAs, BNAs, pesticides, PCBs, pyridine compounds, cyanide, and TAL metals. Samples south of the abandoned railway, were taken on May 13, 2003 and the remaining samples were collected on May 15, 2004. Sample locations are shown on Figure 3.11. Analytical results are presented and discussed in Section 6.0. As these samples were collected by the U.S. EPA, CRA did not validate the results.

3.11 VILLAGE OF MAYBROOK SUPPLY WELLS

The Village of Maybrook serves a population of approximately 3,000 within the Town of Montgomery and a portion of County Route 4 (or Maybrook Road). An estimated average of 277,027 gallons was delivered to customers served by the Village of Maybrook in 2003. The water supply has been obtained from seven bedrock wells of an average depth of 357 feet. An eighth well was put into service in 2003 providing an additional 94,000 gallons per day.

The Village of Maybrook conducts annual testing on all water supply wells. Samples are analyzed for VOCs, polychlorinated organic compounds (POCs), Group II and III metals, and synthetic organic compounds. An annual report is produced presenting the results for this sampling round.

In addition to the annual sampling program, due to the close proximity of the Site to the Village of Maybrook wells 1, 2 and 3 on County Route 4, the village is required to monitor these wells on a quarterly basis for VOCs, POCs, and Site-Related pyridine compounds.

Per a telephone conversation with Matthew Thorp, Chief Operator, on March 16, 2005, no detections of Site-related parameters have been found for the samples collected from wells 1, 2, and 3 up to the end of 2004.

4.0 REGIONAL SETTING

The following provides an overview of the physiography, geology and hydrogeology of the region surrounding the Site.

4.1 REGIONAL PHYSIOGRAPHY

The Former Lagoon Site lies within the Valley and Ridge physiographic province of the Appalachian Region, which is characterized by low rolling relief. In general, the topography of the area is typified by relatively low-lying northeast-southwest trending ridges and valleys.

The Valley and Ridge province has been described as follows:

> ***"The Valley and Ridge province is underlain by alternating layers of hard sandstone and soft shale that were compressed and crumpled by pressure exerted from the southeast. The long axis of the wrinkle-like folds in the crumpled rock layers consequently trend northeast-southwest. Different rates of erosion of these tilted layers of hard and soft rock give rise to the sequence of narrow ridges and valleys typical of this province."*** (Frimpter, 1985).

4.2 REGIONAL GEOLOGY

4.2.1 REGIONAL BEDROCK GEOLOGY

The bedrock geology of the Appalachian region is quite complex. The bedrock underlying the Site and surrounding area belongs to the Normanskill Formation, which consists primarily of shale with interbeds of argillite and siltstone (see Figure 4.1). The fine-grained sediments, which have consolidated (lithified) to form the shale, were deposited in marine waters during Middle Ordovician time.

Two periods of intense deformation have structurally altered the somewhat incompetent bedrock (Rich, 1986). These tectonic forces were exerted from the southeast and have resulted in a general northeast trend to the structural lineaments (e.g. faults, folds) and general pattern of the various bedrock formations through the region (see Figure 4.1).

The intense structural deformation followed by glacial loading and unloading has resulted in a high degree of fracturing in the shale bedrock. The degree of fracturing (and weathering) is greatest in the upper portion of bedrock with the frequency of fracturing decreasing with depth.

Current topographic expression is characterized by a series of low relief valleys and ridges through the region which are generally controlled by the underlying topography of the bedrock surface, which generally trend northeast-southwest.

4.2.2 REGIONAL OVERBURDEN GEOLOGY

The unconsolidated deposits that form the overburden are generally thin (average approximately 20 feet) but may be thicker in bedrock depressions (valleys). The deposits are primarily glacial in origin having been transported and deposited by glacial activity (Frimpter, 1985).

Unstratified "ground moraine" deposits of till (an unsorted mixture of boulders, gravel, sand, silt and clay) deposited directly from the glaciers overlie the bedrock in most places (see Figure 4.2). Greater thicknesses of till (up to 200 feet) occur in "drumlins" (elongated hills) which were deposited directly beneath the moving ice and are oriented parallel to the direction of ice flow (Frimpter, 1985). C.A. Rich reported the direction of glacial flow to be south-southwest as evidenced by glacial striations on the shale bedrock (Rich, 1986).

During the period of glacial retreat, meltwaters carried and deposited much of the debris brought along with the glaciers. These deposits may be divided into three categories. The first is referred to as ice-contact deposits (also known as stratified glacial drift) and comprises the landforms known as kames and eskers. These were deposited in close proximity to the glaciers and are dominated by moderately sorted deposits of sand and gravel (see Figure 4.2).

The second category of deposit is referred to as outwash, and relates to those sediments carried further away from the ice sheet by meltwaters. These deposits are generally stratified and well sorted, and generally consist of fine sand through gravel sized grains.

The third category of deposit is termed glaciolacustrine, and relates to fine-grained deposits (silt and clay sized particles) which slowly settled out of glacial meltwaters in ponded lakes.

The glacial deposits of sand and gravel generally overlie the till and bedrock, and in turn may be overlain by glaciolacustrine silt and clay deposits.

4.3 REGIONAL HYDROGEOLOGY

4.3.1 OVERBURDEN

"Among the aquifers of Orange and Ulster Counties, the glacially derived sand and gravel deposits in the valleys possess the greatest potential as groundwater reservoirs." (Frimpter, 1985). Streams often flow across, and are in hydraulic contact with, these sand and gravel aquifers. Water may move in either direction between the surface water bodies (streams and lakes) and groundwater aquifers, depending on the relative hydraulic heads which may reverse between wet and dry seasons (or periods). These sand and gravel aquifers may be surficial water table aquifers being recharged by vertical infiltration of precipitation, or they may be confined by an overlying, relatively impermeable silt and clay layer. In the later case, recharge may be largely lateral from upgradient areas as in valley settings. The water table tends to parallel the ground surface in a subdued manner and generally fluctuates in elevation from spring (high) to late summer (low).

If an overlying, low-permeability layer is present, the aquifer may be confined. This implies that if a well was drilled into the aquifer, the water level would rise to a position higher than the top of the aquifer body. This often occurs in valley settings.

The sand and gravel aquifers are often utilized for high capacity municipal or industrial supply wells. These aquifers are often developed with infiltration galleries or wells close to surface waters to induce infiltration and increase production. Where no sand and gravel aquifer occurs in the overburden, a large diameter (36-inch) well bored into till generally provides sufficient quantities of water for individual households.

The quality of the water in the sand and gravel was described by Frimpter:

> *"Water produced from sand and gravel aquifers in Orange and Ulster Counties is generally hard, and in a few areas dissolved iron and manganese are present in quantities greater than the limits recommended by the U.S. Public Health Service. Water treatment can generally reduce the iron and manganese to acceptable concentrations."* (Frimpter, 1985).

It is therefore possible that groundwater in the overburden may not be suitable for domestic use in some areas due to naturally occurring compounds.

4.3.2 BEDROCK

The bedrock aquifer provides a significant portion of the groundwater for domestic uses in the area. In 1985, Frimpter notes:

> *"It is estimated that about one-half of the 300,000 inhabitants of Orange and Ulster Countries depend on bedrock aquifers for their domestic water supplies. Although bedrock generally yield sufficient quantities of water for home or farms needs, it rarely can be depended upon for larger industrial or municipal supplies. Only about eight percent of the population is served by public water supplies obtained from bedrock aquifers."* (Frimpter, 1985).

However, the permeable zones, which yield the water, are not homogeneously distributed in the rock, as noted by Frimpter.

> *"The consolidated rocks of the two-county area have virtually no primary or intergranular porosity for groundwater storage or transmittal. Groundwater obtained from the bedrock aquifers comes from fractures or cracks in the rock. In a few instances, zones of high porosity and permeability have been found in the bedrock of the two-county area. These zones are primarily due to faulting or fracturing..."*(Frimpter, 1985).

Recharge to bedrock aquifers occurs from surface waters in direct contact with the bedrock (no overburden present), or more commonly as groundwater in the overburden migrates downward into the weathered, fractured upper portion of the bedrock. The groundwater then follows a complex path of cross-cutting and interconnected fractures to greater depth.

> *"The porosity and permeability of the unconsolidated sediments overlying the bedrock aquifer have a great effect on the rate of recharge. Thick layers of till with low permeability allow only very slow recharge to the underlying bedrock aquifer, whereas water-saturated sand and gravel overlying the bedrock acts as a source of rapid recharge. Wells tapping bedrock beneath sand and gravel have adequate yields at shallow depths because of the abundant supply of recharge water in the sand and gravel. Many bedrock wells with high yields tap aquifers overlain by saturated sand and gravel". (Frimpter, 1985).*

Frimpter (1985) provides a comparison of productivity between shale aquifer wells and wells screened in sand and gravel aquifers. In terms of gallons per minute of well yield per foot (gpm/ft) of well exposed to the aquifer, the sand and gravel aquifers have much greater productivity (21.9 gpm/ft versus 0.5 gpm/ft).

Frimpter also describes the quality of the water in the bedrock aquifer as being acceptable, but not ideal for domestic use.

> ***"Water obtained from the bedrock aquifers is generally of good chemical quality but is often moderately to very hard. Deeper wells in shale frequently yield water containing hydrogen sulfide. Wells tapping the red shale and sandstone formations in southeast Orange County commonly yield water containing undesirable quantities of iron, and other consolidated rock aquifers occasionally yield water with high iron concentrations."*** (Frimpter, 1985).

4.4 GROUNDWATER USE

A historical water well inventory was conducted by C.A. Rich (Rich, 1986). In June 1995, CRA updated the water well inventory to include residents with water wells located within the immediate vicinity of the Site. The water well locations that were identified are presented on Figure 3.2 and a summary of the well descriptions is presented in Table 3.17. Forty-two wells were identified within a two-mile radius of the Site. Of these 42 wells, 12 are overburden wells, which are completed in either sand and gravel aquifers (large diamond (bored) wells) or in till. The overburden wells range in depth from 6 feet to 32 feet. Bedrock wells reportedly obtain water from the shale formation(s). Bedrock well depths range from 35 feet to 399 feet. Two private wells, number 36 and number 39, ranging in depth from 40 to 50 feet; were identified approximately 50 feet north and 50 feet southwest (across Beaverdam Brook) of the Site, respectively. The property owners of number 36 and number 39 were asked by CRA to identify the well construction details (i.e., depth, age, screened material, usage, etc.). Due to the age of these wells (number 36 was constructed in 1958 and number 39 was constructed in the 1800's) water well records are not available. Therefore, it is unknown whether these wells are screened in the overburden or bedrock. According to the owners, these wells are not used for potable water. Reported yields for all the bedrock wells range from 5 to 630 gallons per minute.

Wells of particular importance for this investigation include the Town of Maybrook municipal supply wells and the infiltration gallery located north of County Road No. 4.

The Town of Maybrook currently uses 8 bedrock wells for their municipal water supply. Wells 1, 2 and 3 are in close proximity to the Site. Town Wells 1 and 2 (well numbers 11 and 12 in the inventory) were constructed in 1957, but were not in use until 1993. Each of these wells is 300 feet deep and the wells have a combined well yield of 200 gpm. Town Well 3 (well number 10 in the inventory) was constructed in 1975 and is 300 feet deep with a well yield of approximately 190 gpm. Town Well 3 is currently used as a municipal water source producing approximately 100 gpm. The groundwater from the town wells is treated with chlorine and a sequestering agent to treat high levels of naturally-occurring manganese and iron.

All wells require annual testing of VOCS, POCs, nitrates, group I and II inorganics, and synthetic organic chemical (which includes pesticides). Wells 1, 2 and 3 require additional quarterly sampling for VOCs, POCs and Site-specific pyridines due to the proximitiy to the Site.

An infiltration gallery is located north of the municipal supply wells adjacent to Beaverdam Brook. The infiltration gallery was constructed in the 1930's and consists of three holding pits approximately 18 feet in depth. Water flows into the pits from Beaverdam Brook where it is pumped into a wet well located within the pumphouse building. The infiltration gallery is not currently in use

Other wells to be noted in the vicinity of the Site are bedrock wells (commercial wells numbered 24, 25, and 26 and public supply well number 27). It is expected, that on a larger scale, flow in the bedrock aquifer from the Site will be to the east following the Otter Kill Valley.

5.0 SITE SETTING

The purpose of this section is to provide a description of the geology and hydrogeology that has been identified at the Site.

5.1 SITE GEOLOGY

The Site geology, as described in the following sections, is based upon the data obtained from pre-RI investigations, 39 boreholes drilled at the Site during the Phase I/II RI, the 15 test pits completed for the Treatability Study, the boreholes completed for the installation of the eight wells in 2001, and the 62 boreholes installed for the soil investigations conducted in 2003. Utilizing this information, four hydrogeologic cross-sections were developed. The cross-section locations are presented on Figure 5.1. The four cross-sections identified on Figure 5.1 are presented as Figures 5.2 through 5.5.

The geologic units at the Site are divided into two primary units. These are:

1. Overburden
   - topsoil
   - fill
   - silty sand and gravel; and
2. Bedrock
   - weathered and fractured shale; and
   - shale.

5.1.1 OVERBURDEN

Ground surface topography is generally bedrock controlled in that the ground surface generally follows the bedrock surface topography. The overburden thickness at the Site is also related to bedrock topography in that it is generally thinner (or absent) over bedrock ridges, while greater thicknesses of overburden have been deposited in bedrock depressions and valleys (see Figure 5.2 through 5.5). The overburden is absent in areas of bedrock outcrops and ranges to greater than 20 feet in thickness in the southern portion of the Site.

The overburden is quite variable in texture (grain size) and occurs as discrete layers indicating varying depositional environment and composition. For the most part, the overburden comprises deposits of sand and gravel, which overlie the upper, weathered portion of the bedrock. Stratigraphic logs indicate that the lower portion of this overburden is often coarser grained (gravel often reported) relative to the upper portion.

The regional surficial geology map (Figure 4.2) indicates a relatively widespread outwash sand and gravel deposit along the Otter Kill and in the vicinity of the Site. There may be some variation to this interpretation of outwash on the Site as most stratigraphic intervals described in the logs indicate rather poor sorting of grain sizes. This would indicate a fairly high rate of glacial wasting with the glacial debris being "dumped" without the winnowing action of water currents necessary to produce well-sorted (uniform grain size) deposits more typical of outwash. The Site overburden stratigraphy likely includes ice-contact deposits.

It is possible that some of the sandy overburden deposits are actually "till" as they often contain some silt and/or clay as described in the borehole logs. C.A. Rich (Rich, 1986) describes ***"a thin veneer of unconsolidated glacial deposits composed of a heterogeneous mixture of till-like materials"*** on Site.

Grain size analyses of surficial soil samples indicate primarily sand composition with varying amounts of gravel or silt (and clay). The stratigraphic logs also indicate localized occurrences of topsoil (silt) up to two feet thick.

The overburden appears to be highly variable over short distances as indicated by the stratigraphic logs. It is uncertain whether this is actually the case or if there is a lack of consistency in the descriptions between the various investigating consultants. An example of this is the variation between borehole BH1-91 (CRA, 1992) and monitoring well MW-2 (Ground/Water Technology, Inc., 1983) located in the northern portion of the Site (see Appendices C and D). One log reports a thick silt layer while the other log describes a layer of fill overlying sand and gravel.

A fine-grained layer (interpreted here as silt and clay till) generally overlying the bedrock surface (see Figure 5.2) is reported in several stratigraphic logs (e.g. BH5-91, DW-1 (converted to DW-1-95), MW-7, MW8U-95, and MW-1U-91). This may be an example of the glacial till deposited on valley slopes as described by Frimpter (Frimpter, 1985). A similar layer is reported at BH1-91 in the northern portion of the Site.

In the vicinity of the former lagoons, the native overburden was initially excavated to form the wastewater disposal lagoons. Based on the depth to bedrock reported at various boreholes and test pits located in the former lagoon areas, and the presence of immediately adjacent bedrock outcrops, it appears that the soft, weathered and fractured upper portion of bedrock was also excavated. Examples of this are boreholes MW-3 and MW-2D-91. Following the cessation of disposal activities, the lagoons were filled in with shale fill and overburden materials. The limits of the shale-filled former lagoons (excavations) are depicted in the cross-sections (Figures 5.2 to 5.4). Sharp changes in overburden material between boreholes are attributed to filled excavations versus areas of native ground. Sand fill was also apparently used as backfill and generally overlies the shale fill.

5.1.2 BEDROCK

Bedrock underlying the thin overburden beneath the Site is comprised of massive to bedded shale of the Normanskill Formation. Interbeds of sandstone are also reported in the on-Site stratigraphic logs.

The upper portion of the shale bedrock is highly weathered and fractured as indicated by the bedrock corehole logs. The fracture attitude in the upper portion is mainly vertical and very frequent. Other fracture orientations noted were parallel to, and perpendicular to, the bedding plane orientation. The fractures were often infilled or lined with quartz. In addition, quartz veins were also reported. The fracture surfaces were often iron stained and/or weathered. Occasionally, chalcopyrite, pyrite, and clay mineral fracture fillings were also reported. The shale contained visible amounts of chlorite in places indicating the low-grade metamorphism that the bedrock has undergone.

C.A. Rich (Rich, 1986) determined the weathered, highly fractured upper zone of bedrock to be from one to three feet in thickness. Beneath this, the shale bedrock is dark grey and more competent (harder). Within the lower, unweathered bedrock, fractures were found at approximately 10 to 30-foot spacings (Rich, 1986).

The bedrock monitoring well logs indicate that the upper portion of the shale bedrock is fractured and weathered. The shale becomes more competent with depth. Bedrock outcrops occur at six locations on Site. The locations of these outcrops are presented on Figure 5.6. Outcrop mapping indicates the shale beds strike northeast (N20°E to N79°E).

The shale beds generally dip to the southeast at 40 to 50 degrees (below horizontal). In general, the strata were determined to strike N30°E and dip 40°SE.

An anticlinal structure (convex-up fold) was mapped near the center of the Site. A normal fault with drag features is indicated along the west limit of the Site, south of the abandoned railway grade. The downthrown side was to the east corresponding with the dip of the strata.

Bedrock topography is presented on Figure 5.14. The figure shows that the surface of the bedrock at the site is roughly saddle-shaped, with two ridges trending northeast-southwest, one to either side of the lagoon area, controlling the overburden groundwater flow on Site.

5.2 SITE HYDROGEOLOGY

5.2.1 HYDROSTRATIGRAPHIC UNITS

The hydrogeologic characterization of the Site is complex due to the irregular ground and bedrock topography, and the complex vertical and spatial distribution of varying native deposits and fill materials comprising the overburden. In general, however, three hydrostratigraphic units were defined for the Site. Two are aquifers, and one is an aquitard. Historical and present water level data (see Table 3.9) are used in this evaluation.

The three hydrostratigraphic units, in order of increasing depth, are:

1. the water table aquifer (Shallow Aquifer);
2. the localized Overburden Aquitard Unit; and
3. the Bedrock Aquifer.

The Shallow Aquifer groundwater elevations measured on July 12, 2001 (representative of the dry season) and June 3, 2002 (representative of the wet season) were contoured and are shown on Figure 5.7 and Figure 5.8, respectively. The groundwater elevation contours in the Shallow Aquifer indicate a flow divide occurring in an east-west direction, with the axis running across former Lagoon 4. Therefore, the groundwater flow in the Shallow Aquifer is divided into two flow components: north (north-northeast flow) and south (southwest-southeast flow).

The Bedrock Aquifer groundwater elevations measured on July 12, 2001 and June 3, 2002 were contoured and are shown on Figure 5.9 and Figure 5.10, respectively. The groundwater elevation contours in the Bedrock Aquifer also indicate a flow divide in an east-west direction, with the axis near former Lagoon 4. As well, the groundwater flow in the Bedrock Aquifer indicates two major flow components: north (north-northwest flow) and south (southwest flow) and a minor flow component in the west direction. The current and historical groundwater elevations for all RI monitoring wells are summarized in Table 3.9.

Water Table Aquifer (Shallow Aquifer)

The Water Table Aquifer (Shallow Aquifer) unit consists of the more permeable, outwash sand and gravel deposits which constitute the bulk of the overburden. Also included in the Shallow Aquifer is the uppermost weathered interval of bedrock due to its highly weathered, fractured and permeable nature, and the direct hydraulic connection with the overlying granular deposits. The shale fill in the former lagoon areas is also included in this unit.

Based on the hydrographs for selected Shallow Aquifer monitoring wells presented in Appendix F, the high water table (wet season) typically occurs in January to early June and the low water table (dry season) typically occurs in late June to August. The groundwater elevations in the Shallow Aquifer in the wet season are typically 3 to 6 feet higher than groundwater elevations in Shallow Aquifer in the dry season.

The Shallow Aquifer is generally found under unconfined conditions with the exception of the southwest portion of the Site.

Aquitard Unit

The Overburden Aquitard Unit is a localized, fine-grained deposit of silt and clay (till) within the overburden, predominantly occurring through the south end (MW-7, MW1U-91, SW-8, SW-9, MW-11U-01 and DW-1-95) of the Site and discontinuous in the north end (MW8U-95, MW-10U-01 and MW-9U-01) of the Site. Where present, this layer acts as a partial hydraulic barrier between the overlying sand deposits and the upper, fractured interval of bedrock. Groundwater may become perched atop this layer and, therefore, must be considered when evaluating the configuration of the water table aquifer unit. This aquitard unit may cause confined conditions to be present in the underlying upper bedrock interval. Also, the Overburden Aquitard Unit will reduce direct vertical recharge to the underlying zones where it is present. The Overburden Aquitard Unit ranges in thickness from 3 feet (MW-7) to 13 feet (SW-9).

Bedrock Aquifer

The Bedrock Aquifer occurs within the more competent portion of the shale bedrock which underlies the highly weathered/broken upper several feet of the bedrock surface. Fracturing is still frequent, although decreasing with depth (see Appendix D). Shallow and deep bedrock wells (from 26 feet BGS at MW4D-91 to 120 feet BGS at DW-1-95) were installed across the Site. Stratigraphic and instrumentation logs for the bedrock wells installed at the Site indicate that the shallow and deep bedrock wells are screened in similar lithology (Shale Bedrock). The water level data for the shallow and deep bedrock wells across the Site also exhibit similar equipotential hydraulic head. Therefore, it is expected that the shallow and deep bedrock wells are interconnected and are considered to be monitoring the Bedrock Aquifer.

Minor fluctuations in the groundwater elevation in the Bedrock Aquifer occur seasonally. Examination of the hydrographs for the selected Bedrock Aquifer monitoring wells presented in Appendix F indicate that the highest water elevations occurred in May 1993 and January 1995 and the lowest groundwater elevations occurred in August 1991 and July 1995.

5.2.2 HYDRAULIC GRADIENTS

Shallow Aquifer

As stated in Section 5.2.1, the groundwater flow in the Shallow Aquifer is divided into the northern (north-northeast flow) and southern (southwest-southeast flow) flow components.

Examination of the groundwater contours (Figure 5.7 and Figure 5.8) indicates that the horizontal hydraulic gradients are marginally steeper in the northern portion of the Site (0.032 feet/foot), than in the southern portion of the Site (0.020 feet/foot). The horizontal hydraulic gradient calculations are provided in Appendix P.2 and the results are summarized in Table 5.1.

Vertical hydraulic gradients (VHG) between the Shallow Aquifer and Bedrock Aquifer were calculated in the northern, southern, and lagoon area of the Site. The VHG calculations are presented in Appendix P.2 and the results are summarized in Table 5.1. The VHG between the Shallow and Bedrock Aquifer is generally downward.

The VHG calculated in the northern portion of the Site are based on well nests MW9U-01/MW9D-01, MW10U-01/MW10D-01, MW-5U-95/MW-5D-95 and SW-2/DW-2-95. Positive gradient values indicate downward flow, whereas negative numbers indicate upward flow. During the seasonally low groundwater conditions (July - August), the average VHGs at wells MW9U-01/MW9D-01, MW10U-01/MW10D-01, MW-5U-95/MW-5D-95 and SW-2/DW-2-95 were –0.058 feet/foot, -0.012 feet/foot, 0.090 feet/foot and 0.036 feet/foot, respectively. At seasonally high groundwater conditions (May – June), the average VHGs at wells MW9U-01/MW9D-01, MW10U-01/MW10D-01, MW-5U-95/MW-5D-95, and SW-2/DW-2-95 were –0.077 feet/foot, -0.013 feet/foot, 0.096 feet/foot, and 0.044 feet/foot. This indicates that VHGs are consistently downward near the center of the Site (SW-2), and are consistently upward further north of the Site (MW9U-01), and are variable near the northern Site boundary (MW10U-01).

The VHGs calculated in the southern portion of the Site were based on well nests SW-9/DW-1-95 and MW11U-01/MW11D-01. During the seasonally low groundwater conditions, the average VHGs at SW-9/DW-1-95 and MW11U-01/MW11D-01 were 0.113 feet/foot and 0.0059 ft/foot, respectively. At seasonally high groundwater conditions, the average VHG at SW-9/DW-1-95 was 0.095 feet/foot and at MW11U-01/MW11D-01 the VHG was 0.0158 feet/foot. This indicates that in the southern portion of the Site, the VHG is consistently downward.

In the area of the former lagoons, the VHG was calculated for well nest MW-4/MW-2D-91. The average VHG under seasonally low groundwater conditions for MW-4/MW-2D-91 was 0.103 feet/foot. At seasonally high groundwater elevations, the VHG for MW-4/MW-2D-91 was 0.126 feet/foot. This also indicates that near the center of the Site, the VHG is consistently downward.

Bedrock Aquifer

The groundwater flow in the Bedrock Aquifer is divided into two flow components: north (north-northwest flow) and south (southwest flow) with a minor flow component in the west direction.

The horizontal gradients in the northern and southern portion of the Site are 0.025 feet/foot and 0.025 feet/foot, respectively as shown by the groundwater contours presented on Figures 5.9 and 5.10. The difference in the hydraulic gradient between the northern and southern Bedrock Aquifer flow zones is comparable to the difference seen in the Shallow Aquifer.

5.2.3 HYDRAULIC CONDUCTIVITY

Shallow Aquifer

The hydraulic conductivities of the Shallow Aquifer have been estimated by single-well response tests conducted during the RI as presented in Table 3.14

The hydraulic conductivities calculated from the single-well response tests in the northern and southern components of the shallow aquifer were 9.85×10^{-4} cm/s and 8.75×10^{-4} cm/s, respectively. The hydraulic conductivity that was calculated in the Shallow Aquifer in the former lagoon area was 1.07×10^{-4} cm/s. Since these values are similar in magnitude, it is assumed that an overall hydraulic conductivity of 8.75×10^{-4} cm/s is representative of the Shallow Aquifer across the Site.

Bedrock Aquifer

The hydraulic conductivity of the Bedrock Aquifer has been estimated by a number of methods during the RI. The methods used include:

i) short-duration pumping tests on wells MW-1D-91, MW-2D-91, MW-3D-91, and MW-4D-91 (June and July 1995);
ii) single-well response tests on wells MW-1D-91, MW-2D-91, MW-3D-91, MW-4D-91, MW-5D-95, MW-6D-95, DW-1-95, and DW-2-95 (June and July 1995);
iii) water injection tests (packer tests) on well DW-1-95; and
iv) the constant rate packer flow tests on wells MW-1D-91, MW-2D-91, MW-3D-91, and MW-5D-95 (June 2001).

The hydraulic conductivities calculated from the above tests are comparable (see Tables 3.14, 3.15, and 3.16) and are representative of the Bedrock Aquifer at the Site with the exception of one value calculated at DW-2-95. The hydraulic conductivity calculated for well DW-2-95 (5.22×10^{-7} cm/s) is low in comparison to the results for the water injection tests and the caliper logs (Rich 1986 and Appendix H).

The geometric mean hydraulic conductivities calculated for the former lagoon area and the southern flow component of the Bedrock Aquifer are 4.29×10^{-4} cm/s and 1.18×10^{-4} cm/s, respectively. The hydraulic conductivity representing the Bedrock Aquifer in the northern flow component is estimated to be 1.95×10^{-5} cm/s. An overall geometric mean of the hydraulic conductivity in the Bedrock Aquifer across the Site is estimated to be 1.93 x 10^{-4} cm/s.

5.2.4 GROUNDWATER VELOCITY

Shallow Aquifer

The horizontal groundwater velocity was estimated for the Shallow Aquifer using the following equation:

$$v = \frac{Ki}{n}$$

where:

v = velocity (feet/day)
K = hydraulic conductivity (feet/day)
i = horizontal hydraulic gradient (feet/foot)
n = effective porosity

This equation represents the velocity of groundwater flow through a homogeneous porous medium. Hydraulic conductivities were estimated by single-well response tests in the northern and southern component of the Shallow Aquifer. The hydraulic conductivity representing both components is estimated to be 8.75 x 10^{-4} cm/s. Horizontal hydraulic gradients in the northern and southern flow components of the Site are estimated to be 0.036 feet/foot and 0.020 feet/foot, respectively. Due to the variability of material in the Shallow Aquifer (i.e., sand, gravel, silt, clay, and shale), a weighted average effective porosity was determined. Mean porosities based on the U.S. EPA (EPA 1985) document entitled "A Screening Procedure for Toxic and Conventional Pollutants in Surface and Groundwater - Part II", were utilized to determine the weighted average effective porosity in the Shallow Aquifer. The effective porosities determined for the northern and southern components of the Site are 36 percent and 34.4 percent, respectively. Based on the above equation, the groundwater velocities in the northern and southern flow components are estimated to be 0.25 ft/day and 0.14 ft/day, respectively.

Bedrock Aquifer

Hydraulic conductivities for the northern and southern flow component of the Bedrock Aquifer were estimated by single-well response tests, short-term pumping tests, and water injection tests. The hydraulic conductivities representing the north and south flow components are estimated to be 1.95×10^{-5} cm/s and 1.18×10^{-4} cm/s, respectively. Horizontal hydraulic gradients in the northern and southern flow components of the Site are estimated to be 0.025 ft/foot and 0.025 ft/foot, respectively. The effective porosity of the shale ranges from 1.4 to 9.7 percent (EPA, 1985). An average mean porosity of 6 percent was assumed to be representative of the shale. Based on the above equation, the groundwater velocities for the northern and southern flow components are estimated to be 0.023 ft/day and 0.023 feet/day, respectively.

5.3 HYDROGEOLOGIC CONCEPTUAL MODEL

Based on the discussion and findings in Section 5.2, a hydrogeologic conceptual model of the Site has been developed. In general, there are two aquifers on Site through which the bulk of groundwater flow occurs. The Shallow Aquifer comprises the saturated portion of the sandy overburden deposits and the underlying upper few feet of weathered, fractured bedrock. The Bedrock Aquifer comprises the more competent shale bedrock underlying the highly weathered fractured zone.

A localized till unit (primarily in the south end of the Site) acts as an overburden aquitard unit. However, due to its limited areal extent, this till unit does not significantly affect general groundwater flow on Site.

In general, infiltration to the groundwater occurs in the area of the former lagoons and enters the Shallow Aquifer. Groundwater flow within the Shallow Aquifer occurs to the north and south (see Figures 5.7 and 5.8). A downward VHG exists between the Shallow Aquifer and the Bedrock Aquifer. Therefore, the Bedrock Aquifer is also partially recharged from the lagoon area. Groundwater flow in the Bedrock Aquifer also has a northern and southern horizontal component and a minor component which flows west towards Beaverdam Brook. Vertical flow to deeper portions of the Bedrock Aquifer is also a significant flow component.

A schematic diagram of the conceptual flow system is shown on Figure 5.11.

The Hydrological Evaluation of Landfill Performance computer model (HELP, Version 2.0) was utilized to estimate the average annual infiltration at the Site. The results of the computer modeling are presented in Appendix N. Based on NYSDEC comments to the Phase I RI (April 12, 1993) the HELP model was re-run for the Phase II RI utilizing Site-specific input parameters (i.e., in situ hydraulic conductivity, weighted average porosity, etc.). These results indicate that the average annual infiltration for the Site is approximately 11 inches/year.

Based on an estimated 11 inches/year of surface infiltration over the northern flow component area of 6.6 acres (289,300 ft^2) and southern flow component area of 12.7 acres (553,000 ft^2), the average daily infiltration over these areas is approximately 5,500 gal/day and 10,500 gal/day, respectively. Surface infiltration is the only source of recharge to the Shallow Aquifer.

The volume of groundwater flow through the Shallow and Bedrock Aquifer was estimated using the following form of Darcy's Law:

$$Q = KiA$$

where:

K = hydraulic conductivity (ft/day)
i = horizontal hydraulic gradient (ft/foot)
A = cross-sectional area (ft^2)
Q = horizontal groundwater flow (ft^3/day)

The groundwater flow in the Shallow Aquifer is calculated using the following input parameters (Section 5.2):

<u>Northern Flow Component:</u>

K = 2.79 feet/day (9.85×10^{-4} cm/s)
i = 0.036 ft/foot
A = 1,440 ft^2 to 4,800 ft^2
(assuming average saturated thickness of 3 feet in dry season and 10 feet in wet season and average cross-sectional width of 480 feet)

<u>Southern Flow Component:</u>

K = 2.48 ft/day (8.75×10^{-5} cm/s)
i = 0.020 ft/foot

A = 2,340 ft^2 to 7,020 ft^2
(assuming average saturated thickness of 3 feet in the dry season and 9 feet in the wet season and an average cross-sectional width of 780 feet)

The calculated flow to the north ranges between 1,100 and 3,600 gal/day and to the south ranges between 900 and 2,600 gal/day in the Shallow Aquifer.

Groundwater flow within the Bedrock Aquifer is also divided into the northern flow component and southern flow component as depicted on Figures 5.9 and 5.10. Applying Darcy's equation of flow, the horizontal flow in the Bedrock Aquifer is calculated using the following input parameters:

Northern Flow Component:

K = 0.0552 feet/day (1.95 x 10^{-5} cm/s)
i = 0.025 feet/foot
A = 27,600 ft^2 to 44,400 ft^2
(The saturated thickness is assumed from the deepest well, DW-2-95 as 92 feet in the dry season and 148 feet in the wet season. The average cross-sectional width is 300 feet.)

Southern Flow Component:

K = 0.544 feet/day (1.92 x 10^{-4} cm/s)
i = 0.025 feet/foot
A = 30,360 ft^2 to 47,380 ft^2
(The saturated thickness is assumed from the deepest well, DW-1-95 as 66 feet in the dry season and 103 feet in the wet season. The average cross-sectional width is 460 feet.)

The calculated flow to the north ranges between 290 and 460 gal/day and to the south ranges between 3,090 and 4,820 gal/day in the Bedrock Aquifer. Discharge of northward flow in the Bedrock Aquifer is towards Beaverdam Brook. Discharge of southward flow is towards Otter Kill Valley.

A groundwater flow budget is provided in Table 5.2. The difference between the Shallow Aquifer groundwater flow and the surface infiltration ranges between 13,800 and 118,000 gal/day. These values represent the vertical recharge to the bedrock.

5.4 SURFACE WATER HYDROLOGY

5.4.1 REGIONAL SURFACE WATER HYDROLOGY

The Site is located within a 4.5 square mile watershed consisting of Beaverdam Brook and its tributaries, which discharges to the Otter Kill. The watershed area of the Otter Kill, at the confluence of Beaverdam Brook and the Otter Kill, is approximately 32 square miles. The confluence is located approximately 500 feet south of the Site.

Beaverdam Brook flows south into and through the northwestern portion of the Site. It drains several wetland areas north of the Site and terminates at the Otter Kill, approximately 500 feet south of the property. The Otter Kill flows generally toward the southeast. Approximately one mile east of the Site it is dammed to form Brown's Pond, a small impoundment formerly used for agricultural purposes. Brown's Pond is identified on Figure 3.2. The Otter Kill meets Cromline Creek to form Moodna Creek about one mile west of the village of Washingtonville. Moodna Creek drains into the Hudson River, which is approximately 8 miles to the east.

As flow records are not available for the Otter Kill, the flow rate was estimated by using flow data for the Wallkill River at Gardiner, New York, as presented in "Water Resources Data, New York, Water Year 1984, Volume 1," USGS water data report NY-84-1 (USGS NY-84-1). The Wallkill River is located approximately two to three miles west of the Site and flows northward into the Hudson River. The Gardiner station is the closest river gage to the Site, located approximately 18 miles to the north. The drainage area of the Wallkill River at the gage station is 711 square miles. The mean flow for the Wallkill River was adjusted, based upon the relative drainage areas contributing to the flow rates, to estimate the mean flow in the Otter Kill at the confluence of the Otter Kill and Beaverdam Brook.

The average discharge at the Gardiner station over a 60-year period is 1,067 cfs, as reported in USGS NY-84-1. The average flow for the Otter Kill, at the confluence with Beaverdam Brook, therefore, is estimated to be approximately 48 cfs.

5.4.2 SITE SURFACE WATER HYDROLOGY

The Site is situated on rolling topography of low to moderate relief. Two hills located near the center of the property are the high points of the Site with an elevation of approximately 400 feet AMSL. Between these hills is a topographic "saddle" where the lagoons formerly existed. The rest of the Site generally slopes downward from the two hills. The low point on the property is approximately 345 feet AMSL at Beaverdam Brook, near the northwestern property boundary. The six former lagoon locations have been filled in with sand and shale fragments quarried at the Site, and now appear to be at or near their former natural grade.

Surface drainage within the Site is topographically controlled. There are no artificial impervious surfaces on Site, therefore surface drainage generally follows a natural pattern radially away from the central part of the Site as presented on Figure 5.12.

An additional phase of surface water investigation was conducted during the Phase II RI to further delineate potential points of groundwater discharge to the surface waters.

The surface water level monitoring stations in Beaverdam Brook, Otter Kill, and the northeast swamp are presented on Figure 5.13. Surface water elevations presented on Figure 5.13 were measured concurrently with the Site groundwater elevations on July 24, 1995 and August 14, 1995. Surface water elevations were also measured on June 28, 1995. High and low surface water elevations were observed on June 28, 1995 and August 14, 1995, respectively. The surface water elevations are presented in Table 5.3. The greatest change in surface water elevations between high and low surface water elevations were noted at stations WEII-6, WEII-7, and WEII-9. Consistent with the regional surface water flow pattern, Beaverdam Brook flows from north to south, along the west boundary of the Site, towards Otter Kill which is south of the Site.

Site groundwater flow in the Shallow and Bedrock Aquifer is divided into a northern and southern component (see Section 5.2). Examination of the Site groundwater and surface water elevations indicate that the Site groundwater elevations are higher than surface water elevations measured in Beaverdam Brook, Otter Kill, and the northeast swamp. Groundwater in the northern portion of the Site likely discharges to surface water in Beaverdam Brook (based upon water table groundwater elevations, see Figure 5.7 and 5.8). Groundwater in the southern portion of the Site may discharge to surface water in Beaverdam Brook and Otter Kill. Due to a minor west flow component in the Bedrock Aquifer, groundwater may also discharge towards Beaverdam Brook to the west of the Site.

6.0 CHEMICAL DISTRIBUTION

The following sections present descriptions of the distribution of chemicals in the different media at the Site. Where appropriate, historical data are included to supplement the data obtained during the RI investigation. The data validation of all analytical data generated during the RI and Additional Investigations is presented in Appendix L. All analytical results are organized according to media type and are presented in Appendix I.

6.1 SOIL SAMPLING RESULTS

For the purpose of evaluating the detected compounds, the analytical results were compared to appropriate criteria and the exceedances are presented in the following paragraphs. Organic results were compared to the NYSDEC Recommended Soil Cleanup Objectives from the document "Technical and Administrative Guidance Memorandum (TAGM): Determination of Soil Cleanup Objectives and Cleanup Levels" (TAGM #4046, NYSDEC, 1994). Soil Cleanup Objectives for the pyridine compounds were determined by the NYSDEC in a letter dated August 14, 1996. The letter is provided in Appendix S.

Inorganics were compared to both the generic residential risk-based criterion (RBC) for U.S. EPA Region III (U.S. EPA, October 2004) and background soil concentrations using the methods presented in the Additional Soil Investigation Work Plan. Lead did not have a Region III RBC, so the U.S. EPA Region IX Preliminary Remediation Goal (PRG) for residential soil (U.S. EPA, October 2002) was used.

Surface soil sampling results are screened separately against U.S. EPA Soil Screening Levels (www.epa.gov/ecotox/ecossl/) and the USDOE Preliminary Remediation Goals for Ecological Endpoints (ES/ER/TM-162/R2) in Section 8.0.

Calcium, magnesium, potassium, and sodium do not have RBCs, as they are essential nutrients.

6.1.1 BACKGROUND SOIL SAMPLES

The Additional Investigation included the collection of 32 soil samples in 2003 from the surrounding area to use as background concentrations for the parameters. The samples were analyzed for TAL metals, and cyanide. The upper 0 to 2 ft bgs interval from each borehole was also analyzed for pesticides to further evaluate the source of pesticides detected in soil samples previously collected at the Site. Figure 3.5 presents the sample locations that were selected in conjunction with the U.S. EPA and NYSDEC. The analytical results are presented in Appendix I.1 and are summarized in Table 6.1.

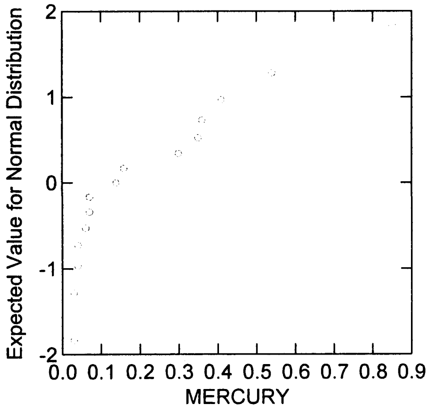
Two pesticides were detected in the background samples that were collected. 4,4'-DDE was detected at 4.0 µg/kg at S-12 and 33 µg/kg at S-15. Both concentrations did not exceed the NYSDEC Soil Cleanup Objective. 4,4'-DDT was detected at 3.7 J µg/kg at S-12 and 33 µg/kg at S-15, also not exceeding the NYSDEC Soil Cleanup Objective.

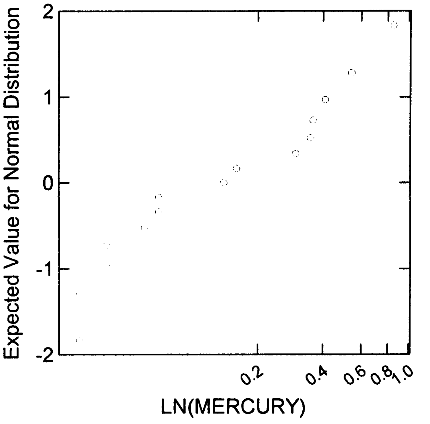
TAL metals were detected in all background samples that were collected. Arsenic and iron were detected at concentrations exceeding the Region III RBCs. Arsenic exceeded the Region III RBCs in all sample locations, at concentrations ranging from 3.3 mg/kg at S-12 to 13.9 J mg/kg at S-18. Iron exceeded the Region III RBCs in all sample locations except S-3 (0-2 ft bgs), S-5 (0-2 ft bgs), S-6 (6-7 ft bgs), S-12 (0-2 ft bgs), S-14 (0-2 ft bgs), and S-18 (0-2 ft bgs). Exceeding concentrations for iron ranged from 24,700 mg/kg at S-11 to 48,500 mg/kg at S-18.

6.1.2 METHYLMERCURY ANALYSIS

As outlined in Section 3.8, mercury speciation analyses (of total mercury and methylmercury) were performed for 15 soil samples selected from the 120 soil samples collected in May 2003 during the Additional Investigation. The samples and results for the U.S. EPA and NYSDEC approved mercury speciation samples are presented in Table 6.2 and Figure 6.4. The percentages of methylmercury versus total mercury ranged from 0.03 to 0.85 percent, with a mean of 0.23 percent methylmercury. Prior to performing statistical calculations, characteristics of the data set were evaluated. Specifically, the data distribution and presence of statistical outliers were tested. A significance level of 0.05 (i.e., 95 percent confidence) was applied for all statistical procedures.

Data distribution was tested using probability plots and the Shapiro-Wilk W-test (U.S. EPA, 2000). From inspection of the probability plots (shown below for raw and log-transformed data), the data appear to be lognormally distributed, as evidenced by the straight line of the individual points on the log-transformed plot. The Shapiro-Wilk test results confirmed this finding that the data are lognormally distributed.





The presence of statistical outliers was tested using probability plots and Dixon's test for outliers (U.S. EPA, 2000). On the probability plot of the log-transformed data (above), the extreme observations (i.e., the highest and lowest percentages of methylmercury) appear to be consistent with the rest of the data (i.e., fit on the line in the plot). The only exception is the lowest point that does not seem to fit on the otherwise straight line. Further evaluation using Dixon's test did not indicate the presence of statistical outliers, and therefore all fifteen methylmercury concentration percentages were retained for subsequent analysis.

Mercury speciation results are used in the HHRA as the basis for risk estimates for methylmercury exposure across the entire Site. The mercury speciation results will be extrapolated to the total mercury concentrations for the remaining investigative samples not subjected to the speciation analysis when calculating exposure terms. To provide a conservative estimate representing a maximum exposure concentration scenario, the mercury speciation data have been used to calculate an upper tolerance limit (UTL) on the methylmercury vs. total mercury ratio. This UTL represents the upper limit on the 95th percentile of the mercury speciation data with 95 percent confidence. That is, if additional samples from lagoon soils were submitted for mercury speciation analysis, a maximum of one in twenty of these samples could have a percentage of methylmercury higher than the UTL.

Upper tolerance limits have been recommended by U.S. EPA (1989, 1992, 2001) and the Texas Natural Resource Conservation Commission (TNRCC) (1992) for use as tools to provide upper bounds on measured parameters, for example, to establish an upper bound on background concentrations of a chemical. These UTLs provide estimates that are both conservative and representative of the maximum concentration. UTLs achieve higher than the specified coverage in most cases since they are based on observed values, and not on assumptions. UTL calculation details are provided in U.S. EPA, 1989.

A UTL for the mercury speciation data was calculated using the mean and standard deviation of the log-transformed data, and applying a statistical tolerance coefficient (k) from a standard table (Hahn and Meeker, 1991). The resulting UTL was 2.4 percent methylmercury.

The mercury speciation data collected for the Site represent a well-characterized data set. The fifteen samples that were used for speciation represent the entire range of detected total mercury concentrations for the 120 soil samples collected in May 2003. For the purposes of estimating an exposure term for methylmercury, the UTL approach will provide a conservative, yet still representative, estimate of the Site-wide methylmercury concentration. This is further demonstrated by the fact that the calculated UTL of 2.4 percent is more than 10 times greater than the mean of the selected samples (0.23 percent) and approximately three times greater than the maximum observed methylmercury concentration (0.85 percent) for the speciated samples.

Based on the results of the mercury speciation analysis, CRA recommends that the UTL value of 2.4 percent methylmercury be used to represent a conservative methylmercury percentage for soil samples subjected to total mercury analysis for the purposes of calculating an exposure estimate for methylmercury in the HHRA. The remaining 97.6 percent will be evaluated as inorganic mercury. (CRA Memorandum dated March 12, 2004). U.S. EPA concurred with the speciation analysis in a letter transmitted electronically on May 25, 2004.

6.1.3 SURFACE SOILS

Surface soil samples were collected from each of the six former lagoons and the Site access road during the Phase II RI as discussed in Section 3.2.3. The eight investigative samples were analyzed for TCL VOCs, SVOCs, pesticides/PCBs, TAL metals, cyanide, TOC and Site-specific parameters. Analytical results are presented in Appendix I.1 and are discussed in the following paragraphs.

A summary of detected parameters is presented in Table 6.3. Sampling locations for VOC and SVOC showing detected parameters are presented on Figure 6.1. Tentatively identified compounds (TICs) are presented in Appendix I.1.

VOCs

VOCs were detected at low levels in six of the eight surface soil samples as presented on Figure 6.1. Tetrachloroethene was detected at concentrations ranging from 3 J μg/kg to

51 J µg/kg at locations SSII-1, SSII-3, SSII-5, SSII-6, and SSII-7. Acetone was detected at low levels ranging from 1 J µg/kg to 18 µg/kg at locations SSII-2, SSII-5, and SSII-6. Xylene was detected at a concentration of 2 J µg/kg at location SSII-7. No other VOCs were detected in the surface soil samples. No VOCs were detected in samples collected from locations SSII-4 and SSII-8.

As presented in Table 6.3, the detected VOCs did not exceed the Soil Cleanup Objectives in any of the surface soil samples.

SVOCs

SVOCs were detected in samples collected from seven of the eight sample locations with the most compounds detected at location SSII-7 situated on the Site access road. Seven SVOCs (phenanthrene (86 J µg/kg), fluoranthene (110 J µg/kg), pyrene (220 J µg/kg), benzo(a)anthracene (150 J µg/kg), chrysene (230 J µg/kg), bis(2-ethylhexyl)phthalate (59 J µg/kg), and benzo(a)pyrene (200 J µg/kg)) were detected at location SSII-7.

Bis(2-ethylhexyl)phthalate was detected at concentrations ranging from 54 J µg/kg to 130 J µg/kg from sample locations SSII-2, SSII-3, SSII-6, SSII-7, and SSII-8.

Fluoranthene was detected in samples collected from locations SSII-1, SSII-2, SSII-4, SSII-6, and SSII-8 at concentrations ranging from 61 J µg/kg to 110 J µg/kg.

Other SVOCs that were detected in other locations than the previously mentioned, include 4-methylphenol (160 J µg/kg at SSII-6), pyrene (59 J µg/kg at SSII-2 and 85 J µg/kg at SSII-4), benzo(a)pyrene (57 J µg/kg at SSII-2), and di-n-octylphthalate (640 J µg/kg at SSII-6).

Site-specific parameters alpha-picoline, 2-amino-pyridine, and pyridine were not detected in any of the surface soil samples.

There were no exceedances of the Soil Cleanup Objectives except for an estimated value of benzo(a)pyrene (200 J µg/kg) at SSII-7.

Pesticides

The majority of the pesticides detected were found at sample location SSII-6, situated close to Lagoon 6, at concentrations ranging from 0.89 J µg/kg to 220 J µg/kg. These pesticides include 4'4-DDE, 4'4-DDT, beta-BHC, delta-BHC, dieldrin, endosulfan I, endosulfan sulfate, endrin aldehyde, endrin keytone, heptachlor, heptachlor epoxide, and methoxychlor.

Pesticides were not detected in samples collected from the other locations with the exception of SSII-5 (endrin (2.7 J µg/kg), 4,4'-DDT (2.5 J µg/kg).

All pesticides that were detected were reported at concentrations below the Soil Cleanup Objectives.

PCBs

Aroclor-1254 was detected in four of the eight sample locations at the following concentrations:

Sample	*Location*	*Concentration (µg/kg)*
SSII-4	Lagoon 4	30 J
SSII-5	Lagoon 5	34 J
SSII-6	Lagoon 6	4800 J/8900 J
SSII-8	Site access road	32 J

The concentrations reported for sample location SSII-6 exceeded the NYSDEC Soil Cleanup Objective. Aroclor-1254 was not detected, nor any other PCB, at locations SSII-1, SSII-2, SSII-3, or SSII-7.

Metals

All samples exhibited concentrations of arsenic and iron greater than the Region III RBCs. Concentrations ranged from 6.0 mg/kg to 10.4 mg/kg for arsenic, and 23,800 mg/kg to 34,200 mg/kg for iron. However, these ranges are consistent with the background samples which also exceeded the Region III RBCs (see Section 6.1.1). Total mercury was reported at concentrations higher than the Region III RBC for methylmercury at sample location SSII-6 (23.4 J/13 J mg/kg). These concentrations are also greater than background levels.

Total Organic Carbon

TOC concentrations ranged from 8,160 mg/kg to 87,400 mg/kg for the eight surface soil samples as presented on Table 6.3. The average TOC concentration for the eight samples is 27,871 mg/kg (2.7 percent).

6.1.3.1 SURFACE SOILS SUMMARY

No VOCs were detected at concentrations exceeding the Recommended NYSDEC Soil Cleanup Objectives. Benzo(a)pyrene was the only SVOC detected at an estimated concentration exceeding the NYSDEC criteria. The contaminant was found at sample location SSII-7 at a concentration of 200 J µg/kg. Site-specific parameters such as pyridine, 2-aminopyridine, and alpha-picoline were not detected in the surface soil samples. PCB exceedances included Aroclor-1254 which was detected at a maximum concentration of 8,900J µg/kg at SSII-6. Total mercury exceeded the Region III RBC for methylmercury at SSII-6 (23.4 J mg/kg), and in all samples, arsenic (with a maximum concentration of 10.4 mg/kg) and iron (with a maximum concentration of 34,200 mg/kg) exceeded the Region III RBC but were within the range of background concentrations.

6.1.4 SUBSURFACE SOILS

Subsurface soil samples were collected during the Phase I and II RI in 1991 and 1995, the Treatability study in 1996 and for the Additional Investigation in 2003. The analytical results for these samples are presented in Appendix I.1 and discussed in the following sections.

Similarly to the surface soil samples, organic results were compared to the Recommended NYSDEC Soil Cleanup Objectives, and inorganics were compared to both U.S. EPA Region III RBCs and site-specific background soil concentrations.

6.1.4.1 FORMER LAGOON AREAS

Soil samples selected for chemical analysis were analyzed for TCL VOCs, SVOCs, pesticides/PCBs, TAL metals, cyanide, and Site-specific parameters with the exception of the Additional Investigation in 2003, for which pesticides/PCBs, TAL metals, and cyanide were analyzed. Sampling locations are presented on Figures 3.1 and 3.6. A summary of detected parameters is presented in Tables 6.4 to 6.6 and organic parameter results are summarized on Figures 6.2 and 6.3. Additionally, mercury speciation was performed on selected samples from the 2003 Additional Investigation and the results are presented on Figure 6.4 and Table 6.2.

Lagoon 1

A total of twenty-four investigative soil samples were collected at or in the vicinity of Lagoon 1. Four samples were collected in 1991 from four test pits: TP-11A, TP-12, TP-15, and TP-46, four samples were collected in 1996 from two test pits: L1-TP1 and L1-TP3, and 19 samples were collected in 2003 from six sample locations: L1-1, L1-2, L1-3, L1-4, L1-5, L1-6, L1-7, and L1-8.

The VOCs detected in the nine samples analyzed for VOCs were 2-butanone, 4-methyl-2-pentanone, acetone, benzene, chlorobenzene, carbon disulfide, ethylbenzene, toluene, and xylene. Of these, exceedances occured for acetone, benzene, toluene and xylene. Acetone concentrations exceeded the Soil Cleanup Objective in TP-12 (450 µg/kg) and TP-11A (300 J µg/kg). Benzene was found to exceed the Soil Cleanup Objective at six locations: TP-11A (210 µg/kg), TP-12 (120 µg/kg), TP-15 (160 J µg/kg), TP-46 (330JD/300 µg/kg), L1-TP1 (120 µg/kg) and L1-TP3 (160/190 µg/kg). Toluene exceeded the Soil Cleanup Objective at TP-11A with a concentration of 26,000 µg/kg. Xylene was found to exceed the Soil Cleanup Objective in four locations: TP-12 (2100 µg/kg), TP-11A (4500 µg/kg), TP-46 (3,800D/7,700 µg/kg) and L1-TP3 (1300 µg/kg). Total petroleum hydrocarbon concentrations ranged from 253 mg/kg at TP-11A to 4,980 mg/kg at TP-15.

SVOCs that were detected in the nine samples were 2-aminopyridine, 2-methyl naphthalene, 2-picoline, bis(2-ethylhexyl)phthalate, naphthalene, and pyridine. Bis(2-ethylhexyl)phthalate was detected at a concentration of 60,000 J µg/kg at TP-46, exceeding the Soil Cleanup Objective. 2-Aminopyridine was detected at a concentration exceeding the Soil Cleanup Objective at sample locations TP-46 (750 µg/kg), L1-TP1 (2900J µg/kg), and L1-TP3 (610J/720J µg/kg). Alpha-picoline and pyridine exceeded the Soil Cleanup Objective in the 1985 sample L-1 with a concentration of 7,600J µg/kg and 3,300J µg/kg, respectively. The other detected parameters did not exceed the Soil Cleanup Objectives.

Fifteen soil samples from 10 sample locations were analyzed for pesticides and PCBs. Pesticide concentrations were detected in seven of the samples at four locations (TP-15, L1-TP1, L1-TP3 and L1-4). Detected pesticides were found at concentrations exceeding background, but below Soil Cleanup Objectives: 4,4'-DDD (9.6 µg/kg at L1-TP3), 4,4'-DDE (2.2 J ug/kg and 3.1 J ug/kg), aldrin (9.9 µg/kg at TP-15), alpha-BHC (4 µg/kg at L1-TP1), alpha-chlordane (1 J µg/kg at L1-TP3), endosulfan I (0.97J/1.1 J µg/kg at L1-TP3), endrin (7.3 J and 2.1 J µg/kg at L1-TP3), gamma-chlordane (5.4 and 1.4 J/2.7J µg/kg at L1-TP3), and heptachlor epoxide (1.7 J and 1.4 J at L1-TP3). Aroclor-1254 was the only PCB detected, and was found at concentrations below the Soil

Cleanup Objective at locations TP-11A (780 μg/kg), TP-15 (260 μg/kg) and L1-TP3 (58 J/100 J μg/kg).

All samples, except TP-46, exceeded the Region III RBC of 0.43 mg/kg for arsenic. However, the maximum background arsenic concentration detected was 13.9 J mg/kg, which exceeded all but the TP-11A sample (87.5 mg/kg). This value was believed to be suspect due to the elevated level of aluminum reported at this location (294,000 mg/kg) which exceeded the Region II RBC.

Manganese was detected above the Region III RBC and background at L1-1 (1620 mg/kg), and L1-8 (1640 J mg/kg). Total mercury exceeded the Region III RBC for in L1-TP3 with a concentration of 79.9 mg/kg. In the case of iron, all samples except the sample taken at TP-11A, and two samples taken from L1-5, exceeded the Region III RBC. Similarly to arsenic, the maximum iron concentration found in the background samples exceeded all iron concentrations detected in Lagoon 1. The remaining parameters detected in the Lagoon 1 samples were below the Region III RBCs.

Lagoon 2

A total of twenty-five subsurface soil samples were collected from Lagoon 2. Three soil samples were collected from two test pits in 1996: L2-TP1 and L2-TP2. In 2003, 22 samples were collected from 11 sample locations: L2-1, L2-2, L2-3, L2-4, L2-5, L2-6, L2-7, L2-8, L2-9, L2-10, and L2-11.

Only the 1996 samples were analyzed for VOCs and SVOCs. Ten VOCs were detected in the samples: 1,2-dichloropropane, 2-butanone, 4-methyl-2-pentanone, acetone, benzene, carbon disulfide; chlorobenzene, ethylbenzene, styrene, toluene, and xylene. Five of the detected VOCs in Lagoon 2 exceeded the Soil Cleanup Objectives in two locations. At L2-TP1, benzene was detected at 1,200J/1,300J μg/kg, chlorobenzene was detected at 5,600 J/3,900 J μg/kg, ethylbenzene was detected at 32,000/22,000 μg/kg, toluene was detected at 7,700 μg/kg, and xylene was detected at 300,000/240,000 μg/kg. At L2-TP2, detected compounds above the Soil Cleanup Objectives were benzene (13,000 μg/kg), chlorobenzene (12,000 μg/kg), ethylbenzene (13,000 μg/kg) in L2-TP2, toluene (7,200 μg/kg), and xylene (43,000 μg/kg).

Three SVOCs were detected in the four samples from L2-TP1 and L2-TP2. 2-Aminopyridine was detected in L2-TP1 and L2-TP2 at 2,200 J μg/kg and 19,000 J μg/kg, respectively. 2-Methyl naphthalene and naphthalene were also detected, but concentrations were below the Soil Cleanup Objectives.

A number of pesticides were detected in samples collected from Lagoon 2. 4,4'-DDD, 4,4'-DDE, 4,4'-DDT, aldrin, alpha-BHC, dieldrin, endosulfan I, endosulfan II, endosulfan sulfate, endrin, gamma-chlordane, heptachlor epoxide, and methoxychlor were all detected in a number of samples. Only sample location L2-TP1 displayed concentrations of aldrin (77 J µg/kg), dieldrin (140 J and 170 J µg/kg), endrin (390 J and 300 J µg/kg) and heptachlor epoxide (44 J and 36 J µg/kg) that were above the Soil Cleanup Objectives.

Aroclor-1254 and Aroclor-1260 were detected at concentrations ranging from 78 to 15,000 µg/kg and 310 J to 4,300 J µg/kg, respectively. Of these PCBs, Aroclor-1254 exceeded the Soil Cleanup Objective at sample location L2-TP1.

Inorganics detected in Lagoon 2 above the Region III RBCs were arsenic, iron, manganese, and mercury. Total mercury was detected in 19 samples with two of them being above the Region III RBC at 51.3 J mg/kg (L2-TP1) and 30.1 mg/kg (L2-7). Manganese was only detected above the Region III RBC in one sample at 1,640 mg/kg in L2-TP1. Arsenic was detected above the Region III RBC in all samples ranging from 3.5 J mg/kg to 12.3 J µg/kg. Iron was detected above the Region III RBC in all samples but one from L2-2 and two from L2-4. All arsenic, manganese, and iron concentrations were within the range of concentrations found in the background samples.

Lagoon 3

A total of forty soil samples were collected at or in the vicinity of Lagoon 3. Two samples were collected from three test pits in 1991: one sample was collected at TP-6, and the second sample was a composite of samples collected at TP-23 (7 ft bgs) and TP-24 (5 ft bgs). In 1996, six soil samples were collected from three test pits: L3-TP1, L3-TP2 and L3-TP3. In 2003, 32 soil samples were collected from sixteen sample locations: L3-1, L3-2, L3-3, L3-4, L3-5, L3-6, L3-7, L3-8, L3-9, L3-10, L3-11, L3-12, L3-13, L3-14, L3-15, and L3-16.

A total of eight VOCs were detected in the subsurface soil samples collected from the Lagoon 3 area: 2-butanone, acetone, benzene, carbon disulfide, chlorobenzene, ethylbenzene, toluene, and xylene. Only benzene exceeded the Soil Cleanup Objectives at L3-TP3 with a concentration of 67 J µg/kg. Total petroleum hydrocarbons were detected at 10.8 mg/kg (TP-6) and 13.9 mg/kg (TP-23/TP-24). No VOCs were detected at sample L3-TP1.

2-Aminopyridine was the only SVOC detected at a concentration exceeding the Soil Cleanup Objective. The compound was reported at concentrations ranging from

720 µg/kg to 74,000 µg/kg in sample locations TP-6, L3-TP1, L3-TP2, and L3-TP3. Concentrations of 4,4'-DDD, 4,4'-DDE, alpha-BHC, alpha-chlordane, endosulfan I, endosulfan II, endosulfan sulfate, endrin, endrin aldehyde, gamma-chlordane, and heptachlor epoxide, were detected in the three samples from L3-TP3. All concentrations for these parameters fell below its Soil Cleanup Objectives.

PCBs detected were Aroclor-1254 and Aroclor-1260. Concentrations were 37 J to 110 µg/kg for Aroclor-1254 and 30 J µg/kg for Aroclor-1260. Neither parameter exceeded its Soil Cleanup Objective.

All inorganics detected in Lagoon 3 were below the Region III RBC with the exception of arsenic, manganese, and iron. All samples exceeded arsenic criteria with concentrations ranging from 3.0 to 13.3 J mg/kg. All samples (except one from each of L3-2, L3-4, L3-5, L3-8, L3-10, L3-11, L3-13, L3-15, and two from L3-14) exceeded iron criteria with a maximum concentration of 35,500 mg/kg. Manganese was detected at a concentration of 1,770 mg/kg in L3-TP3 and 1,650 mg/kg in L3-7. Concentrations of these parameters are consistent with the background concentrations.

Lagoon 4

Twenty-seven investigative soil samples were collected from Lagoon 4. Two composite samples and one other sample were collected in 1991: one composite was from test pits TP-22 (7 ft bgs) and TP-26 (6 ft bgs), and the second was from TP-20 (7 ft bgs) and TP-29 (5 ft bgs); the third soil sample was collected at TP-28. Five samples were collected from test-pits L4-TP1 and L4-TP2 in 1996. Nineteen samples were collected from the following sample locations in 2003: L4-1, L4-2, L4-3, L4-4, L4-5, L4-6, L4-7, L4-8, and L4-9.

2-Butanone, acetone, benzene, carbon disulfide, xylenes, toluene, and chlorobenzene were detected in the soil samples collected from Lagoon 4. The concentrations of these parameters did not exceed the Soil Cleanup Objectives with the exception of acetone. Acetone was detected at 210 µg/kg in TP-28. Total petroleum hydrocarbons were detected at concentrations ranging from 1.3 mg/kg to 8.3 mg/kg.

The compounds 2-aminopyridine and bis(2-ethylhexyl)phthalate were the only SVOCs detected in the samples from Lagoon 4. 2-Aminopyridine was detected at 9 of 11 sample locations at concentrations ranging from 260 J µg/kg to 99,000 J µg/kg. Six of the samples exhibited concentrations exceeding the Soil Cleanup Objective. Bis(2-ethylhexyl)phthalate did not exceed the Soil Cleanup Objective.

Pesticides detected in the samples collected at Lagoon 4 were found at TP-28, and the two 4-6 ft bgs samples collected at L4-TP1. The parameters detected at L4-TP1 were 4,4'-DDD (87 J/110 J μg/kg), aldrin (1.4 J μg/kg), endosulfan I (2.3/3.1 μg/kg), endosulfan II (4.9 J/10 J μg/kg), endrin (3.4 J μg/kg), and heptachlor epoxide (3 J/5.9 J μg/kg). At TP-28, 4,4'-DDD was reported at 30X μg/kg.[1] Concentrations of these parameters did not exceed the Soil Cleanup Objectives.

PCBs were detected in the samples collected at TP-28, the two 4-6 ft bgs samples collected at L4-TP1 and the 3 ft bgs sample collected at L4-TP2. Aroclor-1254 (110 J/210 J μg/kg), and Aroclor-1260 (71/100 μg/kg) were detected at L4-TP1. At TP-28, Aroclor-1254 was reported at 210 μg/kg. L4-TP2 had only Aroclor-1254 reported at 22 J μg/kg. PCB concentrations did not exceed the Soil Cleanup Objectives.

All inorganics detected in Lagoon 4 were below the Region III RBCs with the exception of arsenic, manganese, and iron. All samples exceeded the Region III RBC for arsenic with concentrations ranging from 3.6 to 10.2 J mg/kg. All samples (except one from each of L4-1, L4-4, L4-5, L4-6, L4-8, L4-9, and two from L4-3) exceeded the Region III RBC for iron with a maximum concentration of 34,400 mg/kg. Manganese exceeded the region III RBC with concentrations at 1750 J mg/kg and 1630 mg/kg at L4-TP1 and L4-1 respectively. Concentrations for these parameters are consistent with background concentrations.

Lagoon 5

A total of 27 subsurface soil samples were collected at or in the vicinity of Lagoon 5. Two investigative samples were collected from two test pits in 1991 (TP-2 and TP-4) and three were collected from two test pits in 1996 (L5-TP1 and L5-TP2). Twenty-two samples were collected in 2003 at eleven sampling locations: L5-1, L5-2, L5-3, L5-4, L5-5, L5-6, L5-7, L5-8, L5-9, L5-10, and L5-11.

1,2-Dichloroethene, 4-methyl-2-pentanone, acetone, benzene, carbon disulfide, chlorobenzene, ethylbenzene, toluene, trichloroethene, and xylene were all detected in the soil samples collected from Lagoon 5. The concentrations of these parameters which exceeded the Soil Cleanup Objectives were almost all in the sample from L5-TP2: benzene at 190 μg/kg; chlorobenzene at 6,900 μg/kg; ethylbenzene at 12,000 μg/kg; toluene at 52,000 μg/kg; and xylene at 120,000 μg/kg. The toluene Soil

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[1] X - Compound with spectra that do not meet identification criteria, but is suspected to be present.

Cleanup Objective was also exceeded at TP-4 with a concentration of 2,200 µg/kg. The remaining VOCs did not exceed the Soil Cleanup Objectives.

Only 2-aminopyridine was detected above the Soil Cleanup Objective in four sample locations at concentrations ranging from 470 J µg/kg to 1,800 µg/kg. Other SVOCs that were detected at concentrations below the Soil Cleanup Objective include 2-methylnaphthalene, bis(2-ethylhexyl)phthalate, phenanthrene, and pyridine.

Several pesticides were detected in the samples collected at Lagoon 5. These parameters include 4,4'-DDE, 4,4'-DDT, alpha-BHC, alpha-chlordane, beta-BHC, dieldrin, endosulfan I, endosulfan II, endosulfan sulfate, endrin, endrin aldehyde, endrin keytone, gamma-chlordane, heptachlor epoxide and methoxychlor. Concentrations of these parameters did not exceed the Soil Cleanup Objectives.

PCBs were detected in the samples collected in 1996: L5-TP1 and L5-TP2. Aroclor-1254 (1,200 J µg/kg), and Aroclor-1260 (700 J µg/kg) were detected at L5-TP1. At L5-TP2, Aroclor-1254 was reported at 290 to 910 µg/kg and Aroclor-1260 at 290 J µg/kg. PCB concentrations did not exceed the Soil Cleanup Objectives.

All inorganics detected in Lagoon 5 were below the Region III RBCs with the exception of arsenic, manganese, mercury, and iron. All samples exceeded the Region III RBC for arsenic with concentrations ranging from 3.0 to 13.3 J mg/kg. All samples (except one from each of L3-2, L3-4, L3-5, L3-8, L3-10, L3-11, L3-13, L3-15, and two from L3-14) exceeded the Region III RBC for iron with a maximum concentration of 35,500 mg/kg. Manganese exceeded the Region III RBC at TP-2 (2,470 mg/kg) and L5-6 (1,980 mg/kg). Concentrations for these parameters are consistent with background concentrations. Total mercury was detected above the Region III RBC in L5-TP1 and L5-TP2 at concentrations of 61.7 mg/kg and 26.5 mg/kg, respectively.

Lagoon 6

A total of 12 subsurface soil samples were collected at or in the vicinity of Lagoon 6. One sample was collected in 1991 at TP-41, another sample was collected in 1996 at L6-TP3. The remaining eleven soil samples were collected from seven test pits in 2003: L6-1, L6-2, L6-3, L6-4, L6-5, L6-6, and L6-7.

Only one VOC, at one location (toluene at 1,800J µg/kg (TP-41)), was detected in the analyzed samples, at a concentration exceeding the Soil Cleanup Objectives. TPH at 300 mg/kg was also detected. Three SVOCs, 4-nitrophenol, benzoic acid, and bis(2-ethylhexyl)phthalate were also detected but only benzoic acid at TP-41 (4,400 µg/kg) exceeded the Soil Cleanup Objective.

Several pesticides were detected in the samples collected at Lagoon 6. These parameters include aldrin (190J µg/kg), methoxychlor (120 µg/kg), dieldrin (37X µg/kg), endrin (26J µg/kg), heptachlor epoxide (20X µg/kg and 1J µg/kg), and 4,4'-DDT (20X µg/kg). Of these parameters, only aldrin was found to exceed the Soil Cleanup Objectives.

Aroclor-1254, the one PCB detected in the samples collected at Lagoon 6, did not exceed the Soil Cleanup Objective with concentrations of 9,200J µg/kg at TP-41 and 110 µg/kg at L6-TP-3.

All inorganics detected in Lagoon 6 were below the Region III RBCs with the exception of arsenic and iron. All samples exceeded the arsenic Region III RBC with concentrations ranging from 6.5 to 47.6 mg/kg. All samples except one, L6-4, exceeded the iron Region III RBC with a maximum concentration of 49,200 mg/kg. Concentrations for both parameters are consistent with background concentrations.

6.1.4.2 GENERAL SITE BOREHOLE SOIL SAMPLES

During the Phase I investigation, a total of 19 investigative samples were collected from eight boreholes in 1991, four of which were completed as monitoring wells (one overburden well and three bedrock wells). The locations of these monitoring wells and boreholes are presented on Figure 2.1. No soil sample was collected from the borehole completed for the installation of well MW-3D-91 since groundwater was not encountered. All of the borehole soil samples were analyzed for TCL VOCs and SVOCs, pesticides/PCBs, TAL metals, cyanide, TPH, and Site-specific parameters. Analytical results are presented in Appendix I.1 and are discussed in the following paragraphs.

A summary of detected parameters is presented in Table 6.7 and VOC and SVOC analytical results are presented on Figure 6.5.

VOCs

Of the eight boreholes sampled, chlorobenzene, 1,2-dichloroethane, acetone, and toluene were detected at low concentrations in samples from four of the boreholes. Only one VOC was detected in the sample from each borehole: chlorobenzene at BH-1-91 (22 µg/kg at 8-10 ft bgs), 1,2-dichloroethane at BH-5-91 (7 µg/kg at 4-6 ft bgs), acetone at

MW-1U-91 (ND(11)[2]/19 μg/kg at 8-10 ft bgs), and toluene at MW-4D-91 (13 μg/kg at 0-2 ft bgs, 14 μg/kg at 4-6 ft bgs, and 7 μg/kg at 8-10 ft bgs). No parameter exceeded the NYSDEC Soil Cleanup Objectives.

SVOCs

SVOCs were detected in all eight boreholes, with the most compounds detected at borehole MW-4D-91. Eight SVOCs (fluoranthene (130J μg/kg), benzo(b)fluoranthene (120J μg/kg), pyrene (120J μg/kg), chrysene (110J μg/kg), bis(2-ethylhexyl)phthalate (86J μg/kg), benzo(a)anthracene (62J μg/kg), n-nitrosodiphenylamine (44J μg/kg), and phenanthrene (43J μg/kg) were detected at borehole MW-4D-91. Only bis(2-ethylhexyl)phthalate was detected at all three sampling intervals (68J μg/kg at 0-2 ft bgs, 61J μg/kg at 4-6 ft bgs, and 86J μg/kg at 8-10 ft bgs). Of these parameters, only benzo(b)fluoranthene exhibited a concentration that exceeded the NYSDEC Soil Cleanup Objective.

Bis(2-ethylhexyl)phthalate was detected in all boreholes, except in MW-2D-91. Concentrations did not exceed the Soil Cleanup Objective and ranged from 50J μg/kg at BH-5-91 (0-2 ft bgs) to 600J μg/kg at BH-3-91 (0-2 ft bgs).

2-Aminopyridine was detected at boreholes BH-1-91, BH-5-91, MW-1U-91, and MW-2D-91. Concentrations were found to increase at the deeper sampling intervals of each borehole. Concentrations ranged from 150J μg/kg at BH-5-91 (8-10 ft bgs) to 3,600 μg/kg at BH-1-91 (8-10 ft bgs). Exceedances occurred at the 8-10 ft bgs interval at BH-1-91 (3600 μg/kg), and the 8-10 ft bgs interval at MW-1U-91 (2,100 J/3,500 J μg/kg).

Di-n-octylphthalate was detected at boreholes BH-1-91 and BH-5-91 at concentrations below the NYSDEC Soil Cleanup Objective. This parameter was detected at 370J μg/kg at BH-1-91 (0-2 ft bgs), and at 44J μg/kg at BH-5-91 (4-6 ft bgs).

Phenol was detected at BH-1-91 at a concentration of 350 J mg/kg which exceeded the NYSDEC Soil Cleanup Objective.

Pesticides

No pesticides were detected in any of the borehole soil samples submitted for analysis.

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[2] ND - Compound was not detected at or below the associated value.

PCBs

No PCBs were detected in any of the borehole soil samples submitted for analysis.

Metals

All borehole samples had iron levels that exceeded the Region III RBC. Concentrations ranged from 23,500 to 42,400 mg/kg. The Region III RBC for arsenic was exceeded at all locations except at 4-6 ft bgs at BH-1-91 and 0-2 ft bgs at BH-4-91 where there were no detected concentrations. Detected concentrations for this parameter ranged from 6.1 to 15.2 S[3] mg/kg. The sample taken at BH-5-91 at 0-2 ft bgs exhibited a slight exceedance of the manganese Region III RBC, which was detected at a concentration of 1,670 mg/kg. Concentrations for these parameters are consistent with background concentrations.

6.1.4.3 CURTAIN DRAIN INVESTIGATION SOIL SAMPLING RESULTS

Subsurface soil samples were collected from three test pits TP-49, TP-51, and TP-52 excavated for the investigation of the "Curtain Drain" as described in Section 3.2.6. The samples were collected from immediately beneath the pipe bedding. The "Curtain Drain" investigation soil samples were analyzed for TCL VOCs, SVOCs, pesticides/PCBs, TAL metals, cyanide, TOC, and Site-specific parameters. Analytical results are presented in Appendix I.1 and are discussed in the following paragraphs. A summary of detected parameters is presented in Table 6.8.

VOCs

VOCs were not detected in the subsurface soil samples collected from the test pits with the exception of trichloroethene and tetrachloroethene at estimated concentrations of 1 J µg/kg and 2 J µg/kg, respectively, in the sample collected from test pit TP-52. Both of these parameters did not exceed the NYSDEC Soil Cleanup Objectives.

SVOCs

SVOCs were not detected in the subsurface soil samples collected from the test pits with the exception of fluoranthene and pyrene at estimated concentrations under the NYSDEC criteria of 75 J µg/kg and 55 J µg/kg, respectively in the duplicate sample

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[3] S - The reported value is less than the Contract Required Detection Limit, but greater than the Instrument Detetection Limit.

collected from test pit TP-51. Fluoranthene and pyrene were not detected in the original sample from test pit TP-51.

The compound 4-hydroxy-4-methyl-2-pentanone (diacetone alcohol) was tentatively identified at estimated concentrations ranging from 9,000 JNA[4] µg/kg to 20,000 JNA µg/kg in the soil samples collected from test pits TP-49, TP-51, and TP-52. This tentatively identified compound may be an aldol condensation product of acetone and water generated by the laboratory extraction procedure.

Several other tentatively identified SVOCs were reported for samples collected from the three test pits as presented in Appendix I.1.

Site-specific parameters, alpha-picoline, 2-amino-pyridine, and pyridine were not detected in the soil samples from the test pits.

Pesticides

No pesticides were detected in any of the test pit samples submitted for analysis.

PCBs

No PCBs were detected in any of the test pit samples submitted for analysis.

Metals

Inorganic results for the three test pit sample locations are evaluated relative to background concentrations. Inorganic concentrations for the soil samples collected from the "Curtain Drain" were similar to background concentrations. Arsenic and iron exceeded the Region III RBCs similar to the majority of all other soil samples collected including the background samples.

Total Organic Carbon

Total organic carbon concentrations ranged from 2,180 mg/kg to 5,190 mg/kg for the "Curtain Drain" investigation soil samples. The average TOC concentration for the three sample locations is 3,190 mg/kg (0.32 percent).

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[4] *N - indicates presumptive evidence of a tentatively identified compound.*
A - the associated tentatively identified compound is a suspected aldol-condensation product.

In summary, analytical results from the "curtain drain" investigation show little evidence of contamination. The inorganic exceedances are similar to background concentrations. Only low concentrations of VOCs and SVOCs were detected.

6.1.4.4 SUBSURFACE SOIL SUMMARY

The lagoon soil samples exhibited the following VOCs that exceeded the Recommended NYSDEC Soil Cleanup Objectives:

- acetone
- benzene
- chlorobenzene
- ethylbenzene
- toluene
- xylene

SVOCs that exceeded the Recommended NYSDEC Soil Cleanup Objectives are the following:

- 2-aminopyridine
- phenol
- bis(2-ethylhexyl)phthalate
- pyridine
- aniline
- benzoic acid

Pesticides were found only to exceed the Recommended NYSDEC Soil Cleanup Objectives in Lagoons 2 and 6. These parameters included aldrin, dieldrin, endrin, and heptachlor epoxide. Aldrin, heptachlor epoxide, dieldrin, and endrin were reported at concentrations above TAGM soil cleanup objectives in soil sample RM-25/26 collected from Lagoon 2 in 1996. Aldrin was also reported at a concentration higher then the TAGM value for the soil sample collected from TP-41 in Lagoon 6 in 1991. Pesticides for the remaining 63 investigative soil samples were either not detected or were reported at concentrations below TAGM values.

In accordance with the Additional Soil Sampling Work Plan, to further investigate the presence of the these four pesticides (i.e., aldrin, heptachlor epoxide, dieldrin, and endrin), two subsurface soil samples were collected from each of two borings (L2-5 and L2-10) installed in the area of sample RM-25/26 in Lagoon 2 and two borings (L6-1 and L6-2) installed in the area of TP-41 in Lagoon 6 in 2003. The results for these samples indicate that these pesticides were not detected in these investigative samples with the exception of dieldrin at a concentration of 2.1 J $\mu g/kg$ in the duplicate sample, below the Soil Cleanup Objective.

Several pesticides including dieldrin, 4,4'-DDD, 4,4'-DDE, 4,4'-DDT, and methoxychlor were reported at concentrations below TAGM soil cleanup objectives for the sample collected from drum D001 in Lagoon 1 near Test Pit 17. All detected pesticide concentrations were qualified as suspect in this drum sample, as the detection of these pesticides may be do to pattern interference from the high concentration of Aroclor-1254 which was present in the investigative and duplicate samples. To verify the presence and evaluate whether the pesticides dieldrin, 4,4'-DDD, 4,4'-DDE, 4,4'-DDT, and methoxychlor should be considered COCs, four soil samples from two borings near the former location of drum D001 were collected in 2003 and analyzed for TCL pesticides in accordance with the Additional Soil Sampling Work Plan. The results for these locations, designated L1-3 and L1-4, indicated that dieldrin, 4,4'-DDD, 4,4'-DDE, 4,4'-DDT, and methoxychlor were not detected in the investigative samples with the exception of 4,4'-DDE at a concentration of 4.0 μg/kg in the duplicate sample, below the Soil Cleanup Objective. 4,4'-DDE was also detected in background soil samples.

Pesticides are therefore not considered to be Site-related.

The only PCB detected above the criteria was Aroclor-1254, which was also only found in Lagoon 2.

Inorganics that were reported in soil samples at concentrations exceeding the Region III RBCs in one or more samples included arsenic, manganese, iron, mercury, aluminum, and antimony. However, with the exception of mercury, these parameters were found to be consistent with the observed range of background inorganic concentrations.

6.2 GROUNDWATER QUALITY

Groundwater samples were collected in August 1991 and June 1995 during the Phase I and Phase II RI, respectively. Additional groundwater samples were collected in July 2001 and June 2002 in dry and wet seasons for the Additional Investigation. As well, groundwater sampling of the Site perimeter wells[5] was conducted semi-annually from 1997. Groundwater samples were collected from monitoring wells presented on Figure 2.1, and also from Maybrook town wells 1 and 3 in 1991, and town wells 1, 2 and 3 in 1995. Samples collected in 1991 were analyzed for TCL VOCs, SVOCs, pesticides/PCBs, TAL metals, cyanide, TPH, and Site-specific parameters. Samples collected in 1995 were analyzed for the same parameters except pesticides/PCBs and

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[5] Site perimeter wells include MW-1, MW-5D-95, MW-5U-95, MW-8U-95, MW-9D-01, MW-9U-01, MW-10D-01, MW-10U-01, MW-11D-01, MW-11U-01, MW-12D-01, MW-13D-01, SW-9, and T-2.

TPH. Samples collected in 1995 were also analyzed for chloride. Samples collected in July 2001 were analyzed for TCL VOCs, SVOCs, TAL inorganics, Site-specific compounds (pyridine, 2-aminopyridine, and alpha-picoline), and natural attenuation parameters. The June 2002 groundwater samples were analyzed for TCL VOCs, Site-related pyridine-based compounds, and natural attenuation parameters. Groundwater samples collected for the Interim Semi-Annual Groundwater Sampling Program were analyzed for BTEX and pyridine compounds.

All groundwater analytical results including available historical data are presented in Appendix I.2. A summary of the detected parameters for 1985, 1991, and 1995 is presented in Table 6.9. The detected compounds for the Interim Groundwater Monitoring Program are presented in Table 6.10. The additional groundwater sampling (July 2001 and June 2002) summary of detected parameters is presented in Table 6.11.

The Phase I RI sampling (1991), Phase II RI sampling (1995), and Additional Round of sampling analytical results for organic compounds are presented on Figures 6.7 through 6.9 and Figures 6.14 through 6.16 for the overburden and bedrock wells, respectively.

The analytical results have been compared to Class GA groundwater standards and guidance values for the State of New York published in the Technical and Operational Guidance Series (TOGS 1.1.1), and the Federal Maximum Contaminant Levels (MCLs). Exceedances of the standards and guidance values are highlighted on Table 6.9 through Table 6.11, Figures 6.7 through 6.9 and Figures 6.14 though 6.16.

6.2.1 OVERBURDEN GROUNDWATER QUALITY

The following paragraphs present a description of the constituents present in the overburden and is primarily based on the July 2001 and June 2002 data. The 2001 and 2002 are considered to be most representative of current groundwater quality conditions. Analytical results are summarized in Table 6.11 and presented in Appendix I.2.

VOCs

The distribution of VOCs for 2001 in the overburden groundwater is presented on Figure 6.9. The concentrations of detected VOCs were generally highest at well MW-7 in both the dry and wet seasons in 2001 and 2002 respectively. VOCs detected in this well in 2001 which exceeded the New York State Groundwater Criteria include benzene (1,100/340 μg/L), xylene (520/450 μg/L), ethylbenzene (150/130 μg/L), toluene

(23/23 μg/L), and chlorobenzene (23/23 μg/L). During the 2002 sampling in the wet season, the concentrations of VOCs in MW-7 were also generally the highest of the sampled overburden wells. VOCs detected in 2002 which exceeded the New York State Groundwater Criteria in MW-7 were benzene (790 μg/L), xylenes (210 μg/L), ethylbenzene (70 μg/L), toluene (19 μg/L), chlorobenzene (14 μg/L), and 1,2'-dichloroethane (4.3 μg/L). During both sampling events, the concentrations of benzene in MW-7 also exceeded the Federal MCLs.

Benzene has the highest detection rate of the VOCs. In 1991 and 1995 VOCs were not detected in wells downgradient (SW-9) or cross-gradient (SW-7 and SW-8) from MW-7. In the 2001 dry season, the benzene overburden plume extended slightly to include SW-9 (1.5 μg/L) and SW-8 (63 μg/L). Figure 6.10 presents the benzene distribution in overburden groundwater in July 2001. During the wet season in June 2002, the benzene plume differed slightly since benzene was not detected in the downgradient well SW-9. The difference between the benzene distributions in 2001 and 2002 may be due to the higher groundwater levels (approximately 4 feet higher) in 2002. The benzene distribution in 2002 is shown on Figure 6.11.

VOCs were detected at relatively high concentrations at well MW-3 in Lagoon 5, located just north of the groundwater divide. VOCs at similar concentrations were detected at well SW-4 between Lagoons 4 and 5, between the groundwater divide. In both 2001 and 2002, VOCs which exceeded the New York State Groundwater Criteria at these sampling locations included benzene, chlorobenzene, xylene, and ethylbenzene. VOCs which exceeded the Federal MCLs at these two sampling locations were benzene and chlorobenzene.

VOCs were also detected at well MW-1U-91, located south of Lagoon 3. 1,2-Dichloroethane, benzene, and chlorobenzene were detected at concentrations exceeding the New York State Groundwater Criteria. Dichloroethane and benzene were also detected at well MW-4, located north of Lagoon 3, at concentrations exceeding the New York State Groundwater Criteria. Benzene also exceeded the Federal MCL in the two locations.

Wells MW-2 and SW-2, located downgradient of the lagoons and north of the groundwater divide, had detected concentrations of benzene (96 ug/L) and chlorobenzene (22 ug/L) exceeding the New York State Groundwater Criteria for the 2001 sampling round. During the 2002 sampling event, MW-2 was not sampled. The concentrations which exceeded the New York State Groundwater Criteria in SW-2 were benzene (55 μg/L), chlorobenzene (14 ug/L and 8.6 μg/L), and 1,2-dichloroethane

(1.6 μg/L). Concentrations of benzene exceeded the Federal MCLs in all samples collected from these two locations.

During the 2001 sampling round no exceedances were reported at wells MW-1, MW-5U-01, MW-8U-01, MW-9U-01, MW-10U-01, MW-11U-01, and SW-7. At a few of these wells, the 2002 sampling round yielded some VOCs which exceeded the New York State Groundwater Criteria. In MW-5U-95, methyl tert butyl ether (24 μg/L), in MW-8U-95, benzene (40 μg/L) and in MW-9U-01 methyl tert butyl ether (250 μg/L) and 1,2-dibromo-3-chloropropane (1 μg/L). Methyl tert butyl ether is not considered to be Site-related. The concentrations of benzene and 1,2-dibromo-3-chloropropane also exceeded the Federal MCLs. During the 2002 sampling round, wells MW-10U-01 and MW-11U-01 were not sampled, and no exceedances were reported at wells MW-1, SW-7, SW-9, and SW-10.

SVOCs

The distribution of SVOCs for 2001 in the overburden groundwater is presented on Figure 6.9. The concentrations of detected SVOCs were generally highest at wells MW-7 and MW-2.

Three SVOCs (2-aminopyridine, 4-chloroanaline, and 4-methylphenol) were detected at concentrations above the New York State Groundwater Criteria in July 2001. The exceedances for 2-aminopyridine occurred at monitoring wells MW-2, MW-3, MW-4, MW-7, MW-1U-91, MW-8U-95, SW-2, SW-4, SW-6, and SW-8 at concentrations ranging from 2 J μg/L to 520 μg/L. The distribution of 2-aminopyridine in overburden groundwater is presented on Figure 6.12 and Figure 6.13 for the July 2001 and June 2002 data, respectively. 4-Chloroanaline exceeded criteria only at MW-7 (25/25 μg/L) while 4-methylphenol exceeded criteria only at SW-2 (3 J μg/L).

Metals

Five metals (arsenic, iron, lead, manganese, and sodium) were encountered at levels exceeding the New York State Groundwater Criteria in samples collected from one or more overburden monitoring wells. Arsenic and lead were reported at a concentration slightly exceeding the New York State Groundwater Criteria at one overburden well location each, MW-7 and MW-4, respectively. However, arsenic and lead were not detected at any other location for either sampling event or in samples collected from the bedrock groundwater at concentrations exceeding the applicable criteria. The concentrations of arsenic, iron, lead, manganese, and sodium are considered to be representative of these naturally occurring metals in the area of the Site. Metals that were reported at concentrations exceeding the Federal MCLs during the 2001 sampling round were arsenic at wells MW-1U-91, MW-7, and SW-4, and lead at well MW-4.

6.2.2 BEDROCK GROUNDWATER QUALITY

The following paragraphs present a description of the constituents present in the bedrock groundwater and is primarily based on the July 2001 and June 2002 data. Detected analytical results are presented in Table 6.11 and presented in Appendix 1.2.

VOCs

The distribution of VOCs for 2001 in the bedrock groundwater is presented on Figure 6.16. During the 2001 and 2002 sampling rounds, eight VOCs (1,2-dichloroethane, 2-butanone, acetone, benzene, chlorobenzene, ethylbenzene, toluene, and xylene) were detected at levels exceeding the New York State Groundwater Criteria in samples collected from one or more monitoring wells. In July 2001, the six VOC exceedances were encountered at the following monitoring well locations: acetone (DW-1-95 and DW-2-95); benzene (MW-1D-91, MW-2D-91, DW-1-95, and DW-2-95); chlorobenzene (MW-1D-91 and MW-2D-91); ethylbenzene (MW-2D-91); toluene (MW-5D-95); and xylene (MW-2D-91). In June 2002, seven VOC exceedances were encountered at the following monitoring well locations: 1,2-dichloroethane (DW-1-95); 2-butanone (DW-2-95); acetone (DW-1-95 and DW-2-95); benzene (MW-1D-91, MW-2D-91, DW-1-95, and DW-2-95); chlorobenzene (MW-1D-91 and MW-2D-91); ethylbenzene (MW-2D-91); and xylene (MW-2D-91). Benzene also exceeded the Federal MCL at MW-1D-91, MW-2D-91, DW-1-95, and DW-2-95.

Similar to the overburden, benzene had the most detections of the VOCs detected in the bedrock groundwater. The chemical isoconcentration contours for benzene in the bedrock groundwater at the Site for July 2001 and June 2002 are presented on Figures 6.17 and 6.18, respectively. Similar to the overburden results, there is some variability of the benzene contours from 2001 to 2002. However, within the bedrock, the variations are minimal.

SVOCs

Two SVOCs (2-aminopyridine and bis(2-ethylhexyl)phthalate) were encountered at levels exceeding the New York State Groundwater Criteria in samples collected from one or more monitoring wells. In July 2001, the exceedances for 2-aminopyridine were encountered at monitoring well locations MW-1D-91 and MW-2D-91. In June 2002, the exceedances for 2-aminopyridine were encountered at monitoring well locations DW-1-95 and MW-1D-91. Bis(2-ethylhexyl)phthalate was reported at a concentration slightly exceeding the New York State Groundwater Criteria and Federal MCL at well DW-2-95 for the sample collected in July 2001. Bis(2-ethylhexyl)phthalate was not detected at any other location for either sampling event in samples collected from the bedrock groundwater. The reported concentration of bis(2-ethylhexyl)phthalate at this location may be attributable to lab contamination.

The distribution for 2-aminopyridine in the bedrock groundwater at the Site for July 2001 and June 2002 are presented on Figures 6.19 and 6.20, respectively. There is some variability of the contours from 2001 to 2002. However, within the bedrock, the variations are minimal.

Metals

Four metals (antimony, iron, manganese, and sodium) were encountered at levels exceeding the New York State Groundwater Criteria in samples collected from one or more bedrock monitoring wells. Metals did not exceed the Federal MCLs at any sampling locations. Antimony was reported at a concentration exceeding the New York State Groundwater Criteria at bedrock well MW-9D-01 for the July 2001 sample. However, antimony was not detected in the confirmatory sample collected from well MW-9D-01 in November 2001. In addition, antimony was not detected at any other location for either sampling event for samples collected from the overburden and bedrock groundwater. Based on the absence of antimony in wells throughout the Site, the result can be considered anomalous.

The concentrations of iron, manganese, and sodium are considered to be representative of these naturally occurring metals in the area of the Site.

6.2.3 SUMMARY

The distribution of VOCs and SVOCs in the overburden and bedrock groundwater at the Site is relatively well defined.

The 2001 groundwater samples exhibited the following VOCs that exceeded the New York State TOGS Groundwater Criteria:

- 1,2-dichloroethane;
- 1,2-dibromo-3-chloropropane;
- 2-butanone;
- acetone;
- benzene;
- chlorobenzene;
- ethyl benzene;
- benzoic acid;
- methyl tert butyl ether (MTBE);
- toluene; and
- xylene.

MTBE is not related to the Site as it was detected at off-Site monitoring well locations and the cessation of disposal activities at the Site pre-dates the use of MTBE.

SVOCs that exceeded the New York State Groundwater Criteria are the following:

- 2-aminopyridine;
- 2-methylphenol;
- 4-chloroaniline;
- 4-methylphenol; and
- bis(2-ethylhexyl)phthalate).

The reported exceedance of bis(2-ethylhexyl)phthalate in bedrock groundwater may be attributable to lab contamination.

6.3 TOWN OF MAYBROOK SUPPLY WELLS

In 1995, the Town of Maybrook used wells 1, 2, and 3 as their municipal water supply. Supply wells 1 and 3 were individually sampled in October 1991. The combined flow from supply wells 1, 2, and 3 was sampled in June 1995. Samples collected in 1991 and 1995 were analyzed for TCL VOCs, SVOCs, TAL metals, cyanide, and Site-specific parameters. The investigative and duplicate samples collected in 1995 were also analyzed for chloride. Analytical results are presented in Appendix I.2.

VOCs

Methylene chloride was the only VOC detected at a concentration of 2 J μg/L in the samples collected from supply wells 1 and 2 during the 1991 sampling program.

VOCs were not detected in the samples collected from the combined flow of supply wells 1, 2, and 3 during the 1995 sampling program.

SVOCs

Bis(2-ethylhexyl)phthalate was the only SVOC detected at concentrations of 13 µg/L and 21 µg/L for the samples collected in 1991 from supply wells 1 and 2, respectively.

SVOCs were not detected in the samples collected form the combined flow of supply wells 1, 2, and 3 during the 1995 sampling program.

Inorganics

The inorganics that were detected in the Maybrook supply wells during the 1991 and 1995 sampling rounds were reported at generally low concentrations below applicable Federal and New York State drinking water standards with the exception of naturally occurring high levels of manganese and sodium.

In summary, there is no indication of a Site-related impact at the Town of Maybrook supply wells based on the results of the 1991 and 1995 sampling program.

In addition to this sampling, as discussed in Section 3.11, no detections of Site-related parameters have been found for the samples collected quarterly by the Village of Maybrook from wells 1, 2, and 3 up to the end of 2005.

6.4 SURFACE WATER QUALITY

Surface water sampling locations are presented on Figure 3.3 and the analytical results are summarized in Table 6.12 and on Figure 6.21 and Figure 6.22. Complete analytical tables are presented in Appendix I.3.

Exceedances of applicable New York State ambient surface water criteria are indicated in Table 6.12. Applicable criteria for a Class D surface water were used for Beaverdam Brook and the Northeast Marsh. Class C criteria were used for Otter Kill.

6.4.1 1985 SURFACE WATER SAMPLING RESULTS

Four surface water samples were collected during the 1985 C.A. Rich investigation. Two of these locations, W-1 and W-3, closely correspond to the locations 3 and 4, respectively, of the RI investigation. Of the limited compounds analyzed during the 1985 investigation (see Table 6.12), the only SVOC detected was total pyridines at W-1

(1,400 μg/L), W-2 (2,000 μg/L), W-3 (1,800 μg/L), and W-4 (1,700 μg/L). These did not exceed the surface water criteria. The results for total pyridine for these samples is suspect as the analysis was a colorimetric determination subject to interference and inaccuracy. This uncertainty is supported by the absence of the primary pyridines 2,6-diaminopyridine, 2-aminopyridine, and 2-chloropyridine detected in the surface water samples.

VOCs were not detected in samples collected from any of the locations.

The only metals detected at W-1 during the 1985 investigation were selenium (13 μg/L), sodium (7,800 μg/L), and zinc (74 μg/L). The only metals detected at W-3 during the 1985 investigation were calcium (5 μg/L), selenium (12 μg/L), sodium (5,730 μg/L), and zinc (74 μg/L). All inorganic parameters fell below New York State ambient water quality standards and guidance values.

6.4.2 1991 SURFACE WATER SAMPLING RESULTS

Two rounds of surface water samples were collected during the 1991 investigation. All samples were analyzed for TCL VOCs, SVOCs, pesticides/PCBs, TAL metals, cyanide, TPH, and Site-specific parameters.

The first sampling round was collected during low flow conditions (period of no rainfall) and the second sampling round was collected during high flow conditions (within 48 hours of a rainfall event that produced runoff from the Site). Surface water samples were collected at locations 3, 4, 6, and 7, as presented on Figure 3.3. No samples were collected at locations 1, 2, and 5 due to lack of surface water at these locations at the time of sampling.

Location 3 represents upstream surface water quality in Beaverdam Brook, approximately 20 feet upstream of Orange County Highway 4 which is located on the other side of the highway. Location 4 represents surface water quality in Beaverdam Brook adjacent to the southern end of the Site. Location 6 represents surface water quality in the Otter Kill, approximately 20 feet upstream of the confluence of Beaverdam Brook and the Otter Kill. Location 7 represents surface water quality furthest downstream of the Site, approximately 10 feet downstream of the confluence of Beaverdam Brook and the Otter Kill. Analytical results for low flow and high flow conditions are presented in Appendix I.3. A summary of the analytical results is presented in Table 6.12 and the VOC and SVOC parameter results are summarized on Figure 6.21.

Low Flow Condition

Methylene chloride was reported present at very low concentrations, below criteria in samples from Location 4 (4 J μg/L), Location 6 (4 J μg/L), and Location 7 (3 J μg/L). The presence of methylene chloride at Location 6, which monitors the upstream surface water quality, at concentrations comparable to those detected at locations adjacent to and downgradient of the Site (Locations 4 and 7), indicated that the reported levels of methylene chloride at Locations 4 and 7 do not demonstrate a Site-related impact. The only other VOC compound detected in the surface water samples is toluene which was detected below criteria at a very low concentration of 1J μg/L in the sample from Location 4.

During low flow conditions, no SVOCs, pesticides or PCBs were detected in the samples taken.

The inorganic parameter concentrations for the low flow surface water sampling indicate that the downstream sampling locations (Locations 4 and 7) have similar concentrations to the upstream sampling locations (Locations 3 and 6), indicating that the Site is not impacting the surface water quality with respect to inorganic constituents. However, samples including upstream locations did show exceedances of the New York State Ambient Water Quality Standards and Guidance Values for iron and aluminum, and in one case, copper (23.2 μg/L), which was detected in the upstream sample Location 3.

High Flow Condition

Acetone and toluene were reported present in the duplicate sample from upstream sampling Location 3 at concentrations of 16 μg/L and 12 μg/L during the high flow sampling event. These compounds were not detected in the other sample collected at the same time from this location. Both of these parameters were also detected, at lower concentrations in the samples from downstream Location 4 (acetone at 3 J/<10 μg/L and toluene at 1 J/1 J μg/L) and Location 7 (acetone at 3 J μg/L and toluene at 1 J μg/L). Since these parameters were reported present at higher concentrations in the upstream sample, their presence at Locations 4 and 7 is not considered to be Site-related. These parameters did not exceed the New York State criteria.

No pesticides or PCBs were detected in the high flow samples.

Bis(2-ethylhexyl) phthalate and di-n-octylphthalate were reported present at comparable concentrations in samples from Locations 3, 6, and 7. The maximum of 320 µg/L and 110 µg/L occurred for bis(2-ethylhexyl)phthalate and di-n-octylphthalate respectively, at Location 7, which was comparable to the concentrations at the upstream Location 7, indicating their presence is not considered to be Site-related. All the bis(2-ethylhexyl)phthalate detected concentrations exceeded criteria.

During the high flow event, iron and aluminum were found to exceed the New York State criteria in most cases, including the background samples. Similarly to the low flow surface water sampling, the detected inorganic parameter concentrations for the downstream sampling locations are generally consistent with the upstream concentrations, and hence do not indicate a Site-related impact.

6.4.3 1995 SURFACE WATER SAMPLING RESULTS

Three rounds of surface water samples were collected during the 1995 investigation. All samples were analyzed for TCL VOCs, SVOCs, pesticides/PCBs, TAL metals, cyanide, Hardness, Chloride, and Site-specific parameters.

The first sampling round was collected during a period of low flow on June 5, 1995 (period of no rainfall). The second sampling round, indicating a "medium" flow event, was collected during a period of relatively high water table elevations on June 14, 1995. The third round was collected during high flow conditions (within 48 hours of a rainfall event that produced runoff from the Site) on June 19, 1995. Surface water samples were collected at locations SWII-2, SWII-4, SWII-7, and SWII-9 in accordance with the Phase II RI WP. Surface water sample locations are presented on Figure 3.3.

Location SWII-2 is located in the northeast swamp north of 1991 sample location 2.

Location SWII-4 is situated immediately north of the former railroad line in Beaverdam Brook.

Location SWII-7 is adjacent to 1991 sample location 7 and represents surface water quality downstream of the Site immediately east of the confluence of Beaverdam Brook and Otter Kill.

Location SWII-9 represents upstream surface water quality in Beaverdam Brook approximately 600 feet north of County Highway 4 and upstream of the stream joining Beaverdam Brook from the east. Analytical results are presented in Appendix I.3. A summary of analytical results is presented in Table 6.12. VOC and SVOC parameter results are summarized on Figure 6.22.

Low Flow, Medium Flow, and High Flow Conditions

TCL VOCs, SVOCs, pesticides and PCBs were not detected in any of the surface water samples collected during the three flow conditions with the exception of acetone detected during high flow conditions at location SWII-2. Acetone was reported at a concentration below the New York State criteria of 5 J μg/L for this sampling event.

The inorganic results for locations SWII-2, SWII-4 and SWII-7 were generally consistent with upstream background location SWII-9. Aluminum was detected at SWII-2 during high flow (566 μg/L) and SWII-7 during low flow (65.4 μg/L) compared to background location SWII-9 during high flow (349 J μg/L). Arsenic was detected at concentrations ranging from 1.5 to 3.2 μg/L at all sample locations including SWII-9.

Vanadium, thallium, chromium, and cyanide were each detected at one location but not at background location SWII-9. During high flow conditions, vanadium was detected below the New York State criteria, at location SWII-2 (2.2 μg/L). At location SWII-7, thallium was detected (2.8 μg/L) below New York State criteria. However, chromium (0.70 ug/L) did exceed the New York State Ambient Water Quality Standards and Guidance Values. Cyanide was detected above criteria at SWII-4 during medium flow conditions (54.4 J μg/L) but was not detected in the duplicate for this sample.

6.4.4 SUMMARY

In general, the surface water quality data indicates that the Site has no measurable impact on parameter concentrations in Otter Kill, Beaverdam Brook or the northeast swamp. Comparable concentrations of organics and inorganics were reported at both upstream and downstream sampling locations.

Relatively high naturally occurring iron concentrations exceeded applicable surface water criteria at all 1991 and 1995 sampling locations including background. Aluminum exceeded the surface water standard at all 1991 locations, but was only exceeded in 1995 at the SWII-2 and SWII-9 locations during the high flow event. Cyanide was reported at a concentration of 54.4 J μg/L at 1995 location SWII-4 located in Beaverdam Brook. Cyanide was not detected in the duplicate sample collected at this location. Chromium was detected at the 1995 location SWII-7 at a concentration of 0.70 μg/L exceeding the surface water criteria.

Bis(2-ethylhexyl)phthalate exceeded the surface water standard at 1991 sample location 3 located within Beaverdam Brook upgradient of the Site, and upgradient and downgradient locations 6 and 7 located within Otter Kill. Bis(2-ethylhexyl)phthalate was not detected in samples collected in 1995 from Beaverdam Brook or Otter Kill. The EPA has agreed that bis(2-ethylhexyl)phthalate in surface water is likely a non-site-related parameter.

6.5 SEDIMENT SAMPLING RESULTS

Sediment sample results are summarized in Tables 6.13 and 6.14. Analytical results for VOCs and SVOCs are presented on Figure 6.23 and Figure 6.24. Complete analytical tables are presented in Appendix I.4.

The sediment sampling results have been compared to sediment screening benchmark criteria as described in Section 8.0. The selected benchmark value is the lowest criterion for the available sources. Tables 6.13 and 6.14 list the sources of each criterion in the footnotes.

6.5.1 1985 SEDIMENT SAMPLING RESULTS

Sediment samples collected during the C.A. Rich investigation (Rich, 1986) were analyzed for parameters listed in Appendix 1.4. Locations for which sediment samples were collected during the previous investigation are presented on Figure 3.3.

Sampling locations W-1 and W-3 of the 1985 investigation closely correspond to sampling locations 3 and 4, respectively, of the 1991 investigation. Sample location W-4 is located upstream of the Site at Beaverdam Brook approximately 400 feet north of the Site. Location W-2 is located approximately 400 feet southeast of the Site at Otter Kill. Detected parameters for W-1 to W-4 are presented in Table 6.13.

2-Butanone (7 μg/kg) was detected in the 1985 sediment sample collected from location W-1.

Total pyridines were detected in the 1985 samples collected from locations W-1 (82,500 μg/kg), W-2 (5,720 μg/kg), W-3 (39,940 μg/kg), and W-4 (38,100 μg/kg) (see Table 6.8). These did not exceed the sediment criteria. Similar to the 1985 surface water samples, the total pyridine results are believed to be suspect as the analysis was a colorimetric determination subject to interference and inaccuracy.

Arsenic was detected at a concentration that exceeded the sediment benchmark values at W-3 (14.7 mg/kg).

6.5.2 1991 SEDIMENT SAMPLING RESULTS

One round of sediment sampling was conducted at the Site and samples were analyzed for TCL VOCs, SVOCs, pesticides/PCBs, TAL metals, cyanide, TPH, and Site-specific parameters. The analytical results are presented in Appendix I.4. A summary of the detected compounds is presented in Table 6.13. This data is presented on Figure 6.23.

Location 1

Two samples were collected in close proximity at Location 1, and are designated 1a and 1b. Locations 1a and 1b are approximately 1,880 feet from the northern Site boundary, along Orange County Highway 4, on the south side of the highway. Samples were taken from a stream ditch next to a culvert north of the marshy area of Location 2. The soil was moist, but there was no standing water present. The stream flows to the west where it joins Beaverdam Brook approximately 800 feet to the west of Location 1.

Location 2

Location 2 is approximately 800 feet east of the northern Site boundary. The sample was taken from a relatively dry swamp/marsh area.

Location 3

Location 3 is approximately 20 feet north of Orange County Highway 4, on the west bank of Beaverdam Brook.

Location 4

Location 4 is approximately 25 feet south of the old railway grade, on the west bank of Beaverdam Brook.

Location 5

Location 5 is a drainage swale approximately 50 feet east of where the Site access road meets the old railway grade. It is located in a drainage ditch immediately south of the former lagoons in a wooded area.

Location 6

Location 6 is on the north bank of the Otter Kill, approximately 75 feet upstream of the confluence of Beaverdam Brook and the Otter Kill.

Location 7

Location 7 is on the west bank of Beaverdam Brook at the confluence of the Otter Kill and Beaverdam Brook.

Location 8

Location 8 is a drainage swale at the eastern Site boundary, approximately 100 feet south of the old railway grade, in a clearing within a wooded area.

VOCs

Methylene chloride (56 µg/kg), acetone (130 µg/kg), 2-butanone (38 µg/kg), and total petroleum hydrocarbons (13 µg/kg) were detected in the upstream sediment sample collected from Otter Kill (Location 6). These same parameters, with the exception of 2-butanone which was not detected, were found at similar concentrations at Location 7 indicating the sediment locations along Beaverdam Brook had no impact on Otter Kill. Toluene, which was not detected at Location 6, was detected at Locations 5, 7 and 8 at concentrations of 470 J, 22 and 100 µg/kg, respectively.

Location 3, the upstream sediment location on Beaverdam Brook, also included concentrations of methylene chloride (28 µg/kg) and acetone (13 J µg/kg).

Methylene chloride was detected at all sample locations, at concentrations less than the upstream Otter Kill location, except for Location 2 which had a concentration of 86 µg/kg. However, no samples exceeded criteria. Acetone was detected at Locations 3, 4 and 7 at lower concentrations than the upstream sample taken at Otter Kill.

All acetone and toluene concentrations, with the exception of toluene at Location 7, exceeded the sediment benchmark values.

Total petroleum hydrocarbons (maximum value of 69 mg/kg) were detected in sediment samples from other sediment sampling locations (Locations 1a, 1b, 2, 5, 6, and 8) on or adjacent to the Site.

SVOCs

Several SVOCs were detected in the sediment samples collected from the area of the Site with the majority of the compounds and the highest concentrations reported for samples from Locations 1a, 1b, 5, and 2.

The major compounds detected in these samples are PAHs (maximum value of 27,000 μg/kg fluoranthene) and phthalate esters (maximum value of 9,600 μg/kg bis(2-ethylhexyl)phthalate) which were found at the highest concentrations at Location 1 approximately 1,880 feet from the northern Site boundary. These compounds were also reported present on-Site at Location 5, which is located in a drainage ditch immediately south of the former lagoons, at an average of about 80 percent less than the highest concentrations found at samples taken at Location 1. Furthermore, lower concentrations were found in the sediment samples collected from Locations 2 and 4. Most concentrations detected exceeded the sediment screening benchmark values as highlighted in Table 6.13.

Pesticides

Concentration of pesticides (maximum value of 180 μg/kg 4,4'-DDT) were detected in the sediment sample collected from Location 5, which is located in a drainage ditch immediately south of the former lagoons. The Sediment screening benchmark values were exceeded for 4,4'-DDD, 4,4'-DDE, 4,4'-DDT, aldrin, and methoxychlor.

Pesticides were not detected at the other sample locations.

PCBs

PCBs were not detected at the sample locations.

Inorganics

Sediment screening benchmark values were exceeded for a large portion of the inorganics. Parameters with concentrations detected over the criteria include antimony, arsenic, cadmium, chromium, copper, iron, lead, manganese, nickel, silver and zinc. Exceedances can be found highlighted In Table 6.13.

The inorganic sediment data was evaluated using the data from sampling Locations 3 and 6 as being generally representative of background concentrations. Based upon this evaluation, it was determined that elevated inorganic parameter concentrations are present in the samples from Location 2 (antimony at 256 mg/kg, cadmium at 22.7 mg/kg, copper at 91.6 mg/kg, silver at 66.5 mg/kg); and Location 8 (lead at 134 mg/kg). Manganese was also reported present at elevated concentrations (maximum value of 3,070 mg/kg) at several of the sediment sampling locations.

6.5.3 1995 SEDIMENT SAMPLING RESULTS

One round of sediment samples was conducted at sample locations SDII-2 and SDII-12 to investigate the sediment quality in the northeast swamp and a southwest marshy area within Beaverdam Brook. Sample locations are presented on Figure 3.3. Samples were analyzed for TCL VOCs, SVOCs, pesticides/PCBs, TAL metals, TOC, and Site-specific parameters. The analytical results are presented in Appendix I.4 and are summarized in Table 6.13. VOC and SVOC detects are presented on Figure 6.23.

VOCs

Acetone (ND(29) J/71 J µg/kg), toluene (20 J/140 J µg/kg), and ethylbenzene (6 J/ND(31) J µg/kg) were detected exceeding criteria in the sample from location SDII-2 located in the northeast swamp. Acetone (140 J µg/kg) and 2-butanone (51 J µg/kg) were detected exceeding criteria in the sample collected from location SDII-12 located in the southwest marsh area. These VOCs were previously detected at background locations 3 and 6 during the 1991 investigation.

SVOCs

Several SVOCs, primarily PAHs, were detected at similar concentrations in the two sediment samples SDII-2 and SDII-12 collected in 1995. These SVOCs were reported at concentrations ranging from 110 J µg/kg to 990 J µg/kg. The PAH concentrations were generally one order of magnitude lower at location SDII-2 compared to 1991 sampling locations located approximately 900 feet to the northwest. Most concentrations detected exceeded criteria as presented on Table 6.13.

Pesticides/PCBs

Pesticides and PCBs were not detected in samples collected from the 1995 sample locations SDII-2 and SDII-12 with the exception of 4,4'-DDE reported at a concentration of 11 J at location SDII-2. This parameter exceeded the sediment screening benchmark value.

Inorganics

Inorganic parameter concentrations for samples collected from SDII-2 and SDII-12 were generally consistent with the results for the 1991 background locations 3 and 6. Beryllium and cadmium detected in the 1991 samples, were not detected in the 1995 sediment samples. Arsenic and barium were reported at lower concentrations in the sediment samples collected in 1995 compared to the 1991 samples.

Mercury was reported at concentrations ranging from 0.14 J μg/kg to 0.33 J μg/kg in sediment samples collected from SDII-2 and SDII-12.

Exceedances of the sediment criteria occurred for copper, iron, manganese, mercury, nickel, and zinc.

6.5.4 2003 EPA SEDIMENT SAMPLING RESULTS

The U.S. EPA conducted sediment sampling in Beaverdam Brook in 2003. Twenty-seven samples were collected and analyzed for TCL VOCs, SVOCs, pesticides/PCBs, TAL metals, cyanide, and Site-specific parameters. Sample locations are presented on Figure 3.11. The analytical results are presented in Appendix I.4. A summary of the detected compounds is presented in Table 6.14 and on Figure 6.24.

Sample SE-01 was taken downstream of the pond south of the abandoned railway, while samples SE-02 through SE-11 were taken within the pond. These sample locations are similar to samples 4 and W-3 collected in 1985 and 1991 respectively. Samples SE-12 through SE-23, and SE-26 were taken at various points along Beaverdam Brook. Background samples (SE-24, SE-25, and SE-27) were collected north of the County Highway 4. These samples were considered by the U.S. EPA to be representative of upgradient background conditions. The parameters which were determined to be Site-related are presented below.

VOCs

VOCs which were detected above both the equipment and background samples were acetone, 2-butanone and toluene. Toluene was detected only at SE-18 (3J μg/kg). SE-13 along Beaverdam Brook detected the highest concentrations of both acetone (370J μg/kg) and 2-butanone (150J μg/kg). Acetone and 2-butanone have been detected during the 1995 sampling event in Beaverdam Brook at SDII-12. They were also detected at the background locations 3 and 6 during the 1991 investigation. All acetone concentrations exceeded the sediment screening benchmark values.

SVOCs

Several SVOCs, primarily PAHs, were detected in the 2003 Beaverdam Brook sediment samples. These SVOCs were reported at concentrations ranging from 120 J μg/kg to 3400 μg/kg. Generally the samples with higher SVOC concentrations were located at the south end of Beaverdam Brook in the pond. Exceedances of criteria are highlighted on Table 6.14.

PCBs

PCBs were not detected in the samples collected from the 2003 sampling event.

Pesticides

Two pesticides were detected (4,4'-DDD and 4,4'-DDE) although 4,4'-DDD was detected three times below the background concentrations. Therefore, only 4,4'-DDE is potentially attributable to the Site. The highest detection of 4,4'-DDE was found in sample SE-02 (40 μg/kg) located in the pond just north of the foot bridge. All detected pesticides exceeded the sediment screening benchmark values.

Inorganics

Inorganic parameters which were determined to be above background were arsenic, copper, lead, manganese, selenium, silver, and sodium. The detected results were generally consistent with the samples collected in 1991 and 1995 although copper, sodium, and selenium were slightly elevated compared to the historic results. The maximum copper concentration was 685 μg/kg at SE-11 while sodium detects ranged from 71.7J μg/kg at SE-04 to 442 μg/kg at SE-13. Selenium was detected once at 2.3J μg/kg at SE-04. These maximum detected results are located in the pond. Antimony, arsenic, cadmium, copper, iron, lead, manganese, nickel, zinc, and cyanide all had concentrations exceeding the sediment screening benchmark values.

6.5.5 SUMMARY

In general, the sediment data indicates that there is no concrete evidence that the Site has measurable impact on parameter concentrations in Otter Kill, Beaverdam Brook, or the Northeast Swamp.

VOCs exceeding criteria were acetone and toluene. Acetone was highest at SDII-12 (140 J μg/kg) but comparable to the sample taken upstream of Otter Kill at Location 6 (130 μg/kg). Toluene exceeded criteria in 1991 at Location 5 and 8, and in 1995 at SDII-2.

Numerous SVOCs, primarily PAHs, exceeded criteria at locations 1a, 1b, 2, 4 and 5 in 1991; SDII-2 and SDII-12 in 1995; and SW-16, SE-21 and SE-23 in 2003. The probability that these parameters are Site-related is slim, as the highest concentrations were found north of the Site and in the northeast marsh area. The elevated concentrations at Location 1 suggest contamination from a separate source.

Several pesticides exceeded criteria at Location 5 in 1991. In 1995, 4,4'-DDE exceeded the criteria at SDII-2. In 2003, several samples exceeded the criteria for 4,4'-DDE and 4,4'-DDE.

The inorganics that exceeded the sediment screening benchmark values were antimony, arsenic, cadmium, chromium, copper, iron, lead, manganese, mercury, nickel, silver, zinc, and cyanide. Of these, copper and mercury may be considered Site-related. The remaining inorganics primarily have only one sample location where concentrations were elevated.

7.0 CHEMICAL FATE AND TRANSPORT

Site-related chemicals have been detected in the shallow and deep groundwater, surface water, sediment, surface soil, and subsurface soil at the Site.

The potential chemical migration pathways which exist at the Site include:

- overburden and bedrock groundwater flow;
- surface water runoff;
- atmospheric dispersion from surface water and surface soils; and
- tracking of surface soils.

A general description of the physical and chemical properties of the Site-related chemicals and the influence of these properties on their fate and transport is presented in Section 7.1. The chemical migration potential for each of the identified pathways are evaluated in Sections 7.2 through 7.5. A discussion of the probable behavior and environmental fate of the primary chemical constituents at the Site is presented in Section 7.6.

7.1 PHYSICAL AND CHEMICAL PROPERTIES OF SITE-RELATED COMPOUNDS

Contaminant mobility, a factor in contaminant migration, depends upon the physical and chemical properties of both the contaminants and the media in which they are identified. Properties which affect contaminant mobility include, but are not limited to, solubility, liquid density, vapor pressure and chemical affinity. The partitioning of chemicals between media is controlled by a variety of factors such as adsorption, absorption, volatilization, solubility and chemical affinity.

Chemicals released to a soil medium may be adsorbed by the soil until the adsorptive capacity of the soil is reached. Under continued release(s) of the chemicals, the chemicals may migrate both horizontally and vertically, expanding the area of contaminated soils as the adsorptive capacity of the soil in the vicinity of the release is attained. Similarly, infiltration of precipitation or release(s) of other chemicals may cause the initial chemical to migrate at a rate primarily controlled by the adsorptive capacity of the soil and by the solubility of the initial chemical in the transport media.

Chemicals which have migrated to the groundwater may solubilize in the groundwater to the aqueous solubility limit of the chemical. The solubilized chemical may migrate with the groundwater and adsorb onto adjacent soils. Under continued migration to the groundwater from the soils above the water table, the extent of groundwater contamination may expand as the adsorptive capacity of the soils beneath the water table in the vicinity of the release is attained and exceeded.

When chemicals migrate from soils above the water table to the groundwater at a rate greater than the solubility capacity of the groundwater, the migration of the excess chemicals will result in a separate non-aqueous phase liquid (NAPL). The migration of NAPL is governed primarily by its density. If the chemical's density is less than one, it will tend to float on the surface of the groundwater and may migrate horizontally in the direction of groundwater flow. If the chemical's density is greater than one, it will tend to migrate vertically downward until a low-permeability geologic unit is encountered, at which point the chemical may tend to migrate horizontally in the direction of the surficial slope of the geologic unit. The extent of chemical migration, above or below the water table, may expand as the sorption capacity of the respective geologic unit is attained and exceeded. It is noted that NAPL was not identified in any of the borehole or monitoring well installations, and hence will not be further considered in this evaluation.

The physical and chemical properties of the Site-related chemicals detected in the different media at the Site are presented in Tables 7.1 and 7.2. The chemical properties include: molecular weight, aqueous solubility, vapor pressure, Henry's law constant, sorption coefficient, and specific density. Descriptive ranges of values for chemical and physical properties, and their relation to chemical transport are presented in Table 7.3. Also included in Table 7.3, are lists of examples of Site-related chemicals which exhibit these characteristics.

The molecular weight of a compound is useful for many calculations including: weight/volume unit conversions, molar volume determinations, and estimating Henry's law constants.

Aqueous solubility is an important factor in estimating a chemical's fate and transport in groundwater and surface water. Compounds with high aqueous solubilities have a tendency to desorb from soils and sediment, are less likely to volatilize from water, and tend to be more susceptible to biodegradation. Compounds with a high solubility will generally enter the water table more readily than less soluble compounds. Aqueous solubility is affected by temperature, pH, and other dissolved constituents. Site-related compounds with relatively high solubilities include VOCs with aqueous solubilities ranging from 174 mg/L (ethylbenzene) to miscible in all proportions (acetone). SVOCs detected at the Site generally have relatively low solubilities ranging from 2.6×10^{-4} mg/L to 35.1 mg/L with the exception of benzoic acid (3,400 mg/L), phenol (80,700 mg/L), and pyridine (1.0×10^{6} mg/L). Pesticides and PCBs have low aqueous solubilities ranging from 0.003 mg/L to 0.275 mg/L.

The sorption coefficient (Koc) indicates the tendency of a compound to partition between particles containing organic carbon and water. The sorption coefficient is inversely related to aqueous solubility such that a compound that binds strongly to organic carbon will have a low solubility. Compounds that adsorb onto organic materials in an aquifer are retarded in their movement in groundwater such that the compound migrates at a linear velocity less than the groundwater flow velocity.

Generally, VOCs have relatively low Koc values ranging from 1.2 to 292. SVOCs have a much higher adsorptive capacity with Koc values ranging from 0.6 to 9.8×10^{8}. Pesticides and PCBs have high Koc values ranging from 407 to 4.93×10^{5} and will bind strongly to soils.

The specific density of a compound is equivalent to the density of the substance relative to the density of water. Hydrophobic (low aqueous solubility) compounds with a specific density greater than one will generally tend to sink through the water table as dense non-aqueous phase liquids. Hydrophobic compounds with a specific density less than one will generally tend to float on the water table. Hydrophilic compounds (high aqueous solubility) behave differently. Acetone, for example, with a density of 0.790 does not float on water because it is highly soluble in water. Therefore, the solubility of a substance must be considered in conjunction with the specific density of a compound.

The vapor pressure of a compound provides a semi-quantitative rate at which volatilization will occur from soil and/or water to the atmosphere and/or soil gas. Generally, the Site-related VOCs have relatively high vapor pressures ranging from 6 mmHg to 455 mmHg. The Site-related SVOCs have low vapor pressures and a low potential for volatilization with values ranging from 9.59×10^{-11} mmHg to 20 mmHg. Pesticides and PCBs have a very low potential for volatilization, with vapor pressures ranging from 1×10^{-7} mmHg to 7.7×10^{-5} mmHg.

Henry's law constant provides an indication of the relative volatility of a compound (see Table 7.3). Henry's law constants are greatest for VOCs detected at the Site, and are all greater than 10^{-5} atm-m^{3}/mol. Site-related SVOCs and Pesticides/PCBs have Henry's law constants which are generally low, with values ranging from 2.96×10^{-20} atm-m^3/mol to 2.6×10^{-3} atm-m^3/mol.

7.2 GROUNDWATER CHEMICAL MIGRATION

Site-related chemicals have been detected in the groundwater in both the overburden and bedrock aquifer units in the vicinity and downgradient of the former lagoons, as described in Section 6.4.

Transport pathways in groundwater for Site-related chemicals include the water table aquifer unit and the bedrock aquifer unit, as discussed in Section 5.2.

7.2.1 WATER TABLE AQUIFER UNIT

The Water Table Aquifer unit consists of the overburden and uppermost 2-3 feet of the fractured bedrock. Flow within this aquifer unit is controlled by the divide which runs through Lagoon 4 in an east-west orientation such that groundwater flows either to the north or south of the divide.

The shallow groundwater flow from both the north and the south areas was calculated in Section 5.3 using Darcy's equation given by:

$$Q = KiA$$

where:

Q = the flow rate (ft^3/day)
K = average hydraulic conductivity (ft/day)
i = hydraulic gradient (ft/foot)
A = cross sectional area perpendicular to the flow path (ft^2)

The saturated thickness in the north section was estimated to range from 3 feet (dry season) to 10 feet (wet season). The saturated thickness in the south section was estimated to range from 3 feet (dry season) to 9 feet (wet season).

The length of the perpendicular cross-section flow zones for the northern and southern flow components were estimated to be 480 feet and 780 feet, respectively.

The calculated flow to the north ranges between 1,080 and 3,600 gal/day and to the south ranges between 870 and 2,600 gal/day in the Shallow Aquifer.

7.2.2 BEDROCK AQUIFER UNIT

The Bedrock Aquifer unit consists of the deeper fractured bedrock. Groundwater flows radially within the bedrock aquifer unit. The flow is primarily to the north and south with a minor westerly component.

The groundwater flux in the Bedrock Aquifer unit was calculated in Section 5.3 using Darcy's equation. The flux through the Bedrock Aquifer was estimated to range from 290 to 460 gal/day for the north component and from 3,090 to 4,820 gal/day for the south component.

The shallow bedrock groundwater discharges to the surface west of the Site to Beaverdam Brook, and south of the Site to Otter Kill.

The north component of shallow bedrock groundwater flow may discharge to Beaverdam Brook north of the Site. Another portion of this flow may potentially reach the Town Well 3, the closest of well of the Village of Maybrook wells. Water quality data for the Town wells indicates that the Site is not currently impacting the water supply as previously indicated in Section 3.11.

7.3 CHEMICAL MIGRATION VIA SURFACE WATER

Chemicals present in surface soil and sediments are potentially subject to transport via surface water runoff. Migration may occur by physical transport of the soils or by dissolution. This section presents an evaluation of the potential for migration of chemicals off Site via this pathway.

Surface soil samples collected during the Phase II RI in 1995 indicate the presence of VOCs, SVOCs, and PCBs at very low concentrations and hence, transport of chemicals from the general Site area via surface water runoff, is considered to be minimal.

Two samples, TP-11a and TP-41, collected during the test pit program, were collected at a distance of 1 to 3 feet below the surface and one foot below the surface, respectively. These samples are, therefore, considered representative of surface soil conditions within the lagoons although actual parameter concentrations at the surface would be expected to be somewhat lower due to biodegradation and volatilization. Several VOCs ranging in concentration from 32 µg/kg to 26,000 µg/kg were detected at TP-11a as well as pyridine (3,300J[6] µg/kg) and alpha-picoline (7,600J µg/kg). Toluene (1,800J µg/kg) and benzoic acid (4,400 µg/kg) were reported at TP-41. PCBs were reported at 780 µg/kg and 9,200J µg/kg for samples from TP-11a and TP-41, respectively. In addition, several pesticides were reported from TP-41 at concentrations ranging from 20X[7] µg/kg to 190J µg/kg.

The VOCs detected in samples taken from TP-11a and TP-41 at shallow depths, exhibited relatively high aqueous solubilities and low soil adsorptivities (low K_{oc}). This indicates that these compounds have the potential to be transported via surface water runoff in the dissolved phase in addition to the physical transport of the soil particles. Pesticides and PCBs exhibit a low aqueous solubility and a high adsorptive capacity for soils (high K_{oc}) and hence, have a far less potential for transport in the dissolved phase.

Surface water generally flows across the lagoons in a southerly direction to the base of the abandoned railway grade midway between the Site. Surface water flow tends either east or west following the railway grade, to a culvert to the east, or to Beaverdam Brook to the west. Surface water which travels east either collects in the marsh area north of the culvert, or travels through the culvert to the marsh area south of the abandoned railway grade. Surface water in the marsh area will either infiltrate or evaporate. Sediments in the surface water runoff will likely be deposited in the marsh areas.

Sediment samples were collected from sampling locations 1, 2, 5, and 8 (see Figure 3.3). Locations 1 and 2 are in a marsh area northeast of the Site, and Locations 5 and 8 are situated in a drainage ditch along the southern property boundary. Methylene chloride (43 µg/kg) and toluene (110 µg/kg) were present in the sample from Location 8, and methylene chloride (86 µg/kg) was reported in the sample from Location 2. A wide range of SVOCs primarily phthalates and PAHs, were reported at concentrations ranging from 460J µg/kg to 27,000 µg/kg, for samples obtained from Locations 1 and 2. Methylene chloride (36 µg/kg) and toluene (470 µg/L) were detected in the sample from Location 5 along with several SVOCs at a maximum concentration of 4,600 µg/kg (fluoranthene).

The presence of the chemicals in the drainage ditch sediments (Locations 5 and 8) indicates that chemicals are being transported via surface water runoff in the ditch south

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[6] The "J" qualifier indicates an estimated value.

[7] The "X" qualifier indicates compound with spectra that do not meet identification criteria, but is suspected to be present.

of the Site. The primary group of chemicals that were detected, PAHS, are likely attributable to the abandoned rail bed. Further transport of these chemicals may occur during significant rainfall events with ultimate discharge to Beaverdam Brook and Otter Kill. Chemicals, primarily PAHs unrelated to the Site, present in the marsh area northeast of the Site (Locations 2 and SSII-2) could also be further transported via surface water runoff for ultimate discharge to Beaverdam Brook.

Surface water flow in Beaverdam Brook and Otter Kill has the potential to transport Site-related chemicals away from the Site. Subsurface groundwater flow beneath the Site may discharge to either Beaverdam Brook or Otter Kill.

7.4 ATMOSPHERIC DISPERSION

The potential for atmospheric dispersion of chemicals at the Site is due to the release of chemicals from the surficial soils, exposed sediment and surface water to the atmosphere. The release of chemicals is affected by either volatilization or atmospheric entrainment of chemicals adsorbed onto particulate matter (dust).

The SVOCs detected in the surface soil at the Site, have a relatively low level of volatilization as indicated by the vapor pressure and Henry's law constants presented in Table 7.1. Vapor pressures are orders of magnitude below levels for compounds considered volatile with the exception of alpha-picoline and pyridine. Alpha-picoline and pyridine were detected in the sample from TP-11a (Lagoon 1). Pyridine has a high aqueous solubility (1.0×10^6 mg/L) and a high affinity for soil, thereby minimizing the potential for volatilization.

The pesticides and PCB (Aroclor-1254) compounds detected in sample TP-41 have a very low potential for volatilization due to their low vapor pressures and high adsorption capabilities (high Koc).

Volatile compounds detected in the surface soils and exposed sediment have a relatively high potential for volatilization as indicated by the high vapor pressures (6 to 455 mm Hg) and Henry's law constants ($>10^{-5}$ atm.m^3/mol) presented in Table 7.1. However, VOCs were detected at generally low concentrations (less than 500 µg/kg) in exposed sediment and the resulting atmospheric concentrations would be negligible.

VOCs detected at location TP-11a have somewhat higher concentrations (32 µg/kg to 26,000 µg/kg) and may potentially impact the air quality in the immediate vicinity of this sampling point. However, field measurements of organic vapors taken during the test pit excavation program indicated readings at background levels in the breathing zone. During test pit excavations, the highest vapor reading measured was 7 ppm for the subsurface sludge sample submitted for analysis from location TP-11a. Based on these organic vapor results, it is concluded that atmospheric concentrations of Site-related chemicals is negligible.

The potential for atmospheric entrainment of chemicals adsorbed onto particulate matter is considered minimal. The central and east central portions of the Site where the former lagoons were located have a cover consisting primarily of grass, immature herbaceous plants and immature pioneer tree species. This cover would tend to minimize the amount of airborne particulates and stabilize the surface soils at the Site. Any particulate dispersion would be minimized by the surrounding heavily wooded area.

7.5 CHEMICAL MIGRATION VIA INADVERTENT TRACKING

Tracking of chemicals at the Site would be caused by trespassers or wildlife walking across soils or through surface water contaminated with Site-related chemicals or potentially by vehicular traffic through the Site. Site access from County Road 4 is restricted by a locked gate. Vehicular traffic is therefore minimized. During the Site investigation, tire tracks or any other evidence of vehicular traffic was not observed. Although the Site is not restricted to trespassers, pedestrian traffic is considered minimal. Evidence of wildlife at the Site has been noted. Deer and rabbits may potentially track surface soils and surface water across the Site. However, the total mass of chemicals potentially transported via this pathway would be negligible.

7.6 ENVIRONMENTAL FATE

The basic physical and chemical properties of all detected Site-related organic and inorganic constituents are presented in Tables 7.1 and 7.2, respectively.

The probable behavior and environmental fate of all constituents can be assessed to some extent by evaluating the physical and chemical properties of the constituent. The mobility and persistence of these constituents are of primary importance in this evaluation. Mobility is the potential for a chemical to migrate away from the Site. Persistence is a measure of how long a chemical will remain in the environment. Factors that affect the mobility and persistence of Site-related constituents include, but are not limited to:

1) physical properties;
2) chemical properties;
3) moisture levels;
4) presence of microorganisms;
5) water chemistry;
6) pH; and
7) Reduction and Oxidization (REDOX) potential.

Water solubility is the maximum concentration of a compound that can dissolve in water at a specific temperature and pH. Compounds with high solubilities generally exhibit increased mobility in aqueous systems.

Vapor pressure and Henry's Law constants provide an indication of the volatility of a compound. High vapor pressures and Henry's Law constants indicate a greater tendency for a compound to volatilize. Compounds with high Henry's Law constants generally do not persist in surface water or surface soil environments.

The organic carbon partitioning coefficient (Koc) indicates the tendency of a compound to be adsorbed to organic materials in soils or sediments. High Koc values generally indicate lower mobility. For example, PCBs have very high Koc values and are virtually immobile in organic rich soils.

The behavior of inorganic compounds is dependent upon numerous factors including: water chemistry, pH, and REDOX potential. Generally, inorganic compounds will precipitate as pH and alkalinity increase. Inorganic compounds may also become complexed depending upon the chemical environment making them complicated to analyze.

The general degree to which the chemical properties affect the environmental fate of Site-related constituents is summarized in Table 7.3.

The following subsections provide descriptions of the four main contaminant groups identified at the Site: BTEX, PAHs, Site-specific parameters (alpha-picoline, 2-amino-pyridine, and pyridine), and inorganics.

7.6.1 BTEX

BTEX compounds are mono-aromatic hydrocarbons often found together at industrial sites, in the soil and groundwater media. BTEX compounds are some of the most prevalent organic compounds found in the environment due to their widespread use in solvents and in other petroleum products including automotive gasoline.

7.6.2 CHEMICAL IDENTITY

Chemical properties of BTEX compounds are presented on Table 7.1.

Benzene is a colorless liquid with a sweet odor that readily evaporates into air and dissolves easily in water. Automotive gasoline typically contains benzene at concentrations ranging from 1 to 4.5 percent.

Xylene is also a clear colorless liquid with a sweet odor. There are three xylene isomers: meta-xylene, ortho-xylene, and para-xylene (m-, o-, p-xylene). Total xylenes refers to the three isomers. Xylene evaporates easily but does not mix well with water. Automotive gasoline typically contains xylenes at concentrations ranging from 5.5 to 9 percent.

Ethylbenzene is a colorless liquid with chemical properties very similar to xylene. However, ethylbenzene is less readily sorbed onto soil particles in a groundwater system than xylenes (lower K_{oc}) and therefore, ethylbenzene is more mobile. Automotive gasoline typically contains ethylbenzene at concentrations ranging from 1 to 1.5 percent.

Toluene is a colorless liquid with a sour or burnt characteristic odor. The chemical properties of toluene generally lie in the middle range of the BTEX compounds with an aqueous solubility lower than benzene yet higher than xylene and ethylbenzene. Similarly, the vapor pressure and Koc values generally lie in the middle range of the BTEX compounds. Toluene typically occurs in automotive gasoline at concentrations of 4.0 to 6.5 percent.

7.6.2.1 ENVIRONMENTAL FATE

BTEX compounds released to the environment are subject to environmental fate processes including volatilization, sorption and biodegradation.

BTEX compounds have relatively low Koc values and high vapor pressures and therefore, are subject to transport from a soil medium into the atmosphere or the water column via volatilization or flushing, respectively. Of the BTEX compounds, benzene has the highest potential for mobility from the soil medium (K_{oc} of 78 ml/g, vapor pressure of 95.2 mm Hg, Henry's law constant of 5.48×10^{-3} atm-m^3/mol) whereas xylenes have the lowest potential mobility (K_{oc} of 240 ml/g, vapor pressure of 10 mm Hg, Henry's law constant of 7.04×10^{-3}atm-m^3/mol).

The rate of leaching of a chemical from soil to the aqueous phase is dependent upon the chemical concentration in the soil, the temperature, the chemical solubility, and the contact time between the source and the groundwater. The presence BTEX compounds in the groundwater beneath the Site indicates that leaching from the soils is occurring.

Chemicals present in a groundwater system can be transported via advection and hydrodynamic dispersion. Advection refers to the process of chemical transport by the bulk flow of groundwater and depends upon the groundwater velocity.

Hydrodynamic dispersion refers to the combined processes of molecular diffusion and kinematic dispersion. Diffusion is the process whereby chemicals move from regions of high concentration to regions of low concentration. Kinematic dispersion is the process of mechanical mixing due to variations in pore velocities. Both diffusion and dispersion result in a dilution of the chemical. Diffusion is dominant at low fluid velocities while dispersion is dominant at medium to high fluid velocities.

The transport of chemicals in a groundwater system is also affected by adsorption/desorption processes. Adsorption, or sorbing, refers to the binding of chemical molecules present in the aqueous phase to subsurface material. Desorption, the reverse of adsorption, refers to the release of chemical molecules from a bound state to a free aqueous phase. While adsorption reduces the aqueous phase concentration, desorption increases it.

Adsorption/desorption processes are typically assumed to be adequately described as a reversible, instantaneous, linear equilibrium isotherm approximated by the following expression:

$$S = K'C$$

where:

S = is the sorbed concentration (M/M);
K' = is the equilibrium partition coefficient (dimensionless); and
C = is the dissolved concentration (M/M).

Organic sorption refers to the sorption of organic compounds due to hydrophobic partitioning. In this case, S has units of chemical mass per unit mass of soil (M/M), C has units of chemical mass per unit mass of water (M/M) and K' is represented by the dimensionless "partition coefficient" K_p. Research into organic sorption has shown (Hamaker and Thompson, 1972, de Marsily, 1986) that K_p is highly related to the organic content of the subsurface material as expressed by:

$$K_p = K_{oc} f_{oc}$$

where:
K_{oc} represents the organic carbon partitioning coefficient (dimensionless); and
f_{oc} represents the fraction of organic carbon (dimensionless).

The organic carbon partitioning coefficient is a measure of the tendency for organics to be adsorbed by the subsurface media. K_{oc} is chemical specific and is largely independent of soil properties. The fraction of organic carbon represents the mass concentration (mass of carbon per unit mass of soil) of organic carbon in the porous medium and is thus a property of the subsurface material.

The impact of sorption on chemical transport may be represented by a retardation factor R. The retardation factor is the reciprocal of the ratio of the chemical transport velocity to the average linear groundwater velocity. For example, if it takes one year for a volume of groundwater to travel from a point A to another point B it would take 1*R years for a sorbing solute to travel from point A to point B.

The retardation factor may be calculated as follows [de Marsily, 1986].

Organic sorption:

$$R = 1 + \frac{\rho_b}{n\rho_w} K_p$$

where:
ρ_b represents the dry bulk density of the porous medium (M/L^3);
ρ_w represents the fluid density (M/L^3); and
n represents the total porosity (dimensionless).

Based upon the chemical properties for BTEX compounds presented in Table 7.1 and an assumed f_{oc} value of 0.01 (1 percent), a dry bulk density of 1.6 g/cm^3, and a porosity value of 0.3, the retardation factors for these compounds are estimated to be:

- benzene - 5.2;
- toluene - 8.0;
- ethylbenzene - 9.4; and
- xylenes - 13.8.

Due to their volatility, BTEX compounds do not persist to any significant extent in surface water. The rate of volatilization from surface waters is dependent upon the turbulence and water depth of the surface water body. BTEX compounds, as a group, also have a low potential for bioconcentration and therefore do not bioconcentrate to any significant extent in aquatic organisms.

BTEX compounds, once present in the groundwater, can also be attenuated via volatilization into the vadose zone and ultimately the atmosphere. As discussed previously, BTEX compounds have relatively high Henry's Law constant values and vapor pressures. Therefore, volatilization to the vadose zone would be a significant process for reducing chemical concentrations in the groundwater. BTEX compounds present in the atmosphere are subject to degradation process such as reaction with hydroxyl radicals (benzene, ethylbenzene) or phototransformations (xylenes).

Under certain conditions, BTEX compounds present in the groundwater readily biodegrade with reported half-life values ranging from 11-37 days (xylenes) to 68-110 days (benzene) in naturally occurring soil-groundwater systems (Dragun, 1988). Hence, under conditions with sufficient oxygen and nutrient sources, biological degradation can also be a significant mechanism for removal of BTEX from the groundwater.

In summary, BTEX compounds in soils are relatively mobile and subject to transport via groundwater and/or volatilization to the atmosphere. However, a natural attenuation process such as biological degradation is significant for reducing concentrations of BTEX compounds in groundwater or atmospheric air.

7.6.3 PAHS

PAHs are a group of chemicals formed during the incomplete combustion of coal, oil, gas, garbage, or other organic substances. Some PAHs are used in medicines and others are used to make dyes, plastics and pesticides. PAHs are found throughout the environment in the air, water, and soil. As pure chemicals, PAHs generally exist as colorless, white, or pale yellow-green solids. Most PAHs do not occur alone in the environment, rather they are found as mixtures of two or more PAHs. PAHs can occur either in the air attached to dust particles, or in soil or sediment as solids. PAHs can also be found in substances such as crude oil, coal tar pitch, creosote, and road/roofing tar. PAHs generally exhibit low solubility in water, but some may readily volatilize in air.

7.6.3.1 CHEMICAL IDENTITY

The following 14 PAHs are considered as a group in this review of the chemical and physical properties of PAHs.

- acenaphthene;
- anthracene;
- benz(a)anthracene;
- benzo(a)pyrene;
- benzo(b)fluoranthene;
- benzo(g,h,i)perylene;
- benzo(k)fluoranthene;
- chrysene;
- dibenzo(a,h)anthracene;
- fluoranthene;
- fluorene;
- indeno(1,2,3-cd)pyrene;
- phenanthrene; and
- pyrene.

The chemical properties of PAHs that were detected at the Site are presented in Table 7.1.

7.6.3.2 ENVIRONMENTAL FATE

PAHs released to the environment are subject to a number of environmental fate processes including volatilization, sorption, biodegradation, and bioaccumulation. PAH compounds tend to be removed from the water column by volatilization to the atmosphere, adsorption to particulates or sediments, or by being accumulated by, or sorbed into, aquatic organisms. Adsorption of PAHs to soil and sediments increases with increasing organic carbon content, and is also directly dependent on particle size.

The environmental fate of PAHs is described by U.S. EPA in the document "Ambient Water Quality Criteria for Polynuclear Aromatic Hydrocarbons" which states:

> ***"PAH will adsorb strongly onto suspended particulates and biota and their (PAH) transport will be determined largely by the hydrogeologic condition of the aquatic system. PAH dissolved in the water column will probably undergo direct photolysis at a rapid rate. The ultimate fate of those which accumulate in the sediment is believed to be biodegradation and biotransformation by benthic organisms (U.S. EPA, 1979)".***

This is restated in the U.S. EPA document "Health Effects Assessment for Polycyclic Aromatic Hydrocarbons (PAHs)" which states:

> ***"The predominant mechanism that is likely to dictate the fate of most PAHs in aquatic media is sorption to particulate matter and subsequent sedimentation and microbial degradation."***

Since PAHs are most likely to stay in the sediment or soil, microbial degradation is the most likely ultimate environmental fate in contrast to photolysis and volatilization. Compounds with four cyclic rings or less are most amenable to microbial degradation. Benzo(a)pyrene (five cyclic rings) has a half-life in soil inoculated with bacteria of less than eight days. The half-life reported for benzo(a)pyrene in soil reported in Superfund Public Health Evaluation Manual (SPHEM, 1986) is 420 to 480 days but there is no reference to the microbial content in the soil.

In summary, PAHs in soils and sediment would be relatively immobile thereby precluding significant concentrations in water or plants.

7.6.4 PYRIDINE COMPOUNDS

Three parameters (2-aminopyridine, alpha-picoline and pyridine), specific to the operations at the Harriman Site, were prevalent in the soils and groundwater samples collected at the Site. 2-aminopyridine and alpha-picoline are variations of the general pyridine compound which is used in the synthesis of vitamins and drugs and is also used in food flavorings, solvents, water-proofing and rubber chemicals. In addition, other pyridine-based TICs were identified in the investigation that may be present at the Site.

7.6.4.1 CHEMICAL IDENTITY

Pyridine is a colorless liquid with a characteristic burnt, pungent odor. Chemical properties of 2-aminopyridine, alpha-picoline, and pyridine are presented in Table 7.1. Some of the chemical specific properties for alpha-picoline and 2-aminopyridine are not available in the existing literature as these are relatively uncommon compounds. The pyridine compounds are generally characterized by very high aqueous solubilities and vapor pressures and low Koc values. Hence, pyridines are considered to be very mobile in a groundwater system.

7.6.4.2 ENVIRONMENTAL FATE

Pyridine compounds released to the environment are subject to environmental fate processes including volatilization, sorption, and biodegradation.

Similar to BTEX compounds, pyridines have high vapor pressures and low Koc values and therefore, are subject to transport from a soil medium into the atmosphere or the water column via volatilization or flushing, respectively. Scientific studies indicate that pyridine (and related compounds) can also be biodegraded in soil.

Pyridine compounds present in the atmosphere may be slowly photodegraded by hydroxyl radicals. However, due to their high water solubility, vapor phase pyridines tends to dissolve in water vapor such as clouds and rain drops.

The magnitude of the Henry's Law Constant values for pyridine indicates that pyridine in the aqueous phase will not readily volatilize to the atmosphere. Pyridine present in the aqueous phase will be transported in the groundwater via advection and hydrodynamic dispersion as discussed for BTEX compounds in Section 7.6.2.1.

The transport of pyridine compounds in the groundwater is affected by adsorption/desorption processes. Utilizing the relationship for organic sorption as presented in Section 7.6.2.1, the retardation factors for the Site-specific pyridine parameters were estimated to be:

alpha-picoline - 1.6;
2-amino-pyridine - 1.03; and
pyridine - 1.4.

Based upon these values, it is apparent that organic sorption onto soil particles has a relatively limited effect on attenuation of pyridine compounds in a groundwater system. Limited studies indicate that biodegradation may be the most important mechanism that can degrade pyridine in water.

Although not experimentally measured at this time, pyridine is expected to have a low bioconcentration factor suggesting that it is not bioconcentrated by aquatic plants or animals and therefore, pyridines are not biomagnified in terrestrial or aquatic food chains.

In summary, pyridine compounds in soils are relatively mobile and subject to transport via groundwater, yet less likely to volatilize to the atmosphere. Biological degradation is significant for reducing concentrations of pyridine compounds in groundwater and in atmospheric vapor.

7.6.5 INORGANICS

This section presents an evaluation of environmental fate for the following inorganics detected at the Site:

- iron;
- antimony;
- copper;
- lead;
- manganese;
- thallium;

- mercury; and
- arsenic;

The following sections present a review of the properties of each of the metals to determine and evaluate their properties in the environment that are important in determining their environmental fate.

7.6.5.1 IRON

The Earth's core is thought to be composed mainly of iron (Fe), with about 10 percent occluded hydrogen. Iron is the fourth most abundant element in the Earth's crust. Iron is a heavy ductile magnetic metallic element and is silver-white in pure form but readily rusts.

Iron is used in construction and tools and armament. Iron metal is usually alloyed with other metals and carbon for commercial uses. Steel is iron with a small percentage of carbon (<1.5 percent). Cast iron generally has 3 percent or greater carbon content and is very brittle and non-ductile. Wrought iron is pure iron crystals with thin layers of silicous slag resulting in grayness similar to wood.

Chemical Identity

Iron is a naturally occurring metallic element. Its atomic number is 26 and it has a density of 7.86 g/cm^3. Pertinent physical and chemical properties of iron are presented in Table 7.2.

Environmental Fate

Iron is mostly likely present in the atmosphere as particulates and is removed from the atmosphere by dry and wet deposition. The principal compounds of iron are ferrous (+2) and ferric (+3). In general, ferrous and ferric forms are mutually interconvertible. Ferrous compounds are more stable than ferric when ionized, less stable when covalent. A large proportion of iron salts are water soluble. Iron is often found in groundwater (in the form of ferrous iron) in concentrations ranging from zero to 10 ppm.

Speciation

In soils, oxidizing and alkaline conditions promote precipitation of iron, whereas acid reducing conditions promote the solution of iron compounds. Ferrous iron converts to a solid bluish-green ferrous hydroxide at pH >8.5. In the presence of oxygen, ferrous iron oxidizes to ferric iron, and ferric hydroxide forms a yellowish-orange solid (commonly called yellow boy), which precipitates at pH >3.5. In oxygen-poor conditions where iron is primarily in the ferrous form, enough alkalinity must be added to raise the solution pH to 8.5 before ferrous hydroxide precipitates.

7.6.5.2 ANTIMONY

Antimony (Sb) is a natural metal, extremely brittle, and consists of a flaky crystalline texture. It is not abundant, but is found in over 100 mineral species. It is sometimes found native, but more frequently as the sulfide stibnite.

Antimony greatly increased the hardness and mechanical strength of lead. It is often used in the flame retardant industry, glass, batteries, fireworks, and explosives.

Chemical Identity

The physical and chemical properties of antimony are presented in Table 7.2. Antimony has an atomic weight of 121.75 g/mole and an atomic number is 51. Antimony is insoluble in water.

Environmental Fate

Antimony occurs naturally in soils, groundwater, and surface water. It may enter the groundwater through natural weathering of rock, industrial production, municipal water disposal, or manufacturing processes.

Antimony is oxidized to antimony trioxide by reaction with atmospheric oxidants. The fate of antimony in waters is difficult to determine since the dissolved state is important and generally, total antimony in water is low. Antimony compounds may undergo photochemical reactions, but this fate has been reported to be insignificant. Antimony may be reduced and methylated by microorganisms and become mobilized. Antimony is known to form coprecipitates with hydrous iron, manganese, and aluminum oxides in soil and sediment. Methylated antimony compounds may be formed in waterlogged soil.

Speciation

Dissolved antimony discharged to natural waters tend to precipitate with sulfides or oxides, which are insoluble.

7.6.5.3 COPPER

Copper (Cu) is a naturally occurring element which is one of the oldest metals ever used and has been one of the important materials in the development of civilization. Uses include:

- electrical wiring;
- alloys;
- plating; and
- pesticide salts.

The principal copper ores include:

- cuprite;
- chalcocite;
- chalcopyrite;
- malachite; and
- azurite.

Chemical Identity

Copper is malleable, ductile and a good conductor of heat and electricity. The properties of copper metal are presented below:

• Atomic Weight	63.54 g/mole;
• Atomic Number	29;
• Density	$8.9\ g/cm^3$;
• Melting Point	108.34°C;

- Boiling Point 2595°C; and
- Oxidation States +1, +2.

Environmental Fate

In water, the amount of copper is dependent upon the pH and hardness. Concentrations from a few ppb to ppm have been reported (Stegavik, 1975). Natural copper concentrations in soil range from 2 to 100 mg/kg (Bower, 1966).

Speciation

Copper is speciated to a high degree in the presence and absence of organic complexing agents.

7.6.5.4 LEAD

Lead (Pb) is a naturally occurring bluish-gray metal found in only limited quantities in the Earth's crust. Lead and its compounds can be detected in all parts of the environment. Lead has a wide variety of uses including:

- batteries;
- ammunition;
- metal products; and
- chemical manufacturing.

Chemical Identity

The physical properties of lead are presented in Table 7.2.

Metallic lead is stable in dry air. In moist air, however, it quickly forms lead monoxide, which in turn produces lead carbonate with carbon dioxide in air. In general, the properties of inorganic lead compounds are similar to those of the alkaline earth metals. The nitrate, chlorate, and acetate salts are water soluble, chloride salt is slightly soluble and the sulfate, carbonate, chromate, phosphate, and sulfide salts are insoluble. Lead forms stable compounds with organic compounds, especially under anaerobic conditions, which are generally insoluble in water.

Environmental Fate

It is believed that combustion of leaded gasoline is responsible for over 90 percent of the lead in the environment. Lead particles are removed from the atmosphere by wet or dry deposition. Lead is extremely persistent in both water and soil. The specification of lead depends upon pH, temperature and the presence of humic materials. Lead is largely associated with the suspended solids and sediments (immobile forms) in aquatic systems.

Lead has the tendency to form compounds of low solubility. Hydroxide, carbonate, sulfide, and sulfate may act as solubility controls in precipitating lead from water.

The amount of lead that remains in solution depends largely upon the pH and the water dissolved salt content. Equilibrium calculations show that at pH>5.4 the solubility of lead is 30 µg/L (hard water) or 500 µg/L (soft water). The presence of sulfate and carbonates will limit these levels. In the groundwaters of the upper sand aquifer, the majority of any aqueous phase lead, which may leave the former lagoons, would be readily precipitated as $PbCO_3$ (solid).

Speciation

Lead exists in three oxidation states ($0, 2^+$ and 4^+). Lead can precipitate as a number of compounds including $PbS0_4$, $PbC0_3$, $Pb(OH)_2$, PbS, and $Pb_3(PO4)_2$.

Lead readily forms complexes with organic ligands, which tend to increase the amount of lead which can be dissolved in water. Lead is readily adsorbed to numerous solid surfaces including organic matter, clay, silica, iron, and manganese oxides.

7.6.5.5 MANGANESE

Manganese is an abundant element comprising about 0.1 percent of the earth's crust. It is a gray-white or silvery brittle metallic element, occurring in several allotropic forms, found worldwide, especially in the ores pyrolusite and rhodochrosite and in nodules on the ocean floor. Manganese is used as a binding agent in red brick and as an anticorrosive in most steel alloys.

Chemical Identity

Manganese is a naturally occurring metallic element. Its atomic weight is 54.94 g/mol and it has a density of 7.21 g/cm^3. Pertinent physical and chemical properties of manganese are presented in Table 7.2. Manganese is highly soluble and decomposes in water.

Environmental Fate

The chemical fate and transport of manganese in water is a function of the solubility of the specific chemical form, which in turn is a function of pH, oxidation-reduction potential, and the characteristics of available anions. Manganese is often transported in the aquatic environment adsorbed to suspended sediments.

The oxidation state of manganese in soils may be altered by microbial activity. It was observed that manganese (II) in suspensions of silt or clay loams was oxidized by microorganisms, leading to the precipitation of manganese minerals. Other studies have shown that bacteria and microflora can increase the mobility by increasing dissolution of manganese in subsurface environments.

Speciation

High concentrations of sulfide ions can cause manganese precipitation. Manganese sulfides become very insoluble in water.

7.6.5.6 THALLIUM

Thallium is a soft, bluish-white metal that is widely distributed in trace amounts in the earth's crust. It is usually found in combination with other elements primarily oxygen, sulfur and the halogens. Thallium compounds are used in the manufacture of alloys, electrical apparatus, catalysts for industrial organic reactions and optical equipment (e.g., spectrometers), and in photographic and ceramic formulations. Thallium is quite stable in the environment because it is neither transformed nor biodegraded.

Chemical Identity

Thallium is a naturally occurring element. Its atomic weight is 204.38 g/mol and it has a density of 11.85 g/cm^3. Pertinent physical and chemical properties are presented in Table 7.2. Thallium is highly insoluble in water.

Environmental Fate

Thallium may occur in aqueous systems as relatively soluble salts (thallium chloride, sulfate, carbonate, bromide, and hydroxide) associated with suspended solids and in combination with organic material. Thallium may partition from water to soils. Metallic thallium oxidizes slowly in air. Thallium released to the atmosphere is deposited onto soil or water surface by wet and dry deposition.

Speciation

Thallium ion forms complexes in solution with halogens, oxygen and sulfur.

7.6.5.7 MERCURY

Mercury occurs in the earth's crust primarily in the form of sulfides (i.e., cinnabar). Mercury is a heavy, toxic and volatile silvery conducting metal, which is liquid at room temperature. It is associated by most people with thermometers and barometers, but is also used as an industrial chemical in a range of processes (e.g., chlorine production). Most of the mercury in the environment stems from natural sources (e.g., volcanoes, glaciers, oceans and seas).

Chemical Identity

Mercury has an atomic weight 200.6 g/mol, unique (for metals) in that it remains liquid under all but very extreme temperatures. It has a density of 13.53 g/cm^3. Pertinent physical and chemical properties of mercury are presented in Table 7.2. Mercury is has a water solubility of 0.28 mg/L.

Environmental Fate

Natural sources, such as geological weathering, contribute about the same order of magnitude of mercury to the environment as do the direct and indirect sources related to human activities.

The primary form of atmospheric mercury, metallic mercury vapor is oxidized by ozone to other mercury forms and is removed from the atmosphere by precipitation. Elemental mercury is oxidized to inorganic mercury under natural conditions. The overall residence time of elemental mercury have been reported to be 6 days to 2 years.

The main atmospheric transformation process of organomercurials is reported to be photolysis.

Mercury is transported to water systems via surface runoff and atmospheric precipitation. Inorganic mercury sorbed to particulate material is not readily desorbed. Therefore, sediments are important repositories for inorganic forms. The most important transformation process of mercury in water is biotranformation. Any form of mercury entering waters can be microbially converted to methylmercuric ions. Volatile elemental mercury may be formed through the demethylation of methylmercury or the reduction of inorganic mercury. Abiotic reduction of inorganic mercury to metallic mercury in aqueous solutions can also occur, particularly in the presence of soluble humic substances. This reduction is enhanced by light and can occur under both aerobic and anaerobic conditions.

Mercury compounds in soils may undergo the same fate process as those seen in waters. The formation and degradation of organic mercurials occurs through biotransformation and may also undergo abiotic processes. Mercuric mercury usually forms various complexes with chloride and hydroxide ions in soils.

Speciation

In the presence of elemental sulfur that may be abundant in anoxic sediments, some of the methylmercury may be methylated to form volatile dimethylmercury, which diffuses into the water column and evaporates into the atmosphere. Under slightly more oxidizing conditions than those required for methylation, methylmercury is demethylated by marine bacteria. Under more strongly reducing conditions in marine sediments, most of the inorganic mercury precipitates as highly insoluble mercuric sulfide.

7.6.5.8 ARSENIC

Arsenic is a metalloid occurring naturally in the earth's crust and fossil fuels. It is used in the production of glass, enamels, ceramics, oil, cloth, linoleum, electrical semiconductors, pigments, fireworks, pesticides, fungicides, veterinary pharmaceuticals, and wood preservatives. It is a known human carcinogen, and bioaccumulates to toxic levels.

Chemical Identity

Arsenic has an atomic weight of 74.92 g/mol and a specific density of 5.727. The atomic number for this metal is 33.

Arsenic may occur as a semi metallic element (As^o), arsenate (As^{5+}), arsenite (As^{3+}), or arsine (As^{3-}).

Environmental Fate

When released to the atmosphere primarily as arsenic trioxide or as arsines, these arsenic compounds undergo oxidation to the pentavalent state. Arsenic in the atmosphere is typically a mixture of trivalent and pentavalent forms. Photolysis is not an important fate for arsenic compounds in the atmosphere.

Arsenic in aquatic environments undergo complex series of transformation, including oxidation-reduction reactions, ligand exchange, precipitation and biotransformation. Arsenate is the predominant form of arsenic in the aquatic environment. Aquatic microorganisms could reduce arsenate to arsenite and a variety of methylated arsenicals.

The arsenic cycle in soils is complex, with many biotic and abiotic processes controlling its overall fate. Arsenicals applied to soils may be methylated by microorganisms to arsines which are lost through volatilization and organic forms may be mineralized to inorganic forms. Organoarsenical pesticides (i.e., MMA, DMA) are metabolized by bacteria to alkylarsines, arsenate and MMA.

Speciation

Through precipitation, arsenic is removed as either calcium or ferric arsenate. Arsenites can also be precipitated, but they are generally more soluble and less stable then arsenates. Arsenite-containing effluent is generally oxidized prior to precipitation to ensure that the arsenate predominates.

8.0 ECOLOGICAL RISK ASSESSMENT

8.1 INTRODUCTION

The Ecological Risk Assessment (ERA) presented herein consists of an evaluation of potential risk to ecological receptors, if any, posed by COPCs in and around the vicinity of the Site. The study area for the ERA is presented on Figure 1.2.

An ERA was initially completed for this Site following guidance provided in the NYSDEC (Fish and Wildlife Impact Analysis for Inactive Hazardous Waste Sites [FWIA]) (NYSDEC, 1994). Appendix D provides the previous FWIA report which completed Step 1 of the FWIA process. It consists of a site description and mapping of the Site and fish and wildlife resources within two miles of the Site. An analysis of fish and wildlife values is also included and a note that no observations of stressed vegetation were made. Step 2 of the FWIA, Contaminant Specific Analysis, will be addressed in this ERA.

The ERA presented herein consists of a revision of the original ERA based on current U.S. EPA guidance to the extent possible, which includes: "Ecological Risk Assessment Guidance for Superfund, Process for Designing and Conducting Ecological Risk Assessments" (U.S. EPA, 1997) and "Framework for Ecological Risk Assessment" (U.S. EPA, 1992c). In the original ERA, Steps I and II of the FWIA were roughly equivalent to Steps 1 and 2 of the U.S. EPA's 8-step process (U.S. EPA 1997). Steps taken in conducting the ERA are documented according to the U.S. EPA process, where possible.

An ecological field investigation was conducted in May 1995 by Fine Line Technical Services Inc. (Fine Line) of Medina, New York. The purpose of the field investigation was to provide a qualitative description of fish and wildlife resources in and around the vicinity (two-mile radius) of the Site. The results of the field investigation are documented in a report prepared by Fine Line. A copy of the Fine Line report is provided as Appendix D. A summary of the report is included in the description of the environmental setting (Section 8.2.1.1).

A screening level ecological risk assessment (SLERA) was conducted using the guidance of the NYSDEC and U.S. EPA, where appropriate, and is comprised of the following steps:

Step 1: Screening-Level Problem Formulation and Ecological Effects Evaluation; and

Step 2: Screening-Level Exposure Estimate and Risk Calculation.

U.S. EPA guidance (1997) recommends that, based on the results of a screening assessment, a decision be made on whether to stop the assessment or proceed to a more comprehensive baseline ecological assessment (BERA). Based on the results of the screening assessment, it was determined that a baseline assessment was necessary to adequately evaluate the potential for risk to ecological receptors. The rationale for this decision is presented in Section 8.3.4. The BERA conducted as part of this ERA includes the following steps:

Step 3: Baseline Risk Assessment Problem Formulation; and

Step 7: Risk Characterization.

Steps 4 and 5 of the U.S. EPA 8-step process typically involve additional analyses and may require collection of data to fill in data gaps identified in the screening assessment. Step 4 is the development of a Work Plan and Sampling and Analysis Plan. Step 5 is verification of the feasibility of any additional studies. Step 6 is the site investigation and data analysis step. Because no additional field investigations or laboratory testing was conducted, Steps 4, 5, and 6 were not necessary.

The risk characterization conducted as Step 7 includes food chain modeling for terrestrial receptors and mammalian and avian piscivores. U.S. EPA collected and analyzed sediments from Beaverdam Brook in 2003. These data are evaluated in the screening assessment.

Step 8 of the 8-step process is risk management. Recommendations for management of ecological risk are provided in Section 8.8 of this document.

Sections 8.2 and 8.3 of this ERA report present the SLERA (Steps 1 and 2). Section 8.4 presents the BERA (Steps 3, 7 and 8).

8.2 STEP 1 - SCREENING-LEVEL PROBLEM FORMULATION AND ECOLOGICAL EFFECTS EVALUATION

8.2.1 SCREENING-LEVEL PROBLEM FORMULATION

8.2.1.1 ENVIRONMENTAL SETTING

8.2.1.1.1 OVERVIEW

The Site is a 29.3 acre parcel of undeveloped, primarily forested land. A general description and relevant background information for the Site is provided in Sections 2.0 through 6.0 of this RI Report and Section 9.1 in the Human Health Risk Assessment portion of this document. Additional information for this ERA was provided by an ecological site investigation conducted by Fine Line in May 1995. A copy of the Fine Line Report is provided as Appendix D. In addition to the Fine Line investigation, CRA conducted a site reconnaissance and delineated jurisdictional wetlands and watercourses on the Site in May 2003. CRA's Wetlands Delineation Report is included as Section 3.9 of this RI Report.

A natural resource map was prepared by Fine Line as part of the ecological field investigation (Figure 2.1 of Appendix D). The natural resource map is comparable to the topographic map identified in the NYSDEC guidance for preparation of a FWIA. The map identifies the location of the Site; natural features such as streams, open water, and freshwater wetlands; and fish and wildlife resources in the area within two miles of the perimeter of the Site.

An aerial photograph of the Site identifying cover types on and immediately adjacent to the Site is provided as Figure 2.4 of the Fine Line Report in Appendix D. As shown on Figure 2.4, there are no buildings or other structures on the Site. Because the Site is undeveloped, it provides potentially significant habitat for wildlife.

No areas of obvious contamination or stressed vegetation as a result of contamination were observed on or in the vicinity of Site during the ecological field investigation (Fine Line) or Site reconnaissance (CRA).

8.2.1.1.2 ON-SITE RESOURCES

8.2.1.1.2.1 TERRESTRIAL HABITATS

Four primary types of terrestrial habitats are present on the Site: Appalachian oak-hickory forest, successional northern hardwood forest, successional shrub land, and successional oldfield. Figure 2.4 of the Fine Line Report (Appendix D) identifies the approximate locations of these four terrestrial habitats on and adjacent to the Site.

Appalachian oak-hickory forest typically occurs on well drained soils or ridges of upper slopes. Black cherry (*Prunus serotina*), red oak (*Quercus rubra*), white oak (*Quercus alba*), red maple (*Acer rubrum*), sugar maple (*Acer saccharum*), American elm (*Ulmus americnum*), shagbark hickory (*Carya ovata*), white ash (*Fraxinus america*) are dominant overstory species. Common shrubs in this forest type include gray-stem dogwood (*Cornus foemina*), flowering dogwood (*Cornus florida*), barberry (*Berberis canadensis*), choke cherry (*Prunus virginiana*), and blackhaw (*Virburnum pruniflolium*).

Successional northern hardwood forest occurs in areas that have been disturbed by timbering, farming, or other anthropogenic disturbances. Canopy species tend to be tolerant of sunny conditions and the shrub and herbaceous species are characteristic of successional shrub and old field communities. Common canopy species in this forest type include quaking aspen (*Populus tremuloides*), cottonwood (*Populus deltoides*), red maple, black cherry, American elm, shagbark hickory, white ash, green ash (*Fraxinus pennsylvanicus*), and sycamore (*Platanus occidentalis*). Silky dogwood (*Cornus amomum*), gray-stem dogwood, blackberry (*Rubus allegheniensis*), staghorn sumac (*Rhus typhina*), and honeysuckle (*Lonicera tartarica*) are common understory species. Herbaceous species typically include garlic mustard (*Alliaria officinalis*), enchanters nightshade (*Circaea quadrisiculata*), moneywort (*Lysimachia nummularia*), and trout lily (*Erythronium americanum*).

Successional old field is the dominant habitat on those areas of the Site that have been disturbed by filling, mining operations, and other anthropogenic activities. The plant communities in successional old field habitats are dominated by grass and forb species that tend to colonize recently disturbed areas. Dominant herbaceous species include goldenrod (*Solidago altissima*), cinquefoil (*Potentila simplex*), birdfoot trefoil (*Lotus corniculatus*), Queen Anne's lace (*Daucus carota*), ox-eye daisy (*Chrysantheum leucanthemum*), milkweed (*Asclepius syriaca*), evening primrose (*Oenothera biennis*), and hawkweed (*Hieracium pilosella*). Trees and shrubs, which are relatively few in number in this type of habitat, include quaking aspen, gray birch (*Betula populifolia*), staghorn sumac, multiflora rose (*Rosa multiflora*), gray-stem dogwood, and honeysuckle. Successional old field is the dominant habitat type in the area of the lagoons.

The Fine Line Report (see Appendix D) provides additional information on these terrestrial habitats.

8.2.1.1.2.2 WETLANDS

As part of the Site reconnaissance conducted in support of this ERA, CRA formally delineated the extent of jurisdictional wetlands on and adjacent to the Site. CRA's Wetlands Delineation Report is included as Section 3.9 of this RI Report. CRA identified and delineated three areas of wetlands, designated as Wetland BB, Wetland CC, and Wetland DD, within the boundaries of the Site. The total acreage of these three wetlands is 2.0 acres. Wetland AA and Wetland FF were also identified and delineated, however, both of these wetlands are off-Site. Wetland AA is southeast of the Site. Wetland FF is southwest of the Site and adjacent to the Otter Kill. The locations of the on-Site and off-Site wetlands are identified on Figures 3.9 and 3.10.

Wetland AA is an off-Site palustrine forested wetland adjacent to the southeastern corner of the Site. Dominant species in the canopy are pin oak (*Quercus palustris*) and red maple. Dominant species in the sapling/shrub stratum are red maple, southern arrowwood (*Viburnum dentatum*), green ash, silky dogwood, red chokeberry (*Aronia arbutifolia*), black cherry, black gum (*Nyssa sylvatica*), and poison ivy (*Toxicodendron radicans*). Both the tree and sapling/shrub covers in this wetland are dense. Dominant taxa in the herbaceous stratum are marsh blue violet (*Viola cucullata*), enchanters nightshade, jewelweed (*Impatiens capensis*), sedge (*Carex sp.*), and fern.

Soil in Wetland AA is mapped as Alden silt loam, which is listed as a hydric soil on the National Hydric Soils List and the New York Hydric Soils List. At the time of the Site reconnaissance, the soil was saturated to the surface. Depth of free water was 2 inches below ground surface. Evidence of vernal ponds within the forested wetland was also observed.

Wetland BB is a palustrine forested wetland in the southeastern portion of the site, south of the right of way of the abandoned railroad. Dominant species in the canopy are red maple, pin oak, swamp white oak (*Quercus bicolor*), slippery elm (*Ulmus rubra*), and silky dogwood. Dominant species in the sapling/shrub stratum are green ash, red maple, silky dogwood, southern arrowwood, witch hazel (*Hamamelis virginiana*), multiflora rose, and poison ivy. Dominant species in the herbaceous stratum are jack-in-the-pulpit

(*Arisaema atrorubens*), wild geranium (*Gerrnium maculatum*), sensitive fern (*Onoclea sensilibis*), marsh blue violet, skunk cabbage (*Symplocarpus foetidus*), and trout lily.

Soil in Wetland BB is mapped as Alden silt loam. At the time of the Site reconnaissance, soil was saturated to the surface. Depth of free water was 2 inches below ground surface. Several vernal pools, supporting tussock sedge (*Carex stricta*), were also observed.

Wetland CC is a palustrine forested wetland associated with Beaverdam Brook, which flows along the western boundary of the Site. Wetland CC is south of the concrete bridge in the west central portion of the site. Dominant species in the canopy are swamp white oak and green ash. Dominant species in the sapling/shrub stratum are wild black current (*Ribes americanum*), choke cherry, southern arrowwood, and poison ivy. Dominant taxa in the herbaceous stratum are skunk cabbage, sensitive fern, tussock sedge, arrow arum (*Peltandra virginica*), and sedges.

Soil in Wetland CC is mapped as Halsey silt loam, which is listed as a hydric soil on the National Hydric Soils List and the New York Hydric Soils List. At the time of the Site reconnaissance, soil was saturated to the surface. Free water was at the ground surface. Stained leaves, indicating the presence of inundation, were also observed.

Wetland DD is a palustrine forested wetland associated with Beaverdam Brook in the northwestern portion of the Site. It is separated from Wetland CC by a concrete bridge in the west-central portion of the Site. Dominant species in the canopy are swamp white oak and pin oak. Dominant species in the sapling/shrub stratum are swamp white oak, pin oak, red maple, silky dogwood, multiflora rose, and honeysuckle. Dominant taxa in the herbaceous stratum are sensitive fern, purple loosestrife (*Lythrum salicaria*), grass-leaved goldenrod (*Solidago graminifolia*), steeplebush (*Spiraea tomentosa*), Kentucky bluegrass (*Poa pratensis*), fragrant bedstraw, iris (*Iris sp.*), sedge (*Carex sp.*), and grass (Gramineae).

Soil in Wetland DD is mapped as Halsey silt loam. At the time of the Site reconnaissance, the soil was saturated to the surface. Free water was 2 inches below ground surface. Stained leaves, indicating the presence of inundation, were also observed.

Wetland FF is a palustrine forested wetland associated with Beaverdam Brook immediately west of the southwestern Site boundary. Wetland FF is a continuation of Wetland CC, but is physically separated from Wetland CC by the abandoned rail right-of-way. Vegetation, soil, and hydrology of Wetland FF are similar to Wetland CC.

Wetland FF is referred to as the Southwest Marsh throughout the remainder of this document.

The wetland area referred to as the Northeast Marsh is not located on-Site and was not included in the CRA wetland delineation because it is on the north side of Route 4 adjacent to Beaverdam Creek upstream of the Site and is outside the area of impact associated with the Site.

8.2.1.1.2.3 WATERCOURSES

Beaverdam Brook is a perennial watercourse that enters the Site near the northwestern property boundary, flows south, and exits the Site at the west-central property boundary. Beaverdam Brook on and adjacent to the Site is typical of a marsh headwater stream. The segment adjacent to the Site, near its confluence with Otter Kill, includes typical backwater sloughs and rocky substrates. Beaverdam Brook is generally five to ten feet wide, with a depth of two feet or less. Substrates include cobbles, stones, gravel, sand, and silt. Observations of debris drift and high water marks indicated that the flow regime is variable, and surrounding areas may be seasonally inundated.

Beaverdam Brook flows into Otter Kill. The portion of Otter Kill in the vicinity of the Site is a medium size stream with varied habitat characteristics. The stream channel is approximately 30 feet wide with a slightly meandering pattern. The course of the channel is well defined with steep to gently sloping banks composed of sandy loam and silty clay soils. Substrates include stone, rock, gravel, and mud. Flow conditions and water depths in this area are seasonally variable, with high water conditions occurring in spring and fall. Under base conditions, flow is sluggish and without distinct riffle and pool areas. Water depths in mid-channel are three feet or less. Floodplain areas show evidence of periodic high water conditions, including debris drift and high water marks on streamside vegetation.

The portion of Beaverdam Brook on and adjacent to the Site and the Northeast Marsh Area are classified by NYSDEC as Class "D" surface water. The portion of Otter Kill adjacent to the Site boundary is classified by NYSDEC as a Class "C" surface water.

8.2.1.1.2.4 SPECIES OF CONCERN

As part of the ecological field investigation, Fine Line contacted the NYSDEC's Significant Habitat and Natural Heritage Program regarding historical or current records for species of concern (e.g., threatened and endangered species) within the vicinity of the Site. Responses from NYSDEC indicate that there are no known occurrences of wildlife species that are rare or endangered, or species of special concern, within two miles of the Site boundary.

CRA contacted the U.S. Fish and Wildlife Service (USFWS) regarding the potential occurrence of federally-listed species of concern on or adjacent to the Site. In a letter dated December 30, 2005, the USFWS responded that there is no critical habitat within the project area. The USFWS also stated the Site is located within two miles of known roosts and within 30 miles of known hiberbacula for the Indiana bat (*Myotis sodalis*) and within 10 miles of known sites for the bog turtle (*Ckemmys muhlenbergii*). A copy of the December 30, 2005 letter from the USFWS is provided as Appendix E.

8.2.1.1.2.5 WILDLIFE

A description of wildlife expected within each habitat type is presented in the Fine Line Report and summarized in Table 8.1. Information on potential wildlife utilization of on-site habitats was obtained through review of standard natural history references; observations during the ecological field investigation; and from NYSDEC databases, including Region 3 Bureau of Wildlife, Region 3 Division of Fisheries, and the New York Natural Heritage Program, Wildlife Resources Center.

The wildlife/habitat associations presented in Table 8.1 represent the best professional judgment of Fine Line. Information regarding the actual occurrence of wildlife species on and in the vicinity of the Site is not available. No netting, trapping, subsurface examination of substrate, or other collecting procedures were required nor conducted during the field investigation.

8.2.1.1.2.6 FISH

The Fine Line Report stated that limited data are available for fish populations and communities within the immediate vicinity of the Site. The Fine Line Report does, however, identify species of fish historically collected by NYSDEC in Beaverdam Brook and the Otter Kill watershed. Fish species identified NYSDEC as a result of a survey conducted in 1936 are summarized in Table 8.1. The fish survey conducted by NYSDEC in 1936 is not necessarily representative of the species that presently occur in Beaverdam Brook and the Otter Kill watershed. However, the results of the survey are useful in identifying the species that historically occurred. No additional surveys for fish were identified during the preparation of this RI Report.

8.2.1.1.3 OFF-SITE RESOURCES

8.2.1.1.3.1 VEGETATION COVER TYPES 0.5 MILE RADIUS FROM PERIMETER OF THE SITE

Natural fish and wildlife resource communities that occur within a one-half mile radius of the perimeter of the Site include components of riverine, palustrine wetland, and terrestrial systems. These components are identified as distinct community types based on descriptions of the New York Natural Heritage Program. The ecological field investigation report prepared by Fine Line (see Appendix D) presents a detailed description of natural fish and wildlife resource communities that occur within a one-half mile radius of the perimeter of the Site.

Natural vegetative cover types, and locations of field survey observation points in the area within a one-half mile radius of the Site perimeter, are shown on Figure 2.3 of the Fine Line report in Appendix D. The map was prepared from interpretation of aerial photographs, review of agency resource information, and observations made during the field investigation in May 1995.

Two areas immediately adjacent to the Site boundaries, the Southwest Marsh and the Northeast Marsh Area, are evaluated as part of this ERA. The Southwest Marsh is a palustrine forested wetland associated with Beaverdam Brook immediately west of the southwestern Site boundary. The Southwest Marsh is a continuation of Wetland CC identified in the Wetlands Delineation Report (Section 3.9), but is physically separated from Wetland CC by the abandoned rail right-of-way. Vegetation, soil, and hydrology of the Southwest Marsh are similar to Wetland CC (see Section 8.2.1.1.2.2).

The Northeast Marsh is an upstream continuation of the Wetland DD drainage. Dominant tree species are swamp white oak and pin oak. Dominant species in the sapling/shrub stratum are swamp white oak, pin oak, red maple, silky dogwood, multiflora rose, and honeysuckle. Dominant taxa in the herbaceous stratum are sensitive fern, purple loosestrife, grass-leaved goldenrod, steeplebush, Kentucky bluegrass, fragrant bedstraw, iris, sedge, and grass.

8.2.1.1.3.2 RESOURCES WITHIN TWO MILES FROM THE PERIMETER OF THE SITE

Natural resources within two miles of the Site include a number of New York State (NYS) freshwater wetlands. Fifteen NYS freshwater wetlands occur, in whole or in part, within two miles of the Site. Boundaries of mapped NYS freshwater wetlands are shown on Figure 2.1 of the Fine Line Report (see Appendix D). State wetlands are classified according to a classification system developed by NYSDEC as outlined in Part 664.5. This system establishes four separate classes that rank wetlands according to their ability to perform wetland functions and provide wetland benefits.

State wetland MB-18 is identified as a Class I wetland. State wetlands MB-1, MB-7, MB-16, MB-32, GO-6, GO-12, GO-13, and GO-14 are identified as Class II wetlands. State wetlands MB-3, MB-4, MB-17, MB-30, MB-31, and MB-33 are identified as Class III wetlands.

A number of mapped federal wetlands also occur in the area within two miles of the Site. These wetlands were identified from the United States Fish and Wildlife Service, National Wetland Inventory Maps titled "Maybrook, NY" and "Goshen, NY" (portions of which are presented on Figure 2.2 of the Appendix D). Federally regulated wetlands are classified according to ecological systems, cover type, and water regime.

Portions of the Walkill River, Otter Kill, Beaverdam Brook, and a number of classified tributaries and open water areas occur within a two-mile radius of the Site boundary. These resources are identified on Figure 2.1 of the Fine Line Report in Appendix D and are described in the Fine Line Report.

8.2.1.1.3.3 RESOURCES MORE THAN TWO MILES DOWNSTREAM FROM THE SITE

No additional natural resources have been mapped between two and four miles downstream of the Site perimeter. The closest mapped resources occurring more than four miles downstream include State wetland MB-35 and Moodna Creek.

8.2.1.1.4 VALUE OF HABITAT TO ASSOCIATED FAUNA

A detailed description of fish and wildlife resource values are provided in the Fine Line Report. In general, natural communities in the vicinity of the Site have not been significantly altered by anthropogenic activities. Acreage adjacent to, and in the vicinity of the Site, is partially developed as agricultural, recreational (Otter Kill County Club), and residential. Resident and migratory wildlife that may utilize these resources include a variety of species of fish, waterfowl, amphibians, reptiles, birds, and mammals (See Table 8.1).

8.2.1.1.5 VALUE OF RESOURCES TO HUMANS

Resources in the vicinity of the Site have recreational and economic value to humans. The extent to which these resources are utilized is largely undocumented. Some areas in the vicinity are controlled and used by hunting clubs. The occurrence of tree stands and discarded spent cartridges observed in some locations indicate that these areas are utilized for hunting deer, wild turkey, and other game species.

Potential recreational opportunities of the Site area include use by bicycles, ATVs, and snowmobiles; fishing; observation of wildlife; and scientific study. The intended future land use of the Site is a Nature Conservatory that will undoubtedly attract naturalists to the area.

All areas in the Site vicinity have been logged at some time in the past. No evidence of commercial timbering was observed during the field reconnaissance. Some hardwood tree species in the area are marketable, though the potential for future commercial timbering in the area is likely to be driven by economic forces.

8.2.1.2 KNOWN AND SUSPECTED CONTAMINANTS

A description of the nature and extent of contamination in various Site media is presented in Section 6.0 of this RI Report. Based on investigations completed to date, Site-related constituents have been identified in the following media:

- surface soil on Site within the area of the former lagoons;
- surface water of the Northeast Marsh Area, Beaverdam Brook, and Otter Kill; and
- sediment of the Northeast Marsh Area, Beaverdam Brook, and Otter Kill.

Site-related constituents have also been detected in subsurface soil in the vicinity of the former lagoons and in groundwater. Because ecological receptors are not exposed to subsurface soil and groundwater, these media are not evaluated in this ERA. Soil samples evaluated in this ERA are from 0 to 2 feet below ground surface (bgs).

Chemical constituents detected in surface soil, surface water, and sediment are primarily metals. Aluminum, arsenic, barium, beryllium, chromium, cyanide, copper, iron, lead, manganese, nickel, thallium, vanadium, and zinc have been detected in all three media. Acetone, toluene, and bis(2-ethylhexyl)phthalate have also been detected in all three media. Several VOCs (2-butanone, methylene chloride, and tetrachloroethene), SVOCs (4-methylphenol, n-nitrosodiphenylamine, and phenol), BTEX constituents (ethylbenzene and xylene), PAHs (phenanthrene, fluoranthene, pyrene, benzo(a)anthracene, chrysene, benzo(b)fluoranthene, and benzo(a)pyrene), phthalates (di-n-octylphthalate), pesticides (4,4'-DDE, 4,4'-DDT, and endrin aldehyde), and PCBs (Aroclor 1254) have been detected in one or more media.

8.2.1.3 CONTAMINANT FATE AND TRANSPORT

The vegetative communities on the Site have the potential to support a diversity of ecological receptors. In addition, Beaverdam Brook, Otter Kill, the Northeast Marsh Area, and Southwest Marsh Area are either on or in close proximity to the Site. There are several potential mechanisms for transport of chemical constituents from one or more sources to habitats that potentially support ecological receptors.

One source of transport is via surface flow during storm events. Constituents dissolved in stormwater or adsorbed to particles suspended in stormwater may be transported from source areas to habitats within or adjacent to the Site. Surface flow may also transport constituents from source areas to the surface water and sediments of Beaverdam Brook, Otter Kill, the Northeast Marsh Area, or the Southwest Marsh. It is expected that VOCs are released to the atmosphere during this mechanism of transport and are not deposited in habitats away from source areas.

Wind is another potential mechanism for transport of chemical constituents from source areas to ecological receptors. Constituents transported by wind may be deposited in terrestrial habitats, wetlands, or the surface water of Beaverdam Brook, Otter Kill, the Northeast Marsh Area, and Southwest Marsh Area. The fate of constituents reaching these habitats depends on the chemical and physical properties of the constituents and the interaction of the constituents with biological properties (e.g., microbial communities) of the habitats.

Chemical constituents may also be transported from source areas via groundwater. Constituents may have historically been introduced into groundwater beneath the Site, or they may have leached from soil into the groundwater. Groundwater has the potential to migrate from the Site and discharge into Beaverdam Brook or other nearby surface waters (e.g., wetlands). The fate of constituents in watercourses and wetlands is dependent on the chemical and physical properties of the constituent and the interaction of the constituents with the biological properties of the habitats. VOCs likely volatilize to the atmosphere. Hydrophobic constituents likely leave solution and become bound to organic matter in the sediments or soil of the wetlands. Other less hydrophobic constituents may remain in solution.

8.2.1.4 POTENTIALLY COMPLETE PATHWAYS

8.2.1.4.1 OVERVIEW

The purpose of the exposure pathway analysis is to identify the potential primary pathways of contaminant migration and exposure to terrestrial and/or aquatic receptors. Exposure pathways identify how ecological receptors can come in contact with or ingest COPCs from a source media. The ERA evaluated potential risks associated with COPCs identified in abiotic media and emphasizes routes of exposure associated with direct contact and ingestion. Additionally, potential exposure to COPCs in biotic media (i.e., food chain) was considered for certain chemical exposure scenarios. U.S. EPA guidance (1997) specifies that the number of exposure pathways requiring evaluation in an ERA can be reduced to one or a few critical exposure pathways, which reflect the maximum exposures of receptors within the ecosystem, or which constitute exposure pathways to ecological receptors sensitive to a specific contaminant.

8.2.1.4.2 TERRESTRIAL EXPOSURE PATHWAYS

The terrestrial exposure pathways at the Site involve receptor exposure to contaminants originating in Site surface soil. Both direct soil contact exposure pathways and indirect food chain exposure pathways are considered in this ERA. Terrestrial portions of the Site consist of a diversity of cover types that potentially support different types of ecological receptors. Much of the Site is vegetated by forested uplands and to a lesser extent includes successional old field and shrub communities. Successional communities on the Site include the former lagoon areas that have been altered by filling. The former lagoon areas are dominated by opportunistic plant species that quickly colonize recently disturbed areas (i.e., successional old field). The terrestrial habitats potentially support a diversity of plant, invertebrate, bird, reptile, and small mammals indigenous to the area.

Terrestrial soil invertebrates and terrestrial plants are potentially exposed to contaminants in soils primarily through direct contact. Plants are exposed to contaminants in soil primarily through absorption from soil solution by the root (Efroymson et al., 1997a). Plant leaf tissue can also absorb airborne contaminants that have vaporized from soil or from airborne particulates that settle on their surfaces. The air pathways were considered minor for the purposes of conducting the ERA, and as a result, were not evaluated as potential exposure routes for plants. Terrestrial invertebrates are often exposed to contaminants in soil through direct contact and, for some organisms such as earthworms, through soil ingestion. These soil invertebrate exposure pathways were considered in the ERA.

Terrestrial wildlife can be exposed to chemicals in surface soil through direct contact and through secondary exposure via ingestion of prey, which has bioaccumulated COPCs (food chain exposure). Some chemicals can potentially bioaccumulate in the tissues of plants and animals in direct contact with soils. When affected plant and animal tissues are consumed by higher trophic animals, the chemicals become available to exert their toxic effect on the consumer. Because bioaccumulative chemicals can exert a magnified toxic effect on higher trophic species that may not be observed in lower trophic species, potential food chain exposure pathways involving bioaccumulating chemicals were evaluated to reflect the maximum exposures of receptors within the terrestrial ecosystem.

Secondary exposure pathways for terrestrial wildlife include the inhalation of chemicals volatilized from soil, the incidental ingestion of chemicals in soil, and the dermal absorption of chemicals in soil. Inhalation of chemicals was assumed to be negligible at the Site because most of the VOCs, the chemicals most likely to present a risk through inhalation exposure, rapidly volatilize from soil to air, where they are quickly diluted and dispersed. Sample (1997) suggests that VOCs are generally lost from soil before significant exposure can occur. In addition, there is a general lack of species-specific data available to assess the inhalation pathway. As a result, the inhalation exposure pathway was not evaluated in the ERA.

Whereas methods are available to assess dermal exposure to humans, data necessary to estimate dermal exposure are generally lacking for wildlife (U.S. EPA, 1993a). The fur of mammals and feathers of birds reduce the likelihood of significant dermal exposure by limiting contact of skin with contaminated media (Sample, 1997). Therefore, dermal exposure was assumed minor relative to other pathways and was not evaluated in the ERA.

Incidental ingestion of soil may occur while foraging (i.e., ingestion of soil invertebrates or plant roots) or while grooming, which involves cleaning fur and feathers by licking. This pathway may be significant for certain receptors, and was included in the evaluation where appropriate exposure data are available.

8.2.1.4.3 AQUATIC EXPOSURE PATHWAYS

The aquatic environment in the near vicinity of the Site includes Beaverdam Brook, which flows into the larger Otter Kill. A marsh area southwest of the Site is located immediately upstream of where Beaverdam Brook flows into Otter Kill. For the purposes of the ERA, evaluation of Beaverdam Brook includes the Southwest Marsh Area. In addition, an intermittent aquatic environment exists in the Northeast Marsh Area, which is also evaluated in this ERA. In these aquatic environments, sediment-dwelling (benthic) organisms represent a critical trophic level in the food chain. These organisms are in immediate contact with sediments and therefore are exposed to any sediment-bound contaminants. Direct contact exposure to chemicals in the sediment by benthic invertebrates was considered a complete exposure pathway for the aquatic environments in the Site area.

Otter Kill is classified by NYSDEC as a Class "C" surface water, suitable for fish propagation and survival. The portion of Beaverdam Brook that flows through the Site area is classified by NYSDEC as Class "D" surface water. Because of conditions such as intermittency of flow and stream bed condition, Class D waters are not conductive to propagation of a game fishery, but shall be suitable for fish survival. Thus the segments of Beaverdam Brook and Otter Kill in the Site vicinity are likely to support various fish species that may utilize the waters as habitat for feeding and cover and, in the case of Otter Kill, for propagation. The Fine Line Report (1995) concludes that limited data are available regarding the fish species in the Otter Kill and Beaverdam Brook. Fish identified by the NYSDEC during surveys conducted in 1936 are summarized in Table 8.1.

Direct contact exposure to chemicals in the surface water by fish was considered a complete exposure pathway for the Beaverdam Brook/Otter Kill aquatic environment in the Site area.

The presence of surface water in the Northeast Marsh Area is intermittent with periodic conditions of little or no free standing water. As a result, the presence of significant fish populations in the Northeast Marsh Area is expected to be low, if any, and therefore, a fish exposure pathway to chemicals in the intermittent surface waters is probably incomplete. However, for the purpose of the screening assessment it is assumed that this area is capable of supporting fish populations.

Exposure pathways involving food chain interactions are potentially complete in the aquatic environments near the Site. For certain constituents, the potential for bioconcentration from sediment and surface water to aquatic receptors exists. Subsequently, affected fish, for example, may be consumed by aquatic or semi-aquatic wildlife, resulting in the bioaccumulation of these constituents. Thus, food chain pathways in the aquatic environment were considered in the ERA.

8.2.1.4.4 PATHWAY SUMMARY

The terrestrial exposure pathways that warrant consideration as part of the ERA include:

- direct contact with potentially impacted surface soil by invertebrates, plants and wildlife; and
- ingestion of potentially impacted terrestrial invertebrates, plants and/or animals by terrestrial wildlife.

The aquatic exposure pathways that warrant consideration as part of the ERA include:

- direct contact with potentially impacted surface water by fish;
- direct contact with potentially impacted sediments by benthic macroinvertebrates; and
- ingestion of potentially impacted aquatic receptors by aquatic wildlife.

8.2.1.5 ASSESSMENT ENDPOINTS AND MEASUREMENT ENDPOINTS

8.2.1.5.1 ASSESSMENT ENDPOINTS

Assessment endpoints for the SLERA are identified in Table 8.2. The assessment endpoints for soil focus on ecologically functional groups of organisms that are in direct and constant contact with soil or ingest organisms that inhabit soil: microorganisms, soil invertebrates, plants, and wildlife. The assessment endpoint for the surface waters of Beaverdam Brook/Otter Kill and the Northeast Marsh Area is protection of aquatic organisms in the water column. The assessment endpoint for the sediment of Beaverdam Brook and the Northeast Marsh Area is the protection of benthic macroinvertebrates that inhabit sediments.

The selected assessment endpoints are intentionally broad. Once specific constituents of ecological concern (COECs) are identified (i.e., Step 3), specific assessment endpoints will be developed to reflect the mechanisms of ecotoxicity and ecological receptors most sensitive to each COEC. COECs are a subset of the COPCs.

8.2.1.5.2 MEASUREMENT ENDPOINTS

For the screening-level assessment, concentrations of all chemical constituents detected in soil, surface water, and sediment are evaluated. To ensure a level of conservatism commensurate with a screening-level assessment, maximum concentrations will be compared to conservative ecological screening values (ESVs). Table 8.2 identifies the measurement endpoints associated with each of the assessment endpoints identified in Section 8.2.1.5.1.

8.2.2 SCREENING-LEVEL ECOLOGICAL EFFECTS EVALUATION

8.2.2.1 ECOLOGICAL SCREENING VALUES

8.2.2.1.1 OVERVIEW

This step in the SLERA involves identifying appropriate contaminant exposure levels that represent conservative thresholds for adverse ecological effects. In the screening-level assessment, these conservative thresholds are referred to as ecological screening values (ESVs). The use of ESVs, which are ambient concentrations believed to be protective of ecological receptors, is a method for screening chemical constituents detected at a site to identify those that actually have a potential to pose unacceptable risk. Exceedance of a conservative ESV indicates only that ecotoxicological effects are possible. It does not indicate or imply that impact or injury is occurring. Therefore, ESVs should not be used as indicators of ecological risk or defining remedial goals. In order to maintain a conservative approach in the screening-level process, constituents without an applicable screening benchmark were retained as COPCs.

8.2.2.1.2 SURFACE SOIL ESVS

Consistent with guidance from the U.S. EPA Region II Biological Technical Assistance Group (BTAG) (CRA conversation with Christopher Stitt, BTAG, May 28, 1999), COPCs in soil were identified by comparing the maximum concentrations of chemicals detected in surface soils to ESVs developed by Oak Ridge National Laboratory (ORNL) and ESVs available from the Netherlands (Dutch values). Based on comments received from U.S. EPA in July 2005, preliminary remedial goals (PRGs) for ecological endpoints (Efroymson et al., 1997b) and ecological soil screening levels (ECO-SSLs)(USEPA, 2005b) were also considered as ESVs. Consistent with the conservatism of the SLERA, the lowest available benchmark for a given constituent from the sources consulted (ORNL, Dutch values, PRGs, ECO-SSLs) was selected as the ESV. Table 8.3 presents a summary of the soil ESVs for constituents detected in surface soil, and identifies the ESVs applied in the screening assessment. The following Sections briefly describe the ORNL, Dutch soil ESVs, PRGs, and ECO-SSLs.

8.2.2.1.2.1 ORNL SOIL ESVS

ORNL soil ESVs were considered for three functional groups of soil organisms. The first group is soil microorganisms and microbial processes (Will and Suter, 1995). The second group, also reported by Will and Suter (1995), is soil invertebrates. The earthworm is identified as the representative species for soil invertebrates. The third group reported by Efroymson et al., (1997a) is terrestrial plants. For each group, ESVs are based on data provided by toxicity studies in the field or in laboratory settings.

The ORNL ESVs are conservative in practice, and in some cases, lower than background concentrations due to the bioavailability of the chemical applied in the tests. As a result, exceedance of these conservative benchmarks indicates the potential for adverse effects, but does not indicate that such effects are occurring or even likely. The authors of the ORNL benchmarks have assigned a level of confidence to each benchmark based on the number of literature values used in their derivation. There is high confidence in a benchmark based on more than 20 reported literature values, whereas confidence is moderate for benchmarks based on 10 to 20 values, and low for benchmarks based on fewer than 10 reported literature values.

8.2.2.1.2.2 DUTCH SOIL ESVS

The Dutch soil screening values consist of three types of values: target values, intervention values, and intermediary values. Target values are the soil concentrations ultimately aimed for, and essentially represent background or reference conditions. Intervention values are concentrations that potentially pose risk to human and ecological receptors. Soil with a chemical concentration above an intervention value represents a potential for impact or injury to the ecosystem because the no observed effect concentration (NOEC) for vital life-functions of species (i.e., survival, growth, and reproduction) and microbial and enzymatic processes is exceeded (RIVM, 1994). Secondary food chain effects to mammals and birds were also considered in the derivation of the Dutch intervention values. The intermediary value is the arithmetic mean of the target and intervention values. For the screening assessment, intermediary values are used. For those substances without a target value, one-half of the intervention value was used (RIVM, 1994). The intervention value is 50 mg/kg.

For certain classes of compounds, notably phthalates and PAHs, the Netherlands identifies a total group ESV. To calculate a total group concentration, the detected concentrations for each individual constituent within a group were summed for each sample. If the summed concentration for the constituent class was below the total group criterion, then none of the individual constituents within the class was identified as COPCs.

8.2.2.1.2.3 PRELIMINARY REMEDIAL GOALS

Preliminary remediation goals (PRGs) for soil were developed for ecological risk assessments for the ORNL site (Efroymson et al., 1997b). They are based on the lowest ecological benchmark for earthworms, plants, and wildlife. Benchmarks for earthworms and plants are taken from Will and Suter (1995) and Efroymson et al. (1997a), respectively. Wildlife PRGs are based on exposure estimates equivalent to wildlife endpoint-specific and contaminant-specific lowest observed adverse effects levels (LOAELs). Efroymson et al. (1997b) calculate PRGs for six species of wildlife present at the ORNL site: short-tailed shrew (mammalian insectivore), white-footed mouse (mammalian omnivore), red fox (mammalian carnivore), white-tailed deer (mammalian herbivore), American woodcock (avian insectivore), and red-tailed hawk (avian carnivore). The PRG of wildlife is the lowest value of the six species.

Preliminary remediation goals correspond to concentrations that would have minimal effects on individual organisms, which would have minimal effects on ecological populations and communities. Preliminary PRGs represent the minimal and acceptable effects on general ecological assessment endpoints.

8.2.2.1.2.4 ECOLOGICAL SOIL SCREENING LEVELS

Ecological soil screening levels (ECO-SSLs) are risk-based concentrations of common contaminants in soil that are protective of ecological receptors that are in direct contact with soil or receptors that ingest biota that live on or in soil (USEPA, 2005b). The ECO-SSLs were developed by a multi-stakeholder work group consisting of representatives of federal and state regulatory and resource agencies, industry, academe, and the consulting industry, with USEPA as the lead agency. In addition to the expertise within the work group, the ECO-SSLs have undergone external peer review.

Ecological soil screening levels have been developed for four functional groups of organisms: soil invertebrates, plants, avian wildlife, and mammalian wildlife. The ECO-SSLs are based on standardized procedures for literature review, selection of toxicity data, and data evaluation. Screening values for soil invertebrates and plants are derived from toxicity tests, whereas screening values for avian and mammalian wildlife are based on generic food chain models.

The ECO-SSLs are intended to the used in Step 2 (screening assessment) of the 8-step USEPA process for conducting ecological risk assessments. The guidance for developing ECO-SSLs (USEPA, 2005b) clearly states that these benchmarks are intentionally conservative and are not intended to be used as clean-up levels.

8.2.2.1.3 SURFACE WATER

The portion of Beaverdam Brook on and adjacent to the Site and the Northeast Marsh Area is classified by NYSDEC as Class "D" surface waters. The portion of Otter Kill adjacent to the Site boundary is classified by NYSDEC as a Class "C" surface water.

The following available surface water criteria/standards and screening benchmarks were considered in the screening assessment to identify COPCs:

i) New York State Ambient Water Quality Standards (AWQS) for Class C or D waters (NYSDEC, 1998a);

ii) U.S. EPA Recommended Water Quality Criteria (RWQC) for Protection of Aquatic Life (U.S. EPA, 2004e); and

iii) National Oceanic and Atmospheric Administration (NOAA) Screening Quick Reference Table (SquiRT), (Buchman, 1999).

Consistent with direction from U.S. EPA, priority for selection of an ESV was given to the NYSDEC AWQS. If a NYSDEC criterion was not available for a given constituent, then the U.S. EPA RWQC was used. If neither a NYSDEC AWQC nor a USEPA RWQC were available, the benchmark from NOAA SquiRT was used as the ESV. Tables 8.4 and 8.5 present summaries and the identification of the surface water ESVs for constituents detected in Beaverdam Brook/Otter Kill and the Northeast Marsh Area, respectively. The following paragraphs briefly describe each surface water benchmark source.

8.2.2.1.3.1 NEW YORK STATE AWQS

The NYS AWQS are expressed separately for Class "C" and Class "D" waters. These water classifications are adopted by NYSDEC to describe the best usage of each body of water. To be conservative, surface water concentrations in Beaverdam Brook were compared to NYS class C standards, the same as Otter Kill. Concentrations in the Northeast Marsh Area were compared to NYSDEC Class D standards.

8.2.2.1.3.2 U.S. EPA RWQC

RWQCs are derived from the analysis of toxicity test data conducted under controlled laboratory conditions, exposing aquatic organisms to known concentrations of a chemical. The chronic RWQC represents an acceptable chemical concentration in surface water for long-term chemical exposures.

8.2.2.1.3.3 NOAA SQUIRT VALUES

Ecological benchmarks for surface waters identified in the NOAA SquiRT tables are based on U.S. EPA ambient water quality criteria for the protection of aquatic organisms available at the time of publication (Buchman 1999). Benchmarks are generally available for acute (criteria maximum concentration (CMC)) and chronic (criteria continuous concentration (CCC)) effects. Because CCC values are most representative of potential effects due to long-term exposure, CCC values were given priority over CMC values.

8.2.2.1.4 SEDIMENT

If available, NYSDEC values for benthic aquatic life criteria (NYSDEC, 1998b), were used as sediment ESVs. If NYSDEC values were not available, alternative screening values were considered from several other sources. Alternative sources for applicable sediment benchmarks include:

- the U.S. EPA's Sediment Ecotox Threshold Values (U.S. EPA, 1996);
- the NOAA Sediment Quality Guidelines (Long et al., 1995);
- the Ontario Ministry of the Environment's (MOE) Sediment Quality Guidelines (MOE, 1993); and
- the ORNL's Sediment Screening Benchmarks (Jones et al., 1997).

Table 8.6 presents the COPC identification process for sediments. Consistent with the conservatism of the SLERA, the lowest available sediment benchmark from the above sources was selected as the ESV if a NYSDEC value was not available. The following Sections briefly describe each sediment benchmark source.

NYSDEC Sediment Criteria

Following the NYSDEC approach, sediment concentrations were compared to the following sediment criteria categories:

- protection of benthic aquatic life from chronic toxicity; and
- protection of wildlife from toxic effects of bioaccumulation.

For non-polar organic chemicals, NYSDEC derived sediment ESVs using the equilibrium partitioning (EqP) method. This method is based on the theory that toxics in sediments will exert their effect, either toxicity or bioaccumulation, to the extent that the chemical becomes freely bioavailable in the sediment interstitial (pore) water. The fundamental assumption of this approach is that concentrations in pore water and concentrations in sediment exist in equilibrium, and that the concentrations are related by the chemical-specific organic carbon partition coefficient (K_{OC}). Thus, for a particular chemical, the sediment criterion is derived by back-calculating from the chemical's AWQC using the chemical-specific K_{OC}. The application of the EqP approach requires four major assumptions:

1. partitioning of the organic chemical between organic carbon and the pore water is at equilibrium;
2. the sensitivity of the benthic species and the AWQC test species (mostly water column species) are similar;
3. the levels of protection offered by the AWQC are appropriate for benthic species; and
4. chemical exposures are similar regardless of feeding type or habitat.

The NYSDEC sediment criteria for metals are based upon procedures and data developed by the Ontario MOE (Persaud et al., 1992), and NOAA (Long and Morgan, 1990). The NYSDEC sediment criteria for metals are based on two levels of protection as defined by the MOE. The Lowest Effect Level (LEL) is a concentration that can be tolerated by the majority of benthic organisms, but still causes toxicity to a few species. The Severe Effect Level (SEL) is a concentration at which pronounced impacts to sediment-dwelling community can be expected. Both MOE and NOAA developed sediment guidelines that correspond to roughly the same effect levels. The NYSDEC LEL and SEL criteria are a combination of the MOE guideline values and the NOAA sediment criteria. The lowest concentration from either MOE or NOAA, in each of the two effect levels, was selected as the NYSDEC LEL and SEL screening criteria.

U.S. EPA Sediment Ecotox Threshold Values

The U.S. EPA has derived Ecotox Threshold (ET) values for 67 chemicals for use as ESVs (U.S. EPA, 2003). ETs are defined as media-specific contaminant concentrations above which there is sufficient concern regarding adverse ecological effects to warrant further site investigation.

For any one chemical, ETs may be available that represent an U.S. EPA sediment quality criteria (SQC) or an U.S. EPA sediment quality benchmark (SQB), in addition to a sediment Effects Range - Low (ER-L) value.

U.S. EPA has adopted published SQCs for use as ETs. However, SQCs have been derived for only a handful of constituents using the EqP method. For constituents without an available SQC, available SQBs were adopted as ETs. The derivation of SQBs follows the same EqP approach used to derive the SQCs, except that a Tier II surface water value is substituted for the AWQC in the back-calculation. For chemicals without an available SQC or SQB, available ER-L values where adopted as ETs. The ER-L values represent the lower effect level derived by NOAA (Long et al., 1995), described in more detail in the following section.

NOAA Sediment Guideline Values

Under the NOAA approach, the guideline values were determined by assembling compiled toxicity data to determine the lower 10th and 50th percentile concentrations among the data range. The ER-L values are concentrations equivalent to the lower 10th percentile of the screened data, and identify the low end of the range of concentrations at which effects were observed or predicted. These values are intended to represent concentrations above which adverse sub-lethal effects occur or are predicted for sensitive life stages and/or species. The Effects Range-Median (ER-M) values are concentrations equivalent to the 50th percentile concentration in the screened available data. The NOAA guideline values were revised in 1995 using an updated data set by Long et al. (1995).

The ER-L and ER-M values are not to be construed as NOAA standards or criteria. These values were used to rank sites with regard to the potential for adverse ecological effects, assuming that the sites in which the average chemical concentrations exceeded the ER-L and ER-M values would have the highest potential for effects.

To determine the reliability or accuracy of each guideline value, Long et al. (1995) quantified an incidence of adverse effects actually occurring within each effects range, expressed as a percent. For both the ER-L and ER-M to be viewed as reliable, Long determined that the incidence of adverse effects from concentrations below the ER-L should be less than 25 percent, and that the incidence of adverse effects from concentrations above the ER-M be greater than 75 percent. The majority of guidelines derived by Long conform to these criteria; however, mercury and nickel, for example, have relatively weak relationships between their concentrations and the incidence of effects (Long et al. 1995).

Because ER-L and ER-M values were developed for estuarine sediment, they were used as ESVs only if freshwater sediment benchmarks were not available.

Ontario MOE Sediment Guidelines

The Ontario MOE has established sediment guidelines for the LEL and the SEL. The LEL and SEL are based on the long-term effects that contaminants may exert on sediment dwelling organisms. The LEL indicates a level of contamination that has no effect on the majority of sediment dwelling organisms. The SEL identifies a concentration at which adverse effects to most sediment dwelling organisms are likely to occur.

The MOE guidelines are based on the Screening Level Concentration (SLC) approach. The SLC is an effects-based approach applicable primarily to benthic organisms. Both the LEL and SEL are derived using field data on the co-occurrence in sediments of benthic infaunal species and different concentrations of contaminants (MOE, 1993). The SLC is an estimate of the highest concentration of a contaminant that can be tolerated by a specific proportion of benthic species. In general, the LEL values are concentrations equivalent to the lower 5th percentile of the SLC data, and are indicative of the low end of the range of concentrations at which effects were observed or predicted. The SEL values are concentrations equivalent to the upper 95th percentile of the SLC data, and are indicative of the upper end of the range of concentrations at which effects were observed or predicted. The LEL guidelines compare reasonably well with the ER-L guidelines developed by NOAA (Long et al., 1995).

ORNL Sediment Screening Benchmarks

ORNL has derived a set of sediment screening benchmarks for use in ecological risk assessment. The ORNL sediment screening benchmarks were derived for non-ionic organic chemicals using the EqP approach. In many cases, Tier II or secondary water quality benchmarks derived by the ORNL have been used in the EqP approach to estimate the sediment screening benchmarks.

Organic Carbon Normalized Sediment Criteria

Some of the applicable sediment benchmarks, such as the NYSDEC sediment criteria, MOE SELs, ORNL sediment benchmarks, and the U.S. EPA SQC and SQB criteria, can be adjusted for TOC concentration of Site area sediments. Two sediment samples were analyzed for TOC in 1995. TOC concentrations were 34,300 mg/kg (3 percent) for sediment sample SDII-12 collected from the southwest marsh in Beaverdam Brook and 132,500 mg/kg (13 percent) for sediment sample SDII-2 collected from the Northeast Marsh Area. To maintain a level of conservatism commensurate with a screening-level assessment, ESVs were not adjusted for TOC. For any organic constituents carried forward to Step 3, ESVs will be adjusted.

8.3 STEP 2 - SCREENING-LEVEL EXPOSURE ESTIMATE AND RISK CALCULATION

8.3.1 SCREENING LEVEL EXPOSURE ESTIMATE

8.3.1.1 TERRESTRIAL EXPOSURE

The presence of larger transient and/or migratory animals, such as deer or fox, is possible at the Site, although their frequency on Site would be less than animals with smaller home ranges. The Site is not likely to fully support larger animals due to their transient nature and because the size of the Site relative to their home range requirements is relatively small. Larger animal species typically require large habitat areas in order to fulfill the requirements of their life-cycles. However, in keeping with the conservative nature of the SLERA, it is assumed that all potentially exposed species spend 100 percent of their time on Site, and receive constant exposure to the maximum concentrations of Site-related constituents. This assumption of 100 percent area use and exposure is assumed in the screening process to improve confidence and reduce uncertainty when eliminating exposure pathways or chemicals during the screening step of the ERA process.

8.3.1.2 AQUATIC EXPOSURE

The majority of aquatic wildlife species require large home ranges to satisfy their life-cycle requirements in relation to the size of the aquatic environments being evaluated. For example, the area in which piscivores forage for fish is likely to be significantly larger than the Site area along Beaverdam Brook and Otter Kill, or the Northeast Marsh Area. However, in keeping with the conservative nature of the screening process, it is assumed that piscivorous wildlife will obtain 100 percent of their diet from the areas of the Site with maximum concentrations of Site-related constituents.

Surface water in the Northeast Marsh Area is intermittent in nature, observed at times to have no free standing water. However, for the purpose of the screening assessment it is assumed that this area is capable of supporting fish populations.

8.3.2 SCREENING LEVEL RISK CALCULATION

8.3.2.1 OVERVIEW

Screening-level risk was estimated using the hazard quotient approach, which compares maximum detected concentrations in each media with an appropriately conservative ESV. Use of maximum concentrations is consistent with applicable regulatory guidance and the conservatism of the screening step. For each detected chemical constituent in each environmental medium, a hazard quotient was expressed as the ratio of the maximum concentration to the ESV as follows:

$$HQ = \frac{EEC_{Max}}{ESV}$$

where:

HQ = hazard quotient (unitless);
EEC_{Max} = maximum estimated environmental concentration at the Site (mg/kg); and
ESV = ecological screening value (mg/kg).

Hazard quotients are used to identify COPCs in the ERA. An HQ lower than or very near to one indicates that the chemical is unlikely to cause adverse ecological effects, whereas an HQ greater than one indicates that there is a potential for the constituent to cause ecological effects, although further analysis is required to make a meaningful determination. Consequently, constituents with an HQ less than or close to one were eliminated from further evaluation, whereas those constituents with an HQ greater than one were retained as COPCs and carried forward to Step 3 for further evaluation.

8.3.2.2 SURFACE SOIL

Table 8.7 presents the COPC identification process for surface soils. All of the identified VOCs and SVOCs detected in surface soil have HQs less than one with the exception of benzoic acid, which does not have an ESV. Benzoic acid will be carried forward to Step 3 for further evaluation. All other VOCs and SVOCs can be eliminated from further evaluation. Ten TICs, were detected in one or more soil samples. The detected TICs are 1,2-propanedione, 1-phenyl; 1-proponone, 1-(3-pyridinyl); 2-butyl pyridine; 2-ethyl pyridine; bipyridine isomer; two chlorothioxanthenone isomers; dichlorobiphenyl isomer; methyl ester benzoic acid; and methyl phenanthrene isomer. Because these

TICs, which are all SVOCs, do not have ESVs, they will be carried forward to Step 3 for further evaluation.

None of the individual pesticides or groups of pesticides (BHC group, DDT/DDE/DDD group, or DRINs group) have HQs that exceed 1. Therefore, all pesticides in soil are eliminated from further consideration.

The PCB Aroclor–1254 was detected in five of 15 samples, with the maximum estimated concentration (9,200 μg/kg) resulting in a HQ of 25. Consequently, Aroclor-1254 is retained as a COPC and will be carried forward to Step 3.

Several inorganic constituents have HQs that exceed one. These inorganics include aluminum (436), antimony (46), arsenic (4.8), cadmium (12), chromium (152), cobalt (1.5), copper (2.2), iron (212), lead (18), manganese (17), mercury (45,882), nickel (1.4), selenium (4.7), thallium (2.1), vanadium (21), and zinc (70). These 16 inorganics are retained as COPCs and will be carried forward to Step 3 for further evaluation. Four other inorganics - calcium, magnesium, potassium, and sodium - do not have ESVs and will also be carried forward to Step 3.

8.3.2.3 SURFACE WATER

8.3.2.3.1 BEAVERDAM BROOK/OTTER KILL

Table 8.8 presents the COPC identification process for the surface waters of Beaverdam Brook/Otter Kill. Chemical constituents detected in Beaverdam Brook/Otter Kill that have an HQ greater than one include the SVOCs bis(2-ethylhexyl)phthalate (533) and di-n-octylphthalate (37). These two phthalates are retained as COPCs and will undergo additional evaluation in Step 3. It should be noted that these phthalates likely are not Site-related as they were detected in similar concentrations in upgradient samples. In addition, these compounds were not detected in surface water samples collected in 1995. The constituents have been carried forward due to the conservative nature of the assessment.

Four SVOC TICs – an ethylmethlbenzene isomer, two ethylmethylcyclohexane isomers, and 4-hydroxy-4-methyl-2-pentanone – were detected in one or more surface water samples. Because these four TICs do not have ESVs, they will also be carried forward for further evaluation.

Three inorganics – aluminum (6.3), cyanide (10), and iron (3.8) have HQs that exceed one. Six additional inorganics (barium, calcium, magnesium, manganese, potassium, and sodium) do not have ESVs. These nine inorganics are retained as COPCs.

8.3.2.3.2 NORTHEAST MARSH AREA

Table 8.9 presents the COPC identification process for the surface water of the Northeast Marsh Area. Acetone was detected in one of three samples, but will be carried forward because an ESV was not available in the sources consulted for the screening assessment. Iron is the only detected chemical in the Northeast Marsh Area surface water with an HQ greater than one. Iron will be carried forward to Step 3 for further evaluation.

Two SVOC TICs – an ethylmethylbenzene isomer and 4-hydroxy-4-methyl-2-pentanone; and six inorganics – barium, calcium, magnesium, manganese, potassium, and sodium do not have ESVs. These eight constituents will also be carried forward to Step 3 for further evaluation.

8.3.2.4 SEDIMENT

8.3.2.4.1 BEAVERDAM BROOK/OTTER KILL

Table 8.10 presents the COPC identification process for sediments of Beaverdam Brook/Otter Kill. Organic constituents detected in Beaverdam Brook/Otter Kill that have an HQ greater than one include acetone (43), benzo(a)pyrene (3.4), benzo(g,h,i)perylene (1.3), benzo(k)fluoranthene (1.9), dibenz(a,h)anthracene (2.2), and indeno(1,2,3-cd)pyrene (1.1). Total PAHs also exceed their group ESV. Those individual PAHs not eliminated will be carried to Step 3. An ESV is not available for benzo(b)fluoranthene. Therefore, benzo(b)fluoranthene will also be carried forward to Step 3.

Several inorganic constituents have HQs that exceed one. These inorganics are antimony (2.6), arsenic (1.8), cadmium (6.7), copper (43), iron (2.2), lead (1.5), manganese (4.0), nickel (2.7), and zinc (1.4). Due to the unavailability of applicable ESVs, aluminum, barium, beryllium, calcium, cyanide, magnesium, potassium, selenium, sodium, thallium, and vanadium are also identified as COPCs in the sediments of Beaverdam Brook/Otter Kill, and will be carried to Step 3 for further evaluation.

8.3.2.4.2 NORTHEAST MARSH AREA

Table 8.11 presents the COPC identification process for sediments of the Northeast Marsh Area. Organic constituents detected in the Northeast Marsh Area that have an HQ greater than one are acetone (8.2), 4-methylphenol (5.3), and phenol (2.3). These three organic constituents are retained as COPCs and will be carried forward to Step 3 for further evaluation.

Several inorganic constituents will also be carried forward to Step 3. Antimony (128), cadmium (38), copper (5.7), lead (2.3), mercury (2.2), nickel (1.3), and silver (67) have HQs greater than 1. Ecological screening values are not available for aluminum, barium, calcium, magnesium, potassium, selenium, sodium, and vanadium. Lead and zinc have HQs of 1.03 and 1.02, respectively. Because these HQs are essentially equal to one, lead and zinc are not identified as COPCs.

PAHs detected in the sediment of the Northeast Marsh Area were excluded from the screening assessment because their presence is considered to be anthropogenic and unrelated to the Site.[2] Therefore, no PAHs are identified as COPCs in the Northeast Marsh Area.

8.3.2.5 IDENTIFICATION OF BIOACCUMULATIVE CHEMICALS OF CONCERN (BCCS)

Chemical constituents that have a significant potential to bioaccumulate in biological tissues were identified for the ecological exposure pathways that involve food chain interactions. The U.S. EPA (1995b) has identified BCCs as part of their rulemaking on water quality guidance for the Great Lakes System. For the screening assessment, detected constituents that are identified as BCCs by U.S. EPA were considered as potential COPCS for the food chain exposure pathways.

For surface soils, the ESVs for organic parameters used in the screening assessment were taken almost exclusively from the Dutch guidance, which generally provided the most conservative ESVs. The Dutch methodology for deriving ecotoxicological criteria takes into consideration potential food chain effects where secondary toxicity to birds, mammals, and worms may occur from exposure to bioaccumulative chemicals. Therefore, detected organic chemicals in surface soil that were eliminated based on an

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[2] The elimination of PAHs in the sediments of the Northeast Marsh for evaluation in the ERA has been discussed with and agreed to by USEPA.

HQ of less than one were not re-evaluated for potential bioaccumulative effects. Detected organic chemicals in surface soil that have an HQ greater than one were included in the BCC identification process. Detected inorganic chemicals in soil were also included in the same BCC identification process.

Barium and cyanide were eliminated from further evaluation in the screening step. The ESVs for barium and cyanide were the PRG for wildlife and Dutch intermediary value, respectively. Because the ESVs for these two constituents consider chain food effects, barium and cyanide will not be re-evaluated for potential bioaccumulative effects.

Mercury in the form of methyl mercury has the potential to biomagnify[2] in the food chain. However, with the exception of surface soils, only total mercury was analyzed in media samples collected during the Site investigations. Mercury was not detected in surface water from Beaverdam Brook, Otter Kill, or the Northeast Marsh Area. The highest detected concentration of mercury from all sediment samples collected was 0.27/0.33 mg/kg from sample SWII-2 and its duplicate taken from the Northeast Marsh Area. Mercury was considered a BCC in the terrestrial food chain, having been detected in surface soils at concentrations in excess of its ESV.

DDE was detected in one Northeast Marsh Area sediment sample at a concentration of 11 μg/kg. The detection of DDE was qualified as estimated and is uncertain considering that DDE was not detected in a duplicate sample. DDE was not detected in any other aquatic media and was not detected in surface soil samples. It is unlikely that the DDE detected in the Northeast Marsh Area is related to the Site or Site activities because the Site is downstream of the Northeast Marsh Area such that DDE bound to soil particles would not be transported from the Site to this location via surface water runoff. Agricultural farmland is common in the areas surrounding the Site and the Northeast Marsh Area, and may be a historical source of pesticides present in some aquatic ecosystems. Consequently, DDE was not evaluated as a BCC for the Northeast Marsh Area aquatic ecosystem.

No BCCs were identified for evaluation in the Beaverdam Brook/Otter Kill or the Northeast Marsh Area aquatic ecosystems.

To avoid incorrectly dismissing risk to consumers of plants, invertebrates, fish, and small mammals, all chemicals identified as COPCs in Step 3 of the screening process (Section 8.4) were evaluated in the baseline assessment using food chain models (Section 8.5), even if the constituent was not classified as a BCC. The COPCs evaluated using food chain models in Section 8.5 are PCBs and mercury in soil;

bis(2-ethylhexyl)phthalate, di-n-octylphthalate, and cyanide in surface water; and copper, selenium, and thallium in sediment.

8.3.3 UNCERTAINTY IN THE SCREENING LEVEL ASSESSMENT

There is a high level of uncertainty inherent in the screening process. However, to avoid incorrectly dismissing the potential for risk to ecological receptors, uncertainty is strongly biased toward overestimating risk and many conservative assumptions are made.

For example, conservative ESVs accepted by the risk assessment community are used to assess the potential for risk. For those constituents with multiple ESVs, the lowest (e.g., most conservative) ESV was selected. Due to the high degree of conservatism in the screening process, it can be concluded with a high level of certainty that concentrations of chemical constituents below their ESVs do not pose risk to ecological receptors. It is, however, uncertain if concentrations above conservative ESVs actually pose an unacceptable potential for risk.

The source, or sources of chemical constituents in on-Site soils and the surface water and sediments of Beaverdam Brook, Otter Kill, and Northeast Marsh Area is uncertain. Metals and PAHs, the primary COPCs, typically come from a multitude of sources, both anthropogenic and natural. The sources of the metals and PAHs evaluated in the screening assessment is uncertain.

Several of the constituents detected in soil, surface water, and sediment do not have ESVs. Therefore, it is uncertain whether or not these constituents pose risk to ecological receptors.

The collection dates of samples evaluated in the SLERA range from 1995 through 2003. It is uncertain whether or not fate and transport processes have attenuated the concentrations of the older samples.

Many of the soil samples evaluated in the screening assessment were collected from areas that have been altered over the years by human activity. Due to the nature of the site investigations conducted over the years, most samples were collected in, or near, suspected source areas for chemical constituents. Utilization of the sample locations by ecological receptors is uncertain. As with all other assumptions considered in this SLERA, a complete exposure pathway was assumed for all sample locations.

Finally, the screening assessment included evaluation of surface water and sediment of the Northeast Marsh Area. Because the Northeast Marsh Area is upgradient of the lagoons, it is uncertain if the COPCs identified in the screening assessment are from the Site or other sources.

8.3.4 SCIENTIFIC/MANAGEMENT DECISION POINT

A decision is made at the end of the screening process on whether it is appropriate to stop the assessment after the screening assessment or proceed into a more comprehensive baseline risk assessment process (U.S. EPA, 1997). To end the evaluation after the screening assessment, there must be adequate information to conclude that ecological risks are negligible and, therefore, no need for remediation on the basis of ecological risk. Based on the screening-level assessment, the majority of Site-related constituents do not occur at concentrations sufficient to pose unacceptable risk to ecological receptors. However, a continuance to a baseline risk assessment process is considered necessary due to the presence of several COPCs in soil, surface water and sediment and two BCCs (Aroclor-1254 and mercury) in the terrestrial ecosystem. The information available at this stage of the ERA process is inadequate to determine whether or not ecological risk is occurring based on exposure to these COPCs and BCCs, and therefore, continuation of the ERA process is warranted.

8.4 BASELINE ECOLOGICAL RISK ASSESSMENT PROBLEM FORMULATION

The baseline risk assessment problem formulation serves to refine the results of the screening assessment, which involved the application of conservative assumptions to estimate potential ecological risk. As part of the baseline problem formulation process, COPCs and exposure pathways and ecosystems potentially at risk are reviewed and refined using Site-specific information, assessment and measurement endpoints are further defined, and a conceptual Site model is derived.

8.4.1 REFINEMENT OF COPCS

8.4.1.1 OVERVIEW

The screening-level assessment conducted in Steps 1 and 2 identified several organic and inorganic chemical constituents as COPCs because (1) their maximum detected concentration exceeds their ESV or (2) an ESV is not available. Step 3 of the U.S. EPA 8-step process provides an opportunity to further screen those constituents that have little or no potential to pose unacceptable risk to ecological receptors. Justification for removal of constituents as COPCs in Step 3 includes elimination of essential nutrients, use of ESVs for surrogate chemicals for those COPCs that were carried forward due to the unavailability of an ESV, consideration of background concentrations for ubiquitous and naturally occurring constituents (e.g., metals), frequency of detection, alternative ESVs, and alternative estimates of exposure concentrations (e.g., 95% upper confidence limit rather than maximum concentration). Evaluation of the COPCS carried forward is evaluated with regard to these factors in the following sections.

A discussion of each constituent identified as a COPC in the screening assessment by medium and location is provided below. Where constituents are eliminated in the refinement process, justification is provided. Constituents that are not eliminated will be carried forward to the baseline assessment as constituents of ecological concern (COECs) for further evaluation.

8.4.1.2 SOIL

8.4.1.2.1 SVOCS

Benzoic acid and ten TIC SVOCs were carried forward because ESVs are not available. In Step 3, ESVs for surrogate chemicals can be used to screen constituents without ESVs provided the chemical structure, mechanisms of fate and transport, and ecotoxicological effects are similar to the COPC. In discussions with U.S. EPA during preparation of the human health risk assessment, surrogates were identified for those TICs without toxicological benchmarks. To provide consistency with the human health risk assessment, surrogates identified for the human health risk assessment were also used as surrogates for the ecological risk assessment, provided that an ecotoxicological benchmark was available. Based on recent discussions with U.S. EPA, those TICs and other SVOCs that lack an ESV and do not have a surrogate with an ecotoxicological benchmark are eliminated from further evaluation due to a high level of uncertainty. This uncertainty is discussed in Section 8.7.4.

For the human health risk assessment, diethyl phthalate was used as a surrogate for methyl ester benzoic acid. The U.S. EPA Region V ESL for diethyl phthalate is 24,800 µg/kg. The maximum concentration detected in soil (5,000 µg/kg) is well below this value.

Diphenylamine was identified as a surrogate for the dichlorobiphenyl isomer. The U.S. EPA, Region V identifies an ESL of 1,010 µg/kg for diphenylamine in soil. The maximum detected concentration of the dichlorobiphenyl isomer (20,000 µg/kg) exceeds the ESL of 1,010 µg/kg. Accordingly, the dichlorobiphenyl isomer is carried forward as a COEC.

For the human health risk assessment, benzaldehyde was identified as a surrogate for 1,2-propanedione, 1-phenyl; biphenol A as a surrogate for the methyl phenanthrene isomer; and terbutryn as a surrogate for 2 butyl pyridine, 2-ethenyl pyridine, and the bipyridine isomer. Ecotoxicological benchmarks are not available for any of these surrogates. Consequently, no further evaluation of 1,2-propanedione, 1-phenyl, the methyl phenanthrene isomer, 2 butyl pyridine, 2-ethenyl pyridine, and the bipyridine isomer was conducted. Appropriate surrogates were not identified for benzoic acid, 1,-propanone, 1-(3-pyridnyl), and the two chlorothioxanthenone isomers. No further evaluation was conducted for these three SVOCs.

8.4.1.2.2 PCBS

Aroclor-1254 was detected in five of 15 soil samples. Concentrations in two of the samples (mean concentration of 6,850 µg/kg for duplicate samples at SSII-6 and 9,200 µg/kg at TP-41, both from Lagoon 6) exceed the ESV for total PCBs. The mean and 95% UCL concentrations (assuming detects have a concentration equal to one-half the detection limit) of Aroclor-1254 for all samples are 1,115 µg/kg and 2,840 µg/kg, respectively. These estimated concentrations are also above the ESV of 510 μg/kg. Accordingly, Aroclor-1254 will be carried forward as a COECs.

8.4.1.2.3 INORGANICS

Twenty inorganic constituents were carried forward from the screening-level assessment. Four of these constituents - calcium, magnesium, potassium, and sodium -can be eliminated from further consideration because they are essential nutrients required by plants and animals for growth, reproduction, and maintenance.

The remaining 16 inorganic constituents are metals that may be derived from a variety of anthropogenic and natural sources. To determine if these inorganic constituents were derived from anthropogenic or natural sources, CRA relied on data obtained during the May 2003 soil sampling event that focused on inorganic parameters. A two-step approach is presented below to determine if concentrations of inorganics are consistent with anthropogenic or natural sources:

- elimination of the inorganic compound if the maximum Site-related concentration is less than the maximum background concentration; and
- elimination of the inorganic compound if it was agreed that, based on further statistical analyses, the inorganic constituent can be considered consistent with background.

As presented in Table 8.12, four inorganics were eliminated from further consideration because their maximum on-site concentrations were less than the maximum background concentration. These metals are aluminum, iron, nickel, and thallium.

For the remaining 12 inorganics, further statistical analyses, including outlier analyses, were conducted. In addition to statistical analyses, historical manufacturing data were relied upon to determine the inorganics that were considered Site-related constituents of concern.

Based on the extensive review of inorganic data and discussions with U.S. EPA during preparation of the human risk assessment, CRA identified those inorganic constituents with on-site soil concentrations that were considered statistically consistent with site-specific background concentrations. Of the remaining 12 inorganics, only copper and mercury may be considered Site-related based on a review of the processes conducted at the Nepera Harriman Facility. This information, along with statistical analyses, supported the determination that at least eight inorganics could be eliminated from further consideration. These inorganic constituents are arsenic, chromium, cobalt, lead, manganese, selenium, vanadium, and zinc. Based on concurrence by U.S. EPA and to maintain consistency with the human health risk assessment, these eight inorganic constituents are eliminated from further evaluation. The remaining four metals carried through the process are antimony, cadmium, and the two Site-related constituents – copper, and mercury.

The screening assessment considered only the maximum concentration of metals detected in surface soil. Measures of central tendency of the on-Site concentrations (mean and 95% UCL of the mean) are more representative of the exposure of terrestrial receptors to Site-related constituents than the maximum concentration.

Table 8.13 presents the maximum, mean, and upper 95 percent UCL for the four remaining metals. The mean concentration of copper (44 mg/kg) is below its ESV, whereas the upper 95 percent UCL (52 mg/kg) is essentially equal to the ESV (50 mg/kg). Given the proximity of the 95 percent UCL to its conservative ESV, copper can reasonably be eliminated as a COPC.

At this stage, three metals (antimony, cadmium, and mercury) remain as COPCs. The ecological benchmarks selected as ESVs are the most conservative (e.g., lowest) concentrations in the sources consulted in the screening assessment. In the problem formulation step, alternative ESVs can be used to evaluate potential risk to ecological receptors posed by concentrations of metals in soil. Crommentuijn et al. (1997) have developed maximum permissible concentrations (MPCs), which take into account both background concentrations and potential effects to ecological receptors, including effects to upper trophic level receptors. To further refine the COPCs that will be carried forward to the BERA, the upper 95 percent UCL concentrations of the remaining three metals were compared to MPCs. The results of this comparison are summarized in Table 8.14. The upper 95 percent UCL concentration of antimony (3.1 mg/kg) is below its MPC (3.5 mg/kg). Accordingly, antimony can be eliminated as a COPC, and will not be carried forward to the BERA.

For cadmium, only a single sample (TP-41 collected in 1991) had a concentration outside the range of background concentrations. The upper 95 percent UCL (1.8 mg/kg) is slightly greater than its MPC (1.6 mg/kg). Both the MCP and ESV, which is the lowest of the benchmarks considered in the screening assessment, are within the range of site-specific background concentrations (0.23 mg/kg to 3.8 mg/kg). The ESV is very close to the lowest detected background concentration. Therefore, neither benchmark is appropriate for Site conditions. As a site-specific ESV, the benchmark identified in Table 8.3 with the lowest concentration above the maximum background concentration was used. This benchmark is 4.0 mg/kg, the ORNL benchmark for plants. The maximum concentration (4.4 mg/kg) is slightly greater than this site-specific ESV; the 95 percent UCL is below this ESV. Accordingly, cadmium is eliminated as a COPC.

The upper 95 percent UCL concentration of mercury (3.9 mg/kg) exceeds its MPC (2.2 mg/kg). Furthermore, all benchmarks considered in the screening assessment (Table 8.3) are outside the range of site-specific background concentrations. Therefore mercury will be carried forward to the BERA.

Table 8.15 and Table 8.16 provide a summary of the organic and inorganic constituents, respectively, in soil retained and eliminated in this problem formulation step. For those constituents eliminated, the reason for elimination is identified.

8.4.1.3 SURFACE WATER

8.4.1.3.1 BEAVERDAM BROOK/OTTER KILL

8.4.1.3.1.1 VOCS

The screening assessment carried acetone forward because an ESV was not available in the sources consulted. Several ecological benchmarks are available in sources other than those consulted for the screening assessment. Benchmarks range from a low of 1,500 µg/L (U.S. EPA Tier II secondary chronic value) to a high of 508,000 µg/L (ORNL lower chronic value for fish). The maximum concentration (3.0 µg/L) is three orders of magnitude lower than the lowest benchmark. Consequently, acetone is eliminated as a COPC.

8.4.1.3.1.2 SVOCS

The phthalates, bis(2-ethylhexyl)phthalate and di-n-octylphthalate, were carried forward because their maximum (and only) detected concentrations exceed their respective ESVs. An alternative ESV (U.S. EPA Region III) of 30 μg/L is available for bis(2-ethylhexyl)phthalate. The maximum concentration (320 μg/L) exceeds the alternative ESV. U.S. EPA Region III does not have an ESV for di-n-octylphthalate. Accordingly, both bis(2-ethylhexyl)phthalate and di-n-octylphthalate are carried forward as COECs.

Four TIC SVOCs were carried forward to Step 3 because they lack ESVs. These TICs are ethylmethylbenzene isomer, two ethylmethycyclohexane isomers, and 4-hydroxyl-2-methyl-2-pentanone. As was done with soil, surrogates identified for the human health risk assessment were used as surrogates for the TIC SVOCs in surface water. If available, ecotoxicological benchmarks for the surrogates were used as ESVs for the TICs. If an ecotoxicological benchmark was not available, no further evaluation was conducted.

Styrene was identified as a surrogate for the ethylmethylbenzene isomer. Two ESVs are available for styrene. The U.S. EPA Region V identifies an ESL of 32 µg/L and U.S. EPA Region VI identifies a screening value of 2,500 µg/L. The maximum concentration of the ethylmethylbenzene isomer (200 µg/L) is above the ESL, but well below the Region VI benchmark of 2,500 µg/L. Because the maximum concentration is below the U.S. EPA Region VI screening benchmark, the ethylmethylbenzene isomer is eliminated as a COPC.

Methyl methacrylate was identified as a surrogate of 4-hydroxyl-2-methyl-2-pentanone. The U.S. EPA Region V identifies an ESL of 2,800 µg/L, whereas the Region VI identifies a screening benchmark of 232,400 µg/L for methyl methacrylate. Because the maximum concentration of 4-hydroxyl-2-methyl-2-pentanone (8 µg/L) is several orders of magnitude less than both benchmarks, it can be eliminated as a COPC.

Cyclohexylamine was identified as a surrogate for the two ethylmethycyclohexane isomers. Ecotoxicological benchmarks are not available for cyclohexylamine. Therefore, further evaluation of the ethylmethycyclohexane isomers was not conducted.

Table 8.17 summarizes the refinement of COECs for the surface water of Beaverdam Brook/Otter Kill.

8.4.1.3.1.3 INORGANICS

The screening assessment identified aluminum, barium, calcium, cyanide, iron, magnesium, manganese, potassium, and sodium as COPCs. Calcium, magnesium, potassium, and sodium can be eliminated as because they are essential nutrients.

Barium and manganese were carried forward because ESVs were not available in the sources consulted for the screening assessment. Crommentuijn (1997) identifies a MPC of 220 µg/L for barium. Because the maximum concentration (26.2) is below the MPC, barium can be eliminated as a COPC.

For manganese, the range of concentrations in Beaverdam Brook/Otter Kill (66 µg/L to 607 µg/L) is very similar to the range of site-specific background concentrations (51 µg/L to 572 µg/L), with the maximum Site-related concentration being somewhat greater than the maximum background concentration. ORNL (Suter and Tsao, 1996) identifies a lowest chronic value (LCV) for daphnids and LCV for fish of 1,100 µg/L and 1,780 µg/L, respectively. Both of these benchmarks are outside the range of site-specific background concentrations and are, therefore, appropriate for the Site. Because the maximum detected concentration of manganese (607 µg/L) in Beaverdam Brook/Otter Kill is below the lowest of the ORNL benchmarks, manganese can be eliminated as a COPC.

Aluminum, cyanide, and iron were identified as COPCs because their maximum concentrations exceeded their respective ESVs. As was the case for manganese, the range of concentrations for both aluminum and iron in Beaverdam Brook/Otter Kill are very similar to site-specific background concentrations. For aluminum, concentrations in Beaverdam Brook/Otter Kill range from 65 µg/L to 629 µg/L. Background concentrations range from 25 µg/L to 591 µg/L. The ESV used in the screening assessment (100 µg/L) is within the range of background concentrations, indicating that the ESV is not appropriate for site-specific conditions. ORNL benchmarks are available for the LCV for daphnids (1,900 µg/L) and LCV for fish (3,290 µg/L), which are both outside the range of site-specific background concentrations. Because the maximum detected concentration of aluminum is well below both benchmarks, aluminum can be eliminated as a COPC.

Concentrations of iron in Beaverdam Brook/Otter Kill range from 152 µg/L to 1,050 µg/L, with a mean concentration of 596 µg/L. Site-specific background concentrations range from 237 µg/L to 977 µg/L, with a mean of 672 µg/L, which is higher than the mean concentration in Beaverdam Brook/Otter Kill (596 µg/L). The ESV used in the screening assessment (100 µg/L) is approximately two times lower than the lowest background concentration. Consequently, the ESV of 100 µg/L is not appropriate for site-specific conditions. Several sources identify 1,000 µg/L as a screening benchmark for iron. These sources include U.S. EPA Regions IV and VI, National RWQC, and the U.S. EPA Office of Solid Waste and Emergency Response (OSWER). The maximum concentration of iron (1,050 µg/L) in Beaverdam Brook/Otter Kill is slightly greater than the alternative ESV of 1,000 µg/L (HQ=1.02). No other detected concentrations of iron exceeded 1,000 µg/L. Based on the similarity to the range of background concentrations and the very slight exceedance of the alternative

ESV by a single sample, iron can reasonably be eliminated as a COPC in the surface water of Beaverdam Brook/Otter Kill.

Cyanide was detected in only one of ten samples, at a concentration of 54.4 µg/L. Cyanide was not detected (detection limit of 10 µg/L) in a duplicate of this sample. The single detected concentration is outside the range of site-specific background concentrations (maximum of 10 µg/L) and greater than all alternative screening benchmarks (5.2 µg/L to 30 µg/L). Accordingly, cyanide is identified as a COEC and will be carried forward to the BERA.

8.4.1.3.2 NORTHEAST MARSH AREA

Based on discussions with U.S. EPA, the Northeast Marsh Area is eliminated as an assessment area in this problem formulation step. The Northeast Marsh is upgradient of any surface or groundwater flow from source areas related to past Site operations and disposal practices. Therefore, transport of contaminants from the Site to the Northeast Marsh Area in either the dissolved or particulate phase in surface water runoff or dissolved in groundwater is unlikely. Furthermore, the Northeast Marsh does not receive groundwater discharge from the Site. Therefore, no complete migration pathways exist between the Site and the Northeast Marsh.

There are other likely sources of COPCs identified in the screening assessment, such as PAHs from County Highway 4 that runs adjacent to the Northeast Marsh. Concentrations of several ubiquitous constituents, such as PAHs, are higher in the sediment/soil of the Northeast Marsh Area than on or immediately adjacent to the Site, whereas Site-related constituents, such as pyridine, have not been detected in the Northeast Marsh Area. Accordingly, the COPCs identified in the screening assessment for both the surface water and sediments of the Northeast Marsh are eliminated from further evaluation in the BERA.

8.4.1.4 SEDIMENTS

8.4.1.4.1 BEAVERDAM BROOK/OTTER KILL

8.4.1.4.1.1 VOCS

Acetone was identified as a COPC for the sediments of Beaverdam Brook/Otter Kill because the maximum detected concentration exceeds the ESV. The ESV used for acetone was an EqP value identified by ORNL. Because this ESV is based on an assumed TOC content of one percent, the ESV can be adjusted for the three percent TOC in Beaverdam Brook/Otter Kill. This adjustment is based on sample SDII-12 collected from the Southwest Marsh in Beaverdam Brook in 1995. The adjustment for TOC produces a revised ESV of 81 μg/kg. The maximum concentration of (370 μg/L) is still greater than the revised ESV (HQ of 4.6). No alternative ESVs were identified in the sources used in the screening assessment. Acetone is a common laboratory contaminant. Based on discussions with U.S. EPA, acetone is eliminated as a COPC in the sediments of Beaverdam Brook and Otter Kill.

8.4.1.4.1.2 SVOCS

The screening assessment identified bis(2-ethylhexyl)phthalate and several PAHs as COPCs. The ESV for bis(2-ethylhexyl)phthalate used in the screening is the NYSDEC chronic value for benthic aquatic life toxicity (199.5 μg/kg). When adjusted for three percent TOC, the ESV is adjusted to 599 μg/kg. The maximum concentration of 340 μg/kg is less than the adjusted ESV. Based on the adjusted ESV, bis(2-ethylhexyl)phthalate can be eliminated as a COPC.

PAHs identified as COPCs include benzo(a)pyrene, benzo(b)fluoranthene, benzo(g,h,i)perylene, benzo(k)fluoranthene, dibenzo(a,h)anthracene, and indeno(1,2,3-cd)pyrene. The maximum concentration of total PAHs (5,120 μg/kg) also exceeds the ESV for total PAHs (4,000 μg/kg). No PAHs were detected in background samples.

Benzo(a)pyrene was detected in two samples at estimated concentrations of 120 μg/kg (Beaverdam Brook) and 470 μg/kg (Southwest Marsh Area). Only the maximum concentration exceeds the ESV. The maximum concentration slightly exceeds the Ontario LEL (HQ=1.3), NOAA ER-L (HQ=1.1), and the ORNL ESV (HQ=1.1 adjusted for three percent TOC). The maximum concentration is below the ESVs for U.S. EPA (1,290 μg/kg adjusted for TOC), NOAA ER-M (1,600 μg/kg), and Ontario SEL (14,400 μg/kg). Based on an HQ approximating 1 for several alternative ESVs and a maximum concentration below median and severe effect levels, benzo(a)pyrene is eliminated as a COPC.

Benzo(b)fluoranthene was detected in two samples at estimated concentrations of 170 μg/kg (Beaverdam Brook) and 420 μg/kg (Southwest Marsh Area). Benzo(b)fluoranthene was carried forward because an ESV is not available from the Ontario MOE or NOAA. U.S. EPA's Assessment and Remediation of Contaminated

Sediments (ARCS) program identifies a threshold effects concentration (TEC) and probable effects concentration (PEC) of 20 µg/kg and 4,000 µg/kg, respectively. U.S. EPA Region V identifies an ecological screening level (ESL) of 10,400 µg/kg. The maximum concentration is below the ARCS PEC and Region V ESVs. Consequently, benzo(b)fluoranthene is eliminated as a COPC.

Benzo(g,h,i)perylene was detected in a single sample from the Southwest Marsh Area at an estimated concentration of 220 µg/kg. This concentration is slightly above its screening ESV, the Ontario LEL (HQ=1.3), but well below the Ontario SEL of 3,200 µg/kg. Based on an HQ approximating 1 for the screening ESV and a maximum concentration below the severe effect levels, benzo(g,h,i)perylene is eliminated as a COPC.

Benzo(k)fluoranthene was detected in a single sample from the Southwest Marsh Area at an estimated concentration of 460 µg/kg. This concentration is slightly above its screening ESV, the Ontario LEL (HQ=1.9), but well below the Ontario SEL (13,400 µg/kg). Based on an HQ close to 1 for the screening ESV and a maximum concentration below the severe effect levels, benzo(k)fluoranthene is eliminated as a COPC.

Dibenzo(a,h)anthracene was detected in a single sample from the Southwest Marsh Area at an estimated concentration of 130 µg/kg. The single concentration is slightly above the Ontario LEL and NOAA ER-L (HQ=2.1), but well below the Ontario SEL (1,300 µg/kg) and NOAA ER-M (260 µg/kg). Based on an HQ close to 1 for the screening ESV and a maximum concentration below the severe effect levels, dibenzo(a,h)anthracene is eliminated as a COPC.

Indeno(1,2,3-c,d)pyrene was detected in a single sample from the Southwest Marsh Area at an estimated concentration of 220 µg/kg. This concentration is slightly above its screening ESV, the Ontario LEL (HQ=1.1), but well below the Ontario SEL (3,200 µg/kg). Based on an HQ approximating 1 for the screening ESV and a maximum concentration below the severe effect levels, indeno(1,2,3-c,d)pyrene is eliminated as a COPC.

One TIC SVOC, an aliphatic hydrocarbon, was carried forward due to absence of an ESV. Ducane, with an ESV of 123,000 µg/kg, can be used a surrogate for this TIC. The single detected concentration of the aliphatic hydrocarbon is estimated at 100 µg/kg. Because this concentration is less than the surrogate ESV, the aliphatic hydrocarbon TIC can be eliminated as COPC.

8.4.1.4.1.3 INORGANICS

The screening assessment identified 20 inorganic constituents as COPCs. Calcium, magnesium, potassium, and sodium can be eliminated because they are essential nutrients.

Aluminum, barium, beryllium, cyanide, selenium, thallium, and vanadium were carried forward because they lack ESVs. Crommentuijn et al. (1997) identify sediment MPCs for barium (300 mg/kg), selenium (2.9 mg/kg), thallium (2.6 mg/kg), and vanadium (56 mg/kg). The maximum concentrations of barium (169 mg/kg) and vanadium (31.6 mg/kg) are below their respective MPCs. Accordingly, barium and vanadium can be eliminated as COPCs. The maximum concentrations of selenium (31 mg/kg) and thallium (18 mg/kg) exceed their MPCs.

As with soil, background concentrations of sediments can be used to determine if concentrations of the remaining metals are from natural or anthropogenic sources. The range and mean background concentrations for the 14 remaining inorganic constituents are presented in Table 8.18. The maximum, mean, and upper 95 percent UCL concentrations of on-site sediments are also presented in Table 8.18. The maximum concentrations of aluminum, antimony, arsenic, beryllium, and cyanide are within the range of Site-specific background concentrations. These five inorganic constituents are eliminated as COPCs.

To further refine the COPCs, the upper 95 percent UCL concentrations of the remaining nine metals were compared to ESVs. The results of this comparison are summarized in Table 8.19. The upper 95 percent UCL concentrations of lead and zinc are below their respective ESVs. Accordingly, lead and zinc can be eliminated as COPCs, and will not be carried forward to the BERA.

At this point, seven inorganics remain as COPCs: cadmium, copper, iron, manganese, nickel, selenium, and thallium. To determine if the ESVs used in the screening assessment are appropriate for site-specific conditions, the ESVs were evaluated relative to the range of site-specific background concentrations. If an ESV was within or below the range of background concentrations, it was identified as non-applicable to the Site, and alternative ESVs greater the maximum site-specific background concentrations were considered. The first priority for alternative ESVs was given the MPCs identified by Crommentiujn et al. (1997), because these benchmarks consider both background and toxicological effects. In the event an MPC was not available, other sources of benchmarks were considered.

The ESV for cadmium (0.6 mg/kg) is within the range of site-specific background concentrations (0.38 mg/kg to 2.4 mg/kg). Therefore, an alternative ESV, the MPC, was considered. The MPC for cadmium is 30 mg/kg, which is outside the range of background concentrations and an order of magnitude greater than the maximum on-site concentration (4.0 mg/kg). Accordingly, cadmium is eliminated as a COPC.

The ESV for copper (16 mg/kg) is within the range of background concentrations (11 mg/kg to 21 mg/kg). The MPC for copper is 73 mg/kg, which is outside the range of background concentrations. Both the maximum (685 mg/kg) and upper 95 percent UCL (201 mg/kg) concentrations exceed the MPC. Therefore, copper is retained as a COEC and is carried forward to the BERA.

The ESV for iron (20,000 mg/kg) is within the range of background concentrations (14,200 mg/kg to 28,400 mg/kg). An MPC is not available for iron. The only other benchmark identified in the screening assessment was the NYSDEC SEL of 40,000 mg/kg. This benchmark is also the Ontario SEL. The maximum detected concentration of iron (44,700 mg/kg) is slightly greater than the SEL, and was the only sample that exceeded the SEL. The upper 95 percent UCL (26,850 mg/kg) is well below the SEL. Therefore, iron can reasonably be eliminated as a COPC.

The ESV for manganese (460 mg/kg) is within the range of background concentrations (192 mg/kg to 506 mg/kg). An MPC is not available for manganese. The only other benchmark identified in the screening assessment was the NYSDEC SEL of 1,100 mg/kg. This benchmark is also the Ontario SEL. The maximum detected concentration of manganese (1,820 mg/kg) is greater than the SEL, but upper 95 percent UCL (911 mg/kg) is below the SEL. Furthermore, manganese is significantly correlated with aluminum (Pearson Product Moment Correlation; $p < 0.01$), suggesting that the concentrations of manganese can be attributed to background sources. Therefore, manganese can reasonably be eliminated as a COPC.

The ESV for nickel (16 mg/kg) is equal to the lowest detected site-specific background concentration. Consequently, the ESV is not appropriate for site-specific conditions. The MPC for nickel is 44 mg/kg, which is outside the range of background concentrations. The maximum on-site concentration of nickel is equal to the MPC. Because the maximum on-site concentration does not exceed the MPC, nickel is eliminated as a COPC.

The ESVs (MPCs) for selenium (2.9 mg/kg) and thallium (2.6 mg/kg) are outside the range of background concentrations and are, therefore, determined to be appropriate for

Site conditions. Because both the maximum and upper 95 percent concentrations of both selenium and thallium exceed their respective ESVs, both inorganic constituents are carried forward to the BERA as COECs.

The summary of COPCs eliminated and retained in Step 3 is presented in Table 8.20. Copper, selenium, and thallium are carried forward to the BERA as sediment COECs.

8.4.1.4.2 NORTHEAST MARSH AREA

As discussed in Section 8.4.1.3.2, the Northeast Marsh Area is eliminated as an evaluation area based on approval by U.S. EPA.

8.4.2 KNOWN ECOLOGICAL EFFECTS

8.4.2.1 ORGANICS

PCBs (Aroclor 1254)

Polychlorinated biphenyls (PCBs) are a group of 209 stable, fat-soluble chemicals. In the past, PCBs have been used in the manufacturing of capacitors, transformers, carbonless copying paper, paints and hydraulic fluid. Aroclor is the commercial name for U.S.-manufactured PCB product mixtures. Different product mixtures are further identified by codes, such as 1248, 1254 and 1260. These codes designate the percent of chlorination of the product mixture. For example, Aroclor 1254 has an average chlorine content of 54 percent. Most PCBs in the environment are similar, but not identical to, Aroclor 1254.

Environmental fate is of great importance in determining the environmental impact of PCBs. Due to its hydrophobic nature, the solubility of Aroclor 1254 is low. Therefore, Aroclor 1254 readily adsorbs to sediment and soil material. Increasing organic carbon content increases adsorption. Because Aroclor 1254 strongly adsorbs to organic rich sediment and soil, it is relatively immobile in the environment. Aroclor 1254 bioaccumulates in aquatic and terrestrial biota. Volatilization is also an important fate process but may occur only at a relatively slow rate.

Exposure of laboratory animals to Aroclor 1254 has resulted in liver and skin disorders. Reproductive and neurological effects have also been associated with long-term exposure of laboratory animals to Aroclor 1254.

Bis(2-ethylhexyl)phthalate

Bis(2-ethylhexyl)phthalate (BEHP) is used in large quantities as a plasticizer for polyvinyl chloride (PVC) and other plastics. Because of its wide use and distribution, as well as its high volatility and persistence, BEHP is essentially ubiquitous in urban and suburban environments.

BEHP released to aquatic systems biodegrades rapidly (half-life of 2-3 weeks) following a period of acclimation. It adsorbs to sediments (log K_{OC} of 4 to 5) and bioconcentrates in aquatic organisms (log BCF of 2 to 4 in fish and invertebrates). Evaporation and hydrolysis are not significant aquatic fate processes.

BEHP released to soil does not evaporate or leach into groundwater. It has a strong tendency to sorb soil particles and is not considered mobile. BEHP is rapidly biodegraded in soil under aerobic conditions following acclimation.

Rodents tend to absorb BEHP more readily than other animal species. The primary target organs for BEHP are the liver and kidneys. Higher doses may affect testes and decrease hemoglobin volume.

Di-n-octylphthalate

Di-n-octylphthalate is a anthropogenic substance used to soften plastics. It is also used in the manufacture of cosmetics and pesticides. Di-n-octylphthalae has relatively low volatility and a tendency to adhere to soil and sediment. It can be broken down into by-products through microbial processes, photolysis, and chemical reactions with other chemicals. Di-n-octylphthalate bioaccumulates in some aquatic organisms, such as fish and oysters.

Relatively little is known regarding the effects of di-n-octylphthalate on fish and wildlife. Relatively low doses appear to have little adverse effects. Ingestion of high doses for short (14 days or less) or intermediate (15 – 365 days) periods of time has resulted in death or adverse effects on the kidneys of laboratory animals.

8.4.2.2 INORGANICS

Copper

Copper naturally occurs in the earth's crust, primarily as sulphides and oxides. Copper is found in trace amounts in all plant and animal life and is believed to be essential for nutrition. Metallic copper is prepared by smelting and electrolytic refining. Approximately half of all copper production is used in electrical equipment. Copper salts also function as pesticides for fungi or algae (1 to 2 percent solution) and as herbicides.

Copper in soil is strongly adsorbed and tends to remain in the upper few inches of soil. Sandy soils with low pH have the greatest potential for leaching. In most temperate soil, the pH, organic matter, and ionic strength of the soil solutions are the key factors affecting adsorption.

In natural waters, most copper is complexed or tightly bound to organic matter; little is present in the free (hydrated) form. Because of chemical conditions in most natural waters, bioavailable copper is typically available at low levels. Sediment is an important sink and reservoir for copper.

Copper sulfide and other copper compounds are effective algaecides. Single-cell and filamentous algae and cyanobacteria are particularly susceptible to copper. Sublethal effects include reduction in photosynthesis and grown, loss of photosynthetic pigments, disruption of potassium regulation.

At high concentrations, copper can have adverse effects on fish, invertebrates, and amphibians, with adverse effects in embryos and tadpoles. Copper bioconcentrates in several different organ systems in fish and mollusks, but it does not biomagnify.

Small quantities of copper are essential for plant growth and photosynthesis. Some plant species have a great tolerance to increased concentrations of copper and can accumulate moderate concentrations of copper in their tissues. Adverse effects of copper on plants include reduction in root and shoot weight and leaf chlorosis.

Copper toxicosis in mammals is generally not significant. Mammals generally have a great ability to store copper in the liver without resultant damage. At high doses, necrosis of the liver and kidneys and fetal mortality may occur. In birds, high doses of copper may adversely affect growth.

Cyanide

Cyanides are a group of compounds based on a common structure formed when elemental nitrogen and carbon are combined. Cyanides are produced by certain bacteria, fungi, and algae and occur naturally in fruits (i.e., apple pits), roots (i.e., cassava roots), and leaves of numerous plants. Environmental concentrations of cyanide are primarily due to industrial releases.

Free cyanides readily degrade in the open environment, but persist in groundwater. In soils where cyanide levels are high enough to be toxic to microorganisms, it may leach into groundwater. At low concentrations, cyanide biodegrades under aerobic conditions with the initial formation of ammonia, which is converted to nitrate in the presence of nitrifying bacteria. Under anaerobic conditions, cyanides denitrify to gaseous nitrogen.

Biodegradation is a significant fate process in aquatic systems. The extent of adsorption of soluble metal cyanides to suspended solids and sediments increases with decreasing pH and increases with increasing clay and organic material contents of water. Adsorption is insignificant relative to volatilization and biodegradation. Cyanides do not bioaccumulate in aquatic organisms. There is no evidence of biomagnification of cyanides in aquatic food chains.

Adverse effects of cyanide on aquatic plants are unlikely at concentrations that cause acute effects to most species of fish and invertebrates. In higher terrestrial plants, elevated cyanide levels inhibited respiration and ATP production eventually leading to death. At lower concentrations, effects include inhibition of germination and growth.

Fish tend to be the most sensitive taxonomic group to cyanide. Sublethal effects include reduced reproductive capacity, impaired swimming ability, altered growth, and necrosis of the liver. Mammals are less sensitive to cyanide than fish and are relatively tolerant of intermittent sublethal exposures. In birds, cyanide appears to adversely affect respiratory functions.

Mercury

Mercury occurs as elemental mercury and as inorganic and organic compounds. Mercury occurs in the earth's crust primarily in the form of sulfides (i.e., cinnabar). Natural sources, such as geological weathering, contribute about the same order of magnitude of mercury to the environment as do the direct and indirect sources related to human activities.

Mercury is transported to aquatic ecosystems via surface runoff and atmospheric precipitation. Inorganic mercury sorbed to particulate material is not readily desorbed. Therefore, sediments are important repositories for inorganic forms. Elemental mercury is oxidized to inorganic mercury under natural conditions. Inorganic mercury can be methylated by both aerobic and anaerobic bacteria into forms that are readily taken up by benthic biota.

Aquatic organisms bioaccumulate methylmercury from the water column. Methylmercury in fish, aquatic birds, and mammals is the culmination of a complex food chain beginning with the methylation of inorganic mercury, usually in sediments. Biomagnification in fish has been observed to range between 5,000 and 100,000 times the concentrations in water. Once inside the cell, methylmercury binds rapidly to sulfhydryl groups of amino acid compounds. The combination of the lipophilic properties and the affinity for the sulfhydryl groups results in rapid accumulation in the muscles and fat tissues until methyl mercury is metabolized and removed by the liver, kidney, and spleen. Methylmercury is more slowly metabolized and eliminated than it is accumulated in most biota resulting in the bioaccumulation of great quantities which are passed on to their predators.

Plants vary in their ability to take up mercury and can develop a tolerance to high mercury concentrations in their tissues. Plants are generally less sensitive than animals to both inorganic mercury and methylmercury.

Upper trophic level fish, birds, and mammals are particularly susceptible to adverse effects because of the biomagnification of methylmercury. Effects in birds include delayed testicular development, altered mating behavior, reduced fertility, reduced survivability and growth in young, and gonadal atresia. Effects in mammals include ataxia, asphagia, tremors, and diminished coordination.

Selenium

Selenium is ubiquitous in the earth's crust, although it generally occurs at trace levels. Natural sources of selenium in the environment result from leaching and weathering of parent bedrock material, volatilization by plants and bacteria, and from volcanic emissions. Selenium has semiconductor and photoelectric properties that make it useful in electronics, photographic exposure meters, and rectifiers for home entertainment equipment. Elevated concentrations of selenium in aquatic systems occur in regions where the soil is naturally rich in selenium.

In soil, pH and redox potential are determining factors in the transport and partitioning of selenium. Elemental selenium is essentially insoluble and represents a major inert sink for selenium introduced into the environment. In soils, elemental selenium and inorganic selenium compounds are methylated by microorganisms and subsequently volatilized to the atmosphere. Water soluble organic selenium compounds are readily taken up by plants.

Selenium exists in surface water in inorganic (selenite and selenate) and organic forms. The response of aquatic organisms depends on its form. Organic forms generally are more toxic and bioaccumulate faster than either selenite or selenate. Aquatic organisms can convert selenium to both inert and soluble forms. Under natural conditions selenium biomagnifies in aquatic organisms. However, the degree of biomagnification is not likely to exceed one order of magnitude. Adverse effects of selenium on aquatic organisms include loss of equilibrium, liver damage, reproductive failure, reduce growth, chromosomal aberrations, and necrosis of the ovaries.

Selenium is an essential micronutrient for many organisms that can be both beneficial and toxic within a relatively narrow range of doses. In plants, selenium deficiencies can lead to a variety of diseases. Absorption of selenium by vegetation is determined by plant species, age, season of the year and the concentration of soluble selenium in the root zone. Soil with high sulfates reduce the uptake of selenium and may prevent injury. Too much selenium in an animal's diet can cause a variety of disorders (i.e., nutritional myopathies, liver narcosis, pancreatic fibrosis) that can lead to death.

Thallium

Thallium is a soft, bluish-white metal that is widely distributed in trace amounts in the earth's crust. It is usually found in combination with other elements primarily oxygen, sulfur and the halogens. Thallium compounds are used in the manufacture of alloys, electrical apparatus, catalysts for industrial organic reactions and optical equipment (e.g., spectrometers), and in photographic and ceramic formulations. Thallium is quite stable in the environment because it is neither transformed nor biodegraded.

Thallium may occur in aqueous systems as relatively soluble salts (thallium chloride, sulfate, carbonate, bromide, and hydroxide) associated with suspended solids and in combination with organic material. The major fates of thallium in the aquatic environment are transport as soluble complexes, sorption to clay minerals, and bioaccumulation. Thallium partitions from water to soils and sediments at a pH of 8.1. In fish, sublethal effects of thallium are reduction in the growth of larval fish and percent embryo hatchability.

Thallium is completely absorbed from the gastrointestinal tract. In animals, distribution of thallium from the bloodstream is rapid and widespread. Thallium accumulates in the kidneys, heart, brain, bone, skin, and blood.

8.4.3 REFINEMENT OF EXPOSURE PATHWAYS AND ECOSYSTEMS POTENTIALLY AT RISK

Terrestrial Ecosystem

The terrestrial portions of the Site consist of a variety of cover types, consisting mainly of forest and shrub that have the potential to support an abundance and diversity of ecological receptors indigenous to the area. The presence of larger transient and/or migratory animals is possible in these covered areas, although their occurrence on and utilization of the Site would be less than animals with smaller home ranges. Thus, for most higher species of mammals and birds, their utilization of the Site and, thus, exposure to Site-related chemicals is expected to be significantly less than the 100 percent assumed in the screening assessment. For sedentary terrestrial receptors, such as plants and earthworms, the assumption of 100 percent exposure on Site is reasonable.

It was also assumed in the screening assessment that potential terrestrial receptors would spend 100 percent of their time on Site within the area of maximum contamination. The maximum concentration of almost every organic COEC and BCC in surface soil was detected in one of two samples collected from the Lagoon 6 area, as were the maximum detected concentrations of chromium, copper, iron, mercury, nickel, and vanadium. Lagoon 6 is a small area in relation to the size of the Site measuring approximately 0.23 acre. Consequently, it is unrealistic to assume that larger transient wildlife will spend 100 percent of their lifetimes within this area, considering that some wildlife have home ranges in the hundreds of hectares. Wildlife species often require large home ranges and territories in order to obtain food, water and shelter. Although it is reasonable to apply some sort of an exposure factor for the wide ranging higher wildlife species, it was conservatively assumed that these species spend 100 percent of their time on Site, thus receiving an average Site-wide exposure to BCCs.

Evaluation of the potential effects of BCCs is more complicated because many are not directly toxic to either plants or soil invertebrates. PCB toxicity occurs primarily to upper levels of the food chain after the PCBs have biomagnified, but organisms in lower trophic levels may also be exposed to toxic levels of PCBs via direct ingestion of soil, and to a lesser extent, by consumption of vegetation. Because PCBs and mercury do not readily bioaccumulate in plants, herbivorous and omnivorous species face less exposure than carnivorous species. Nevertheless, herbivores are evaluated in the BERA. Exposure to BCCs by predators of soil dwelling invertebrates and to predators of small vertebrates are also evaluated in the BERA.

The exposure pathways that are evaluated in the baseline assessment for the terrestrial exposure areas of the Site include:

- exposure to soils by soil dwelling invertebrates, via direct contact;
- exposure to soils by plants, via direct contact;
- exposure to predators of soil dwelling invertebrates, via their prey and incidental soil ingestion;
- exposure of herbivores, via ingestion of plant tissue and incidental soil ingestion; and
- exposure to predators of small vertebrates.

Aquatic Ecosystem

The direct contact exposure pathways identified in the screening assessment for the aquatic environments remain complete. For COECs identified for surface water and sediments, potential risk to piscivores foraging on fish in Beaverdam Brook/Otter Kill is evaluated. Benthic invertebrates in Beaverdam Brook/Otter Kill are assumed to be essentially sedentary, and thus the assumption of 100 percent exposure within these specific areas is reasonable.

It was conservatively assumed that fish are localized to the evaluation area of Beaverdam Brook/Otter Kill, and thus the assumption of 100 percent exposure within the entire area is reasonable.

The exposure pathways that are evaluated in the baseline assessment for the Beaverdam Brook/Otter Kill aquatic habitat in the Site area include:

- exposure to sediment by benthic invertebrates, via direct contact;
- exposure to surface water by fish, via direct contact; and
- exposure to predators of fish via their prey and incidental sediment ingestion.

8.4.4 SELECTION OF ASSESSMENT AND MEASUREMENT ENDPOINTS

Assessment endpoints identify the ecological values to be protected at the Site, whereas measurement endpoints are measurable responses to a stressor that are related to the valued characteristics chosen as the assessment endpoints (U.S. EPA, 1992c). Measurement endpoints are used in cases where the assessment endpoint cannot be directly measured or evaluated. For example, the ecological health of benthic macroinvertebrates (an assessment endpoint) can be evaluated using aquatic toxicity data (measurement endpoints) derived from appropriate field or laboratory tests.

Selected assessment endpoints are often based on the protection of a valued component of the ecosystem. It is often necessary to identify appropriate indicator species as a representative of an ecological guild (e.g., shrew as a representative of mammalian insectivores; red-tailed hawk as a representative of avian carnivores). The relatively large number of potential receptor species identified for the Site precludes an assessment of potential risks for each and every species. Potential risks to representative ecological guilds were assessed by evaluating the potential risk to indicator species for which relevant ecotoxicity data are available.

The criteria used in the ERA for the selection of indicator species were described by the U.S. EPA (1992c), and are as follows:

i) Species Presence: Species are only to be selected as indicator species if they are found or expected to be present at or adjacent to the Site. There are several factors to consider if the species has not been documented to be present at the Site, including the presence of appropriate habitat, the availability of food, and seasonality of receptor presence (e.g., migratory species may be present only during part of the year).

ii) Ecological Relevance: Indicator species should be selected that reflect important trophic positions within the ecosystem and are functionally related to other species. Selection of ecologically relevant receptor species requires an understanding of the structure and function of the ecosystem at the Site. For example, changes in some species are known to have a controlling influence on the abundance and distribution of many other species in its community.

iii) Susceptibility to Chemicals Present at the Site: In addition to ecological relevance, indicator species should be selected that are sensitive to the chemicals detected at the Site.

iv) Policy Goals and Societal Values: Indicator species should be chosen that reflect policy goals and societal values. Examples of species the society values include federally endangered or threatened species, species with economic value (e.g., fish or game), or species that provide other beneficial uses to people (e.g., aesthetic value).

Assessment Endpoints

Proper assessment endpoints for the terrestrial environment are identified as:

- protection of macroinvertebrates potentially impacted by COECs in soil;
- protection of terrestrial plants potentially impacted by COECs in soil;
- protection of the mammalian and avian consumers of soil macroinvertebrates (i.e., insectivores) potentially impacted by COECs in soil;
- protection of mammalian consumers of terrestrial plants (i.e., herbivores) potentially impacted by COECs taken up from plants;
- protection of the mammalian and avian predators of small vertebrates (i.e., carnivores) potentially impacted by COECs in soil; and
- protection of mammalian consumers of terrestrial plants (i.e., herbivores) potentially impacted by COECs taken up by plants.

Proper assessment endpoints for the combined Beaverdam Brook and Otter Kill in the Site area are identified as:

- protection of benthic macroinvertebrates potentially impacted by COECs in sediment;
- protection of the fish potentially impacted by COECs in surface water; and
- protection of mammalian and avian predators of forage fish potentially impacted by COECs in surface water and sediments.

Measurement Endpoints

Measurement endpoints used to evaluate assessment endpoints for the terrestrial environment include the following:

- the evaluation of potential impact of COEC toxicity on soil macroinvertebrate survival and growth through direct contact exposure using toxicity data from the scientific literature;
- the evaluation of potential impact of COEC toxicity on plant survival and growth through direct contact exposure using toxicity data from the scientific literature;
- the evaluation of potential impact of COEC toxicity on the reproduction of consumers of soil macroinvertebrates, using toxicity data from the scientific literature and bioaccumulation modeling;
- the evaluation of potential impact of COEC toxicity on the reproduction of herbivores using toxicity data from the scientific literature and bioaccumulation modeling; and
- the evaluation of potential impact of COEC toxicity on the reproduction of predators of small vertebrates, using toxicity data from the scientific literature and bioaccumulation modeling.

Measurement endpoints used to evaluate assessment endpoints for the aquatic environments include the following:

- the evaluation of the impact of COEC toxicity on invertebrate survival and growth through direct sediment contact exposure using toxicity data from the scientific literature;
- the evaluation of the impact of COEC toxicity on fish survival, growth, and/or reproduction through direct surface water contact exposure using toxicity data from the scientific literature; and
- The evaluation of the impact of COEC toxicity on survival, growth, and/or reproduction of avian and mammalian piscivores using toxicity data from the scientific literature.

The assessment and measurement endpoints for the baseline assessment are summarized in Table 8.21.

Indicator Species

In the terrestrial environment, earthworms were selected as an indicator species for soil dwelling invertebrates. Earthworms were selected because of their likely presence in Site soils, because they are maximally exposed as a result of constant contact with and ingestion of soils, because they are prey for many higher trophic organisms, and because there is a relatively large toxicity database.

A specific indicator species was not selected for the terrestrial plants. Instead, relevant toxicity data identified for terrestrial plant species exposed to COECs in soil were used in the baseline assessment.

The shrew was selected as a mammalian indicator species and woodcock was selected as an avian indicator species for the terrestrial insectivore pathway. Shrew and woodcock are common insectivores known to feed on earthworms and are likely to incidentally ingest soils while foraging for food. Shrew and woodcock are susceptible to PCB and mercury toxicity and are ecologically important species in the food chain, preyed upon by higher trophic species.

The short-tailed weasel was selected as a mammalian indicator species, and a red-tailed hawk was selected as an avian indicator species, for the terrestrial carnivore pathway. Short-tailed weasel and red-tailed hawk are likely to be present within the vicinity of the Site area and are likely to be among the most sensitive vertebrates to bioaccumulative chemicals, especially PCBs. Small vertebrates comprise a significant portion of the diet of both the short-tailed weasel and red-tailed hawk. Short-tailed weasel and red-tailed hawk are also ecologically important species, occupying a trophic level at the top of the food chain. Additionally, as with most other higher trophic level species, the red-tailed hawk has high intrinsic value to society.

The meadow vole was selected as a vertebrate indicator species for the terrestrial herbivore pathway. The meadow vole is a small herbivorous mammal that burrows in soil. Its small body size, high ingestion rate, small home range, and borrowing lifestyle make it a conservative indicator species for herbivorous vertebrates.

A specific indicator species was not selected for benthic macroinvertebrates inhabiting Beaverdam Brook/Otter Kill. Instead, relevant toxicity data identified for benthic invertebrates exposed to COECs in sediment were used in the baseline assessment. Benthic invertebrates are likely present in Beaverdam Brook/Otter Kill.

The fathead minnow was selected to represent aquatic fish in direct contact with Beaverdam Brook/Otter Kill surface waters. Fish were chosen for evaluation in the baseline assessment because they are ecologically significant species interacting directly with surface waters while representing prey for higher trophic species. The fathead minnow was selected to represent fish because it is a common species in many aquatic ecosystems, is a fairly sensitive species that is used in many regulatory toxicity testing programs, and because there exists a large body of aquatic toxicity data for this species in the scientific literature.

Mink and belted kingfisher were selected as indicator species for the mammalian piscivore and avian piscivore, respectively. Both indicator species are likely to occur in the area and forage on fish in Beaverdam Brook/Otter Kill. Both species are relatively sensitive to the COECs that will be evaluated in the BERA. Both species also have relatively high intrinsic value to humans.

8.4.6 SITE CONCEPTUAL MODEL

The conceptual model is the primary output of the Problem Formulation step for the ERA. The conceptual model represents the potential exposure pathways being evaluated including the relationship between selected assessment and measurement endpoints.

Figures 8.1 and 8.2 present Site conceptual models for the on-Site terrestrial environment and the Beaverdam Brook/Otter Kill aquatic environment respectively. Each conceptual model includes the media source of COECs, the potentially complete exposure pathways, the selected assessment endpoints, the selected receptor species, and the exposure route. The conceptual models indicate how the assessment endpoints being evaluated may become potentially impacted through exposure to the COECs on Site or in the Site vicinity.

The relationship between exposure to a COEC and toxicological effect for the identified indicator species is determined as part of the Exposure and Effects Analysis presented is Section 8.5. The relationship between the COEC concentration and the ecotoxicological effect is defined by the measurement endpoint. The Exposure and Effects Analysis Section includes a description of the exposure models used in the baseline assessment to estimate exposure point concentrations, and includes a description of exposure factors and assumptions, where applied. Measurement endpoints were compared to COEC concentrations in Site media or to estimated concentrations in prey tissue in order to determine if the assessment endpoints are at risk, as part of the Risk Characterization in Section 8.6.

8.5 STEP 6 - EXPOSURE AND EFFECTS ANALYSIS

8.5.1 EXPOSURE ANALYSIS

8.5.1.1 APPLICATION OF REASONABLE UPPER-BOUND EXPOSURE CONCENTRATIONS

To estimate risk to ecological receptors in the risk characterization step of the baseline assessment, measurement endpoint values were compared to media exposure point concentrations for each potentially complete exposure pathway. The exposure media include abiotic media such as soil and surface water, and biotic media such as animal tissue. In order to maintain a conservative approach, maximum detected concentrations were used to identify COECs in the screening assessment. In the baseline assessment, more realistic and Site-specific exposure assumptions were used where applicable to evaluate potential risk to ecological receptors.

Use of maximum detected exposure point concentrations to characterize risk for mobile environmental receptors (i.e., fish and wildlife species) is a conservative practice and tends to overestimate actual ecological risk. Consequently, use of reasonably derived upper-bound mean exposure concentrations is regarded as a valid approach in assessing risk to these receptors (U.S. EPA, 1992c). The upper-bound exposure concentration is defined as the highest exposure that could reasonably occur for a given exposure pathway at a Site, and is intended to account for both uncertainty in the contaminant concentration and variability in exposure parameters (U.S. EPA, 1992a). The upper-bound concentrations used in the baseline assessment were equal to the 95 percent upper confidence level (95 percent UCL) of the mean concentrations or the maximum concentrations, whichever was lower for a particular constituent. For relatively sedentary species (i.e., plants and soil/sediment invertebrates), use of maximum detected concentrations to estimate potential risk was determined to be a conservative but reasonable assumption, consistent with applicable regulatory guidance.

8.5.1.2 ESTIMATION OF COEC CONCENTRATIONS IN FOOD ITEMS

To estimate the toxicological effects of COECs in soil (PCBs and mercury) on shrew, woodcock, meadow vole, short-tailed weasel, and red-tailed hawk, COEC concentrations in the tissue of their prey were estimated using soil-to-tissue uptake factors (UFs) or regression models.

For insectivores (shrew and woodcock), it was conservatively assumed that earthworms were their primary source of food and COEC exposure. Incidental soil ingestion was identified as a secondary COEC source. Concentrations of PCBs and mercury in earthworm tissue, in mg/kg dry weight, can be estimated from soil concentrations using the log-linear regression models derived by Sample et al. (1998b), as follows:

$$\mathrm{Ln}\,[\text{earthworm PCB}] = 1.410 + 1.361 * \mathrm{Ln}[\text{soil PCB}]$$

$$\mathrm{Ln}\,[\text{earthworm Hg}] = 0.0781 + 0.3369 * \mathrm{Ln}[\text{soil Hg}]$$

Estimated concentrations of PCBs and mercury were multiplied by 0.2 to convert dry weight to wet weight concentrations.

For upper trophic level carnivores (weasel and red-tailed hawk), it was assumed that small mammalian vertebrates were their primary source of food and COEC exposure. The information on bioaccumulation of COEC from soil to terrestrial mammals is not extensive (Hebert, et al., 1994). However, Sample et al. (1998a) have derived 90th percentile UFs for mercury for insectivores (1.046), herbivores (0.024), and omnivores (0.13). As a conservative assumption, the UF of 1.046 for insectivores was used to estimate concentrations of mercury in prey items using the equation:

$$[\text{small mammal Hg}] = 1.046 * [\text{soil Hg}].$$

Some data are available on soil-to-small mammal bioaccumulation of PCBs. As summarized in Table 8.22, based on several studies, the UF from soil to small mammals ranges from approximately 0.001 to 5.9, with an arithmetic mean of 1.55.

For a number of reasons, the higher and lower UF values are probably not applicable to conditions at the Site. The wide range of values is probably due to Site-specific differences in bioavailability and the ultimate source of contamination. For example, the high UF values for PCBs found in Hebert et al. (1994) are probably due to the following factors. The soils sampled were from non-industrial, rural areas in which the sources of PCBs were most likely to be atmospheric inputs as opposed to a spill or to disposal activities into the ground. In such cases, the soil to animal UF will be higher than at spill or disposal Sites because the primary source, atmospheric deposition, also blankets the vegetation. In these cases, the PCBs on vegetation, as opposed to the PCBs in the soil, may be the primary source of PCBs to the mammals. This exposure situation is much different from that likely to occur at the Site where the small rodent's primary exposure is the PCBs in the soil, with minimal amounts coming from PCB residues on the plants.

Given the uncertainty and range of values, the arithmetic mean value of 1.55 was chosen as the UF for PCBs in Site soils to evaluate the mammalian and avian carnivore exposure pathways. Concentrations of PCBs in small mammals were calculated using the equation:

$$[\text{small mammal PCBs}] = 1.55 * [\text{soil PCBs}].$$

For herbivores (meadow vole), it was assumed the entire diet consisted of plants from the Site and that exposure to COECs was through consumption of plant tissue and incidental soil ingestion. Concentrations of mercury in plant tissue was calculated using the log-linear regression equation (U.S. DOE, 1998):

$$\text{Ln}\,[\text{plant Hg}] = -0.996 + 0.544 * \text{Ln}[\text{soil Hg}].$$

Percent moisture in terrestrial plants ranges from 70 to 90 percent (Sample et al, 1997). Estimated concentrations of mercury were multiplied by 0.2 to convert dry weight to wet weight concentrations, which assumes 80 percent moisture content.

Uptake of PCBs by plants from soil was calculated using an UF of 0.01. This value was taken from the Screening Level Ecological Risk Assessment Protocol Hazardous Waste Combustion Facilities (U.S. EPA, 1999b). The equation used to calculate PCB concentrations in plants is:

$$[\text{plant PCBs}] = 0.01 * [\text{soil PCBs}].$$

The soil concentrations of PCBs and mercury used to calculate concentrations in food tissue were the 95 percent UCL or the maximum concentration, whichever was lower. Risk estimates for the arithmetic mean concentrations were also included for both pathways for comparative purposes. The dichlorobiphenyl isomer, which was identified as a COEC in the problem formulation step, has a similar chemical structure to PCBs and, thus, the potential to have similar ecotoxicological effects. Therefore, the concentrations of PCBs and the dichlorobiphenyl isomer within individual samples were summed to derive exposure concentrations for PCBs. Based on the sum of PCBs and the dichlorobiphenyl isomer, the 95 percent UCL concentration and mean concentration for PCBs were 6.14 mg/kg and 2.5 mg/kg, respectively. The rationale for using upper-bound concentrations, instead of maximum concentrations is that the shrew, woodcock, weasel, red-tailed hawk, and meadow vole will forage for food not just from the soil area with the highest COEC concentrations, but from their entire home range, and therefore, will average any exposure to affected prey. Additionally, the small vertebrate prey of the weasel and the red-tailed hawk are not concentrated just the area where the highest soil COEC concentrations were detected, but instead are distributed throughout the entire Site area. Accordingly, the mean concentration best represents the exposure concentrations of PCBs and mercury. Use of upper-bound concentrations provides reasonable certainty that exposure concentrations of BCCs are not underestimated for these pathways.

Tables 8.23 and 8.24 present the calculations of the residue concentrations for PCBs and mercury estimated in prey tissue of earthworms for the shrew and woodcock, respectively. Tables 8.25 and 8.26 present the calculations of the residue concentrations for PCBs and mercury estimated in prey tissue of small mammals for the weasel and hawk, respectively. Table 8.27 presents the calculations of the residue concentrations for PCBs and mercury estimated in plant tissue ingested by meadow vole.

To evaluate potential risk to piscivores (mink and belted kingfisher), concentrations of COECs in fish were estimated for both surface water and sediments. For COECs in surface water (bis(2-ethylhexyl)phthalate, di-n-octylphthalate, and cyanide), concentrations in fish were estimated using water to fish bioaccumulation factors identified by U.S. EPA (1999b). Bioaccumulation factors were 70 for bis(2-ethylhexyl)phthalate, 9,400 for di-n-octylphthalate, and 633 for cyanide. Tables 8.28 and 8.29 present the calculations of the residue concentrations for the surface water COECs estimated in fish tissue ingested by mink and belted kingfisher, respectively.

For COECs in sediments, concentrations of copper and selenium in fish were estimated using sediment to fish BCFs identified by Pascoe et al. (1996). These BCFs produce a dry weight concentration in fish. To convert to wet weight, a water content of 75 percent was assumed (Sample et al., 1997). The wet weight concentration of copper and selenium in fish was calculated using the equations:

$$[\text{fish Cu}] = [\text{sediment Cu}] * 0.10 * 0.25$$

$$[\text{fish Se}] = [\text{sediment Se}] * 1.0 * 0.25$$

where [fish Cu] and [fish SE] are the wet weight concentrations of copper and selenium, respectively, in fish. [sediment Cu] and [sediment Se] are the concentrations of copper and selenium, respectively, in sediment.

For thallium, concentrations in fish were estimated using a two-step process. In the first step, concentrations of thallium in benthic macroinvertebrates were estimated using the sediment to invertebrate BCF factor of 0.9 identified by U.S. EPA (1999b). In the absence of published uptake factors from macroinvertebrates to fish, it was conservatively assumed that the uptake factor from macroinvertebrates to fish was one. This assumption is consistent with U.S. EPA (1999b). This approach produces a wet concentration of thallium in fish using the equation

[fish Tl] = [sediment Tl] * 0.9 * 1.0

Where [fish Tl] is the wet weight concentration of thallium in fish and [sediment Tl] is the concentration of thallium in sediment.

Tables 8.30 and 8.31 present the calculations of the residue concentrations for copper, selenium, and thallium in fish tissue ingested by mink and belted kingfisher, respectively.

8.5.1.3 ESTIMATION OF COEC DOSE FOR WILDLIFE INDICATOR SPECIES

8.5.1.3.1 OVERVIEW

The COEC exposure to each wildlife indicator species was estimated by calculating a species-specific COEC dose using the modeled prey tissue concentrations, the species prey consumption rates, and the incidental sediment or soil ingestion rates, if applicable.

The COEC dose to consumers of soil-dwelling invertebrates was estimated by the following equation for total exposure:

> Total Dose = [food] * consumption rate + [soil] * incidental soil consumption rate.

The bracketed terms (e.g., [soil]) refer to the concentration of the COEC in that medium and other values are self-explanatory. Based on a conservative approach, absorption efficiency was assumed to be 100 percent for all pathways. PCBs and mercury were not identified as COECs in surface water, so exposure via drinking water was not evaluated.

8.5.1.3.2 INSECTIVORES

The shrew and the woodcock were assumed to eat only earthworms from the Site (i.e., area use and diet exposure were both assumed to be 100 percent). According to

U.S. EPA (1993a), shrew eat about 0.62 grams of food for each gram of body weight per day, whereas woodcock eat about 0.77 grams of food for each gram of body weight per day. Incidental soil ingestion for the shrew and woodcock was assumed to be 13 percent (Sample, et al., 1997) and 10 percent (U.S. EPA, 1993a) of food consumption, respectively. Thus, the total daily dose of each COEC for the shrew and the woodcock were estimated as follows:

- $\text{Dose}_{\textit{SHREW}}$, mg/kg/day = ([Worm], mg/kg * 87 percent + 13 percent * [Soil], mg/kg) * 0.62 kg/kg body wgt./day; and
- $\text{Dose}_{\textit{WOODCOCK}}$, mg/kg/day = ([Worm], mg/kg * 90 percent + 10 percent * [Soil], mg/kg) * 0.77 kg/kg body wgt./day.

Tables 8.23 and 8.24 present the estimated mean and upper bound COEC dose values used in the baseline assessment for shrew and woodcock.

8.5.1.3.3 CARNIVORES

The COEC dose to consumers of small vertebrate mammals was estimated by the following equation for total exposure:

Total Dose = [food] * consumption rate.

The bracketed term [food] refers to the concentration of the COEC in the rodent tissue. Based on a conservative approach, absorption efficiency was assumed to be 100 percent for all pathways. PCBs and mercury were not identified as COECs in surface water, so exposure via drinking water was not evaluated. In addition, the incidental soil ingestion pathway was not included, because soil ingestion for a top carnivore is considered insignificant in comparison to exposure from the food chain.

Weasel and red-tailed hawk were assumed to eat only small mammals from the Site (i.e., area use and diet exposure were both assumed to be 100 percent). According to U.S. EPA (1993a), weasels eat about 0.31 grams of food for each gram of body weight per day, whereas red-tailed hawks eat about 0.11 grams of food for each gram of body weight per day. Thus, the total daily dose of each COEC for the weasel and the red-tailed hawk were estimated as follows:

- $\text{Dose}_{\textit{WEASEL}}$, mg/kg/day = ([Small Mammal], mg/kg * 0.31 kg/kg body wgt./day; and

- Dose$_{RED\text{-}TAILED\ HAWK}$, mg/kg/day = ([Small Mammal], mg/kg * 0.11 kg/kg body wgt./day.

Tables 8.25 and 8.26 present the estimated mean and upper bound COEC dose values used in the baseline assessment for weasel and red-tailed hawk, respectively.

8.5.1.3.4 HERBIVORES

The COEC dose to herbivores was estimated by the following equation for total exposure:

> Total Dose = [food] * consumption rate + [soil] * incidental soil consumption rate.

The bracketed terms (e.g., [soil]) refer to the concentration of the COEC in that medium and other values are self-explanatory. Based on a conservative approach, absorption efficiency was assumed to be 100 percent for all pathways. PCBs and mercury were not identified as COECs in surface water, so exposure via drinking water was not evaluated.

Meadow vole were assumed to eat only plant tissue from the Site (i.e., area use and diet exposure were both assumed to be 100 percent). According to U.S. EPA (1993a), meadow voles eat about 0.316 grams of food for each gram of body weight per day. Incidental soil ingestion was assumed to be 2.4 percent of food consumption (U.S. EPA, 1993a). Thus, the total daily dose of PCBs and mercury for the meadow vole was estimated as follows:

- DoseVOLE, mg/kg/day = ([plant], mg/kg * 97.6 percent + 2.4 percent * [soil], mg/kg) * 0.316 kg/kg body wgt./day.

8.5.1.3.5 PISCIVORES

8.5.1.3.5.1 SURFACE WATER

Ingestion of COECs in surface water (bis(2-ethylhexyl)phthalate, di-n-octylphthalate, and cyanide) was estimated using the following general equation for total exposure:

Total Dose = [food] * fish consumption rate + [water] * surface water consumption rate.

The bracketed terms (e.g., [food]) refer to the concentration of the COEC in that medium and other values are self-explanatory. Because none of the COECs in surface water were identified as COECs in sediment, ingestion of COECs through incidental ingestion of sediment was considered to be insignificant. Based on a conservative approach, absorption efficiency was assumed to be 100 percent.

Mink and belted kingfisher were assumed to consume only fish taken from Beaverdam Brook/Otter Kill. For mink, the food ingestion rate and water ingestion rate were assumed to be 0.16 gram per gram body weight per day and 0.11 gram per gram body weight per day, respectively. For belted kingfisher, the food ingestion rate and water ingestion rate were assumed to be 1.23 gram per gram body weight per day and 0.11 gram per gram body weight per day, respectively. Based on these assumptions, the total doses of surface water COECs ingested by mink and belted kingfisher were calculated as:

- DoseMINK, mg/kg/day = [Fish], mg/kg * 0.16 kg/kg body wgt./day + [Surface Water], mg/L * 0.11 kg/kg body wgt./day, and
- DoseKINGFISHER, mg/kg/day = [Fish], mg/kg * 1.23 kg/kg body wgt./day + [Surface Water], mg/L * 0.11 kg/kg body wgt./day.

8.5.1.3.5.2 SEDIMENT

For COECs in sediment (i.e., copper, selenium, and thallium), ingestion by piscivores was estimated using the general equation:

Total Dose = [food] * consumption rate + [sediment] * incidental sediment consumption rate + [water] * surface water consumption rate.

The bracketed terms (e.g., [sediment]) refer to the concentration of the COEC in that medium and other values are self-explanatory. Based on a conservative approach, absorption efficiency was assumed to be 100 percent for all pathways.

Food ingestion and water ingestion rates for mink and belted kingfisher for sediment COECs were the same as used for surface water COECs. Incidental ingestion of sediment by mink was assumed to be 1 percent. Incidental ingestion of sediment by belted kingfisher was assumed to be negligible. Based on these assumptions, the total doses of sediment COECs ingested by mink and belted kingfisher were calculated as:

- DoseMINK, mg/kg/day = ([fish], mg/kg * 99 percent + 1 percent * [sediment], mg/kg) * 0.16 kg/kg body wgt./day + [Surface Water], mg/L * 0.11 kg/kg body wgt./day, and, and
- DoseKINGFISHER, mg/kg/day = ([fish], mg/kg * 1.23 kg/kg body wgt./day + [Surface Water], mg/L * 0.11 kg/kg body wgt./day.

Surface water concentrations were available for copper and thallium, but not for selenium. Therefore, the estimated dose for selenium does not include ingestion via drinking surface water.

8.5.2 TOXICITY ANALYSIS

The toxicity analysis step of the ERA involves the evaluation of relationships between the COECs and their ecological effects. The relationship between the COEC and their ecological effect is referred to as the measurement endpoint. Measurement endpoints were used to estimate risk to ecological receptors and, when combined with ecological exposure estimates, allow for an evaluation of the assessment endpoints that are to be protected. Measurement endpoint values were identified that correspond to each indicator species and the COECs they are exposed to through the species-specific exposure pathways. The following sections present a description of the measurement endpoint data used to estimate risk in the ERA.

8.5.2.1 MEASUREMENT ENDPOINT DATA FOR WILDLIFE INDICATOR SPECIES

All measurement endpoint values for terrestrial and piscivorous wildlife indicator species were obtained from the ORNL (Sample, et al., 1996). The measurement endpoints are based on the No Observed Adverse Effect Level (NOAEL) and the Lowest Observed Adverse Effect Level (LOAEL) for the indicator species evaluated. Measurement endpoint values represent COEC dose concentrations in animal tissue presumed not to be hazardous to the consumer. Use of NOAEL- and LOAEL-based measurement endpoints is a conservative approach. As described by Sample et al. (1996), NOAEL-based benchmarks represent concentrations believed to be nonhazardous for the specific wildlife species whereas LOAEL-based benchmarks represent concentrations at which adverse effects are likely to be expressed. All measurement endpoints used in the baseline assessment were based on chronic reproductive endpoints.

Information sources that the ORNL reviewed to identify appropriate toxicological data included the U.S. EPA Terrestrial Toxicity database (TERRETOX), USFWS reports, EPA assessment and criteria documents, and the ATSDR toxicity profiles. In addition, information was selected from many referred journals such as Environmental Toxicology and Chemistry, Archives of Environmental Contamination and Toxicology, and the Journal of Wildlife Management. Due to the scope of literature reviewed by the ORNL for the purposes of deriving their wildlife benchmarks, it was determined that these benchmarks were based on a significant amount of the currently available scientific data, and therefore, were appropriate for use as measurement endpoints.

The general method used by the Sample et al., (1996) to develop measurement endpoints is based on EPA methodology for deriving human toxicity values from animal data. This methodology involves using experimentally derived NOAELs or LOAELs to estimate NOAELs or LOAELs for wildlife by scaling doses to body size. NOAELs and LOAELs are daily dose levels normalized by the body weight of the test animals (mg of chemical per kg body weight per day). Normalization by body weight allows comparisons across species with the consideration for differences in body size. Studies have shown that numerous physiological functions such as metabolic rates, as well as responses to toxic chemicals, are a function of body size (Sample et al., 1996). Therefore, by knowing the dose that causes an effect in a mammalian test species (i.e., mouse), a dose level for a mammalian wildlife species (i.e., shrew) can be estimated for the same effect by using the relationship between their body sizes.

Recent research suggests that physiological scaling factors developed for mammals may not be appropriate for interspecies extrapolation to birds (Sample et al., 1996). A review of appropriate studies by the ORNL led to the conclusion that a scaling factor of 1 was considered most appropriate for interspecies extrapolation among birds. Therefore, if a LOAEL was available for an avian test species (i.e., Japanese quail), then the equivalent LOAEL for an avian wildlife species (i.e., woodcock) was assumed to be the same.

Measurement endpoints for PCBs identified for shrew were 0.067 mg/kg/day for the NOAEL and 0.67 mg/kg/day for the LOAEL, whereas those for mercury were 0.07 mg/kg/day for the NOAEL and 0.35 mg/kg/day for the LOAEL.

Measurement endpoints for PCBs identified for weasel were 0.14 mg/kg/day for the NOAEL and 0.69 mg/kg/day for the LOAEL, whereas for mercury only a NOAEL of 1.0 mg/kg/day was available for a reproductive endpoint.

Measurement endpoint PCB values identified for the woodcock and red-tailed hawk were 0.18 mg/kg/day for the NOAEL and 1.80 mg/kg/day for the LOAEL, while those for mercury were 0.006 mg/kg/day for the NOAEL and 0.064 mg/kg/day for the LOAEL.

Measurement endpoints for PCBs identified for meadow vole were 0.051 mg/kg/day for the NOAEL and 0.511 mg/kg/day for the LOAEL. For mercury, measurement endpoints were 0.054 mg/kg/day for the NOAEL and 0.269 mg/kg/day for the LOAEL.

Potential risk to mammalian and avian piscivores was considered for bis(2-ethylhexyl)phthalate, di-n-octylphthalate, and cyanide in surface water and copper, selenium, and thallium in sediments. For mink, the measurement endpoints for bis(2-ethylhexyl)phthalate are 7.6 and 76 mg/kg body weight/day for the NOAEL and LOAEL, respectively. For belted kingfisher, only a NOAEL of 1.1 mg/kg body weight/day is available.

Neither a NOAEL nor a LOAEL was available for di-n-octylphthalate for either mink or belted kingfisher.

For cyanide, a NOAEL of 49.7 mg/kg body weight/day was identified for mink. A LOAEL was not available. Neither a NOAEL nor LOAEL was available for belted kingfisher for cyanide.

For sediment COECS, measurement endpoints for mink exposed to copper were 11.7 mg/kg/day for the NOAEL and 15.4 mg/kg/day for the LOAEL. For selenium, the measurement endpoints for the NOAEL and LOAEL were 0.154 mg/kg/day and 0.254 mg/kg/day, respectively. For thallium, measurement endpoints were 0.006 mg/kg/day and 0.058 mg/kg/day for the NOAEL and LOAEL, respectively.

For belted kingfisher measurement endpoints for copper were 47 mg/kg/day for the NOAEL and 61.7 mg/kg/day for the NOAEL. For selenium, measurement endpoints were 0.40 mg/kg/day and 1.0 mg/kg/day for the NOAEL and LOAEL, respectively. For thallium, neither a NOAEL nor LOAEL were available for the belted kingfisher, or any other avian receptor.

The input parameters for the food chain models and Toxicity Reference Values are provided in Appendix T.

8.5.2.2 MEASUREMENT ENDPOINT DATA FOR TERRESTRIAL SOIL INVERTEBRATES

There are limited data in the scientific literature that can be used as measurement endpoints for the purpose of characterizing risks to soil invertebrates based on exposure to the soil COECs. Information sources reviewed to identify appropriate toxicological data included the U.S. EPA's TERRETOX database, USFWS reports, and EPA assessment and criteria documents. Table 8.32 identifies the measurement endpoints for soil invertebrates.

Measurement endpoints for the protection of earthworms due to the dichlorobiphenyl isomer and Aroclor 1254 are not available. According to Efroymson et al. (1997b), sufficient data to develop benchmarks for PCBs are not available.

Two sources identify measurement endpoints for mercury. The benchmark concentration identified by both ORNL (Will and Suter, 1996) and U.S. EPA Region 6 is 0.1 mg/kg.

8.5.2.3 MEASUREMENT ENDPOINT DATA FOR TERRESTRIAL PLANTS

As is the case for terrestrial invertebrates, there are limited data in the scientific literature that can be used as measurement endpoints for the purpose of characterizing risks to terrestrial plants based on exposure to the soil COECs. Information sources reviewed to identify appropriate toxicological data included the U.S. EPA's PHYTOTOX database, USFWS reports, and EPA assessment and criteria documents. Table 8.33 presents the measurement endpoints identified for terrestrial plants.

An ecological benchmark for plants is not available for the dichlorobiphenyl isomer. For Aroclor 1254, ORNL identifies 40,000 ug/kg as the ecological benchmark for the protection of plants. This value is used as the measurement endpoint for Aroclor 1254.

ORNL and U.S. EPA Region 6 identify 0.3 mg/kg as the ecological benchmark for the protection of plants for mercury. This value is used as the measurement endpoint for mercury.

8.5.2.4 MEASUREMENT ENDPOINT DATA FOR AQUATIC BENTHIC INVERTEBRATES

The COEC identification process for sediments involves comparison of maximum detected chemical concentrations in sediment to the most conservative sediment benchmarks available. Sediment benchmarks represent conservative measurement endpoints based on ecotoxicological data collected from a wide range of benthic invertebrate species. The sediment benchmarks used in the screening assessment are purposely conservative to identify chemical concentrations that have even the slightest potential to impact benthic invertebrate communities. However, exceedance of a single sediment benchmark value does not demonstrate that unacceptable risk is present. ORNL (Jones at al., 1997) recommends the use of multiple benchmarks for determining potential ecological effects to receptors. This approach provides a better indication of the likelihood and nature of effects. Jones et al. (1997) state that exceedance of only one conservatively estimated benchmark may provide weak evidence of real effects, whereas exceedance of multiple benchmarks of varying conservatism may provide stronger evidence of real effects.

As part of their sediment benchmark derivation process, the benchmark authors determined the chemical concentration range and the effects of those concentrations within the range, in essence, a range of measurement endpoints. A lower benchmark was then chosen to represent a lower percentile of that range (e.g., the NOAA ER-L or the MOE LEL). In some cases, an upper or median benchmark value was also established. For instance, NOAA derived an Effects Range Median (ER-M) benchmark that corresponds to the 50th percentile of the NOAA chemical data set range, and the MOE derived a Severe Effect Level (SEL) that corresponds to the 95th percentile of its respective chemical data set range.

For NOAA, concentrations equivalent to and above the ER-L, but below the ER-M, represent a possible effects range within which effects would occasionally occur, whereas concentrations equivalent to and above the ER-M represent a probable-effects range within which effects would frequently occur (Long et al., 1995). For the MOE, the LEL indicates a level of sediment contamination that can be tolerated by the majority of benthic organisms, while the SEL indicates a level at which pronounced impacts to the sediment-dwelling community can be expected (MOE, 1993). MOE offers little guidance on how to evaluate contaminant concentrations that fall between the usually large range of the LEL and SEL concentrations.

Table 8.6 presents the sediment benchmark values that were considered as part of the COEC selection process in the screening assessment for the Beaverdam Brook/Otter Kill

Area. In Section 8.4, Site-specific benchmarks were defined as the lowest benchmark presented in Table 8.6 greater than the maximum background concentration. To characterize risk to benthic invertebrates in the baseline assessment, ESVs from other sources are considered. These sources include the U.S. EPA's ARCS program (U.S. EPA, 1996), Canadian Council of Ministers of the Environment (CCME), MacDonald et al., (2000), U.S. EPA Office of Solid Waste and Emergency Response (OSWER), U.S. EPA Regions IV, V, and VI, and Washington State Department of Ecology. The ESVs from these sources for COPCs in sediments, along with the ESVs used in the screening assessment are summarized in Table 8.34. Three inorganics in sediment were carried forward to the baseline assessment for further evaluation: copper, selenium, and thallium. Of these three inorganics, an ESV is available only for copper. The range of all available ESVs was used as the measurement endpoint for copper. In the absence of ESVs, potential risk to benthic macroinvertebrates due to selenium and thallium is qualitatively addressed.

8.5.2.5 MEASUREMENT ENDPOINT DATA FOR AQUATIC FISH SPECIES

There are data in the scientific literature that can be used as measurement endpoints for the purpose of characterizing risks to fish based on exposure to the surface water COECs. Information sources that were reviewed to identify appropriate toxicological data included the U.S. EPA's AQUIRE database, USFWS reports, and other assessment and criteria documents in the scientific literature. Table 8.35 presents the measurement endpoint data identified for the fathead minnow or the generic fish group, and indicates the measurement endpoint concentration used to characterize risks to the fish of Beaverdam Brook and Otter Kill.

Lowest chronic values (LCVs) and test EC_{20}s derived by the ORNL (Suter and Tsao, 1996) were considered as appropriate measurement endpoint values for all surface water COECs. LCVs are potential lower benchmark values, whereas EC_{20} values are concentrations that cause less than a 20 percent reduction in (1) the weight of young fish per initial female fish in a lifecycle or partial life-cycle test or (2) the weight of young per egg in an early life-stage test (Suter and Tsao, 1996). The EC_{20} values are protective of fish populations and are roughly equivalent to LCVs.

8.6 STEP 7 - RISK CHARACTERIZATION

8.6.1 GENERAL

The risk characterization step of the ERA is intended to provide a narrative and tabular summary of the potential Site-related ecological risk for each evaluated exposure pathway and indicator species or ecological guild. Risk was characterized by comparing measurement endpoint values to Site-specific upper-bound (vagile receptors) or maximum detected (sedentary receptors) media concentrations. If an upper-bound or maximum concentration did not exceed its respective measurement endpoint value, then adverse ecological effects to the indicator species as a result of the COEC exposure, were determined not likely to occur. If an upper-bound or maximum concentration was in exceedance of its respective measurement endpoint value, then adverse effects to the indicator species, as a result of COEC exposure, may be occurring.

In evaluating the potential for risk, the risk characterization process considered the exposure environment, the individual and population exposure characteristics, and the spatial variation in receptor exposure. Additionally, the basis of the measurement endpoint value, the magnitude of the hazard quotient (HQ), and the conservatism of the assessment must also be considered.

8.6.2 RISK ESTIMATES FOR INSECTIVORES

Measurement endpoint values were identified for the shrew and woodcock, both insectivores, exposed to COECs through the consumption of earthworms and the incidental soil ingestion pathway. The measurement endpoint value represents either a NOAEL or LOAEL COEC dose level, which was compared to Site-specific COEC dose levels in order to estimate potential risk. Site-specific dose levels were estimated for both a mean and upper-bound exposure. Thus, four HQs were calculated for each COEC for the shrew and the woodcock, which represent the ratios of the mean to NOAEL, mean to LOAEL, upper-bound to NOAEL, and upper-bound to LOAEL. Each HQ was calculated as follows:

$$\text{HQ} = \text{COEC Dose}_{EST}/\text{COEC Dose}_{ME}$$

where:

HQ = hazard quotient (unitless);
COEC Dose_{EST} = estimated mean or upper-bound COEC dose level (mg/kg/day); and
COEC Dose_{ME} = measurement endpoint NOAEL or LOAEL COEC dose level (mg/kg/day).

Tables 8.23 and 8.24 present the HQ risk calculations for the shrew and the woodcock, respectively. An HQ of less than or close to one indicates that the indicator species is likely not at risk from the COEC exposure pathway, while an HQ greater than one indicates that the receptor may be at risk from COEC exposure. It should be noted that the baseline assessment has been conducted assuming 100 percent bioavailability and 100 percent area use for all indicator species. Consequently, HQ exceedances should not be viewed as clear indications of ecological impact. Instead, the varying exposure levels and the uncertainties in the ERA process should be considered.

Based on the estimated upper-bound dose levels and the NOAELs, HQs for the shrew exposed to PCBs (plus the dichlorobiphenyl isomer) and mercury were 63 and 7.1, respectively. These HQs indicate that risk to mammalian insectivores may be occurring from COEC exposure. However, the NOAELs used to estimate these HQs represent levels at which no effects are expected to occur, and thus are a conservative estimator of risk. When LOAELs are compared to the upper-bound dose levels, the HQs for PCBs and mercury drop to 6.3 and 1.4, respectively. Using the mean concentration and the LOAEL, the HQs for PCBs and mercury become 2.6 and 0.78, respectively. Thus, based on typical mean exposures, risk to mammalian insectivores may not be of concern at the Site, whereas risk from upper-bound exposures may be causing unacceptable risk. The NOAEL (0.07 mg/kg/day) and LOAEL (0.35 mg/kg/day) for mercury are based on the more toxic organic form. Analyses for speciation of mercury in soil indicates that that mercury at the Site is the less toxic inorganic form. The NOAEL for inorganic mercury is 2.86 mg/kg/day (Sample et al., 1996). Based on this NOAEL for inorganic mercury, HQs for the mean and upper-bound exposure estimates are 0.094 and 0.175, respectively. Therefore, it can be concluded that mercury does not pose an unacceptable potential for risk to mammalian insectivores. As discussed in greater detail in the uncertainties Section (see Section 8.7.2), the potential risk to receptors from PCBs in surface soils results from exposure to surface soils while spending time in the Lagoon 6 area. The remainder of the Site poses no risk to receptors from exposure to PCBs or mercury in surface soils.

Based on the estimated upper-bound dose levels and the NOAELs, HQs for the woodcock exposed to PCBs and mercury were 40 and 89, respectively. These HQs indicate that risk to avian insectivores may be occurring. However, as mentioned above, the use of NOAELs to estimate risks is a conservative practice. When LOAELs are compared to the upper-bound dose levels, the HQs for PCBs and mercury drop to 4.0 and 8.4, respectively. Using the mean concentration term and the LOAEL the HQs for PCBs and mercury become 1.2 and 4.8, respectively. The NOAEL (0.006 mg/kg/day) and LOAEL (0.064 mg/kg/day) for mercury are based on the more toxic organic form. The NOAEL and LOAEL for inorganic mercury are 0.45 mg/kg/day and 0.90 mg/kg/day, respectively (Sample et al., 1996). Based on the NOAEL for inorganic mercury, HQs for the mean and upper-bound exposure estimates are 0.69 and 1.2, respectively. Based on the LOAEL for inorganic mercury, HQs for the mean and upper-bound exposure estimates are 0.34 and 0.60, respectively. Thus, based on typical mean exposures, risk to avian insectivores from PCBs and mercury may not be of concern at the Site. Upper-bound exposures to PCBs and mercury may be causing unacceptable risk to the avian insectivores. The potential PCB and mercury risk to these receptors is present only upon exposure to Lagoon 6 area surface soils.

8.6.3 <u>RISK ESTIMATES FOR SMALL VERTEBRATE CARNIVORES</u>

Measurement endpoint values were identified for short-tailed weasel and red-tailed hawk exposed to COECs through the consumption of small mammals. As was discussed for the insectivores, HQs were calculated by comparing NOAEL and LOAEL measurement endpoint values for the COECs to mean and upper-bound Site-specific COEC dose levels. Tables 8.25 and 8.26 present the HQ risk calculations for the weasel and the red-tailed hawk, respectively.

Based on the estimated upper-bound dose levels and the NOAELs, HQs for weasel exposed to PCBs and mercury were 21 and 1.3, respectively. These HQs indicate that risk to small rodent mammalian carnivores may be occurring from PCB exposure, and possibly limited risk from exposure to mercury. However, as mentioned previously, the use of NOAELs to estimate risks is a conservative practice. When LOAELs are compared to the upper-bound dose levels, the HQ for PCBs drop to 4.3. Using the mean concentration term and the LOAEL the HQ for PCBs becomes 1.7. A LOAEL is not available for mercury. Based on typical mean exposures, risk to mammalian carnivores of small rodents may not be of concern at the Site, whereas risk from both upper-bound and mean exposures to PCBs may be causing unacceptable risk. The potential PCB risk to these receptors is present only upon exposure to Lagoon 6 area surface soils. If data from Lagoon 6 are excluded, the HQ for the most conservative scenario (95 percent UCL soil concentration and NOAEL), is 0.24. The remainder of the Site poses no risk to these receptors from exposure to PCBs in surface soils.

Based on the estimated upper-bound dose levels and the NOAELs, HQs for red-tailed hawk exposed to PCBs and mercury were 5.8 and 75, respectively. These HQs indicate that risk to avian carnivores of small rodents may be occurring. However, as mentioned previously, the use of NOAELs to estimate risks is a conservative practice. When LOAELs are compared to the upper-bound dose levels, the HQs for PCBs and mercury drop to 0.58 and 7.0, respectively. Using the mean concentration term and the LOAEL, the HQs for PCBs and mercury become to 0.24 and 3.0, respectively. The NOAEL (0.006 mg/kg/day) and LOAEL (0.064 mg/kg/day) for mercury are based on the more toxic organic form. The NOAEL and LOAEL for inorganic mercury are 0.45 mg/kg/day and 0.90 mg/kg/day, respectively (Sample et al., 1996). Based on the NOAEL for inorganic mercury, HQs for the mean and upper-bound exposure estimates are 0.42 and 1.0, respectively. Based on the LOAEL for inorganic mercury, HQs for the mean and upper-bound exposure estimates are 0.21 and 0.5, respectively. Therefore, it can be concluded that mercury does not pose an unacceptable potential for risk to avian carnivores. Based on typical average exposure, risk to avian insectivores to PCBs may not be of concern at the Site. Upper-bound exposures to PCBs may pose a slight risk to the avian carnivores. Because the HQs are close to unity, the likelihood of significant risk is low considering the conservatism of the evaluation. The potential PCB risk to these receptors is present only upon exposure to Lagoon 6 area surface soils.

8.6.4 RISK ESTIMATES FOR HERBIVORES

Measurement endpoint values were identified for meadow voles exposed to COECs through the consumption of plants. Hazard quotients were calculated by comparing NOAEL and LOAEL measurement endpoint values for the COECs to mean and upper-bound Site-specific COEC dose levels. Table 8.27 presents the HQ risk calculations for the meadow vole.

Based on the estimated upper-bound dose level and the NOAELs, the HQs for meadow vole exposed to PCBs and mercury were 4.6 and 1.4. These HQs indicate that risk to small mammalian herbivores may be occurring from PCB exposure, and possibly limited risk from exposure to mercury. However, as mentioned previously, the use of NOAELs to estimate risks is a conservative practice. When LOAELs are compared to the upper-bound dose levels, the HQ for PCBs and mercury drop to 0.46 and 0.29, respectively. The HQs for PCBs and mercury drop to 0.19 and 0.16, respectively, when mean exposure is considered. The NOAEL (0.054 mg/kg/day) and LOAEL

(0.269 mg/kg/day) for mercury are based on the more toxic organic form. The NOAEL for inorganic mercury is 2.18 mg/kg/day (Sample et al., 1996). Based on this NOAEL for inorganic mercury, HQs for the mean and upper-bound exposure estimates are 0.02 and 0.035, respectively. Therefore, it can be concluded that mercury does not pose an unacceptable potential for risk to mammalian herbivores. Based on typical mean exposures, risk to small mammalian herbivore due to PCBs is not of concern at the Site.

8.6.5 RISK ESTIMATES FOR PISCIVORES

8.6.5.1 SURFACE WATER COECS

Of the thee COECs identified for surface water (bis(2-ethylhexyl)phthalate, di-n-octylphthalate, and cyanide), measurement endpoints are available only for bis(2-ethylhexyl)phthalate and cyanide. The HQ risk calculations for these two surface water COECs for mink and belted kingfisher are presented in Table 8.28 and Table 8.29, respectively.

For bis(2-ethylhexyl)phthalate, the HQ for the upper- bound exposure dose for mink is 0.15 for the NOAEL. For belted kingfisher, HQs for the mean dose and upper-bound dose are 3.4 and 7.9, respectively, for the NOAEL. A LOAEL is not available for belted kingfisher. These HQs indicate that for bis(2-ethylhexyl)phthalate, there is no potential for risk to mammalian piscivores and some potential for risk to avian piscivores for both the upper-bound and lower-bound exposure doses.

Measurement endpoints are not available for di-n-octylphthalate. Consequently, the potential for risk to mammalian and avian piscivores could not be evaluated. However, di-n-octylphthalate, as well as bis(2-ethylhexyl)phthalate, was detected only once in ten samples from Beaverdam Brook/Otter Kill. Furthermore, bis(2-ethylhexyl)phthalate was detected in several background surface water samples at concentrations similar to that detected at sample location 7 in 1991. In addition, the 1991 data for the phthalates are suspect due to the kerosene-like hydrocarbon contamination in the original analysis and the reanalysis of the samples. Bis(2-ethylhexyl)phthalate and di-n-octylphthalate are common lab contaminants, as demonstrated by the presence of these phthalate esters in lab blanks associated with the surface water samples and several groundwater samples. These phthalate compounds were not detected in lab blanks or the investigative surface water samples collected in 1995. The issue of potential laboratory error was presented to U.S.EPA / NYSDEC in a letter dated March 23, 2001. Based on the infrequency of detection and questionable results for the 1991 samples in which the two phthalates were detected, it can reasonably be concluded that neither di-n-octylphthalate nor bis(2-ethylhexyl)phthalate pose risk to either mammalian or avian piscivores.

For cyanide, the HQ for the upper- bound exposure dose for mink is 0.039 for the NOAEL. Consequently, there is no risk to mammalian piscivores due to cyanide. Measurement endpoints are not available for belted kingfisher. Cyanide was detected in only one of ten samples (sample SWII-4 for medium flow condition) at a concentration of 54.4 ug/L, but was not detected in a duplicate sample at a detection limit of 10 ug/L. This non-detect duplicate result puts the positive detection into question. Cyanide was also not detected in samples collected at this same sampling location (SWII-4) during low and high flow conditions at about the same time period. Consequently, the infrequent presence of cyanide in Beaverdam Brook/Otter Kill, in addition to the background conditions and temporal results at the Site, puts into question any potentially significant risks to avian piscivores due to cyanide.

8.6.5.2 SEDIMENT COECS

Potential risk to mammalian and avian piscivores were evaluated for copper, selenium, and thallium in sediments of Beaverdam Brook/Otter Kill. The results for this evaluation are presented in Table 8.30 and Table 8.31 for mink and belted kingfisher, respectively.

For copper, the HQs for both the mean and upper-bound exposure doses of mink were below one, indicating a limited potential for risks. Hazard quotients range from 0.12 for the mean exposure concentration and LOAEL to 0.24 for the upper 95 percent UCL exposure dose and LOAEL. For belted kingfisher, the HQs range from 0.26 mean exposure concentration and LOAEL to 0.52 for the upper-bound exposure dose and NOAEL. These results suggest a limited potential for risk to both mammalian and avian piscivores due to exposure to copper in the sediments of Beaverdam Brook/Otter Kill.

For mink, the HQ for the upper-bound exposure dose and NOAEL is 3.2 for selenium. The HQ for the upper bound dose and LOAEL is 1.9. For the mean exposure dose, the HQs for the NOAEL and LOAEL are 2.2 and 1.3, respectively. These HQs indicate that, under the conservative area-use and bioconcentration factors, there is a potential for risk to mammalian piscivores due to exposure to selenium in sediments.

For belted kingfisher, the HQs for selenium for the upper-bound exposure dose are 31 and 12 for the NOAEL and LOAEL, respectively. For the lower-bound exposure dose, the HQs drop to 21 and 8.4 for the NOAEL and LOAEL, respectively. These HQs indicate that the potential for risk to avian piscivores exists due to selenium in the sediments of Beaverdam Brook/Otter Kill.

For thallium, measurement endpoints are available only for mink. Measurement endpoints are not available for belted kingfisher or any other avian species. Hazard quotients for the upper-bound exposure dose are 140 and 14 for the NOAEL and LOAEL, respectively. For the mean exposure dose, the HQs are 97 and 10 for the NOAEL and LOAEL, respectively. These HQs suggest a potential for risk to mammalian piscivores.

8.6.6 RISK ESTIMATES FOR THE SOIL INVERTEBRATE COMMUNITY

Measurement endpoint values were identified for the soil invertebrate community exposed to COECs in surface soils on Site. HQs were calculated by comparing measurement endpoint values to the maximum detected surface soil concentrations in order to estimate potential risk. Table 8.32 presents the HQ risk calculations for the soil invertebrate community.

The HQ for mercury is 182. The maximum (18.2 mg/kg) concentration of mercury exceeds the measurement endpoint (0.1 mg/kg). Concentrations of mercury in 13 of the 21 samples were greater than 0.1 mg/kg. Samples with the highest concentrations were collected from Lagoon 6. Exclusive of samples from Lagoon 4 (0.81 mg/kg), Lagoon 5 (1.6 mg/kg), and Lagoon 6, concentrations of mercury were relatively close to 0.1 mg/kg (0.12 mg/kg to 0.31 mg/kg). Samples with the highest concentrations of mercury also had the highest concentrations of Aroclor 1254. The potential for risk to soil invertebrates from mercury exposure may be present in localized areas where soil concentrations exceed 0.10 mg/kg. In general, the main environmental concern about earthworms and high soil mercury concentrations is that the metals will accumulate to concentrations toxic to predators, rather than the metals being toxic to earthworms (Beyer, 1990). Potential risk to predators that consume earthworms was evaluated in Section 8.6.2.

Measurement endpoint values were not identified for the dichlorobiphenyl isomer and Aroclor-1254. Therefore, it is unknown whether the soil invertebrate community is at risk from the exposure to these COPCs in surface soils. The dichlorobiphenyl isomer is a TIC and thus its actual presence in Site soils is questionable. It is noted that all of the TIC COPCs were detected in two soil samples (TP-41 and SSII-6) collected from the Lagoon 6 area on Site. Additionally, the maximum concentrations of Aroclor-1254 and mercury, as well as most other organic chemicals, were detected in one of these two samples.

Table 8.36 presents the surface soil dataset for the Site excluding data from the Lagoon 6 area, as well as the measurement endpoints identified in Section 8.4 and HQs for the maximum, mean, and 95 percent concentrations. The Lagoon 6 area is comprised of approximately 0.23 acre, or less than 1 percent of the total Site area. Maximum concentrations were taken to represent the exposure concentrations in order to characterize risk to the soil invertebrate community. As a result, it is apparent that potential risk to the soil invertebrate community resulting from exposure to organic COECs and to inorganic mercury is a concern only for those receptors inhabiting soils within the Lagoon 6 area. Potential risk to the greater soil invertebrate populations outside of the Lagoon 6 area on or in the vicinity of the Site are insignificant. Based on the MPC as the measurement endpoint identified in Section 8.4, the HQs for the maximum, mean, and 95 percent UCL concentrations for mercury are 0.73, 0.11, and 0.20, respectively.

8.6.7 RISK ESTIMATES FOR THE TERRESTRIAL PLANT COMMUNITY

Measurement endpoint values were identified for the terrestrial plant community exposed to COECs in surface soils on Site. HQs were calculated by comparing measurement endpoint values to the maximum detected surface soil concentrations in order to estimate potential risk. Table 8.33 presents the HQ risk calculations for the terrestrial plant community.

For mercury, the estimated HQ is 61. This HQ is based on the maximum concentration of 18.2 mg/kg, which was detected in sample SSII-6 from Lagoon 6. Concentrations of mercury in three other samples exceeded the measurement endpoint of 0.3 mg/kg: sample TP-41 from Lagoon 6 (6.9 mg/kg), sample SSII-5 from Lagoon 5 (1.6 mg/kg), and sample SSII-4 from Lagoon 4 (0.81 g/kg). There is a potential for risk to plants inhabiting the surface soils of the Lagoon 6 area where the two highest concentrations of mercury were detected. There is also a potential risk to plants in localized areas outside of Lagoon 6 where soil concentrations of mercury exceed 0.3 mg/kg (SSII-5 at Lagoon 5 and SSII-4 at Lagoon 4). The remainder of the Site poses little or no risk to the terrestrial plants from exposure to mercury in surface soils.

The maximum detected concentration of Aroclor 1254 in surface soil (9,200 ug/kg) is an order of magnitude below the benchmark for protection of terrestrial plants

(40,000 ug/mg). Therefore, it can be concluded that Aroclor 1254 does not pose risk to terrestrial plants.

A measurement endpoint value was not identified for the bipyridine isomer. Therefore, it is unknown whether plant community is at risk from the exposure to this COEC in surface soils. This constituent was detected in sample TP-41 from Lagoon 6, where the highest concentrations of mercury and PCBs were also detected.

As was discussed for the soil invertebrates, the maximum concentrations of Aroclor 1254, dichlorobiphenyl isomer, and mercury, as well as most other organic chemicals, were detected in samples collected from the Lagoon 6 area. As a result, it is apparent that potential risk to the terrestrial plant community resulting from exposure to organic COECs and to inorganic mercury is a concern only for those receptors inhabiting soils within the Lagoon 6 area. Potential risk to plants outside of the Lagoon 6 area on or in the vicinity of the Site are insignificant.

It is noted that there were no signs of stressed vegetation or other adverse effects to plant growth at the Site. This is evidence in support of the conclusion that the thriving plant community on Site has generally not been significantly impacted by the present surface soil conditions.

8.6.8 RISK ESTIMATES FOR AQUATIC BENTHIC INVERTEBRATES

The potential for risk to benthic macroinvertebrates was evaluated using multiple benchmark values, which provides an indication of the likelihood and nature of effects. An exceedance of only one conservative benchmark provides weak evidence of real effects, whereas exceedances of multiple benchmark values of varying conservatism may provide stronger evidence of potential effects (Jones et al., 1997). As a result, in order to provide a stronger indication of the likelihood of potential effects, a comparison of sediment concentrations was conducted using additional sediment screening benchmarks of varying conservatism. The additional sediment screening benchmarks considered are identified in Table 8.34. To evaluate the potential for risk, the maximum detected sediment COEC concentrations were compared to the range of sediment screening benchmarks. The number and locations of sediment concentrations outside the range of sediment screening benchmarks were also considered.

Table 8.37 presents the sediment HQ calculations for the Beaverdam Brook/Otter Kill area. Sediment screening benchmarks for copper range from 16 mg/kg (U.S. Region V ESL) to 390 mg/kg (Washington No Effects Level (NEL)). Eighteen of the 26 sediment samples analyzed for copper exceed the low-end benchmark (16 mg/kg), whereas only four samples exceed the high end benchmark (390 mg/kg). The four samples that exceed the Washington NEL are SED02 (433 mg/kg), SED09 (457 mg/kg), SED10 (462 mg/kg), and SED11 (685 mg/kg). These results indicate that there may be risk to benthic macroinvertebrates due to copper at some locations within the Beaverdam Brook/Otter Kill area.

The source of the elevated concentrations of copper in Beaverdam Brook/Otter Kill sediments does not appear to be attributable to the Site. A soil-to-sediment chemical transport pathway is likely present at the Site. However, copper was not identified as a COEC for on-Site soil. Concentrations of copper detected in the surface water of Beaverdam Brook/Otter Kill generally fall in the same range as those detected at background surface water locations. Thus, it does not appear that the Site surface soils are the primary source of the elevated concentrations of copper detected in the sediment of Beaverdam Brook/Otter Kill.

The only sediment screening benchmarks for selenium and thallium are MPCs identified by Crommentuijn et at (1997). The benchmarks for selenium and thallium are 2.9 mg/kg and 2.6 mg/kg, respectively. Selenium was detected in three of 19 samples. Only the sample collected at SED05 exceeded the MPC. Thallium was detected in 12 of 24 samples. The maximum concentration (18.1 mg/kg), which exceeds the MPC of 2.6 mg/kg, was detected in sample SED09. The concentration of thallium in sample SED20 (2.9 mg/kg) slightly exceeds the MPC. Concentrations of thallium in all other samples were below the MPC. Therefore, any risk benthic to macroinvertebrates due to selenium and thallium is very localized.

Statistical correlation can be used to evaluate the potential sources of selenium and thallium in sediment. Aluminum and iron were eliminated as COPCs because their on-Site concentrations are within the range of background concentrations. As shown in Figure 8.3, there is a very strong statistical correlation between iron and aluminum (Pearson product-moment correlation, $p < 0.001$). The correlations between selenium and aluminum ($r=0.405$; $p < 0.05$) and thallium and aluminum ($r=0.441$; $p < 0.05$) are statistically significant. Based on the significant correlation, it can reasonably be concluded that selenium and thallium are from natural sources.

8.6.9 RISK ESTIMATES FOR AQUATIC FISH COMMUNITY

Measurement endpoint values were identified for the fish, specifically for the fathead minnow, which may be exposed to COECs in the surface water of Beaverdam Brook and

Otter Kill. HQs were calculated by comparing measurement endpoint values to the upper-bound surface water concentrations, represented by the lower of the 95 percent UCL concentration or the maximum concentration. Table 8.35 presents the HQ risk calculations for the fish community.

The HQ for di-n-octylphthalate is below 1 (0.35), indicating that the risk to fish from exposure to this COEC is not a concern at the Site. The estimated HQs for bis(2-ethylhexyl)phthalate (1.9) and cyanide (2.3) are in exceedance of one, indicating that unacceptable risk to the fish community may be occurring in these waters. However, both cyanide and bis(2-ethylhexyl)phthalate were detected only once in ten samples from Beaverdam Brook/Otter Kill. In addition, bis(2-ethylhexyl)phthalate was detected in several background surface water samples at concentrations similar to that detected at sample location 7 in 1991. As discussed in Section 8.6.5.1, phthalates are a common laboratory contaminant and their actual presence in the surface water of Beaverdam Brook/Otter Kill and, thus, the potential for risk to fish is questionable. Similarly, the presence of cyanide and subsequent potential for risk to fish due to cyanide is questionable.

Evaluation of risk to fish using benchmarks for surface water does not take into account potential risk due to consumption of invertebrates in the water column or sediment that may have bioaccumulated COECs. Although the concentrations of COECs can be calculated for aquatic vertebrates, exposure factors (e.g., consumption rates) and TRVs are generally not available for fish. Due to the absence of reliable exposure factors and TRVs, a quantitative evaluation of risk is not possible. However, the potential risk due to the consumption of aquatic invertebrates can be qualitatively addressed.

For surface water, the presence of all three COECs (bis(2-ethylhexyl)phthalate, di-n-octylphthalate, and cyanide) is questionable. As discussed previously, each COEC was detected in only one of ten samples. The presence of the two phthalates is most likely due to laboratory contamination. The presence of cyanide is brought into question by the absence of a detectable concentration in the duplicate of the one sample in which cyanide was detected. For the sediment COECs (copper, selenium, and thallium), the BCFs for sediment to benthic invertebrates are relatively low. The BCFs range from 0.1 or copper to 1.0 for selenium. Based on the questionable presence of the three surface water COECs and relatively low BCFs for the three sediment COECs, the potential for risk to fish due to consumption of aquatic invertebrates is likely to be low.

8.7 UNCERTAINTIES ANALYSIS

8.7.1 APPLIED DATASET

The surface soil dataset consisted of surface soil samples collected from 0 to 3 inches and borehole soil samples collected from 0 to 2 feet. Because soil data from 0 to 1 foot is considered most appropriate for assessing ecological concerns, the deeper soil sample depth of 0 to 2 feet introduces an unknown degree of uncertainty into the ERA. The 0 to 2 feet soil samples were added to the surface soil dataset to allow for better spatial representation of Site conditions than would have been allowed if only the 0 to 3 inch soil samples were used to characterize surface soil conditions. The use of these additional deeper soil samples is intended to reduce the uncertainty of using a smaller more limited dataset, thus providing a more complete sample set to adequately assess potential ecological risk.

8.7.2 LAGOON 6 SITE AREA

The Lagoon 6 area comprises an area of approximately 0.23 acre, or less than one percent of the total Site area. The maximum detected concentration for almost every detected organic chemical was observed in samples collected from the Lagoon 6 area. Table 8.36 presents a summary of the surface soil dataset with two Lagoon 6 samples removed. A comparison of Table 8.36 to Table 8.7 reveals that if data from Lagoon 6 were removed from the dataset, only inorganic parameters would be identified as COECs, the majority of which are present at concentrations comparable to background chemical conditions. The potential risk estimated for wildlife, plants, and invertebrates exposed to surface soils results primarily from exposure to mercury and PCBs. Without exposure to Lagoon 6 surface soil, potential risk to these receptors from mercury and PCBs becomes insignificant.

8.7.3 EXPOSURE ASSUMPTIONS

In general, conservative measurement endpoint values were applied to exposed species without adjusting for Site-specific area-use factors. The conservatism is enhanced by applying these factors to the most exposed individuals. Chemical bioavailability and absorption efficiencies were all assumed to be 100 percent, and the diet was assumed to consist totally of contaminated food items. These assumptions exaggerate estimates of risk. In addition, sampling was generally biased toward known or suspected source areas, which inflates estimates of most reasonable exposure. On the other hand, no interspecies application factors were used which could underestimate risk as there may be more sensitive species not represented by the often-limited toxicological data. Also, some pathways (i.e., inhalation, dermal absorption, etc.) were not considered in the evaluation, which might underestimate risk slightly.

Area use is an important factor to consider when estimating the extent of exposure to a particular COEC. The assumption that all indicator species spend 100 percent of their time at the Site in, for example, the Lagoon 6 area, will overestimate exposure and risk for the indicator species. Soil invertebrate and plant receptors are relatively immobile in soils, and therefore, it is reasonable to assume that they are exposed only to those areas of maximum contamination. However, for large, free-ranging vertebrate receptors, such as mink, belted kingfisher, and red-tailed hawk, it is reasonable to assume that they will not spend 100 percent of their time on the Site, or on the Site at all. The home ranges for the higher trophic species can be very large and may encompass areas much larger than the Site. For example, mink may have an average territory size ranging from 30 to 1,900 acres, whereas red-tailed hawk may have an average territory size ranging from 150 to 400 acres (U.S. EPA, 1993a).

The Site area consists of approximately 30 acres, in which it was assumed all wildlife exposure occurred, although more reasonable exposure assumptions could be applied to estimate exposure concentrations. The length of shoreline of Beaverdam Brook/Otter Kill with the highest concentrations of COECs is approximately 400 feet, which is substantially smaller than the length of stream within the foraging range of mink and belted kingfisher. The wildlife exposure models for red-tailed hawk, mink, and belted kingfisher could be modified using reasonable area-use factors that reflect the proportion of Site habitat in relation to the average home range size or length of stream within foraging range. The modified exposure factors are calculated as follows:

- red-tailed hawk exposure factor = 30 acres/150 to 400 acre home range = 0.2 to 0.75.
- mink exposure factor = 400 feet/4,045 to 8,630 feet of foraging range = 0.049 to 0.099.
- belted kingfisher exposure factor = 400 feet/1,280 to 7,185 feet of foraging range = 0.056 to 0.31.

Table 8.38 summarizes the application of the exposure factors to the risk estimates calculated previously in Tables 8.26 (red-tailed hawk), 8.28 (mink), and 8.31 (belted kingfisher). The TRV for red-tailed hawk identified in Table 8.38 is based on the more toxic organic form of mercury. Risk to ecological receptors is dependent on the size of their habitat range; the larger the range, the less COEC exposure, and the lower the risk. Site-specific data are not available for home range sizes of red-tailed hawk, mink, and belted kingfisher. The exposure-adjusted HQ ranges for red-tailed hawk exposed to PCBs and mercury range from 0.05 to 0.44 and from 0.60 and 5.2, respectively. Thus, using reasonably estimated exposure factors, the risk to avian carnivores from PCB exposure at the Site is not a concern. For mercury, the exposure-adjusted HQs for red-tailed hawk are greater than one for the maximum exposure factor (0.75) and greater than one for both the minimum (0.2) and maximum (0.75) exposure factors. These results indicate that there may be risk to red-tailed hawk from mercury. However, as discussed in Section 8.6, analyses for speciation of mercury indicate that mercury in soil is the less toxic inorganic form. When a NOAEL based on inorganic mercury is considered (0.45 mg/kg/day), the HQ for the upper-bound ingestion scenario (95 percent UCL) and maximum area use (75 percent) is 0.75.

For mink, HQs adjusted for foraging area range from a low of 0.001 for cyanide in surface water to a maximum of 1.4 for thallium in sediments. For belted kingfisher, HQs adjusted for foraging area range from a low of 0.04 for copper in sediments to a maximum of 3.4 for thallium in sediments. Therefore, using reasonable area-use factors, the potential for risk to mammalian and avian piscivores is limited.

Additional exposure characteristics to consider involve the population densities of the indicator species. For the mink and red-tailed hawk, population densities typically range from 0.01 to 0.1 animals/ha and 0.0010 to 0.0018 animals/ha, respectively (U.S. EPA, 1993a). The density of breeding pairs of belted kingfisher ranges from two to six breeding pairs per 10 km of stream length. Therefore, in the Site area it could be expected that only one breeding mink, one breeding red-tailed hawk, and one breeding pair of belted kingfisher would be present on the entire area. Consequently, because of the small exposure area associated with the Site, significant risk to the entire populations of mink, red-tailed hawk, and belted kingfisher populations is not expected due to the relatively low population densities and small area of the Site.

8.7.4 EVALUATION OF TENTATIVELY IDENTIFIED COMPOUNDS (TICS)AND OTHER ORGANICS WITHOUT ECOLOGICAL BENCHMARKS

To maintain a conservative approach in the ERA, detected TICs in all media were evaluated in the ERA. In most cases, ecotoxicological data are not available for TICs, and therefore, surrogate chemical toxicity data were applied where appropriate and available. The surrogates identified in the problem formulation and BERA were selected in conjunction with U.S. EPA during preparation of the human health risk assessment. Ecological benchmarks were available for the surrogates for dichlorobiphenyl isomer (diphenylamine), methyl ester benzoic acid (diethyl phthalate), the ethylmethylbenzene isomer (cyclohexylamine), and the 4-hydroxy-4-methyl-2-pentanone (methyl methacrylate). The degree to which the surrogates accurately represent the mechanisms of the fate and transport and mechanism of ecotoxicity of the TICs is uncertain.

The surrogates identified for 1,2-propanedione, 1-phenyl (benzaldehyde), 2-butyl pyridine (terbutryn), 2-ethenyl pyridine (terbutryn), the bipyridine isomer (terbutryn), the methyl phenanthrene isomer (biphenol A), and the two ethylmethylcyclohexane isomers do not have screening values or other benchmarks appropriate for the evaluation of risk to ecological receptors. Appropriate surrogates were not identified for benzoic acid, 1-propanone, 1-(3-pyridinyl), and the two chlorothioxanthenone isomers. No further evaluation was conducted for these organic constituents. The actual toxicity of these organic constituents to ecological receptors exposed to soil, surface water, and sediment is uncertain.

8.7.5 MEASUREMENT ENDPOINT VALUES

Measurement endpoint values could not be identified in the ERA for some COECs. COECs without measurement endpoint values included:

- di-n-octylphthalate for mammalian and avian piscivores,
- thallium for avian piscivores,
- the dichlorobiphenyl isomer for mammalian and avian piscivores, mammalian and avian carnivores, and mammalian herbivores (assumed to have a similar mechanism of toxicity as PCBs),
- the dichlorobiphenyl isomer; and Aroclor 1254 for soil invertebrates, and
- the dichlorobiphenyl isomer for terrestrial plants.

Thus, uncertainty of an unknown magnitude is related to this data gap. However, background data were applied in the ERA to qualitatively evaluate the potential risk posed from exposure to COECs without measurement endpoints, and provided at least some indication of potential risk.

8.8 ECOLOGICAL RISK ASSESSMENT SUMMARY

The ERA conducted for the Site consisted of a screening level assessment and subsequent baseline assessment. The results of the screening assessment identified the need to conduct a more comprehensive baseline assessment based on the identification of COPCs.

The exposure pathways that were quantified as part of the baseline assessment included:

- mammalian and avian insectivores exposed to COECs through the consumption of earthworms and through the incidental ingestion of surface soils;
- mammalian and avian carnivores exposed to COECs through the consumption of small vertebrate prey;
- mammalian herbivores exposed to COECs through the consumption of terrestrial plants and through incidental ingestion of surface soils;
- mammalian and avian piscivores exposed to COECs in surface water and sediments through the consumption of fish from Beaverdam Brook and Otter Kill;
- soil invertebrates exposed to COECs in Site surface soils;
- terrestrial plants exposed to COECs in Site surface soils;
- aquatic benthic invertebrates exposed to COECs in sediment of Beaverdam Brook and Otter Kill; and
- fish exposed to COECs in surface water of Beaverdam Brook and Otter Kill.

The results of the risk characterization indicated that, based on typical mean exposures to PCBs and mercury, risk to mammalian insectivores is most likely not a concern at the Site, whereas risk from upper-bound exposures may pose marginal potential for risk. The potential risk to the mammalian insectivores from PCB and mercury exposure is primarily due to the concentrations of these COECs in the Lagoon 6 surface soils. However, considering the population densities of potential receptors and the fact that mammalian insectivores have a much larger home range than the areas of Lagoon 6, the potential for ecological risks is negligible. In addition, the TRVs for mercury are based on the more toxic organic form of mercury. The predominant form of mercury at the Site is inorganic mercury. If TRVs for inorganic merury are considered, risk from upper bound exposure to mercury is most likely not a concern at the Site. The remainder of the Site poses no risk to these receptors from exposure to PCBs, mercury, or any other parameter detected in surface soils.

For the avian insectivore, risk from mean PCB exposure is most likely not a concern at the Site, while risk resulting from typical mean mercury exposure may be a concern at the Site. Upper-bound exposures to PCBs and mercury may be causing negligible risk to avian insectivores, but only through exposure to surface soils from within the Lagoon 6 area. However, considering the population densities of potential receptors and the fact that avian insectivores have much larger home ranges than the areas of Lagoon 6, the potential for ecological risks is negligible. In addition, the TRVs for mercury are based on the more toxic organic form of mercury. If TRVs for inorganic merury are considered, risk from upper bound exposure to mercury is most likely not a concern at the Site. The remainder of the Site poses no risk to these receptors from exposure to PCBs, mercury, or any other parameter detected in surface soils.

The results of the risk characterization indicated that, based on mean exposures to PCBs, risk to mammalian carnivores is most likely not a concern at the Site, whereas risk from upper-bound exposures to PCBs may pose a marginal potential for risk, only through exposure to surface soils from within the Lagoon 6 area. However, considering the population densities of potential receptors and the fact that mammalian carnivores have much larger home ranges than the areas of Lagoon 6 (i.e., it was assumed in the ERA that ecological receptors spend 100 percent of their time in Lagoon 6), the potential for ecological risks is negligible. The remainder of the Site poses no risk to these receptors from exposure to PCBs or any other parameter detected in surface soils.

For the avian carnivores, risk from mean and upper-bound PCB exposure is most likely not a concern at the Site. The upper-bound estimate of exposure to mercury indicates a potential for risk to avian carnivores. There is relatively low potential for risk based on the mean exposure, with the potential being substantially lower outside of Lagoon 6 area. Hazards quotients for avian carnivores based on the mean concentration of mercury, exlcuding samples for Lagoon 6, are 4.8 and 0.45 for the NOAEL and LOAEL, respectively. These values are an order of magnitude less than the HQs produced by the data set that includes data from Lagoon 6 (31.8 for the NOAEL and 3.0 for the LOAEL). When area use is taken into account, the HQs are reduced to 0.95 to 3.6 for the NOAEL and 0.09 to 0.34 for the LOAEL. These HQs are based on TRVs for the more toxic organic form of mercury. If TRVs for inorganic mercury are considered, the HQs for all scenarios are below one. The remainder of the Site poses no risk to avian carnivores from exposure to PCBs, or any other parameter detected in surface soils.

For mammalian herbivores, there is a slight potential for risk for both PCBs and mercury at the upper-bound exposure. For mean exposure, there is no risk to mammalian herbivores.

For mammalian predators of fish in Beaverdam Brook and Otter Kill, there is no risk due to bis(2-ethylhexyl)phthalate and cyanide in surface water, even at the upper-bound exposure. Similarly, there is no risk to mammalian piscivores due to copper in sediment at either the mean or upper-bound exposure. For selenium and thallium, there is a potential for risk to mammalian piscivores under the assumption of 100 percent area use at both the upper-bound and mean exposure. However, when adjusted for area use, there is little to no potential for risk to mammalian piscivores due to selenium and thallium in the sediments of Beaverdam Brook and Otter Kill.

For avian piscivores, there is a slight potential for risk due to bis(2-ethylhexyl)phthalate at both mean and upper-bound exposure, assuming 100 percent area use. There is no risk when the size of the Site relative to total foraging range is considered. For copper, there is a negligible potential for risk at both the upper-bound exposure and mean exposure. For selenium, there is a potential for risk to avian piscivores at both the upper-bound and mean exposure, assuming 100 percent area use. Selenium does not pose risk to avian piscivores when exposure is adjusted for the size of the Site relative to total foraging area.

The results of the risk characterization indicated that some risk may be occurring to the soil invertebrate community from exposure to mercury in the surface soils of the Lagoon 6 area. The remainder of the Site poses no significant risk to these receptors from exposure to COECs in surface soils.

The results of the risk characterization indicate that some risk may be occurring to the terrestrial plant community from exposure to mercury in surface soils. The potential risk posed by mercury exposure to the plant community is due to mercury levels detected in the Lagoon 6 area, which comprises less than one percent of the Site area. The remainder of the Site poses no significant risk to these receptors from exposure to mercury or any other parameter detected in surface soils.

The results of the risk characterization indicate that some risk may be occurring to the benthic invertebrate community in Beaverdam Brook/Otter Kill from exposure to elevated levels of copper, selenium, and thallium in sediments. Any risk to benthic invertebrates is limited to small isolated patches of sediment, as only a small number of sampling locations had concentrations outside the range sediment benchmarks. The source of these three inorganics in the sediment is most likely not related to historical Site activities involving the lagoons. Concentrations of these inorganics in Site surface soils are generally consistent with background surface soil conditions, and thus, do not appear to be the source of the COEC levels detected in the Beaverdam Brook/Otter Kill sediments.

The results of the risk characterization indicate that some risk may be occurring to the aquatic fish community in Beaverdam Brook/Otter Kill from exposure to cyanide and bis(2-ethylhexyl)phthalate in surface water. However, both cyanide and bis(2-ethylhexyl)phthalate were detected only once in ten samples collected from

Beaverdam Brook/Otter Kill. In addition, bis(2-ethylhexyl)phthalate was detected at comparable levels in background surface water samples, and is therefore, likely unrelated to the historic Site activities and/or may be the result of lab error. The cyanide detection is questionable due to a non-detect result in a duplicate sample, and two non-detect results from surface water samples collected from the same location close in time to the positive result sampling event. Consequently, although some risk is indicated for fish in Beaverdam Brook/Otter Kill from exposure to cyanide and bis(2-ethylhexyl)phthalate, it appears that the presence of these two COECs in the surface water are unrelated to former Site activities.

In summary, the results of this ERA indicate that risks or impacts to ecological receptors exposed to Site-related chemicals are negligible. The majority of identified risk to ecological receptors is associated with Lagoon 6 surficial soils, which comprise 1 percent of the total Site area.

9.0 HUMAN HEALTH RISK ASSESSMENT

OVERVIEW

The Human Health Baseline Risk Assessment (HHRA), presented herein, supersedes the HHRA completed for the Site, as part of the previous RI Report (CRA, 1996) dated March 18, 1996.

A human health baseline risk assessment is an evaluation of the risks, or potential risks, to the public health and welfare, posed by the Site, if left unremediated. It provides an assessment of the contaminants of concern, potential exposure pathways, potential human receptors, and an analysis of the risks resulting under the Site conditions. The purpose of the baseline risk assessment is to define potential Site-related and background exposures and estimated impacts on human health.

The HHRA provides an examination of the potential public health risks and/or hazards, if any, which could result from the exposure to chemicals reported in soils, sediments, surface water, and groundwater (including those which may be mobilized to air) at, or in the vicinity of, the Site. Based on the analytical data, Lagoon 6 had been identified as a potential "hot spot" in the previous HHRA, and thus, was evaluated as a separate area of concern apart from the rest of the Site soil area. Consistent with the format of the previous HHRA, Lagoon 6 has been evaluated as a separate area of concern apart from the rest of the Site soil area. Table 1.0 of Appendix Q.1 presents the selection of exposure pathway scenarios evaluated in the HHRA.

The risk assessment presented herein is consistent with current guidance, including, but not limited to the following:

- U.S. EPA Risk Assessment Guidance for Superfund (RAGS), Volume 1, Human Health Evaluation Manual (Part A), Interim Final, EPA/540/1-89/002, December 1989;
- U.S. EPA Calculating the Concentration Term: Supplemental Guidance to RAGS, EPA Publication 9285.7, May 1992;
- U.S. EPA Calculation Upper Confidence Limits for Exposure Point Concentrations at Hazardous Waste Sites, OSWER 9285.6-10 December 2002.
- U.S. EPA Exposure Factors Handbook, EPA/600/P-95/002Fa, August 1997;
- U.S. EPA Human Health Evaluation Manual, Supplemental Guidance: Standard Default Exposure Factors, OSWER Directive 9285.6-03, March 25, 1991;

- U.S. EPA RAGS Part D, Standardized Planning, Reporting, and Review of Superfund Risk Assessments, Interim Guidance, Publication 9285.7-01D, January 1998;
- U.S. EPA Role of Background in the CERCLA Cleanup Program, OSWER 9285.6.07P, May 2002;
- U.S. EPA Integrated Risk Information System (IRIS) database, 2004;
- U.S. EPA Region 3 Risk Based Concentration (RBC) Table, October 8, 2004; and
- U.S. EPA RAGS Part E: Supplemental Guidance for Dermal Risk Assessment, EPA/540/R/99/005, July 2004.

ORGANIZATION OF THE BASELINE RISK ASSESSMENT REPORT

The organization of the HHRA follows the general outline presented as Exhibit 9-1 of RAGS, which includes the following main sections:

- Characterization of Exposure Setting;
- Identification of Chemicals of Potential Concern;
- Exposure Assessment;
- Toxicity Assessment; and
- Risk Characterization.

The characterization of the exposure setting includes descriptions of the following general physical Site characteristics:

- climate;
- meteorology;
- geologic setting;
- vegetation;
- hydrogeology;
- hydrology;
- land use (current and future); and
- identification of populations using the Site.

The findings of the physical investigations of this Site are presented previously in Sections 3.0, 4.0, 5.0, and 6.0. This HHRA utilized the results of the RI to identify potential exposure pathways, and chemical concentrations, which were used to quantify the health risks, associated with complete exposure pathways.

The identification of COPCs involves several objective approaches that numerically evaluate the presence, distribution, concentration, and toxicity of chemicals detected in Site and background media and, by applying selected criteria, identifies those chemicals which are most likely to pose the majority of potential health risks.

The exposure assessment includes the identification of exposure pathways. Exposure pathways consist of the following four elements:

1. a source and mechanism of chemical release;
2. a retention or transport medium;
3. a point of potential human contact with the contaminated medium; and
4. an exposure route at the contact point.

Due to varying concentrations of constituents, as well as different potentials for human exposure throughout various portions of the Site, the HHRA has been conducted by incorporating an evaluation of each potential contaminant source area (consistent with the U.S. EPA RAGS Guidance). The distribution of contaminants throughout the various media was discussed in Section 6.0. The exposure assessment also includes the determination of the exposure point concentrations and the pathway-specific intake constituents. In general, the guidance provided for determining the quantification of exposure requires that the ***"intake variable values for a given pathway should be selected so that the combination of all intake variables results in an estimate of the reasonable maximum exposure for that pathway"*** **(RAGS, 1989a page 6-19)"**. This is referred to as the Reasonable Maximum Exposure (RME).

The toxicity assessment involves the identification or derivation of toxicity factors in order to evaluate the potential of the COPCs to cause adverse health effects, and to estimate the relationship between contaminant exposure and the increased likelihood of adverse health effects.

The risk characterization involves the quantification of potential carcinogenic and non-carcinogenic risks for each potential exposure pathway based on the integration of the exposure and toxicity assessments.

The HHRA for the Site is based on the following assumptions:

1. no remedial actions will be conducted;
2. access to the Site is unrestricted; and
3. the future land use of the Site is anticipated to be a Nature Conservancy, but future residential land use was also evaluated.

It should be noted that a security fence and gate were erected at the access point to the Site from County Highway No. 4 during the initiation of the RI. An additional Site security fence and associated "No Trespassing" warning signs were erected around the perimeter of the former wastewater disposal lagoons in 1995.

9.1 GENERAL SITE DESCRIPTION AND BACKGROUND

The Site description and background is presented in Section 1.0.

The primary Site-related information that is applicable to this evaluation is summarized as follows:

i) The Site is located in the Town of Hamptonburgh, Orange County, New York on the southern side of Orange County Highway 4. The Site is located approximately 1.5 miles southwest of the Town of Maybrook, as presented on Figure 1.1.

ii) The Site is a 29.3-acre parcel of mostly wooded land with no existing buildings or other structures. Approximately 5 acres of the total property area was physically affected by lagoon operations. The Site also includes a portion of Beaverdam Brook and a portion of an abandoned railroad bed, as presented on Figure 1.2.

iii) Access from Orange County Highway 4 is via a gravel road leading past the former lagoon area to an abandoned railway bed. A security fence and gate were erected at the access point to the Site from Orange County Highway 4 during the initiation of the RI. A Site security fence and associated "No Trespassing" warning signs were erected around the perimeter of the former wastewater disposal lagoons in 1995.

iv) The northern portion of the Site (north of the railroad) is in the AR-5 single family residential/agricultural district (Town of Hamptonburgh Zoning Map, February 24, 1997 revision). This is the portion of the property that includes the former lagoons. The AR-5 district permits agricultural operations, commercial forestry, outdoor recreation facilities, and single family dwelling units, not to exceed one dwelling on each lot. The southern portion of the Site (south of railroad) is in the SR-2 suburban residence district and permits agricultural operations, single and two-family dwelling units, and recreational parkland.

v) The Site is currently inactive. Historically, the Site was utilized for the disposal of wastewater that resulted from the manufacturing operations at the Nepera, Inc. Facility located in Harriman, New York. Following consultation with the Department of Health that the materials underlying the property would be suitable to allow evaporation or percolation of wastewater, the Site was purchased in 1952. On January 13, 1953, a permit to discharge the wastewaters from the Nepera Harriman Facility was obtained from the NYSDOH. Six lagoons were constructed between 1953 and 1963. These lagoons were used to dispose of wastewater from the Nepera Harriman Facility from 1953 to 1967. Although wastewater disposal operations at the Site ceased in 1967, it has been reported that sporadic unauthorized dumping of domestic refuse occurred during the 1970s.

vi) The Site is situated on rolling topography of low to moderate relief. Two hills located near the center of the property are the high points of the Site with an elevation of approximately 400 feet AMSL. Surface drainage within the Site is topographically controlled. There are no artificial impervious surfaces on Site, therefore surface drainage generally follows a natural pattern that moves radially away from the central part of the Site. Drainage flows towards Beaverdam Brook to the west, towards Otter Kill to the south, and towards the ditch along Orange County Highway 4 to the north and northeast.

vii) The Site is located within a 4.5 square mile watershed consisting of Beaverdam Brook and its tributaries, which discharges to Otter Kill. The watershed area of Otter Kill, at the confluence of Beaverdam Brook and Otter Kill, is approximately 32 square miles. The confluence is located approximately 500 feet south of the Site.

viii) Beaverdam Brook flows south into, and through, the northwestern portion of the Site property and approximately 400 feet west of the former lagoon area. It drains several wetland areas north of the Site and terminates at Otter Kill, approximately 0.25 miles south of the Site. Otter Kill flows generally toward the southeast. Approximately one mile east of the Site, it is dammed to form Brown's Pond, a small impoundment formerly used for agricultural purposes.

ix) A marsh/swamp area designated the Northeast Marsh Area exists approximately 800 feet from the northeast corner of the Site as shown on

Figure 1.2. A second marsh/swamp designated the Southwest Marsh Area is located southwest of the Site as shown on Figure 1.2.

x) The nearest residential properties to the Site are located approximately 250 feet and 450 feet from the western Site boundary and the northeast corner of the Site boundary. These properties represent the sparse distribution of residential properties in the low-density rural, agricultural land, which dominates the land use surrounding the Site.

xi) Properties are developed to the northeast of the site. Practices on one property in particular have been observed that may contribute to soil and groundwater impacts (i.e., MTBE).

9.2 IDENTIFICATION OF CHEMICALS OF POTENTIAL CONCERN

This section presents the process for establishing the COPCs which include chemicals most likely to be related to the Site, and to pose the greatest potential risk to public health. In general, detected constituents were identified as COPCs based upon their concentrations, and known toxicity characteristics.

Human health risk-based criteria (RBC) for residential land use (U.S. EPA, 2004a) were used as screening toxicity values to identify COPCs in soil through comparison to the maximum detected concentration of each chemical in soil. RBC criteria for carcinogenic chemicals are based on a risk of 1×10^{-6}, while RBC criteria for non-carcinogenic chemicals are based on a hazard index (HI) of 1. To maintain a conservative approach, the non-carcinogenic RBC criteria were adjusted to reflect an HI of 0.1 to take into consideration exposure to multiple chemicals.

U.S. EPA RAGS states that if a parameter was detected in less than five (5) percent of the total samples, then the parameter can be eliminated from further evaluation in the risk assessment. To maintain a conservative approach, all parameters reported at concentrations greater than the RBC were identified as COPCs for the investigated medium.

To maintain a conservative approach, TICs, except for unknowns, were also retained and evaluated as COPCs. Therefore, all reported TICs, with the exception of unknowns, were evaluated in the HHRA.

The analytical results for samples collected from borehole, test pit, surface soil, surface water and sediment samples were previously summarized and discussed in Section 6.0.

A description of the COPC identification process for the various media are discussed in the following subsections:

9.2.1 Surface Soils - On Site;
9.2.2 Subsurface Soils - On Site;
9.2.3 Sediments - Northeast Marsh Area;
- Southwest Marsh Area;
- Beaverdam Brook;
- Otter Kill;

9.2.4 Surface Water - Northeast Marsh;
- Beaverdam Brook;
- Otter Kill;

9.2.5 Groundwater - Onsite;
- Offsite; and

9.2.6 Air.

9.2.1 COPCs IN SURFACE SOIL-ON SITE

Contaminant concentrations from all lagoons and throughout the Site were evaluated to develop exposure point concentrations in the HHRA. However, when evaluated as one dataset, the distribution of contaminant concentrations from all areas were found to be neither normal, lognormal, nor non-parametric. Therefore, the data were evaluated for potential outliers, and this evaluation suggested that the data collected from Lagoon 6 needed to be assessed separately from the rest of the surface soil data.

Surface soil samples were collected from each of the six lagoons and along the Site access road, as identified on Figures 3.1 and 3.6. In addition, surface soil data was collected as part of the Phase I RI, which included some surface soil data from test pits and soil borings. Thus, for the purpose of assessing surface soil exposures in the HHRA, data collected during the Phase I RI, which included TP-41 (1 ft bgs) and soil borings in the 0 to 2 ft bgs range, were included in the surface soil datasets, in addition to the surface soil samples (0 to 3 inches bgs) collected during the Phase II RI. In 2003, additional surface soil samples were collected to characterize the inorganics on Site.

9.2.1.1 SITE SURFACE SOILS

The detection frequencies, range of detection limits, and rationale for COPC selection for the Site surface soils are summarized in Table 2.1 of Appendix Q.1 for detected parameters in the surface soil dataset.

Organic COPCs in Site surface soils were determined through comparison to screening toxicity values. The majority of organic chemicals with an applicable screening toxicity value were not retained as COPCs in Site surface soils because most maximum detected concentrations were below their respective screening toxicity values, with the exception of benzo(a)pyrene which was identified as a COPC. Because there is no RBC value established for TPH, it was included as a COPC.

Inorganic COPCs in Site surface soils were determined by comparison of the maximum detected concentration to the screening toxicity value; aluminum, antimony, arsenic, chromium, iron, manganese, mercury, thallium, and vanadium were retained as COPCs.

Although no criteria were available for calcium, magnesium, potassium, and sodium they were classified as essential soil nutrients and were therefore not retained as COPCs. This approach is carried throughout Section 9.2 for all media (soil, sediment, surface water, and groundwater).

To be conservative, TICs detected in site surface soils, with the exception of unknowns, were retained in the HHRA as COPCs.

9.2.1.2 LAGOON 6 SURFACE SOILS

The detection frequencies, range of detection limits, and rationale for COPC selection for the Lagoon 6 surface soils are summarized in Table 2.2 of Appendix Q.1, for detected parameters in the Lagoon 6 surface soil dataset.

Organic COPCs in Lagoon 6 surface soil were determined through comparison to screening toxicity values. The majority of organic chemicals with an applicable screening toxicity value were not retained as COPCs in Lagoon 6 surface soils as the maximum detected concentrations were below their respective screening toxicity values, with the exception of aldrin and Aroclor-1254, which were identified as COPCs. There is no screening toxicity value established for TPH, as a result it was included as a COPC.

Inorganic COPCs in Lagoon 6 surface soil were determined by comparison of the maximum detected concentration to the screening toxicity value; aluminum, arsenic, cadmium, chromium, iron, manganese, mercury, thallium, and vanadium were retained as COPCs.

To be conservative, TICs detected in Lagoon 6 surface soils, with the exception of unknowns, were retained in the HHRA as COPCs.

9.2.2 COPCs IN SOIL-ON SITE

The evaluation of Site soils was conducted for two distinct areas, the Lagoon 6 area and the remainder of the Site. Thus, two COPC selection processes were conducted for the Site soils, one inclusive of data collected from the Lagoon 6 area, and the other inclusive of all data collected at the Site, excluding the Lagoon 6 data.

Soil samples were collected from a series of soil borings and test pits during the RI, as discussed previously in Section 3.1 of the RI Report. Additional test pit samples were collected in support of the Treatability Study conducted in 1996. All available soil data, inclusive of surface soil and subsurface soil data, collected as part of the RI Phases and the Treatability study were included in the datasets used to characterize potential risks associated with exposure to soils. In 2003, additional soil samples were collected to characterize the inorganics on Site.

9.2.2.1 SITE SOILS

The detection frequencies, range of detection limits, and rationale for COPC selection for the Site soils are summarized in Table 2.3 of Appendix Q.1, for detected parameters in the soil dataset.

Organic COPCs in Site soils were determined through comparison to the screening toxicity values. The majority of organic chemicals with an applicable screening toxicity value were not retained as COPCs in Site soils because the maximum detected concentrations were below their respective screening toxicity values, with the exception of benzene, bis(2-ethylhexyl)phthalate, benzo(a)pyrene, 2-aminopyridine, aldrin, dieldrin, Aroclor-1254, and Aroclor-1260, which were identified as COPCs. There are no screening toxicity values established for 4-methyl-2-penatanone, TPH, 4-chloro-3-methylphenol, and alpha-picoline, as a result these compounds were also included as COPCs.

Inorganic COPCs in Site soils were determined by comparison of the maximum detected concentration to the screening toxicity value; aluminum, antimony, arsenic, cadmium, chromium, copper, iron, manganese, mercury, thallium, and vanadium were retained as COPCs.

To be conservative, TICs detected in Site soils, with the exception of unknowns, were retained in the HHRA as COPCs.

9.2.2.2 LAGOON 6 SUBSURFACE SOILS

The detection frequencies, range of detection limits, and rationale for COPC selection for the Lagoon 6 soils are summarized in Table 2.4 of Appendix Q.1, for detected parameters in the soil dataset.

Organic COPCs in Lagoon 6 soil were determined through comparison to screening toxicity values. The majority of organic chemicals with an applicable screening toxicity value were not retained as COPCs in Lagoon 6 soils because the maximum detected concentrations were below their respective screening toxicity values, with the exception of aldrin and Aroclor-1254, which were identified as COPCs. TPH was identified as a COPC in Lagoon 6 soil due to the lack of a comparable RBC.

Inorganic COPCs in Lagoon 6 soils were determined by comparison of the maximum detected concentration to the screening toxicity value; aluminum, arsenic, cadmium, chromium, iron, manganese, mercury, thallium, and vanadium were retained as COPCs.

To be conservative, TICs detected in Lagoon 6 soils, with the exception of unknowns, were retained in the HHRA as COPCs.

9.2.3 COPCs IN SEDIMENTS

Sediment sampling was conducted in the following four areas:

- Northeast Marsh Area;
- Southwest Marsh Area;

- Beaverdam Brook; and
- Otter Kill

Sampling results are presented and discussed in Section 6.5.

9.2.3.1 NORTHEAST MARSH AREA

Two sediment samples have been collected from the Northeast Marsh Area. One sediment sample was collected from sample location 2 during the Phase I RI in 1991, and an additional sample was collected from location SDII-2 during the Phase II RI in 1995.

The detection frequencies, range of detection limits, and rationale for COPC selection are summarized in Table 2.5 of Appendix Q.1, for detected parameters in the Northeast Marsh Area sediment. Detected levels of PAHs have been excluded from the evaluation of the Northeast Marsh Area sediments as NYSDEC and U.S. EPA have agreed that the presence of PAH contamination in the Northeast Marsh Area sediments is not attributable to former Site operations. Due to the limited number of samples, all detected chemicals were identified as COPCs in the sediments of the Northeast Marsh Area and were evaluated in the HHRA except for essential nutrients. To be conservative, TICs detected in sediment, with the exception of unknowns, were retained in the HHRA as COPCs.

9.2.3.2 SOUTHWEST MARSH AREA

One sediment sample from SDII-12 was collected from the Southwest Marsh Area during the Phase II RI in 1995. U.S. EPA collected 10 additional sediment samples (SE02 to SE11) from the Southwest Marsh Area in 2003, which have been included in the RA.

The detection frequencies, range of detection limits, and rationale for COPC selection are summarized in Table 2.6 of Appendix Q.1, for detected parameters in the Southwest Marsh Area sediment. Due to the one sample collected by CRA, all detected chemicals were identified as COPCs and were evaluated in the HHRA except for essential nutrients. It should be noted that TICs were not reported in the sediments of the Southwest Marsh Area.

9.2.3.3 BEAVERDAM BROOK

For Beaverdam Brook, one sediment sample was collected adjacent to the Site from location 4 during the Phase I RI in 1991. U.S. EPA collected 18 additional sediment samples (SE01 and SE12 to SE27) from the Beaverdam Brook in 2003 and the results were provided to CRA. These results were also included in the RA.

The detection frequencies, range of detection limits, and rationale for COPC selection are summarized in Table 2.7 of Appendix Q.1, for detected parameters in Beaverdam Brook sediment. Due to one sample collected by the CRA, all detected chemicals were identified as COPCs in the sediments of Beaverdam Brook and were evaluated in the HHRA except for essential nutrients. To be conservative, TICs detected in sediment, with the exception of unknowns, were retained in the HHRA as COPCs.

9.2.3.4 OTTER KILL

For Otter Kill, one sediment sample was collected from sample location 7 during the Phase I RI in 1991. No additional sediment samples were collected from Otter Kill during the Phase II RI.

The detection frequencies, range of detection limits, and rationale for COPC selection are summarized in Table 2.8 of Appendix Q.1, for detected parameters in Otter Kill sediment. Due to the single sample collected, all detected chemicals were identified as COPCs in the sediments of Otter Kill and were evaluated in the HHRA except for essential nutrients. It should be noted that TICs were not reported in the sediments of Otter Kill.

9.2.4 COPCs IN SURFACE WATER

Surface water sampling was conducted in the following three areas:

- Northeast Marsh Area;
- Beaverdam Brook; and
- Otter Kill

Sampling results were previously presented and discussed in Section 6.4 of the RI Report.

9.2.4.1 NORTHEAST MARSH AREA

Surface water samples were not collected from sediment sample location 2 in the Northeast Marsh Area during the Phase I RI since this location did not have sufficient quantities of water. Three rounds of surface water samples corresponding to low, medium, and high flow conditions were collected from location SWII-2 in the Northeast Marsh Area during the Phase II RI.

The detection frequencies, range of detection limits, and rationale for COPC selection are summarized in Table 2.9 of Appendix Q.1, for detected parameters in the Northeast Marsh Area surface water. Due to the limited number of samples, all detected parameters, including TICs, with the exception of essential nutrients were identified as COPCs in the surface water of the Northeast Marsh Area and evaluated in the HHRA.

9.2.4.2 BEAVERDAM BROOK

During the Phase I RI, two rounds of surface water samples corresponding to low and high flow conditions were collected from location 4 and during the Phase II RI, three rounds of surface water samples corresponding to low, medium, and high flow conditions were collected from location SWII-4.

The detection frequencies, range of detection limits, and rationale for COPC selection are summarized in Table 2.10 of Appendix Q.1, for detected parameters in Beaverdam Brook surface water. Due to the limited number of samples, all detected parameters, including TICs, with the exception of essential nutrients were identified as COPCs in the surface water of Beaverdam Brook and evaluated in the HHRA.

9.2.4.3 OTTER KILL

During the Phase I RI, two rounds of surface water samples corresponding to low and high flow conditions were collected from location 7 and during the Phase II RI, three rounds of surface water samples corresponding to low, medium, and high flow conditions were collected from location SWII-7. Both sample locations are immediately downstream from the confluence of Beaverdam Brook and Otter Kill.

The detection frequencies, range of detection limits, and rationale for COPC selection are summarized in Table 2.11 of Appendix Q.1, for detected parameters in Otter Kill surface water. Due to the limited number of samples, all detected parameters, including TICs, with the exception of essential nutrients were identified as COPCs in the surface water of the Otter Kill and evaluated in the HHRA.

9.2.5 COPCs IN GROUNDWATER

The evaluation of groundwater was conducted both on-Site and off-Site. Thus, two COPC selection processes were conducted for Site groundwater, one inclusive of data collected from monitoring wells on-Site and immediately adjacent to the Site, and the other inclusive of all data collected from off-Site residential wells.

9.2.5.1 ON-SITE GROUNDWATER

In 2001, groundwater was sampled from the on-Site and off-Site monitoring wells to provide a comprehensive groundwater quality dataset for evaluation in the HHRA.

The detection frequencies, range of detection limits, and rationale for COPC selection for the on-Site groundwater samples are summarized in Table 2.12 of Appendix Q.1, for detected parameters in the groundwater dataset.

Organic COPCs in on-Site groundwater were determined through comparison to screening toxicity values. The majority of organic chemicals with an applicable screening toxicity value were not retained as COPCs in on-Site groundwater because most maximum detected concentrations were below their respective screening toxicity values, with the exception of benzene, chlorobenzene, ethylbenzene, methyl-tert-butyl ether, vinyl chloride, xylenes, 4-chloroaniline, bis(2-ethylhexyl)phthalate, pyridine, and 2-aminopyridine, which were identified as COPCs. There is no screening toxicity values established for alpha-picoline, as a result this compound was also included as a COPC.

Inorganic COPCs in On-Site groundwater were determined by comparison of the maximum detected concentration to the RBC; antimony, arsenic, barium, chromium, iron, lead, manganese, vanadium, and cyanide were retained as COPCs. Calcium, magnesium, potassium, and sodium they were classified as essential nutrients and were therefore not retained as COPCs.

To be conservative, TICs detected in on-Site groundwater, with the exception of unknowns, were retained in the HHRA as COPCs.

9.2.5.2 OFF-SITE GROUNDWATER

Off-Site groundwater samples were collected in 2003 and 2004, from existing residential wells, by the New York State Department of Health (NYSDOH). The NYSDOH groundwater data consisted of only VOCs, SVOCs, and Site-specific pyridine analytical results. The detection frequencies, range of detection limits, and rationale for COPC selection for the off-Site groundwater samples are summarized in Table 2.13 of Appendix Q.1, for detected parameters in the groundwater dataset.

Organic COPCs in off-Site groundwater were determined through comparison to screening toxicity values. The majority of organic chemicals with an applicable screening toxicity value were not retained as COPCs in off-Site groundwater because most maximum detected concentrations were below their respective screening toxicity values, with the exception of 2-aminopyridine, which was identified as a COPC. Alpha-picoline was identified as a COPC in off-Site groundwater due to the lack of a comparable screening toxicity value.

9.2.6 AIR

The air pathway was evaluated using modeled air concentrations from both the Lagoon 6 soil sources and the remaining Site soil sources on Site. Two sets of COPCs in air were identified based on VOC concentrations reported in the two separate soil datasets. The soil data utilized to determine the potential emissions consist of all Site-wide soil data. The air assessment includes air emissions and dispersion modeling, as presented in Appendix Q.2.

9.3 POTENTIAL EXPOSURE PATHWAYS

9.3.1 GENERAL

As part of the HHRA process, RAGS requires the characterization of the exposure setting that includes an evaluation of the following:

- physical setting; and
- potentially exposed populations.

The physical setting of the Site was presented previously in Sections 4.0 through 5.0. The consideration of Site-specific factors related to land usage are important in the development of realistic exposure scenarios and quantification of risks in any risk assessment. The current and potential future land uses, which are reasonably expected for the Site and its surroundings, determine the potentially exposed populations. The land uses are discussed in the following subsections.

9.3.1.1 CURRENT LAND USE

The Site, currently owned by Nepera, Inc., is inactive. Historically, the Site was utilized for wastewater disposal. The Site was purchased in 1952 and six lagoons were constructed between 1953 and 1963. These lagoons were used to dispose of wastewaters from a plant located in Harriman, New York under a wastewater discharge permit issued by the NYSDOH in January 1953. Wastewater disposal operations at the Site ceased in 1967.

The Site, a 29.3-acre parcel, is mostly wooded with no existing buildings or other structures. The Site is located in an area that is currently designated as low density rural, residential, and agricultural. The Site is surrounded by farmland and bounded on the north by Orange County Highway 4. Access from Orange County Highway 4 is via a gravel road leading past the former lagoon area to an abandoned railway bed. A security fence and gate were erected at the access point to the Site from Orange County Highway 4 during the initiation of the RI. A Site security fence and associated "No Trespassing" warning signs were installed around the perimeter of the former wastewater disposal lagoons in 1995.

The nearest residential properties to the Site are located approximately 250 feet and 450 feet from the western Site boundary and the northeast corner of the Site boundary, respectively. These properties represent the sparse distribution of residential properties in the low-density rural, agricultural land, which dominates the land use surrounding the Site.

The current potentially exposed population includes persons who may infrequently trespass on the Site for recreational purposes. These trespassers may be hunting, hiking, or fishing. It should be noted that although certain security measures do exist, these measures were not taken into account in the determination of the exposure frequency on Site, for purposes of the HHRA.

9.3.1.2 FUTURE LAND USE

The intended land use for the Site is a Nature Conservancy. The conveyance of the Site as a Nature Conservancy will ensure that the future land use is maintained and restricted to uses consistent with a park setting. As indicated in the Town of Hamptonburgh Master Plan (Garling, 1990), there are no Town-owned general recreation lands. However, the Plan recognizes and identifies the need and future demand for recreational parks of at least 10 acres in size or neighborhood recreational facilities of 3 to 5 acres within new developments. According to the Master Plan, the Town of Hamptonburgh has been collecting fees per lot of newly subdivided land for the purpose of purchasing new recreational lands. In addition, the Master Plan recommends that lands along the Otter Kill and other streams within the Town of Hamptonburgh be included in the category of sensitive resource areas. The Beaverdam Brook flows south into, and through, the northwestern portion of the Site property and approximately 400 feet west of the former lagoon area. It drains several wetland areas north of the Site and terminates at Otter Kill, approximately 0.25 miles south of the lagoon area. The Master Plan recommends that stream setbacks and buffer zones be established for these sensitive resource areas which might also be used for greenways or parkland. The future use of the Site as a Nature Conservancy is, therefore, consistent with the recommendations of the Master Plan for the Town of Hamptonburgh.

It is intended that the Nature Conservancy will consist of a forest-like setting with hiking trails throughout. It is anticipated that it will not include camp areas, educational centers, recreational fields, or a child play areas. Therefore, a future park user was evaluated in the HHRA. A park maintenance worker was also evaluated due to certain work functions (i.e., trail care, litter pickup, etc.) that may be required at the park. It is possible that the development of the Site as a Nature Conservancy may necessitate that some below-grade excavation or construction might be required. As a result, a short-term, future potential construction worker exposure to surface and subsurface soils was evaluated in the HHRA.

In addition to the future Nature Conservancy land use scenario evaluated in the HHRA, a future hypothetical on-Site residential exposure scenario was included to provide baseline risks and hazards associated with the most conservative future potential land use on Site.

9.3.2 EXPOSURE POINTS

To determine whether an exposure to COPCs present in a medium exists, the environmental and human components that lead to human exposure must be evaluated.

An exposure pathway consists of four necessary elements:

i) A source and mechanism of chemical release to the environment;
ii) An environmental transport medium;
iii) A point of potential human contact within the impacted medium (exposure point); and
iv) A human exposure route (ingestion, dermal contact, or inhalation) at the contact point.

Exposure pathways are classified as complete, potential, or incomplete. For an exposure pathway to be complete, the aforementioned four elements must be present, which indicates that the exposure is occurring or will occur in the future. Potential exposure pathways have one element presently missing, which indicates that the exposure pathway may be complete in the future. Incomplete exposure pathways have one or more elements that are missing and are reasonably expected never to be present, thus the pathway is not complete. Table 1.0 of Appendix Q.1 presents a summary of the exposure pathways identified for analysis in the HHRA.

Site investigations to date have identified the presence of chemicals in the following media:

- Site surface soils;
- Site subsurface soils;
- sediments in the Northeast Marsh Area, Southwest Marsh Area, Beaverdam Brook, and Otter Kill;
- surface waters in the Northeast Marsh Area, Beaverdam Brook, and Otter Kill; and
- Groundwater on-Site and off-Site.

The following media and potential human exposures (i.e., complete or potential pathways) have been identified for the Site:

(1) Surface Soil - On Site

- Dermal contact to surface soil by occasional Site visitors (i.e., current trespassers/future park users), by future potential workers (i.e., park maintenance workers), or by future potential residents; and
- Incidental ingestion of surface soil by occasional Site visitors (i.e., current trespassers/future park users), by future potential workers (i.e., park maintenance workers), or by future potential residents.

(2) Soil - On Site

- Dermal contact to soil by future construction workers; and
- Incidental ingestion of soil by future construction workers.

(3) Northeast Marsh Area and Southwest Marsh Area

- Dermal contact to sediment by trespassers, recreational users, and occasional visitors/hikers; and
- Incidental ingestion of sediments by trespassers, recreational users, and occasional visitors/hikers.

(4) Surface Water - Beaverdam Brook, Otter Kill, and Northeast Marsh Area

- Dermal contact of surface water by trespassers, recreational users, and occasional visitors/hikers users.

(5) Groundwater - On and off-Site

- Ingestion of groundwater as a potable water source by future potential residents/recreational users;
- Dermal contact of groundwater as a potable water source by future potential residents/recreational users;
- Dermal contact to groundwater exposed during excavation by future construction worker; and
- Incidental ingestion of groundwater during excavation by future construction worker.

(6) Consumption of Fish - Otter Kill

- Ingestion of fish caught in Otter Kill by recreational anglers.

(7) Air

- Inhalation of volatiles and particulates during excavation by future construction workers; and
- Inhalation of volatiles in groundwater during potable water use by potential future residents both on- and off-Site.

Potential current and future exposure scenarios in Beaverdam Brook and Otter Kill involve recreational use of these surface waters for wading. While wading in Beaverdam Brook and Otter Kill, sediments are not expected to adhere to exposed skin for a prolonged period of time since it is likely that sediments would be washed off by surface water. As such, potential exposure to sediments in Beaverdam Brook and Otter Kill will be minimal and was, therefore, not evaluated in the HHRA.

Descriptions of each of these potential exposure pathways are presented in the following subsections.

9.3.2.1 SURFACE SOIL - ON SITE

Currently, access to the Site is generally restricted[1]. The potential receptors that may come in direct contact with surface soils include occasional Site visitors who gain unauthorized access to the Site. These include trespassers who access the Site for recreational purposes such as hunting or hiking. Persons involved would probably traverse the Site wearing shoes/boots and associated clothing, which reflect exposure patterns that demonstrate only occasional and minimal exposure to surface soils via dermal contact and incidental ingestion.

Under the future land use for the Site as a Nature Conservancy, exposure to Site surface soils by park users was evaluated. It is assumed that the park user will be exposed to Site surface soil over a 30-year RME period that will include exposure both as a child, aged zero to six years (6-year exposure) and as an adult aged six to 30 (24-year exposure). This future exposure pathway includes children and adults exposed to surface soils via dermal contact and incidental ingestion.

Also under the future Site land use as a Nature Conservancy, a park maintenance worker exposure was evaluated. It is assumed that the maintenance worker will be exposed to Site surface soil over a 25-year RME exposure period. This future exposure scenario consists of an adult worker exposed to surface soils via dermal contact and incidental ingestion.

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[1] A security fence and gate were erected at the access point to the Site from County Highway No. 4 during the initiation of the RI. Fencing around the perimeter of the former wastewater disposal lagoons was erected in 1995. Additionally, no trespassing/hunting signs are posted throughout the Site area and perimeter.

The future potential residential exposure scenario evaluated in the HHRA consists of a 30-year RME period, which includes exposure both as a child (6-year exposure duration) and as an adult (24-year exposure duration). This future exposure scenario consists of children and adults exposed to surface soils via dermal contact and incidental ingestion.

Each of the exposure scenarios presented above involving surface soils were evaluated separately for both the Lagoon 6 area and the remaining Site-wide area excluding Lagoon 6.

9.3.2.2 SOILS - ON SITE

Future construction activities may occur during Site development or service, and thus, a construction worker exposure scenario was evaluated in the HHRA. Construction activities on Site are assumed to involve activities such as short-duration trenching to lay cable or pipe, and would involve exposure to both surface and subsurface soil. This exposure pathway includes adult workers exposed to soils via dermal contact, incidental ingestion, and inhalation of volatiles and particulates while trenching.

The construction worker exposure to on-Site soils was evaluated separately for both the Lagoon 6 area and the remaining Site-wide area, excluding Lagoon 6.

9.3.2.3 SEDIMENTS - NORTHEAST MARSH AREA AND SOUTHWEST MARSH AREA

A potential current exposure scenario involving dermal contact to and incidental ingestion of, sediments were evaluated for an occasional visitor. This includes trespassers such as, a hiker or hunter who may be traversing either the Northeast Marsh Area or the Southwest Marsh Area. Separate risk estimates were calculated for these areas.

The future exposure scenario for the Northeast Marsh Area is assumed to be the same as the current occasional visitor exposure, however, the potential future exposure scenario for the Southwest Marsh Area is expected to involve exposure to sediment by a park user from the Nature Conservancy or an on-Site resident.

In both the current and future exposure to sediments in the Northeast Marsh Area, the exposure frequencies are anticipated to be low due to typically dry periods during the summer months and frozen conditions during the winter months.

9.3.2.4 SURFACE WATER - NORTHEAST MARSH AREA, BEAVERDAM BROOK, AND OTTER KILL

Potential current exposure scenarios involving dermal contact to surface water were evaluated for an occasional visitor, such as, a hiker or hunter who may be wading in the Northeast Marsh Area, Beaverdam Brook, and Otter Kill. Separate risk estimates were calculated for each of the areas.

The future exposure scenario for the Northeast Marsh Area is assumed to be the same as the current occasional visitor exposure, however, the potential future exposure scenarios for Beaverdam Brook and Otter Kill are expected to involve exposure to surface waters by a park user from the Nature Conservancy, or an on-Site resident.

In both the current and future exposure to surface water in the Northeast Marsh Area, the exposure frequencies are anticipated to be low due to typically dry periods during the summer months and frozen conditions during the winter months.

9.3.2.5 GROUNDWATER - ON AND/OR OFF SITE

The potential residential exposure scenario evaluated in the HHRA consists of a 30-year RME period, which includes exposure both as a child (6-year exposure duration) and as an adult (24-year exposure duration). This exposure scenario consists of children and adults exposed to groundwater via dermal contact and ingestion as a potable water source, and inhalation of VOCs migrating to indoor air during showering/bathing activities.

The residential exposure scenario presented above involving groundwater was evaluated separately for both on- and off-Site groundwater exposures.

Future construction activities may occur during Site development or service, and thus, a construction worker exposure scenario in contact with groundwater was evaluated in the HHRA. Construction activities on Site are assumed to involve activities such as short-duration trenching to lay cable or pipe, and would involve potential exposure to groundwater. This exposure pathway includes adult workers exposed to groundwater via dermal contact, incidental ingestion, and inhalation of volatiles while trenching.

9.3.2.6 CONSUMPTION OF FISH

During Site-related field activities, individuals were observed fishing in Otter Kill. Thus, the consumption of fish from Otter Kill was considered a complete exposure pathway for both current and future recreational anglers, and was evaluated in the HHRA.

9.3.2.7 AIR

The air pathway was evaluated by modeling air concentrations from potential soil and groundwater sources at the Site. The HHRA addresses three exposure scenarios under the future Site condition for both the Lagoon 6 area and the remaining Site area: i) the inhalation exposure to volatiles migrating to ambient air from soil by construction workers during excavation activities; ii) the inhalation exposure to particulates by construction workers during excavation activities; and iii) the inhalation exposure to volatiles migrating to ambient air from groundwater by construction workers during excavation activities.

9.4 EXPOSURE POINT CONCENTRATIONS

To quantify exposures, potential exposure scenarios were developed using guidance presented in the following U.S. EPA documents:

- "Risk Assessment Guidance for Superfund", [(RAGS) Interim Final, EPA/540/1-89/002, December 1989];
- RAGS Supplemental Guidance "Standard Default Exposure Factors" [OSWER Directive 9285.6-03, March 25, 1991];
- "Exposure Factors Handbook", [EPA/600/P-95/022Fa, August 1997];
- "Superfund Exposure Assessment Manual", [EPA/540/1-88/001, April 1988];
- U.S. EPA RAGS Part E: Supplemental Guidance for Dermal Risk Assessment, EPA/540/R/99/005, July 2004;
- U.S. EPA Calculating the Concentration Term: Supplemental Guidance to RAGS, EPA Publication 9285.7-081, May 1992; and
- U.S. EPA Calculating Upper Confidence Limits for Exposure Point Concentrations at Hazardous Waste Sites, OSWER 9285.6-10, December 2002.

In some instances where U.S. EPA documents did not present necessary assumptions, or where more appropriate scientific data were available, professional judgment was applied to develop conservative assumptions which are representative of the RME and are protective of human health. The exposure scenarios and assumptions are presented in Appendix Q.1 with the related data and risk calculation tables.

Two levels of assumptions are presented. The Central Tendency (CT) assumptions present the average or mean value for the assumptions and approximate the most probable exposure conditions. The RME presents conservative assumptions that generally utilize the 90th to 95th percentile assumptions, depending upon the available data. A detailed description of the statistical procedures utilized to calculate the 95 percent UCL of the mean is presented in Appendix Q.3.

To be conservative, the CT exposure point concentration (EPC) was the same as the RME EPC rather than the average or mean value for all exposure media.

9.4.1 SURFACE SOIL ON SITE

The arithmetic mean, maximum, and 95 percent UCL concentrations for the COPCs identified in Site surface soils and Lagoon 6 surface soils are summarized in Tables 3.1 and 3.2 of Appendix Q.1, respectively.

9.4.2 SOILS ON SITE

The arithmetic mean, maximum, and 95 percent UCL concentrations for the COPCs identified in Site soils and Lagoon 6 soils are summarized in Tables 3.3 and 3.4 of Appendix Q.1, respectively.

9.4.3 SEDIMENTS

The arithmetic mean, maximum, and 95 percent UCL concentrations for the COPCs identified in sediments at the Northeast Marsh Area and Southwest Marsh Area are summarized in Tables 3.5 and 3.6 of Appendix Q.1, respectively.

9.4.3.1 NORTHEAST MARSH AREA

Due to the limited number of samples collected from the Northeast Marsh Area during the RI, the 95 percent UCL concentrations were biased high and thus, were not representative of Site levels. Although assuming long-term contact with maximum concentrations is not consistent with the CT and RME approach, maximum concentrations were used as exposure point concentrations for the carcinogenic risk and hazard index estimates. The arithmetic mean, maximum and 95 percent UCL concentrations for sediment samples collected from the Northeast Marsh Area are summarized in Table 3.5 of Appendix Q.1.

9.4.3.2 SOUTHWEST MARSH AREA

Mean and 95 percent UCL concentrations could not be calculated for the PAH sediment data from the Southwest Marsh Area since there was only one sample collected from this area during the RI. The arithmetic mean, maximum, and 95 percent UCL concentrations for parameters that were detected in sediment collected from the Southwest Marsh Area are summarized in Table 3.6 of Appendix Q.1. The 95 percent UCLs or the maximum reported concentrations (whichever were lower) were used as the exposure point concentrations for both the CT and RME carcinogenic risk and HI estimates.

9.4.4 SURFACE WATER

9.4.4.1 NORTHEAST MARSH AREA

Due to the limited number and variability of data points, the 95 percent UCL concentrations were biased high and thus, were not representative of Site levels. Although assuming long-term contact with maximum concentrations is not consistent with the CT and RME approach, maximum concentrations were used as exposure point concentrations for the carcinogenic risk and HI estimates. The arithmetic mean, maximum, and 95 percent UCL concentrations for surface water samples collected from the Northeast Marsh Area are summarized in Table 3.7 of Appendix Q.1.

9.4.4.2 BEAVERDAM BROOK

The arithmetic mean, maximum, and 95 percent UCL concentrations for parameters that were detected in surface water collected from Beaverdam Brook are summarized in Table 3.8 of Appendix Q.1. The 95 percent UCLs or the maximum reported concentrations (whichever were lower) were used as the exposure point concentrations for both the CT and RME carcinogenic risk and HI estimates.

9.4.4.3 OTTER KILL

The arithmetic mean, maximum, and 95 percent UCL concentrations for parameters that were detected in surface water collected from Otter Kill are summarized in Table 3.9 of Appendix Q.1. The 95 percent UCLs or the maximum reported concentrations (whichever were lower) were used as the exposure point concentrations for both CT and RME carcinogenic risk and HI estimates.

9.4.5 ON AND OFF SITE GROUNDWATER

The arithmetic mean, maximum, and 95 percent UCL concentrations for the COPCs identified in on-Site groundwater and off-Site groundwater are summarized in Tables 3.10 and 3.11 of Appendix Q.1, respectively. The 95 percent UCLs or the maximum reported concentrations were utilized for both CT and RME exposure scenarios.

9.4.6 FISH

The levels of chemical parameters in fish tissues were estimated from surface water using chemical-specific BCFs and the surface water RME concentrations for chemical parameters detected in Otter Kill.

9.4.7 AIR

Air exposure point concentrations were determined for the on-Site future construction worker scenarios based upon modeled air concentrations from potential on-Site soil sources represented separately by the Lagoon 6 area, the remaining Site area, and from potential groundwater sources exposed during excavation activities. The air modeling assessment is presented in Appendix Q.2.

9.5 GENERIC ESTIMATION OF INTAKE

In the HHRA, exposure estimates reflect chemical concentration, contact rate, exposure time, and body weight in a term called "intake" or "dose". A generic equation for calculating chemical intake (U.S. EPA, 1989a) is:

$$I = \frac{C \times CR \times EF \times ED \times CF}{BW \times AT}$$

where:

I = Chemical intake (mg/kg body weight/day)
C = Chemical concentration (e.g., mg/kg for soil)
CR = Contact rate (e.g., mg soil/day)
EF = Exposure frequency (days/year)
ED = Exposure duration (years)
CF = Conversion factors as appropriate (e.g., kg/mg)
BW = Body weight (kg)
AT = Averaging time (period over which exposure is averaged, e.g., days)

Non-Carcinogens

The chemical intake of non-carcinogens is estimated over the appropriate exposure period or averaging time. The averaging time selected depends on the exposure time of the specific population being evaluated.

Carcinogens

A lifetime average daily dose of the chemical is estimated for carcinogens. This prorates the total cumulative intake over a lifetime. An averaging time (AT) of 70 years is used for carcinogens.

9.5.1 EXPOSURE TO SURFACE SOIL - ON SITE

9.5.1.1 CURRENT TRESPASSERS/HIKERS

The current surface soil exposure scenario was developed to reflect infrequent and occasional trespasser exposure patterns that are typical of someone who would gain access to the Site for general recreational purposes such as hunting or hiking. Table 4.1 of Appendix Q.1 presents a summary of the conservative and health-protective assumptions that were used to calculate the current trespasser exposure, as appropriate. The exposure assumptions are described as follows:

- only older children and adults would gain access to the area;
- individual trespasses on Site for an exposure frequency of 5 days (CT) and 10 days (RME) per year;
- individual ingests 100 mg of soil each exposure day (CT and RME) (U.S. EPA, 1991a);
- oral absorption factor of 100 percent (CT and RME);
- 3,400 cm^2 of skin is soiled each exposure day for CT and 3,800 cm^2 for RME (based on direction from the U.S. EPA, 2000);
- soil to skin adherence factor is 0.01 and 0.07 mg/cm^2-event for CT and RME, respectively (U.S. EPA, 2004b);
- individual resides in the area and visits the Site on occasion for an exposure duration of 10 years (CT and RME) for carcinogenic assessment, and an exposure duration of 10 year (CT and RME) for non-carcinogenic assessment;
- individual age-adjusted weight is 35 kg for both the CT and RME (based on direction from the U.S. EPA, 2000);
- dermal absorption factor is chemical specific (U.S. EPA, 2004b); and
- averaging time = 25,550 days (carcinogen)
  = 365 days x ED (non-carcinogen).

9.5.1.2 FUTURE PARKLAND USERS

The expected future use of the Site is a Nature Conservancy and as such, a future parkland scenario was evaluated and is presented herein. Table 4.2 of Appendix Q.1 includes a summary of the conservative and health-protective assumptions that were used to calculate the future parkland user exposure to surface soil, as appropriate. The exposure assumptions are described as follows:

- children and adults would be exposed;
- soil ingestion rates are 100 mg/day for adults (CT and RME) and 200 mg/day for children (CT and RME) (U.S. EPA, 1991a);
- the exposed adult skin surface area is assumed to be 5,700 cm^2 for both the CT and RME (this assumes that an individual is wearing a short-sleeved shirt, shorts and shoes) (U.S. EPA, 2004b);
- the exposed child skin surface area is assumed to be 2,800 cm^2 for both the CT and RME (this assumes that a child is wearing a short-sleeved shirt and shorts but not shoes) (U.S. EPA, 2004b);
- soil to skin adherence factor for adults are 0.01 and 0.07 mg/cm^2-event for CT and RME, respectively (U.S. EPA, 2004b);
- soil to skin adherence factor for children are 0.06 and 0.20 mg/cm^2-event for the CT and RME, respectively (U.S. EPA, 2004b);
- exposure frequency for a child is 52 and 104 days/year for the CT and RME, respectively, based on U.S. EPA (2000) recommendation;
- exposure frequency for an adult is 90 and 180 days/year for the CT and RME, respectively, [based on direction from the U.S. EPA, 2000];
- exposure time is 2 hours/day, which was applied for the ingestion route only for the CT and RMT for child and adult (U.S. EPA, 1997);
- adult resides in the area for an exposure duration of 3 years (CT) and 24 years (RME) (U.S. EPA, 1991a);
- child resides in the area for an exposure duration of 6 years for CT and RME (U.S. EPA, 1991a);
- the absorption factor is 100 percent for the ingestion exposure pathway and chemical specific for the dermal exposure pathway (U.S. EPA, 2004b);
- averaging time for both adult and child (carcinogen) is 25,550 days for CT and RME (U.S. EPA, 1989a);
- averaging time (non-carcinogen) for an adult is 1,095 days for CT and 8,760 days for RME (U.S. EPA, 1989a); and
- averaging time (non-carcinogen) for a child is 2,190 days for CT and RME (U.S. EPA, 1989a).

9.5.1.3 FUTURE PARK MAINTENANCE WORKER

A future maintenance worker exposure was evaluated in the HHRA based on the intended use of the Site as a Nature Conservancy. Table 4.3 of Appendix Q.1 includes a summary of the conservative and health-protective assumptions that were used to calculate the future maintenance worker exposure, as appropriate. The exposure assumptions are described as follows:

- adult maintenance workers would be exposed to surface soils;
- worker would be exposed for 221 days/year (5 days/week for the spring, summer, and fall and 2 days/week for the winter period) for both the CT and RME, based on U.S. EPA (2000) recommendations;
- individual ingests 50 mg soil/day (CT and RME) (U.S. EPA, 1991a);
- the exposed skin surface area is assumed to be 3,300 cm^2 for both CT and RME (this assumes an individual wears short-sleeved shirt, pants, and shoes) (U.S. EPA, 2004b);
- soil to skin adherence factor is 0.02 and 0.20 mg/cm^2-event (CT and RME) (U.S. EPA, 2004b);
- the exposure duration's for the worker are 9 years (CT) and 25 years (RME) (U.S. EPA, 1991a);
- individual weighs 70 kg (CT and RME) (U.S. EPA, 1991a);
- the absorption factor is 100 percent for the ingestion exposure pathway and chemical specific for the dermal exposure pathway (U.S. EPA, 2004b);
- averaging time (carcinogen) for a worker is 25,550 days for CT and RME (U.S. EPA, 1989a); and
- averaging time (non-carcinogen) for a worker is 3,285 days for CT and 9,125 days for RME (U.S. EPA, 1989a).

9.5.1.4 FUTURE RESIDENTS

To maintain a conservative approach to the HHRA, a future residential scenario was evaluated and is presented herein. Table 4.4 of Appendix Q.1 includes a summary of the conservative and health-protective assumptions that were used to calculate the future residential exposure, as appropriate. The exposure assumptions are as follows:

- residents consist of both children and adults;

- soil ingestion rates are 100 mg/day for adults (CT and RME) and 200 mg/day for children (CT and RME) (U.S. EPA, 1991a);
- the exposed adult skin surface area is assumed to be 5,700 cm^2 for both the CT and RME (U.S. EPA, 2004b);
- the exposed child skin surface area is assumed to be 2,800 cm^2 for both the CT and RME (U.S. EPA, 2004b);
- soil to skin adherence factor for adult are 0.01 and 0.07 mg/cm^2-event for the CT and RME, respectively (U.S. EPA, 2004b);
- soil to skin adherence factor for children are 0.06 and 0.20 mg/cm^2-event for the CT and RME, respectively (U.S. EPA, 2004b);
- exposure frequency for residents including children and adults is 350 days/year for CT and RME, respectively (U.S. EPA, 1991a);
- exposure duration for residents assume that individual resides on Site for an exposure frequency of 9 years (CT) and an exposure frequency of 30 years (RME) for the carcinogenic assessment, of which 6 years are spent as a child (U.S. EPA, 1989a);
- body weight: child = 16 kg
  adult = 70 kg;
- the absorption factor is 100% for the ingestion exposure pathway and chemical specific for the dermal exposure pathway (U.S. EPA, 2004b);
- averaging time for both adult and child (carcinogen) is 25,550 days for CT and RME (U.S. EPA, 1989a);
- averaging time (non-carcinogen) for an adult is 1,095 days for CT and 8,760 days for RME (U.S. EPA, 1989a); and
- averaging time (non-carcinogen) for a child is 2,190 days for CT and RME (U.S. EPA, 1989a).

9.5.2 FUTURE EXPOSURE TO SOIL-ON SITE

9.5.2.1 FUTURE CONSTRUCTION WORKER

A future construction worker exposure was evaluated in the HHRA based on the intended conversion of the Site to a Nature Conservancy. A scenario was developed to evaluate potential risk to construction workers exposed to both surface and subsurface soils during short-term construction activities on Site. Table 4.5 of Appendix Q.1 includes a summary of the conservative and health-protective assumptions that were used to calculate the future construction worker exposure, as appropriate. The exposure assumptions are described as follows:

- adult construction workers would be exposed to soils;
- worker would be exposed for 20 days/year and 60 days/year for CT and RME, respectively;
- individual ingests 480 mg soil/day (CT and RME) (U.S. EPA, 1991a);
- the exposed skin surface area is assumed to be 3,300 cm^2 for both the CT and RME (U.S. EPA, 2004b);
- soil-to-skin adherence factor are 0.02 and 0.20 mg/cm^2-event (CT and RME) (U.S. EPA, 2004b);
- the inhalation rate is assumed to be 20 m^3/work day for the CT and RME (U.S. EPA, 1997);
- the exposure duration (ED) for the worker is 1 year (CT and RME);
- individual weighs 70 kg (CT and RME) (U.S. EPA, 1991a);
- the absorption factor is 100 percent for the ingestion exposure pathway and chemical specific for the dermal exposure pathway (U.S. EPA, 2004b); and
- averaging time = 25,550 days (carcinogen)
  = 365 days x ED (non-carcinogen) (U.S. EPA, 1989a).

9.5.3 SEDIMENT EXPOSURE

9.5.3.1 NORTHEAST MARSH AREA SEDIMENTS

The current/future occasional visitor/hiker exposure scenario developed for the Northeast Marsh Area sediments is identical to the current on-Site surface soil trespassers/hikers exposure scenario presented in Section 9.5.1.1. The current/future occasional visitors/hikers exposure scenario details are presented in Table 4.6 of Appendix Q.1.

9.5.3.2 SOUTHWEST MARSH AREA SEDIMENTS

9.5.3.2.1 CURRENT OCCASIONAL VISITORS/HIKERS

Table 4.7 of Appendix Q.1 includes a summary of the conservative and health-protective assumptions that were used to calculate the current occasional visitors/hikers exposure to Southwest Marsh Area sediment. The current occasional visitor/hiker exposure to sediments in the Southwest Marsh Area are identical to the current/future occasional visitor/hiker exposure to Northeast Marsh Area sediment as presented above in Section 9.5.3.1 with the following exceptions:

- exposure frequency is 52 and 78 days/year for the CT and RME, respectively, based on U.S. EPA (2000) recommendation.

9.5.3.2.2 FUTURE RECREATIONAL USERS

The expected future use of the Site is a Nature Conservancy and as such, a future recreational scenario was evaluated and is presented herein. Table 4.8 of Appendix Q.1 includes a summary of the conservative and health-protective assumptions that were used to calculate the future recreational user, exposure to Southwest Marsh Area sediment. The future recreational user exposure to sediments in the Southwest Marsh Area is identical to the future residential exposure to surface soil presented above in Section 9.5.1.4 with the following exception of exposure frequency:

- exposure frequency is 52 and 78 days/year for the CT and RME, respectively, for both children and adults based on direction from the U.S. EPA (2000).

9.5.4 SURFACE WATER EXPOSURE

9.5.4.1 NORTHEAST MARSH AREA

9.5.4.1.1 CURRENT/FUTURE OCCASIONAL VISITORS/HIKERS

The current/future occasional visitor wading scenarios developed for Northeast Marsh Area surface waters are presented in Table 4.9 of Appendix Q.1. The current/future occasional visitors/hikers exposure assumptions are described as follows:

- only older children and adults would gain access to the area;
- the exposed skin area is 3,320 cm^2 (CT and RME), which represents lower legs and feet (U.S. EPA, 1997);
- the chemical permeability across the skin is chemical specific (U.S. EPA, 2004b);
- exposure time is assumed to be 1 hour/day and 2 hours/day (CT and RME respectively);

- exposure frequency is 5 days and 10 days/year for CT and RME, respectively;
- the exposure duration is 10 years for non-cancer hazards for both CT and RME (U.S. EPA, 2000);
- exposure duration is 10 years for cancer risk for CT and RME, as recommended by the U.S. EPA (2000);
- the absorption factor is 100 percent for the ingestion exposure pathway and chemical specific for the dermal exposure pathway (U.S. EPA, 2004b); and
- averaging time = 25,550 days (carcinogen)
  = 365 days x ED (non-carcinogen) (U.S. EPA, 1989a).

9.5.4.2 BEAVERDAM BROOK AND OTTER KILL

9.5.4.2.1 CURRENT TRESPASSERS/HIKERS

The current trespassers/hikers exposure scenarios developed separately for dermal contact while wading in the surface waters of Beaverdam Brook and Otter Kill both utilized the following conservative assumptions as presented in Table 4.10 of Appendix Q.1. The current trespassers/hikers exposure to surface water in Beaverdam Brook and Otter Kill is identical to the current/future occasional visitor exposure to surface water at the Northeast Marsh Area presented above in Section 9.5.4.1 with the following exception of exposure frequency:

- exposure frequency is 52 days/year for CT and 78 days/year for the RME, respectively as recommended by the U.S. EPA (2000).

9.5.4.2.2 FUTURE RECREATIONAL USERS

The future potential recreational exposure scenarios developed separately for dermal contact while wading in the surface waters of Beaverdam Brook and Otter Kill both utilized the following conservative assumptions as presented in Table 4.11 of Appendix Q.1. The future recreational user exposure assumptions are described as follows:

- recreational users exposed include children and adults;
- the exposed skin area is 3,320 cm^2 (CT and RME) for adult, which includes lower legs and feet (U.S. EPA 1997);

- the exposed skin area is 2,965 cm^2 (CT and RME) for children, which includes legs and feet (U.S. EPA 1997);
- exposure time is assumed to be 1 hour/day and 2 hours/day (CT and RME respectively);
- exposure frequency is 52 days and 78 days/year for CT and RME, respectively;
- adult resides near the Site for 3 years (CT) and 24 years (RME) (U.S. EPA, 1989a);
- child resides near the Site for 6 years CT and RME (U.S. EPA, 1989a);
- the chemical permeability across the skin is chemical specific (U.S. EPA, 2004b);
- body weight: child = 16 kg
  adult = 70 kg (U.S. EPA 1991a);
- averaging time for both adult and child (carcinogen) is 25,550 days for CT and RME (U.S. EPA, 1989a);
- averaging time (non-carcinogen) for an adult is 1,095 days for CT and 8,760 days for RME (U.S. EPA, 1989a); and
- averaging time (non-carcinogen) for a child is 2,190 days for CT and RME (U.S. EPA, 1989a).

9.5.5 FISH EXPOSURE

9.5.5.1 CURRENT/FUTURE RECREATIONAL ANGLERS AT OTTER KILL

The current and future exposure scenario developed for the ingestion of fish caught in Otter Kill by recreational anglers utilized the following conservative assumptions as presented in Table 4.12 of Appendix Q.1. The current/future recreational angler exposure assumptions are described as follows:

- children and adults would be exposed;
- individual consumes 365 meals of fish per year for both the CT and RME;

  Note: This value is based on the mean weekly intake of 7 meals of fish/week among individuals who reside in a household that consumes recreationally caught fish.
- a daily fish consumption rate for adults of 6 grams/day and 31.9 grams/day was applied for the CT and RME, respectively, based on U.S. EPA (2000) recommendations;

- a daily fish consumption rate for children of 2 grams/day and 10.6 grams/day was applied for the CT and RME, respectively, based on U.S. EPA (2000) recommendations;
- exposure duration is 9 years (CT) and 30 years (RME), of which 6 years is spent as a child (U.S. EPA, 1991a);
- body weight: child =16 kg
  adult = 70 kg (U.S. EPA, 1991a); and
- averaging time = 25,550 days (carcinogen)
  = 365 days x ED (non-carcinogen) (U.S. EPA, 1989b).

9.5.6 GROUNDWATER

9.5.6.1 CURRENT AND FUTURE RESIDENTS

This scenario is based on the potential for future Site residential development and current off-Site residential groundwater exposure. Table 4.13 of Appendix Q.1 presents a summary of the conservative and health-protective assumptions that were used to calculate the residential exposure, as appropriate. The exposure assumptions are described as follows:

- residents consist of both children and adults;
- water ingestion rates for adults are 1.4 L/day and 2 L/day for CT and RME, respectively (U.S. EPA, 1991a);
- water ingestion rates for children are 0.7 L/day and 1 L/day for CT and RME, respectively (U.S. EPA, 1991a);
- the exposed skin surface area is assumed to be 18,000 cm^2 for adults for both the CT and RME (U.S. EPA, 2004b);
- the exposed skin surface area is assumed to be 6,600 cm^2 for children for both the CT and RME (U.S. EPA, 2004b);
- the inhalation rate for adults is 15 and 20 m^3/day respectively for CT and RME (U.S. EPA, 1991a);
- the inhalation rate for children is 8.8 and 13.6 m^3/day respectively for CT and RME (U.S. EPA, 1997);
- dermal permeability constant, fraction absorbed, and lag time is chemical specific (U.S. EPA, 2004b);

- The volatilization factor (L/m^3) is 0.5 for both CT and RME - the inhalation rate for adults is 15 and 20 m^3/day respectively for CT and RME (U.S. EPA, 1991b);
- exposure frequency for residents including children and adults is 350 days/year for both CT and RME (U.S. EPA, 1991a);
- exposure duration is based on an individual residing on Site for 9 years (CT) and 30 years (RME) for the carcinogenic assessment, of which 6 years are spent as a child (U.S. EPA, 1991a);
- body weight: child = 16 kg (U.S. EPA, 1991a)
  adult = 70 kg (U.S. EPA, 1989a);
- adults spend 0.25 and 0.58 hr/day (CT and RME) in direct contact with groundwater through showering and/or bathing (U.S. EPA, 2004b);
- children spend 0.33 and 1 h/day (CT and RME) in direct contact with groundwater through showering and/or bathing (U.S. EPA, 2004b);
- individual spends 100 percent (CT and RME) of time on Site in direct contact with affected surface soils; and
- averaging time = 25,550 days (carcinogen)
  = 365 days x ED (non-carcinogen) (U.S. EPA, 1989a).

9.5.6.2 FUTURE CONSTRUCTION WORKER

This scenario is based on the potential for future development and potential groundwater exposure of construction workers. Table 4.14 of Appendix Q.1 presents a summary of the conservative and health-protective assumptions that were used to calculate the construction worker exposure, as appropriate. The exposure assumptions are described as follows:

- adult workers would be exposed to groundwater during excavation activities;
- construction worker would be exposed for 20 days/year (CT) and 60 days/year (RME) assuming a respective 1 month and 3 month construction campaign;
- individual ingests (incidental) 0.005 L of groundwater/hour (CT and RME). This ingestion rate was derived from the incidental ingestion rate of 50 mL/hour for an adult while swimming (U.S. EPA, 1989a). Since the workers are not swimming in the exposed groundwater, a factor of 10 was applied to arrive at the reasonable incidental ingestion rate of 0.005 L/hr;
- the exposed skin surface area is assumed to be 3,300 cm^2/day for both the CT and RME (U.S. EPA, 2004b);

- the inhalation rate for a construction worker is 12 m^3/day for CT and 20 m^3/day for RME (U.S. EPA, 1991a);
- dermal permeability constant is chemical specific;
- the construction activities will occur within a one-year period (CT and RME);
- individual weighs 70 kg (CT and RME) (U.S. EPA, 1991a);
- individual spends 100 percent (CT and RME) of time on-Site in direct contact with affected groundwater; and
- averaging time = 25,550 days (carcinogen)
  = 365 days (non-carcinogen) (U.S. EPA, 1989a).

9.5.7 EVALUATION OF ADULT EXPOSURES TO LEAD IN SEDIMENT AND WATER

The best available method for determining hazards associated with adult exposures to lead-impacted soil, sediment, and water is based on the adult lead exposure equation as presented in the U.S. EPA document "Recommendations of the Technical Review Workgroup (TRW) for Lead for an Approach to Assessing Risks Associated with Adult Exposures to Lead in Soil" (U.S. EPA, 2003).

The hazard associated with adult lead exposure stems from the relationship between the lead concentration in site media and the blood lead concentration in the developing fetus of a pregnant woman exposed to site media. Current Office of Solid Waste and Emergency Response (OSWER) guidance calls for the establishment of cleanup goals to limit childhood risk of exceeding 10 μg/dL to 5 percent of the population.

Lead was identified as a COPC in Northeast Marsh and Southeast Marsh sediment, Otter Kill surface water, and Site groundwater. The concentrations of lead in the sediment are well below the U.S. EPA residential action level of the 400 μg/g, as a result the sediments in Northeast Marsh and Southeast Marsh were not evaluated for lead exposure. Dermal contact with Otter Kill surface water was the only exposure pathway evaluated for the current trespasser/hiker. Lead exposure by percutaneous absorption is generally not a significant route of exposure for inorganic lead, as a result surface water lead exposure to the current trespasser/hiker was not evaluated. As such, it is reasonably anticipated that a future construction/utility worker performing excavations and future resident on Site would contact lead via the groundwater. The future construction/utility worker was not evaluated, as the exposure frequency of 60 days/year and incidental ingestion rate of 0.005 L/day will not result in an exceedance of the fetal blood level of 10 μg/dL. The adult resident was not evaluated for consumption of lead in a potable water supply as the child resident, the most sensitive receptor, was evaluated using the Integrated Exposure Uptake Biokinetic (IEUBK) model as presented in the following section.

9.5.8 EVALUATION OF CHILD EXPOSURE TO LEAD IN SEDIMENT AND WATER

Because of the uncertainties in the dose-response relationship between exposures to lead and biological effects, it is unclear whether the non-carcinogenic effects of lead exhibit a threshold response. Therefore, U.S. EPA has not derived a Reference Dose (RfD) for lead in children. Instead, in accordance with the OSWER (1998) guidance entitled "Clarification to the 1994 Revised Interim Soil Lead Guidance for CERCLA Sites and RCRA Corrective Action Facilities", lead was evaluated using the IEUBK model. Using this model, potential risks are evaluated for children based on predicted blood lead levels associated with exposure to lead.

U.S. EPA's Lead and Copper Rule, promulgated June 7, 1991 established an action level of 15 μg/L based on 1-litre first-drawn samples taken at the consumer's taps. This action level was utilized as the surrogate groundwater screening toxicity value in the groundwater screening process. The maximum lead concentration found in on-Site groundwater was 109 μg/L and was therefore above the screening criteria. On this basis, lead found in on-Site groundwater was considered to be a COPC.

U.S. EPA (1998) recommends a 400 μg/g screening level for lead in soils at residential properties. OSWER defines a 'screening level' as a "level of contamination above which there may be enough concern to warrant site-specific study of risks". The maximum lead concentrations found in the on-Site surface soil and Lagoon 6 surface soil were 202 mg/kg and 85.9 mg/kg, respectively, therefore, the surface soil concentrations were well below the screening criteria. On this basis, it was determined that lead found in the surface soil was not considered to be a COPC.

For sediment, the maximum concentration in the Southwest Marsh sediment was 44.5 mg/kg. Lead in the Southwest Marsh sediment was identified as a COPC as there are no applicable human health based sediment screening criterion. As, the lead concentration in the Southwest Marsh sediment is well below the U.S. EPA residential soil screening level the sediment will not be evaluated further.

9.5.9 THE IEUBK MODEL FOR LEAD

The IEUBK model for lead provides a method for predicting blood lead levels in populations exposed to lead in several different media. The impetus for the development of this model was the U.S. EPA's conclusion that a threshold does not exist for many of the non-cancer effects of lead in infants and young children. The IEUBK Model is a simulation model which uses mathematical equations to estimate the lead level in blood (PbB) of a child (or population of children) in the 0 to 7 year age group who are exposed to lead in various media such as soil, air, diet, water, and other potential sources. Young children are considered not impacted when the model predicts that less than 5 percent of children will have blood lead levels greater than 10 μg/dL (U.S. EPA, 2002). The IEUBKwin32 Model 1.0, Build 261 was used to evaluate potential health risks from residential-type exposures to lead by young children.

For this evaluation, the intake of lead was determined for various direct and secondary exposure pathways, including the ingestion of soil, dust particles, and drinking water, inhalation of air and dietary intake (i.e., produce, animal products, fish). Site-specific data for groundwater and sediment were evaluated using the drinking water and alternative source inputs. The media and route exposure point concentrations and intake rates that were used in the IEUBK model are presented in Table 9.1. The use of these highest Site-specific values in the model results in the overestimation of the lead blood level in children.

A default lead concentration in air of 0.100 $\mu g/m^3$ and 2.5 μg/dL for the maternal contribution was used as the input into the IEUBK model. A lead concentration in drinking water (i.e., groundwater) of 4.36 μg/L was taken from the calculated arithmethic average and utilized in determining the inputs for the IEUBK model. The model default values were utilized for the dietary lead intake for each age range.

The IEUBK model is built conceptually using a default exposure frequency of 365 days. In terms of the Site, the exposure frequency to groundwater are 350 days per year (Table 4.13 of Appendix Q.1). Time-weighted average (TWA) media concentrations were calculated as outlined in the OSWER document entitled "Assessing Intermittent or Variable Exposures at Lead Sites" (U.S. EPA, 2003), using the following equation:

$$TWA = C \times N_p / P$$

where:

TWA = Time-weighted average (μg/L for groundwater);
C = Exposure concentration (μg/L for groundwater);
N_p = Number of exposure events within the exposure event period; and
P = Event period.

For the sake of the model, an event period (P) is considered to be one week, or seven days in length. Therefore, the exposure frequency of 350 days/year for groundwater was converted to N_p values of 6.71 days/7 days.

The TWA for groundwater of 4.18 μg/L was used as Site-specific concentrations in the model.

9.6 TOXICITY ASSESSMENT

The toxicity assessment weighs the available evidence regarding the potential for a particular COPC to cause adverse effects in exposed individuals and estimates the extent of exposure and possible severity of adverse effects. To develop toxicity values, two steps are taken: hazard identification and dose-response assessment. The hazard identification determines the potential adverse effects associated with exposure to a COPC. In the dose-response assessment, numerical toxicity values are determined or selected from the available toxicity data.

In the selection of toxicity values, preference has been given to the most recently developed values because these would incorporate the most recent toxicological information and would provide the best basis upon which to assess potential health hazards/risks. Toxicity values were primarily obtained from the U.S. EPA IRIS (Integrated Risk Information System) database, Health Effects Assessment Summary Table (HEAST), and U.S. EPA-NCEA provisional values as presented in U.S. EPA Region III RBC. Toxicity profiles of all COPCs in all media excluding all TICs is presented in Appendix Q.4

9.6.1 NON-CARCINOGENIC HAZARDS

9.6.1.1 TOXICITY INFORMATION FOR NON-CARCINOGENIC EFFECTS

For substances suspected to cause non-carcinogenic chronic effects, the health criteria are usually expressed as chronic intake levels or RfDs [in units of mg/(kg-day)] below which, no adverse effects are expected. In other words, there is a level of exposure to a chemical below which no toxic effects are expected. In contrast to the toxicological model used to assess carcinogenic risk, which assumes no concentration threshold, the non-carcinogenic dose-response model postulates a "threshold".

In this risk assessment, chronic RfDs are used as the toxicity values for non-carcinogenic health effects. A chronic RfD is defined as an estimate (with an uncertainty spanning an order of magnitude or greater) of a daily exposure level for the human population, including sensitive sub-populations, which poses no appreciable risk of deleterious effects over a lifetime of exposure. Uncertainty factors are incorporated into the RfDs to account for extrapolations from animal toxicity data, data quality, and to protect sensitive sub-populations. The basis of an RfD is usually the highest dose level administered to laboratory animals that did not cause observable adverse effects after chronic (usually lifetime) exposure. This is called the No-Observed Adverse Effect Level (NOAEL). The NOAEL is then divided by an uncertainty (safety) factor, and sometimes an additional modifying factor, to obtain the RfD. In general, an uncertainty factor of 10 is used to account for interspecies variation and another factor of 10 to account for sensitive human populations. Additional factors of 10 are included in the uncertainty factor if the RfD is based on the Lowest-Observed Adverse Effect Level (LOAEL) instead of the NOAEL, or if data inadequacies are present (e.g., the experiment for which the RfD was derived had less than lifetime exposure). The LOAEL is the dose level administered to laboratory animals that causes the lowest adverse effect (i.e., liver toxicity - although this is species and chemical-specific) after chronic exposure.

In evaluating the potential hazard from the non-carcinogenic effects of chemicals, the average daily exposure was based on shorter-term (non-lifetime) potential exposure estimates, the length of which is dependent on the exposure scenario.

Table 5.1 of Appendix Q.1 presents the non-cancer toxicity data (RfDs) used to estimate human health effects for oral and dermal exposure. Table 5.2 of Appendix Q.1 presents RfDs used for the inhalation exposure route.

9.6.2 CARCINOGENIC RISKS

9.6.2.1 TOXICITY INFORMATION FOR CARCINOGENIC EFFECTS

Cancer Slope Factors (CSFs) are quantitative risk estimates of carcinogenic potency. Slope factors relate the lifetime probability of excess cancers to the lifetime average exposure dose of a substance. CSFs are estimated using mathematical extrapolation models, most commonly the linearized multistage (LMS) model, and are presented as risk per mg/(kg-day) (i.e., mg carcinogen per kg body weight per day). These models assume low dose-response linearity and thus may not be appropriate for some suspect carcinogens, in particular those that function as promoters. As well, the body's natural repair processes and defense mechanisms may decrease cancer risk at low exposure levels. Thus, the risks at lower exposure levels are likely overestimated using the LMS model. When adequate human epidemiology data are available, maximum likelihood estimates (MLEs) of model parameters are used to generate a CSF. When only animal data are available, the CSF is derived from the largest possible linear slope that is consistent with the data (within the upper 95 percent confidence limit). In other words, the true risk to humans, while not identifiable, is not likely to exceed the upper-bound estimate. This is a conservative estimate, and in some cases a linear slope of zero may be as appropriate for the data (i.e., no carcinogenic risk).

Known or suspect human carcinogens have been evaluated and identified by the Carcinogen Assessment Group using the U.S. EPA Weight-of-Evidence approach for carcinogenicity classification (HEAST, 1997). The U.S. EPA classification is based on an evaluation of the likelihood that the agent is a human carcinogen. The evidence is characterized separately for human and animal studies as follows:

Group A -	Known Human Carcinogen (sufficient evidence of carcinogenicity in humans);
Group B -	Probable Human Carcinogen (B1 - limited evidence of carcinogenicity in humans; B2 - sufficient evidence of carcinogenicity in animals with inadequate or lack of evidence in humans);
Group C -	Possible Human Carcinogen (limited evidence of carcinogenicity in animals and inadequate or lack of human data);
Group D -	Not Classifiable as to Human Carcinogenicity (inadequate or no evidence); and
Group E -	Evidence of Non-carcinogenicity for Humans (no evidence of carcinogenicity in animal studies).

In evaluating the incremental lifetime cancer risks for carcinogens, the average daily exposure is based on lifetime exposure estimates. Lifetime estimates are generally based on exposure for 9 or 30 years of a 70-year lifetime.

The COPCs were classified utilizing the U.S. EPA system. Table 6.1 of Appendix Q.1 presents the cancer toxicity data (CSFs) used in the HHRA to estimate the risk of cancer for the oral and dermal exposure routes. Table 6.2 of Appendix Q.1 presents CSFs for the inhalation exposure route.

9.6.2.3 POTENTIAL RISK FROM CARCINOGENS

A CSF is applied to estimate the potential risk of cancer from exposure to carcinogenic chemicals. The CSF, when multiplied by the lifetime average daily intake, provides an estimate of the probability that the intake will cause cancer during the lifetime of the exposed individual. This increased cancer risk is expressed, for example, as 1×10^{-6} or 1.0E-06 (one in one million increased cancer risk). This is an upper limit estimate of the risk, based on very conservative health-protective assumptions and conservative statistical evaluations of data from animal experiments or epidemiological studies. To state that a chemical exposure causes a 1E-06 added upper limit risk of cancer means that if 1,000,000 people are exposed to this concentration of the chemical for their lifetimes, a maximum of one additional incident of cancer may occur. This is a very conservative estimate, and, in fact, there may be no additional cases of cancer due to the chemical exposure.

In the Federal Register publication (FR Vol. 52, No. 130, page 25700) in which U.S. EPA promulgated Maximum Contaminant Levels (MCLs) for certain volatile organic compounds, it is stated that "the target reference risk range for carcinogens is 10^{-6} to 10^{-4} and the maximum contaminant levels that the U.S. EPA is promulgating in this notice generally fall within this range. U.S. EPA considers these to be safe levels and protective of public health. This is supported by the concept expressed by the "WHO (World Health Organization) 1984 Guidelines for Drinking Water Quality", where it selected a 10^{-5} guideline value, and then explained that the application could vary by a factor of ten (i.e., 10^{-6} to 10^{-4}). This acceptable range of risk is applied to the general population in the U.S. when used as limits to develop Federal MCLs. This range of 1×10^{-6} to 1×10^{-4} that U.S. EPA considers protective of public health for drinking water was used as the target range in this evaluation of exposures to Site-related chemicals.

To maintain a conservative approach, reported TICs were evaluated as COPCs in the quantitative HHRA. In cases where toxicity data was available for a given TIC, it was used to estimate the carcinogenic risk and/or HI. In cases where toxicity data was not available for evaluation of a given TIC, a surrogate chemical for which data or published toxicity factors were available was selected on a structure/activity basis and the toxicity factors applied to the calculations for a given TIC. The surrogate chemical selected for the TICs were provided by U.S. EPA (2005). The surrogate chemicals and the corresponding RfDs and CSFs used to calculate the non-carcinogenic hazard and/or the carcinogenic risk for the TICs are presented in Tables 5.3 and 6.3 of Appendix Q.1, respectively.

9.7 RISK CHARACTERIZATION

The objective of this risk characterization is to integrate information developed in the exposure assessment (Section 9.5) and the toxicity assessment (Section 9.6) into a complete evaluation of the potential human health risks associated with exposure to potentially contaminated surface soil, soil, sediment, surface water, groundwater, and air at the Site. The methods used in this risk characterization are based on U.S. EPA HHRA guidance for human exposures (U.S. EPA, 1989a, 1991a, 1997, 1998, 2002a).

9.7.1 HAZARD ESTIMATES

The potential for non-cancer health effects from exposure to a COPC is evaluated by comparing an exposure level over a specified time period to a RfD for a similar time period. This ratio, termed the hazard quotient, is calculated according to the following general equation:

$$HQ = \frac{CDI}{RfD}$$

where:

HQ = The Hazard Quotient (unitless) is the ratio of the exposure dose of a chemical to a reference dose not expected to cause adverse effects from a lifetime exposure. A hazard quotient equal to or below 1.0 is considered protective of human health.

CDI = The Chronic Daily Intake is the chemical dose calculated by applying the exposure scenario assumptions and expressed as mg/(kg-day). The intake represents the average daily chemical dose over the expected period of exposure.

RfD = The Reference Dose is a daily dose believed not to cause an adverse effect from even a lifetime exposure [mg/(kg-day)]. The RfD is based on experimental data and/or epidemiological studies.

The HI is the sum of HQs for individual COPCs for a specific exposure scenario.

The summation of non-carcinogenic effects is only additive if they pertain to similar target organs. The HIs presented in Section 9.7.4 conservatively sum the non-carcinogenic effects without regard to target organs. HIs summed across similar target organs are presented on the hazard quotient calculation tables in Appendix Q.1.

The calculated HIs resulting from exposure to the COPCs are compared to an HI of 1. A HI equal to or below 1.0 is considered protective of human health over a lifetime and indicates that the exposure scenarios are not of concern. Typically, a HI between 1 and 10 suggests that exposure may reduce the margin of safety inherent in the exposure scenario and may be of possible concern for sensitive individuals. When the HI exceeds 10, there may be substantial concern for potential health effects. While any single COPC with an exposure level greater than the toxicity value will cause the HI to exceed 1, for multiple COPCs the HI can also exceed 1 due to the addition of multiple COPC HQs.

9.7.2 CANCER RISK ESTIMATES

Exposure scenarios may involve potential exposure to more than one carcinogen. To represent the potential carcinogenic effects posed by exposure to multiple carcinogens, it is assumed, in the absence of information on synergistic or antagonistic effects, that these risks are additive. Cancer risks are calculated utilizing the following general equation:

$$\text{Cancer Risk} = \text{LADD} \times \text{CSF}$$

where:

Cancer Risk = Estimated upper bound on additional risk of cancer over a lifetime in a population exposed to the carcinogen for a specified exposure period (unitless).

LADD = The Lifetime Average Daily Dose of the chemical calculated using exposure scenario assumptions and expressed in mg/(kg-day). The intake represents the total lifetime chemical dose averaged over an individual expected lifetime of 70 years.

CSF = The Cancer Slope Factor models the potential carcinogenic response and is expressed as $[\text{mg}/(\text{kg-day})]^{-1}$.

For estimating cancer risks from exposure to multiple carcinogens from a single exposure route, the following equation is used:

$$\mathrm{Risk}_T = \sum_{i=1}^{N} \mathrm{Risk}_i$$

where:

Risk_T = Total cancer risk from route of exposure
Risk_i = Cancer risk for the chemical
N = Number of chemicals

The cumulative carcinogenic risks are presented and discussed in Section 9.7.4. The potential cumulative risks resulting from exposure to the COPCs are compared to the target cumulative risk (TCR) of 1E-04 or 1 in 10,000, the upper level of the 1E-06 to 1E-04 risk range. When a cumulative risk to an individual under the assumed exposure conditions in an exposure unit exceeds 1E-04, remedial actions may be necessary.

9.7.3 RISK QUANTIFICATION SUMMARY

The hazard indices and excess lifetime cancer risks for both Site and non-Site-related COPCs for the various exposure scenarios for on- and off-Site media are presented in the Series 7 and Series 8 tables, respectively, in Appendix Q.1 and summarized below. Note that combined risks from dermal contact, incidental ingestion, and ambient air inhalation exposure are presented for soil and groundwater where appropriate.

9.7.3.1 SURFACE SOILS AND SURFACE SOIL TICS

The non-cancer hazard calculations and calculated lifetime cancer risks for surface soil receptors are presented in Appendix Q.1 and summarized below.

Current Trespassers/Hikers

Media	Route	Exposure	Non-Carcinogenic Hazard Index	HI > 1.0	Appendix Table Reference	Carcinogenic Risk	Risk > 10^{-4}	Risk > 10^{-6}	Appendix Table Reference
Surface Soil	Ingestion Dermal	CT	1.1E-02	No	7.1.CT	8.8E-08	No	No	8.1.CT
		RME	2.2E-02	No	7.1.RME	1.9E-07	No	No	8.1.RME
Surface Soil TICs	Ingestion Dermal	CT	7.8E-06	No	7.2.CT	NC	NA	NA	8.2.CT
		RME	1.6E-05	No	7.2.RME	NC	NA	NA	8.2.RME
Lagoon 6 Surface Soil	Ingestion Dermal	CT	3.8E-02	No	7.3.CT	5.3E-07	No	No	8.3.CT
		RME	8.9E-02	No	7.3.RME	1.2E-06	No	Yes	8.3.RME
Lagoon 6 Surface Soil TICs	Ingestion Dermal	CT	3.5E-04	No	7.4.CT	NC	NA	NA	8.4.CT
		RME	6.9E-04	No	7.4.RME	NC	NA	NA	8.4.RME

Notes:
NC- Not Calculated.
NA- Not Applicable.

Future Parkland Users (Child)

Media	Route	Exposure	Non-Carcinogenic Hazard Index	HI > 1.0	Appendix Table Reference	Carcinogenic Risk	Risk > 10^{-4}	Risk > 10^{-6}	Appendix Table Reference
Surface Soil	Ingestion Dermal	CT	4.3E-02	No	7.5.CT	2.8E-07	No	No	8.5.CT
		RME	9.2E-02	No	7.5.RME	6.6E-07	No	No	8.5.RME
Surface Soil TICs	Ingestion Dermal	CT	3.0E-05	No	7.7.CT	NC	NA	NA	8.7.CT
		RME	5.9E-05	No	7.7.RME	NC	NA	NA	8.7.RME
Lagoon 6 Surface Soil	Ingestion Dermal	CT	2.5E-01	No	7.9.CT	1.8E-06	No	Yes	8.9.CT
		RME	9.7E-01	No	7.9.RME	6.4E-06	No	Yes	8.9.RME
Lagoon 6 Surface Soil TICs	Ingestion Dermal	CT	1.3E-03	No	7.11.CT	NC	NA	NA	8.11.CT
		RME	2.6E-03	No	7.11.RME	NC	NA	NA	8.11.RME

Notes:
NC- Not Calculated.
NA- Not Applicable.

Future Parkland Users (Adult)

Media	Route	Exposure	Non-Carcinogenic Hazard Index	HI > 1.0	Appendix Table Reference	Carcinogenic Risk	Risk > 10^{-4}	Risk > 10^{-6}	Appendix Table Reference
Surface Soil	Ingestion Dermal	CT	8.4E-03	No	7.6.CT	2.5E-08	No	No	8.6.CT
		RME	1.8E-02	No	7.6.RME	9.0E-07	No	No	8.6.RME
Surface Soil TICs	Ingestion Dermal	CT	5.9E-06	No	7.8.CT	NC	NA	NA	8.8.CT
		RME	1.2E-05	No	7.8.RME	NC	NA	NA	8.8.RME
Lagoon 6 Surface Soil	Ingestion Dermal	CT	4.2E-02	No	7.10.CT	1.6E-07	No	No	8.10.CT
		RME	2.5E-01	No	7.10.RME	6.4E-06	No	Yes	8.10.RME
Lagoon 6 Surface Soil TICs	Ingestion Dermal	CT	2.6E-04	No	7.12.CT	NC	NA	NA	8.12.CT
		RME	5.2E-04	No	7.12.RME	NC	NA	NA	8.12.RME

Notes:
NC- Not Calculated.
NA- Not Applicable.

Future Maintenance Worker

Media	Route	Exposure	Non-Carcinogenic Hazard Index	HI > 1.0	Appendix Table Reference	Carcinogenic Risk	Risk > 10^{-4}	Risk > 10^{-6}	Appendix Table Reference
Surface Soil	Ingestion Dermal	CT	1.2E-01	No	7.13.CT	9.1E-07	No	No	8.13.CT
		RME	1.3E-01	No	7.13.RME	3.7E-06	No	Yes	8.13.RME
Surface Soil TICs	Ingestion Dermal	CT	8.7E-05	No	7.14.CT	NC	NA	NA	8.14.CT
		RME	8.7E-05	No	7.14.RME	NC	NA	NA	8.14.RME
Lagoon 6 Surface Soil	Ingestion Dermal	CT	4.5E-01	No	7.15.CT	5.5E-06	No	Yes	8.15.CT
		RME	8.1E-01	No	7.15.RME	2.4E-05	No	Yes	8.15.RME
Lagoon 6 Surface Soil TICs	Ingestion Dermal	CT	3.8E-03	No	7.16.CT	NC	NA	NA	8.16.CT
		RME	3.8E-03	No	7.16.RME	NC	NA	NA	8.16.RME

Notes:
NC- Not Calculated.
NA- Not Applicable.

Future Resident (Child)

Media	Route	Exposure	Non-Carcinogenic Hazard Index	HI > 1.0	Appendix Table Reference	Carcinogenic Risk	Risk > 10^{-4}	Risk > 10^{-6}	Appendix Table Reference
Surface Soil	Ingestion Dermal	CT	3.4E+00	Yes	7.17.CT	1.7E-05	No	Yes	8.17.CT
		RME	3.4E+00	Yes	7.17.RME	1.8E-05	No	Yes	8.17.RME
Surface Soil TICs	Ingestion Dermal	CT	2.4E-03	No	7.19.CT	NC	NA	NA	8.19.CT
		RME	2.4E-03	No	7.19.RME	NC	NA	NA	8.19.RME
Lagoon 6 Surface Soil	Ingestion Dermal	CT	1.2E+01	Yes	7.21.CT	1.0E-04	No	Yes	8.21.CT
		RME	1.4E+01	Yes	7.21.RME	1.1E-04	Yes	Yes	8.21.RME
Lagoon 6 Surface Soil TICs	Ingestion Dermal	CT	1.1E-01	No	7.23.CT	NC	NA	NA	8.23.CT
		RME	1.1E-01	No	7.23.RME	NC	NA	NA	8.23.RME

Notes:
NC- Not Calculated.
NA- Not Applicable.

Future Resident (Adult)

Media	Route	Exposure	Non-Carcinogenic Hazard Index	HI > 1.0	Appendix Table Reference	Carcinogenic Risk	Risk > 10^{-4}	Risk > 10^{-6}	Appendix Table Reference
Surface Soil	Ingestion Dermal	CT	3.8E-01	No	7.18.CT	9.3E-07	No	No	8.18.CT
		RME	3.9E-01	No	7.18.RME	8.5E-06	No	Yes	8.18.RME
Surface Soil TICs	Ingestion Dermal	CT	2.7E-04	No	7.20.CT	NC	NA	NA	8.20.CT
		RME	2.7E-04	No	7.20.RME	NC	NA	NA	8.20.RME
Lagoon 6 Surface Soil	Ingestion Dermal	CT	1.4E+00	Yes	7.22.CT	5.6E-06	No	Yes	8.22.CT
		RME	1.7E+00	Yes	7.22.RME	5.3E-05	No	Yes	8.22.RME
Lagoon 6 Surface Soil TICs	Ingestion Dermal	CT	1.2E-02	No	7.24.CT	NC	NA	NA	8.24.CT
		RME	1.2E-02	No	7.24.RME	NC	NA	NA	8.24.RME

Notes:
NC- Not Calculated.
NA- Not Applicable.

9.7.3.2 SOILS AND SOIL TICS

The non-cancer hazard calculations and calculated lifetime cancer risks for future construction worker soil receptors are presented in Appendix Q.1 and summarized below.

Construction Worker

Media	Route	Exposure	Non-Carcinogenic Hazard Index	HI > 1.0	Appendix Table Reference	Carcinogenic Risk	Risk > 10^{-4}	Risk > 10^{-6}	Appendix Table Reference
Soil	Ingestion Dermal Inhalation	CT	4.1E+01	Yes	7.25.CT	5.0E-05	No	Yes	8.25.CT
		RME	1.2E+02	Yes	7.25.RME	1.5E-04	Yes	Yes	8.25.RME
Soil TICs	Ingestion Dermal Inhalation	CT	1.2E-02	No	7.26.CT	NC	NA	NA	8.26.CT
		RME	3.7E-02	No	7.26.RME	NC	NA	NA	8.26.RME
Lagoon 6 Soil	Ingestion Dermal Inhalation	CT	1.6E+00	Yes	7.27.CT	8.6E-07	No	No	8.27.CT
		RME	4.8E+00	Yes	7.27.RME	2.7E-06	No	Yes	8.27.RME
Lagoon 6 Soil TICs	Ingestion Dermal Inhalation	CT	3.3E-03	No	7.28.CT	NC	NA	NA	8.28.CT
		RME	1.0E-02	No	7.28.RME	NC	NA	NA	8.28.RME

Notes:
NC- Not Calculated.
NA- Not Applicable.

9.7.3.3 SEDIMENTS AND SEDIMENT TICS

The non-cancer hazard calculations and calculated lifetime cancer risks for sediment receptors are presented in Appendix Q.1 and summarized below.

Current/Future Occasional Visitors/Hikers

Media	Route	Exposure	Non-Carcinogenic Hazard Index	HI > 1.0	Appendix Table Reference	Carcinogenic Risk	Risk > 10^{-4}	Risk > 10^{-6}	Appendix Table Reference
Northeast Marsh Area Sediment	Ingestion Dermal	CT	3.5E-02	No	7.29.CT	4.0E-08	No	No	8.29.CT
		RME	7.1E-02	No	7.29.RME	8.6E-08	No	No	8.29.RME
Northeast Marsh Area Sediment TICs	Ingestion Dermal	CT	NC	NA	7.30.CT	NC	NA	NA	8.30.CT
		RME	NC	NA	7.30.RME	NC	NA	NA	8.30.RME

Notes:
NC- Not Calculated.
NA- Not Applicable.

Current Trespassers/Hikers

Media	Route	Exposure	Non-Carcinogenic Hazard Index	HI > 1.0	Appendix Table Reference	Carcinogenic Risk	Risk > 10^{-4}	Risk > 10^{-6}	Appendix Table Reference
Southwest Marsh Area Sediment	Ingestion Dermal	CT	2.0E-01	No	7.31.CT	8.0E-07	No	No	8.31.CT
		RME	3.0E-01	No	7.31.RME	1.4E-06	No	Yes	8.31.RME

Notes:
NC- Not Calculated.
NA- Not Applicable.

Future Recreational Users (Child and Adult)

Media	Route	Exposure	Non-Carcinogenic Hazard Index	HI > 1.0	Appendix Table Reference	Carcinogenic Risk	Risk > 10^{-4}	Risk > 10^{-6}	Appendix Table Reference
Southwest Marsh Area Sediment (child)	Ingestion Dermal	CT	8.7E-01	No	7.32.CT	2.2E-06	No	Yes	8.32.CT
		RME	1.3E+00	Yes	7.32.RME	3.7E-06	No	Yes	8.32.RME
Southwest Marsh Area Sediment (adult)	Ingestion Dermal	CT	9.9E-02	No	7.33.CT	1.2E-07	No	No	8.33.CT
		RME	1.5E-01	No	7.33.RME	1.8E-06	No	Yes	8.33.RME

Notes:
NC- Not Calculated.
NA- Not Applicable.

9.7.3.4 SURFACE WATER AND SURFACE WATER TICs

The non-cancer hazard calculations and calculated lifetime cancer risks for surface water receptors are presented in Appendix Q.1 and summarized below.

Current/Future Occasional Visitors/Hikers

Media	Route	Exposure	Non-Carcinogenic Hazard Index	HI > 1.0	Appendix Table Reference	Carcinogenic Risk	Risk > 10^{-4}	Risk > 10^{-6}	Appendix Table Reference
Northeast Marsh Area Surface Water	Dermal	CT	3.3E-04	No	7.34.CT	7.2E-10	No	No	8.34.CT
		RME	1.3E-03	No	7.34.RME	2.9E-09	No	No	8.34.RME
Northeast Marsh Area Surface Water TICs	Dermal	CT	NC	NA	7.35.CT	NC	NA	NA	8.35.CT
		RME	NC	NA	7.35.RME	NC	NA	NA	8.35.RME

Notes:
NC- Not Calculated.
NA- Not Applicable.

Current Trespassers/Hikers

Media	Route	Exposure	Non-Carcinogenic Hazard Index	HI > 1.0	Appendix Table Reference	Carcinogenic Risk	Risk > 10^{-4}	Risk > 10^{-6}	Appendix Table Reference
Beaverdam Brook Surface Water	Dermal	CT	1.0E-02	No	7.36.CT	6.7E-09	No	No	8.36.CT
		RME	3.0E-02	No	7.36.RME	1.2E-08	No	No	8.36.RME
Beaverdam Brook Surface Water TICs	Dermal	CT	NC	NA	7.37.CT	NC	NA	NA	8.37.CT
		RME	NC	NA	7.37.RME	NC	NA	NA	8.37.RME

Notes:
NC- Not Calculated.
NA- Not Applicable.

Future Recreational Users (Child)

Media	Route	Exposure	Non-Carcinogenic Hazard Index	HI > 1.0	Appendix Table Reference	Carcinogenic Risk	Risk > 10^{-4}	Risk > 10^{-6}	Appendix Table Reference
Beaverdam Brook Surface Water	Dermal	CT	2.0E-02	No	7.38.CT	7.8E-09	No	No	8.38.CT
		RME	5.9E-02	No	7.38.RME	2.4E-08	No	No	8.38.RME
Beaverdam Brook Surface Water TICs	Dermal	CT	NC	NA	7.40.CT	NC	NA	NA	8.40.CT
		RME	NC	NA	7.40.RME	NC	NA	NA	8.40.RME

Notes:
NC- Not Calculated.
NA- Not Applicable.

Future Recreational Users (Adult)

Media	Route	Exposure	Non-Carcinogenic Hazard Index	HI > 1.0	Appendix Table Reference	Carcinogenic Risk	Risk > 10^{-4}	Risk > 10^{-6}	Appendix Table Reference
Beaverdam Brook Surface Water	Dermal	CT	5.0E-03	No	7.39.CT	1.0E-09	No	No	8.39.CT
		RME	1.5E-02	No	7.39.RME	2.4E-08	No	No	8.39.RME
Beaverdam Brook Surface Water TICs	Dermal	CT	NC	NA	7.41.CT	NC	NA	NA	8.41.CT
		RME	NC	NA	7.41.RME	NC	NA	NA	8.41.RME

Notes:
NC- Not Calculated.
NA- Not Applicable.

Current Trespassers/Hikers

Media	Route	Exposure	Non-Carcinogenic Hazard Index	HI > 1.0	Appendix Table Reference	Carcinogenic Risk	Risk > 10^{-4}	Risk > 10^{-6}	Appendix Table Reference
Otter Kill Surface Water	Dermal	CT	5.8E-03	No	7.42.CT	1.1E-08	No	No	8.42.CT
		RME	1.7E-02	No	7.42.RME	3.4E-08	No	No	8.42.RME
Otter Kill Surface Water TICs	Dermal	CT	NC	NA	7.43.CT	NC	NA	NA	8.43.CT
		RME	NC	NA	7.43.RME	NC	NA	NA	8.43.RME

Notes:
NC- Not Calculated.
NA- Not Applicable.

Future Recreational Users (Child)

Media	Route	Exposure	Non-Carcinogenic Hazard Index	HI > 1.0	Appendix Table Reference	Carcinogenic Risk	Risk > 10^{-4}	Risk > 10^{-6}	Appendix Table Reference
Otter Kill Surface Water	Dermal	CT	1.1E-02	No	7.44.CT	1.3E-08	No	No	8.44.CT
		RME	3.4E-02	No	7.44.RME	3.9E-08	No	No	8.44.RME
Otter Kill Surface Water TICs	Dermal	CT	NC	NA	7.46.CT	NC	NA	NA	8.46.CT
		RME	NC	NA	7.46.RME	NC	NA	NA	8.46.RME

Notes:
NC- Not Calculated.
NA- Not Applicable.

Future Recreational Users (Adult)

Media	Route	Exposure	Non-Carcinogenic Hazard Index	HI > 1.0	Appendix Table Reference	Carcinogenic Risk	Risk > 10^{-4}	Risk > 10^{-6}	Appendix Table Reference
Otter Kill Surface Water	Dermal	CT	2.9E-03	No	7.45.CT	1.7E-09	No	No	8.45.CT
		RME	8.7E-03	No	7.45.RME	4.0E-08	No	No	8.45.RME
Otter Kill Surface Water TICs	Dermal	CT	NC	NA	7.47.CT	NC	NA	NA	8.47.CT
		RME	NC	NA	7.47.RME	NC	NA	NA	8.47.RME

Notes:
NC- Not Calculated.
NA- Not Applicable.

Future Recreational Anglers (Child)

Media	Route	Exposure	Non-Carcinogenic Hazard Index	HI > 1.0	Appendix Table Reference	Carcinogenic Risk	Risk > 10^{-4}	Risk > 10^{-6}	Appendix Table Reference
Otter Kill Surface Water	Ingestion	CT	2.0E+00	Yes	7.48.CT	2.1E-05	No	Yes	8.48.CT
		RME	1.1E+01	Yes	7.48.RME	1.1E-04	Yes	Yes	8.48.RME
Otter Kill Surface Water TICs	Ingestion	CT	6.0E-04	No	7.50.CT	NC	NA	NA	8.50.CT
		RME	3.2E-03	No	7.50.RME	NC	NA	NA	8.50.RME

Notes:
NC- Not Calculated.
NA- Not Applicable.

Future Recreational Anglers (Adult)

Media	Route	Exposure	Non-Carcinogenic Hazard Index	HI > 1.0	Appendix Table Reference	Carcinogenic Risk	Risk > 10^{-4}	Risk > 10^{-6}	Appendix Table Reference
Otter Kill Surface Water	Ingestion	CT	1.4E+00	Yes	7.49.CT	7.3E-06	No	Yes	8.49.CT
		RME	7.4E+00	Yes	7.49.RME	3.1E-04	Yes	Yes	8.49.RME
Otter Kill Surface Water TICs	Ingestion	CT	4.1E-04	No	7.51.CT	NC	NA	NA	8.51.CT
		RME	2.2E-03	No	7.51.RME	NC	NA	NA	8.51.RME

Notes:
NC- Not Calculated.
NA- Not Applicable.

9.7.3.5 GROUNDWATER AND GROUNDWATER TICs

The non-cancer hazard calculations and calculated lifetime cancer risks for groundwater receptors are presented in Appendix Q.1 and summarized below.

Future Resident (Child)

Media	Route	Exposure	Non-Carcinogenic Hazard Index	HI > 1.0	Appendix Table Reference	Carcinogenic Risk	Risk > 10^{-4}	Risk > 10^{-6}	Appendix Table Reference
On-Site Ground water	Ingestion Dermal Inhalation	CT	4.2E+02	Yes	7.52.CT	3.6E-04	Yes	Yes	8.52.CT
		RME	6.1E+02	Yes	7.52.RME	5.4E-04	Yes	Yes	8.52.RME
On-Site Ground Water TICs	Ingestion Dermal Inhalation	CT	5.1E+01	Yes	7.54.CT	NC	NA	NA	8.54.CT
		RME	7.5E+01	Yes	7.54.RME	NC	NA	NA	8.54.RME

Notes:
NC- Not Calculated.
NA- Not Applicable.

Future Resident (Adult)

Media	Route	Exposure	Non-Carcinogenic Hazard Index	HI > 1.0	Appendix Table Reference	Carcinogenic Risk	Risk > 10^{-4}	Risk > 10^{-6}	Appendix Table Reference
On-Site Ground Water	Ingestion Dermal Inhalation	CT	1.9E+02	Yes	7.53.CT	7.4E-05	No	Yes	8.53.CT
		RME	2.8E+02	Yes	7.53.RME	8.2E-04	Yes	Yes	8.53.RME
On-Site Ground Water TICs	Ingestion Dermal Inhalation	CT	2.2E+01	Yes	7.55.CT	NC	NA	NA	8.55.CT
		RME	3.2E+01	Yes	7.55.RME	NC	NA	NA	8.55.RME

Notes:
NC- Not Calculated.
NA- Not Applicable.

Future Construction Worker (Adult)

Media	Route	Exposure	Non-Carcinogenic Hazard Index	HI > 1.0	Appendix Table Reference	Carcinogenic Risk	Risk > 10^{-4}	Risk > 10^{-6}	Appendix Table Reference
On-Site Ground Water	Ingestion Dermal Inhalation	CT	1.4E-01	No	7.56.CT	2.8E-07	No	No	8.56.CT
		RME	7.2E-01	No	7.56.RME	1.4E-06	No	Yes	8.56.RME
On-Site Ground Water TICs	Ingestion Dermal Inhalation	CT	1.7E-03	No	7.57.CT	NC	NA	NA	8.57.CT
		RME	1.0E-02	No	7.57.RME	NC	NA	NA	8.57.RME

Notes:
NC- Not Calculated.
NA- Not Applicable.

Current Resident (Child And Adult)

Media	Route	Exposure	Non-Carcinogenic Hazard Index	HI > 1.0	Appendix Table Reference	Carcinogenic Risk	Risk > 10^{-4}	Risk > 10^{-6}	Appendix Table Reference
Off-Site Ground water (child)	Ingestion Dermal Inhalation	CT	3.2E+00	Yes	7.58.CT	NC	NA	NA	8.58.CT
		RME	4.6E+00	Yes	7.58.RME	NC	NA	NA	8.58.RME
Off-Site Ground water (adult)	Ingestion Dermal Inhalation	CT	1.5E+00	Yes	7.59.CT	NC	NA	NA	8.59.CT
		RME	2.1E+00	Yes	7.59.RME	NC	NA	NA	8.59.RME

Notes:
NC- Not Calculated.
NA- Not Applicable.

9.7.4 SUMMATION OF HAZARDS AND RISKS

A given population may be exposed to a chemical from several exposure routes and pathways. The purpose of this section is to identify the risks associated with a population that may be exposed to COPCs through a combination of the evaluated pathways.

RAGS (U.S. EPA 1989a) (at pg. 8-15) states that risks should be combined across exposure pathways only where the following occurs:

a) reasonable exposure pathway combinations are identified; and

b) it appears likely that the same individuals would consistently face the "reasonable maximum exposure" ("RME") by more than one pathway.

As opposed to encouraging the calculation of combined risks from across exposure pathways, RAGS (U.S. EPA 1989a) cautions (at pg. 8-16) that each RME estimate includes many conservative assumptions and combining estimates is not appropriate unless the combination itself is a RME:

> *"For real world situations in which contaminant concentrations vary over time and space, the same individual may or may not experience the RME for more than one pathway over the same period of time. One individual might face the RME through one pathway, and a different individual face the RME through a different pathway. Only if you can explain why the key RME assumptions for more than one pathway apply to the same individual or sub-population should the RME risks for more than one pathway be combined.*

> *In some situations, it may be appropriate to combine one pathway's RME risks with other pathways' risk estimates that have been derived from more typical exposure parameter values". (Emphasis added).*

It is improbable that the same person would experience all potential exposures the same number of times or over the period of years specified in the individual RME scenarios. As a result, it may be inappropriate to add together the estimated risks and hazards for the different exposure routes and pathways because this could result in the exaggeration of an appropriate RME for the summed exposures. Therefore, it is unlikely the summation of the RME estimates would result in an underestimations of risk, and this estimate should be evaluated as a conservative estimate of the likely exposure at the Site.

To maintain a conservative approach to the HHRA, RME risks and hazards for separate exposure routes and exposure pathways were combined to estimate total RME exposures for the same potentially exposed receptor group.

However, it would be inappropriate to sum the results of the exposure scenarios that were evaluated separately for an exposed population in the Site area with those from the Lagoon 6 area, without accounting for the relative percentage of time a receptor would spend in one area versus the other. The exposure scenarios evaluated in the HHRA assumed that the receptor spent all their time in the exposure area being considered in order to receive the chemical dose. Thus, the risks and hazards estimated separately for a receptor group (i.e., trespasser/occasional visitor, parkland/recreational users, construction worker, residents, and maintenance worker) in the Site area (excluding Lagoon 6) and the Lagoon 6 area are not considered additive. Tables 9.1 through 9.11 of Appendix Q.1 present the total combined risks and hazards for the exposed populations evaluated under the current and future Site conditions. Section 9.8.1 provides a discussion of more realistic combined exposures for the evaluated receptors than the overestimation from a simple straight summation of individual RME risks.

9.7.4.1 FORMER LAGOON SITE EXCLUDING LAGOON 6

The cumulative HIs and lifetime cancer risks across all applicable exposure routes for receptors in the former lagoon site area are presented in Appendix Q.1 and summarized below.

Receptor	Media	Exposure	Cumulative Non-Carcinogenic Hazard Index	HI > 1.0	Cumulative Carcinogenic Risk	Risk > 10^{-4}	Risk > 10^{-6}	Appendix Q.1 Table Reference
Trespasser/ Hiker/ Occasional Visitor (Current)	(1)	CT	1.7E+00	Yes	8.3E-06	No	Yes	9.1.CT
		RME	7.9E+00	Yes	3.1E-04	Yes	Yes	9.1.RME
Parkland User/ Occasional Visitor/ Recreational User (Future)	(1)	CT	3.0E+00	Yes	3.1E-05	No	Yes	9.3.CT
		RME	1.2E+01	Yes	4.3E-04	Yes	Yes	9.3.RME
Maintenance Worker (Future)	(2)	CT	1.2E-01	No	9.1E-07	No	No	9.5.CT
		RME	1.3E-01	No	3.7E-06	No	Yes	9.5.RME
Resident/ Occasional Visitor/ Recreational User (Future)	(3)	CT	4.8E+02	Yes	4.8E-04	Yes	Yes	9.7.CT
		RME	7.0E+02	Yes	1.8E-03	Yes	Yes	9.7.RME
Construction Worker (Future)	(4)	CT	4.1E+01	Yes	5.0E-05	No	Yes	9.9.CT
		RME	1.2E+02	Yes	1.5E-04	Yes	Yes	9.9.RME

Notes:

(1) Media include surface soil, surface soil TICs, sediment-Northeast Marsh Area, sediment TICs-Northeast Marsh Area, sediment-Southwest Marsh Area, surface water-Northeast Marsh Area, surface water TICs-Northeast Marsh Area, surface water-Beaverdam Brook, surface water TICs-Beaverdam Brook, surface water-Otter Kill, fish consumption- Otter Kill, and surface water TICs-Otter Kill.

(2) Media include surface soil, and surface soil TICs.

(3) Media include surface soil, surface soil TICs, sediment-Northeast Marsh Area, sediment TICs-Northeast Marsh Area, sediment-Southwest Marsh Area, surface water-Northeast Marsh Area, surface water TICs-Northeast Marsh Area, surface water-Beaverdam Brook, surface water TICs-Beaverdam Brook, surface water-Otter Kill, fish consumption- Otter Kill, surface water TICs-Otter Kill, groundwater, and groundwater TICs.

(4) Media include soil, soil TICs, groundwater, and groundwater TICs.

9.7.4.2 FORMER LAGOON SITE INCLUSIVE OF LAGOON 6

Receptor	Media	Exposure	Cumulative Non-Carcinogenic Hazard Index	HI > 1.0	Cumulative Carcinogenic Risk	Risk > 10^{-4}	Risk > 10^{-6}	Appendix Table Reference
Trespasser/ Hiker/ Occasional Visitor (Current)	(1)	CT	1.7E+00	Yes	8.7E-06	No	Yes	9.2.CT
		RME	8.0E+00	Yes	3.1E-04	Yes	Yes	9.2.RME
Parkland User/ Occasional Visitor/ Recreational User (Future)	(1)	CT	3.2E+00	Yes	3.3E-05	No	Yes	9.4.CT
		RME	1.3E+01	Yes	4.4E-04	Yes	Yes	9.4.RME
Maintenance Worker (Future)	(2)	CT	4.6E-01	No	5.5E-06	No	Yes	9.6.CT
		RME	8.1E-01	No	2.4E-05	No	Yes	9.6.RME
Resident/ Occasional Visitor/ Recreational User (Future)	(3)	CT	4.9E+02	Yes	5.7E-04	Yes	Yes	9.8.CT
		RME	7.1E+02	Yes	1.9E-03	Yes	Yes	9.8.RME
Construction Worker (Future)	(4)	CT	1.7E+00	Yes	1.1E-06	No	Yes	9.10.CT
		RME	5.5E+00	Yes	4.1E-06	No	Yes	9.10.RME

Notes:

(1) Media include Lagoon 6 surface soil, Lagoon 6 surface soil TICs, sediment-Northeast Marsh Area, sediment TICs-Northeast Marsh Area, sediment-Southwest Marsh Area, surface water-Northeast Marsh Area, surface water TICs-Northeast Marsh Area, surface water-Beaverdam Brook, surface water TICs-Beaverdam Brook, surface water-Otter Kill, fish- Otter Kill, and surface water TICs-Otter Kill.

(2) Media include Lagoon 6 surface soil, and Lagoon 6 surface soil TICs.

(3) Media include Lagoon 6 surface soil, Lagoon 6 surface soil TICs, sediment-Northeast Marsh Area, sediment TICs-Northeast Marsh Area, sediment-Southwest Marsh Area, surface water-Northeast Marsh Area, surface water TICs-Northeast Marsh Area, surface water-Beaverdam Brook, surface water TICs-Beaverdam Brook, surface water-Otter Kill, fish- Otter Kill, surface water TICs-Otter Kill, groundwater, and groundwater TICs.

(4) Media include Lagoon 6 soil, Lagoon 6 soil TICs, groundwater, and groundwater TICs.

9.7.4.3 OFF-SITE GROUNDWATER

Receptor	Media	Exposure	Cumulative Non-Carcinogenic Hazard Index	HI > 1.0	Cumulative Carcinogenic Risk	Risk > 10^{-4}	Risk > 10^{-6}	Appendix Table Reference
Resident (Current)	Groundwater	CT	3.2E+00	Yes	NC	NA	NA	9.11.CT
		RME	4.6E+00	Yes	NC	NA	NA	9.11.RME

Notes:

NC- Not Calculated.
NA- Not Applicable.

9.8 SUMMARY OF EXCEEDANCES

The media and receptors for which exceedances in the cumulative screening level HI of 1.0 or risk of 10^{-4} based on the RME exposure assumption were observed, are summarized below.

RECEPTOR	*PRIMARY MEDIA*	*EXCEEDS HI*	*EXCEEDS RISK*
Trespasser/ Hiker/ Occasional Visitor (Current)	Consumption of Fish-Otter Kill	Yes	No
Parkland User/ Occasional Visitor/ Recreational User (Future)	Consumption of Fish- Otter Kill	Yes	No
Parkland User/ Occasional Visitor/ Recreational User (Future)	Lagoon 6 Surface Soil, Southwest Marsh Area sediments, Consumption of Fish- Otter Kill	Yes	No
Resident/ Occasional Visitor/ Recreational User (Future)	Surface Soil, Southwest Marsh Area sediments, Consumption of Fish-Otter Kill, Groundwater, Groundwater TICs	Yes	No
Resident/ Occasional Visitor/ Recreational User (Future)	Lagoon 6 Surface Soil, Consumption of Fish- Otter Kill, Groundwater, Groundwater TICs	Yes	Yes
Construction Worker (Future)	Soil	Yes	Yes
Construction Worker (Future)	Lagoon 6 Soil	Yes	No
Resident (Current)	Off-Site Groundwater	Yes	No

9.8.1 OVERESTIMATION OF THE SUMMATION OF RISKS

The summation of hazards and risks presented in Section 9.7.4 assume that the same individual is exposed to all the various media at the Site and in the Site vicinity over all the exposure periods specified in the individual exposure scenarios. But, the likelihood of this happening is very low, and the summation of hazards and risks in this manner is a very conservative approach that should be qualified through the presentation of a more reasonable and likely combined exposure.

For example, the combined RME hazard estimate for the future adult parkland user assumes the following:

- they are on Site in the lagoon areas exposed to surface soils for 2 hours/day, for 180 days/year,
- they are also exposed to sediment and surface water in the Northeast Marsh for 10 days/year,
- they are also exposed to sediment in the Southwest Marsh for 78 days/year,
- they are also exposed to surface water in Beaverdam Brook for 2 hours/day, for 78 days/year,
- they are also exposed to surface water in Otter Kill for 2 hours/day, for 78 days/year, and
- they also consume fish caught from Otter Kill 365 days/year.

Because the exposure frequency for many of the RME scenarios is high, it can be assumed that the separate individual exposures to the several exposure areas at the Site would have to take place on the same day. Thus, on these frequent multiple exposure days, the parkland user is assumed to spend as much as 6 or 8 hours on Site. In addition, most of the evaluated exposure areas are at a fair distance from one another, and it would not be unreasonable to assume that a parkland user would spend additional time at the Site getting from one place to another in order to receive the exposure represented by the summation of risks and hazards in the previous sections.

If a parkland user is reasonably assumed to spend at most approximately 2 hours per exposure day in the park, it may be appropriate to assume that they spend more time on the park trails then they would wading through the surface water bodies. It is reasonable to assume that 85 percent of their time is spent on land exposed to Site surface soils, while 5 percent of their time is spent in each of the Southwest Marsh, Beaverdam Brook and Otter Kill areas. Therefore, on those days that a parkland user visits several or all areas of the Site, perhaps as many as 78 days/year for an adult, it is reasonable and realistic to assume a lower combined exposure than is represented by the summation of all potential exposure scenarios as presented in the previous sections. This more realistic combined exposure for an individual will result in lower risks and hazards.

9.9 RISK AND HAZARD COPC CONTRIBUTIONS

The contribution of total risk and hazard from each COPC for a specific combined exposure scenario varies depending on the COPC's exposure point concentration and relative toxicity. Frequently, the majority of risk estimated for an exposure scenario can be narrowed to a few COPCs. The following presents a summary of the COPCs contributing the majority of risk and hazard for the combined RME scenarios that exceeded the U.S. EPA established risk and hazard target levels.

Current Trespasser/Hiker/Occasional Visitor to Site Area

Table 10.1.RME of Appendix Q.1 presents the COPCs contributing the majority of the combined risk and hazard for the RME current trespasser/hiker/occasional visitor to Otter Kill surface water media. The cumulative RME risk estimate of 3.1E-04 primarily results from exposure to arsenic through ingestion of fish from Otter Kill. Arsenic accounts for 99 percent of the cumulative risk. The cumulative RME HI of 7.9 primarily results from exposure to arsenic, copper, manganese, and thallium through ingestion of fish from Otter Kill. Arsenic, copper, manganese, and thallium account for approximately 26 percent, 32 percent, 7 percent, and 30 percent, respectively, for a total contribution of 95 percent of the cumulative HI. Although manganese contributes 7 percent to the cumulative HI value of 7.9, the HI for central nervous system effects of 0.52 does not exceed the benchmark of 1. Therefore, it is unlikely that an exposure to average concentrations of manganese through ingestion of fish will result in a health hazard to the current trespasser/hiker/occasional visitor to the Site area.

As shown in Table 9.2, arsenic, manganese, and thallium are consistent with background levels in soil, therefore, there is no Site-related source for these constituents. Cancer risks associated with exposure to arsenic in fish tissue are at the upperbound of the acceptable risk range. However, it should be noted that no other carcinogenic contaminants have been identified in fish tissue, so the cumulative estimated risk is therefore also at the upperbound of the acceptable risk range. In addition to arsenic, copper, and thallium were also identified in fish tissue at exposure point concentrations associated with levels of noncancer hazard that exceed the benchmark of a hazard quotient of 1 by less than an order of magnitude. First, none of these metals have been identified as chemicals of concern (COCs) in on-Site soils or groundwater. Second, as previously stated in Section 9.4.6, it is important to reiterate that the fish tissue concentrations were estimated from surface water RME concentrations in the Otter Kill using chemical-specific bioconcentration factors. For all three of these contaminants, due to a paucity of data that did not allow for an upperbound of the average concentration to be calculated, the RME surface water concentration therefore defaulted to the maximum detected concentration. This means that the fish tissue concentration was estimated using the maximum detected surface water concentration for arsenic, copper, and thallium. This presents a very high-end estimate of the fish tissue concentration for these three contaminants.

Taking into account the fact that none of these inorganics were identified as COCs in on-Site soils or groundwater, that the fish tissue concentrations used to estimate risks and hazard are based on a model that conservatively used the maximum detected surface water concentration and a bioconcentration factor that is likely to present a very high-end estimate of the actual fish tissue concentration, and that the cancer risks are within the upperbound of the acceptable risk range and that the noncancer hazards are within an order of magnitude of the benchmark of a hazard quotient of 1, the risks and hazards presented are likely to be biased high and actual exposure to fish tissue is unlikely to result in an unacceptable risk or hazard to human health.

As shown in Table 9.3, the background concentrations for copper in surface water are higher than copper concentrations adjacent to the Site in Beaverdam Brook and Otter Kill, therefore copper in surface water may be impacted by an up-gradient non-Site-related source.

Current Trespasser/Hiker/Occasional Visitor in Lagoon 6

Table 10.2.RME of Appendix Q.1 presents the COPCs contributing the majority of the combined risk and hazard for the RME current trespasser/hiker/occasional visitor. Similar to the above scenario, the Otter Kill surface water media again has the greatest contribution. The cumulative RME risk estimate of 3.1E-04 primarily results from exposure to arsenic through ingestion of fish from Otter Kill. Arsenic accounts for 99 percent of the cumulative risk. The cumulative RME HI of 8.0 primarily results from exposure to arsenic, copper, manganese, and thallium through ingestion of fish from Otter Kill. Arsenic, copper, manganese, and thallium account for approximately 25 percent, 31 percent, 7 percent, and 30 percent respectively, for a total contribution of 93 percent of the cumulative HI. Although manganese contributes 7 percent to the cumulative HI value of 8.0, the HI for central nervous system effects of 0.52 does not exceeds the benchmark of 1. Therefore, it is unlikely that an exposure to average concentrations of manganese through ingestion of fish will result in a health hazard to the current trespasser/hiker/occasional visitor in Lagoon 6.

Cancer risks associated with exposure to arsenic in fish tissue are at the upperbound of the acceptable risk range. However, it should be noted that no other carcinogenic contaminants have been identified in fish tissue, so the cumulative estimated risk is therefore also at the upperbound of the acceptable risk range. In addition to arsenic, copper and thallium were also identified in fish tissue at exposure point concentrations associated with levels of noncancer hazard that exceed the benchmark of a hazard quotient of 1 by less than an order of magnitude. First, none of these metals have been identified as COCs in on-Site soils or groundwater. Second, as previously stated in Section 9.4.6, it is important to reiterate that the fish tissue concentrations were estimated from surface water RME concentrations in the Otter Kill using chemical-specific bioconcentration factors. For all three of these contaminants, due to a paucity of data that did not allow for an upperbound of the average concentration to be calculated, the RME surface water concentration therefore defaulted to the maximum detected concentration. This means that the fish tissue concentration was estimated using the maximum detected surface water concentration for arsenic, copper, and thallium. This presents a very high-end estimate of the fish tissue concentration for these three contaminants.

Taking into account the fact that none of these inorganics were identified as COCs in on-Site soils or groundwater, that the fish tissue concentrations used to estimate risks and hazard are based on a model that conservatively used the maximum detected surface water concentration and a bioconcentration factor that is likely to present a very high-end estimate of the actual fish tissue concentration, and that the cancer risks are within the upperbound of the acceptable risk range and that the noncancer hazards are within an order of magnitude of the benchmark of a hazard quotient of 1, the risks and hazards presented are likely to be biased high and actual exposure to fish tissue is unlikely to result in an unacceptable risk or hazard to human health.

Similar to the above scenario, arsenic, manganese, and thallium are consistent with background levels in soil as shown in Table 9.4, therefore, there is no Site-related source for these constituents. As shown in Table 9.3, the background concentrations for copper in surface water are higher than copper concentrations adjacent to the Site in Beaverdam Brook and Otter Kill, therefore copper in surface water may be impacted by an up-gradient non-Site-related source.

Future Parkland User/Occasional Visitor/Recreational User to Site Area

Table 10.3.RME of Appendix Q.1 presents the COPCs contributing the majority of the combined risk and hazard for the RME future parkland user/occasional visitor/recreational user exposure to sediment at the Southwest Marsh Area and surface water at Otter Kill. The cumulative RME risk estimate of 4.3E-04 primarily results from exposure to arsenic through ingestion of fish from Otter Kill. Arsenic, via ingestion of

fish, accounts for 99 percent of the cumulative risk. The cumulative RME HI of 12 primarily results from exposure to arsenic, copper, and manganese through ingestion of fish from Otter Kill and thallium through exposure to sediment and ingestion of fish. Arsenic, copper, manganese, and thallium account for approximately 24 percent, 29 percent, 6 percent, and 34 percent, respectively, for a total contribution of 95 percent of the total cumulative HI. Although manganese contributes 6 percent to the cumulative HI value of 12, the HI for central nervous system effects of 0.76 does not exceed the benchmark of 1. Therefore, it is unlikely that an exposure to average concentrations of manganese through ingestion of fish will result in a health hazard to the future parkland user/occasional visitor/recreational user to the Site area.

Cancer risks associated with exposure to arsenic in fish tissue are at the upperbound of the acceptable risk range. However, it should be noted that no other carcinogenic contaminants have been identified in fish tissue, so the cumulative estimated risk is therefore also at the upperbound of the acceptable risk range. In addition to arsenic, copper, and thallium were also identified in fish tissue at exposure point concentrations associated with levels of noncancer hazard that exceed the benchmark of a hazard quotient of 1 by less than an order of magnitude. First, none of these metals have been identified as COCs in on-Site soils or groundwater. Second, as previously stated in Section 9.4.6, it is important to reiterate that the fish tissue concentrations were estimated from surface water RME concentrations in the Otter Kill using chemical-specific bioconcentration factors. For all three of these contaminants, due to a paucity of data that did not allow for an upperbound of the average concentration to be calculated, the RME surface water concentration therefore defaulted to the maximum detected concentration. This means that the fish tissue concentration was estimated using the maximum detected surface water concentration for arsenic, copper, and thallium. This presents a very high-end estimate of the fish tissue concentration for these three contaminants.

Taking into account the fact that none of these inorganics were identified as COCs in on-Site soils or groundwater, that the fish tissue concentrations used to estimate risks and hazard are based on a model that conservatively used the maximum detected surface water concentration and a bioconcentration factor that is likely to present a very high-end estimate of the actual fish tissue concentration, and that the cancer risks are within the upperbound of the acceptable risk range and that the noncancer hazards are within an order of magnitude of the benchmark of a hazard quotient of 1, the risks and hazards presented are likely to be biased high and actual exposure to fish tissue is unlikely to result in an unacceptable risk or hazard to human health.

Again, arsenic, manganese, and thallium are consistent with background levels in soil as shown in Table 9.2, therefore, there is no Site-related source for these constituents. As shown in Table 9.3 the background concentrations for copper in surface water are higher than copper concentrations adjacent to the Site in Beaverdam Brook and Otter Kill, therefore copper in surface water may be impacted by an up-gradient non-Site-related source.

Future Parkland User/Occasional Visitor/Recreational User in Lagoon 6

Table 10.4.RME of Appendix Q.1 presents the COPCs contributing the majority of the combined risk and hazard for the RME future parkland user/occasional visitor/recreational user exposure to Lagoon 6 surface soil, sediment at Southwest Marsh Area, and consumption of fish from Otter Kill. The cumulative RME risk estimate of 4.4E-04 primarily results from exposure to Aroclor-1254 in Lagoon 6 surface soils and arsenic through ingestion of fish from Otter Kill. Arsenic, via ingestion of fish, accounts for 96 percent of the cumulative risk. The contribution to the total RME HI of 13 for children primarily resulted from exposure to Aroclor-1254 in Lagoon 6 surface soil; arsenic, copper, and manganese from consumption of fish from Otter Kill; and thallium in Southwest Marsh Area sediment and consumption of fish. Aroclor-1254, arsenic, copper, manganese, and thallium account for 6 percent, 23 percent, 27 percent, 6 percent, and 31 percent respectively, for a total contribution of 93 percent of the cumulative HI. Although manganese contributes 6 percent to the cumulative HI value of 13, the HI for central nervous system effects of 0.76 does not exceeds the benchmark of 1. Therefore, it is unlikely that an exposure to average concentrations of manganese through ingestion of fish will result in a health hazard to the future parkland user/occasional visitor/recreational user in Lagoon 6.

Cancer risks associated with exposure to arsenic in fish tissue are at the upperbound of the acceptable risk range. However, it should be noted that no other carcinogenic contaminants have been identified in fish tissue, so the cumulative estimated risk is therefore also at the upperbound of the acceptable risk range. In addition to arsenic, copper, and thallium were also identified in fish tissue at exposure point concentrations associated with levels of noncancer hazard that exceed the benchmark of a hazard quotient of 1 by less than an order of magnitude. First, none of these metals have been identified as COCs in on-Site soils or groundwater. Second, as previously stated in Section 9.4.6, it is important to reiterate that the fish tissue concentrations were estimated from surface water RME concentrations in the Otter Kill using chemical-specific bioconcentration factors. For all three of these contaminants, due to a paucity of data that did not allow for an upperbound of the average concentration to be calculated, the RME surface water concentration therefore defaulted to the maximum detected concentration. This means that the fish tissue concentration was estimated using the maximum detected surface water concentration for arsenic, copper, and thallium. This presents a very high-end estimate of the fish tissue concentration for these three contaminants.

Taking into account the fact that none of these inorganics were identified as COCs in on-Site soils or groundwater, that the fish tissue concentrations used to estimate risks and hazard are based on a model that conservatively used the maximum detected surface water concentration and a bioconcentration factor that is likely to present a very high-end estimate of the actual fish tissue concentration, and that the cancer risks are within the upperbound of the acceptable risk range and that the noncancer hazards are within an order of magnitude of the benchmark of a hazard quotient of 1, the risks and hazards presented are likely to be biased high and actual exposure to fish tissue is unlikely to result in an unacceptable risk or hazard to human health.

Future Resident/Occasional Visitor/Recreational User to Site Area

Table 10.5.RME of Appendix Q.1 presents the COPCs contributing the majority of the combined risk and hazard for the RME future resident /occasional visitor/recreational user exposure to Site surface soil, consumption of fish from Otter Kill, and groundwater. The cumulative RME cancer risk of 1.8E-03 primarily results from exposure to arsenic through consumption of fish from Otter Kill and ingestion and dermal contact with groundwater as a potable water source and benzene through ingestion, dermal contact, and inhalation of groundwater as a potable water source. Arsenic from fish and benzene from groundwater account for 39 percent and 57 percent, respectively, for a total of 96 percent of the cumulative cancer risk. Again, arsenic is consistent with background levels in soils as shown in Table 9.2.

Cancer risks associated with exposure to arsenic in fish tissue are at the upperbound of the acceptable risk range. However, it should be noted that no other carcinogenic contaminants have been identified in fish tissue, so the cumulative estimated risk is therefore also at the upperbound of the acceptable risk range. First, arsenic has not been identified as a COC in on-Site soils or groundwater. Second, as previously stated in Section 9.4.6, it is important to reiterate that the fish tissue concentrations were estimated from surface water RME concentrations in the Otter Kill using chemical-specific bioconcentration factors. For arsenic, due to a paucity of data that did not allow for an upperbound of the average concentration to be calculated, the RME surface water concentration therefore defaulted to the maximum detected concentration. This means that the fish tissue concentration was estimated using the maximum detected surface water concentration for arsenic. This presents a very high-end estimate of the fish tissue concentration for this contaminant.

Taking into account the fact that arsenic was not identified as a COC in on-Site soils or groundwater, that the fish tissue concentrations used to estimate risk is based on a model that conservatively used the maximum detected surface water concentration and a bioconcentration factor that is likely to present a very high-end estimate of the actual fish tissue concentration, and that the cancer risks are within the upperbound of the acceptable risk range, the risk presented is likely to be biased high and actual exposure to fish tissue is unlikely to result in an unacceptable risk to human health.

The cumulative RME HI of 7.0E+02 for children primarily results from benzene, 2-aminopyridine, aniline, pyridine-3-ethyl, and pyridine 5-ethyl-2 methyl exposure via ingestion, dermal contact, and inhalation of groundwater as a potable water source. Benzene, 2-aminopyridine, aniline, pyridine-3-ethyl, and pyridine 5-ethyl-2 methyl account for approximately 3 percent, 82 percent, 3 percent, 1.5 percent, and 1 percent, respectively, for a total contribution of 90.5 percent of the cumulative HI.

Ingestion and dermal contact with antimony in groundwater results in a hazard quotient greater than 1.0. Antimony was not detected in any of the groundwater samples collected in July 2001 with the exception of a detection of 14.4 µg/L for well MW-9D-01. This single detection is driving an unreasonable hazard in the HHRA for groundwater. A confirmatory sample for TAL inorganic analysis with the Semi-annual program in November 2001 resulted in a non-detect value of 4.6 µg/L for antimony at MW-9D-01. It would appear that the July 2001 hit was likely anomalous. Further evidence to support the interpretation that the detection of 14.4 µg/L for antimony in the groundwater is anomalous includes the infrequent and low level detections of antimony in other Site media. For example, antimony was only detected in 1 of 16 surface soil samples at a concentration of 12.5 mg/kg, and was not detected in any surface soil samples at Lagoon 6. In the subsurface soil samples, antimony was detected at a frequency of 24 out of 163 samples, for a frequency of slightly less than 15 percent, with a maximum detected concentration of 67.5 mg/kg that is less that the industrial risk-based screening value of 410 mg/kg; the subsurface soil samples at Lagoon 6 indicated antimony was present in only 1 of 13 samples at a concentration of 0.53 mg/kg. Evaluating all of this information on the distribution of antimony at the Site leads to the conclusion that the one detection of antimony in the groundwater of 14.4 µg/L is not likely to be consistent with typical site-wide conditions. In addition, the other parameters, benzene, 2-aminopyridine, aniline, pyridine-3-ethyl, and pyridine 5-ethyl-2 methyl were all detected in the overburden groundwater which will not be used as a future on-Site potable groundwater source. A deed notice will restrict the use of the overburden groundwater aquifer and the appropriate institutional controls presented in the Feasibility Study.

Future Resident/Occasional Visitor/Recreational User in Lagoon 6

Table 10.6.RME of Appendix Q.1 presents the COPCs contributing the majority of the combined risk and hazard for the RME future resident /occasional visitor/recreational user exposure to Lagoon 6 surface soil, consumption of fish from Otter Kill, and groundwater. The cumulative RME cancer risk of 1.9E-03 primarily results from exposure to Arolor 1254 through ingestion and dermal contact with Lagoon 6 surface soil, arsenic through ingestion and dermal contact with Lagoon 6 surface soils, consumption of fish from Otter Kill, and ingestion and dermal contact with groundwater as a potable water source and benzene through ingestion, dermal contact, and inhalation of groundwater as a potable water source. Aroclor-1254 from Lagoon 6 surface soil; arsenic from Lagoon 6 surface soil, fish, and groundwater; and benzene from groundwater account for 2 percent, 42 percent, and 53 percent, respectively for a total of 97 percent of the cumulative cancer risk. Arsenic is consistent with background levels in soils as shown in Table 9.4.

The cumulative RME HI of 7.1E+02 for children primarily results from benzene, 2-aminopyridine, aniline, pyridine-3-ethyl and pyridine 5-ethyl-2 methyl exposure via ingestion, dermal contact, and inhalation of groundwater as a potable water source. Benzene, 2-aminopyridine, aniline, pyridine-3-ethyl, and pyridine 5-ethyl-2 methyl account for approximately 3 percent, 81 percent, 3 percent, 1.5 percent, and 1 percent, respectively, for a total contribution of 89.5 percent of the cumulative HI.

Similar to the explanation provided under the heading Future Resident/Occasional Visitor/Recreational User to Site Area the same point of view pertaining to antimony also applies here.

Future Construction Worker Exposure to Site Area

Table 10.7.RME of Appendix Q.1 presents the COPCs contributing the majority of the combined risk and hazard for the RME future construction worker exposure to Site soil and groundwater. The cumulative RME cancer risk of 1.5E-04 primarily results from exposure to benzene through ingestion, dermal contact, and inhalation of Site soils. Benzene accounts for 93 percent of the cumulative cancer risk. The cumulative RME HI of 1.2E+02 primarily results from benzene, toluene, chlorobenzene, xylenes, 2-aminopyridine, and manganese via ingestion, dermal contact, and inhalation of Site soils. Benzene, toluene, chlorobenzene, xylenes, 2-aminopyridine, and manganese

account for approximately 35 percent, 6 percent, 4 percent, 50 percent, 1 percent, and 2 percent, respectively, for a total contribution of 98 percent of the cumulative HI.

Manganese is consistent with background levels in soil as shown in Table 9.5, therefore, there is no Site-related source for this constituent.

Future Construction Worker Exposure to Lagoon 6

Table 10.8.RME of Appendix Q.1 presents the COPCs contributing the majority of the combined risk and hazard for the RME future construction worker exposure to Lagoon 6 soil and groundwater. The cumulative RME HI of 5.5 primarily results from toluene, Aroclor-1254, aluminum, and manganese exposure via ingestion, dermal contact, and inhalation of Lagoon 6 soils and benzene exposure via incidental ingestion, dermal contact, and inhalation of groundwater. Toluene, Aroclor-1254, aluminum, manganese, and benzene account for approximately 24 percent, 11 percent, 12 percent, 30 percent, and 8 percent, respectively, for a total contribution of 85 percent of the cumulative HI.

Although the HI value of 2.3 for central nervous system effects slightly exceeds the benchmark of 1, the estimated HI value of 2.3 is a sum of the HQs of 0.65 for aluminum and 1.64 for manganese. The HQ for both of these inorganics is based on a limited sample set of 5, so the exposure point concentration defaulted conservatively to the maximum detected concentration. Therefore, it is unlikely that an exposure to average concentrations of these inorganics in the Lagoon 6 area will be associated with HQ values that would exceed the benchmark of 1.

Additionally, aluminum and manganese are consistent with background levels in soils shown in Table 9.6.

Current Off-Site Resident Exposure

Table 10.9.RME of Appendix Q.1 presents the COPCs contributing the majority of the combined risk and hazard for the RME current off-Site resident exposure to groundwater. The cumulative RME HI of 4.6 for children primarily results from 2-aminopyridine exposure via ingestion and dermal contact with groundwater as a potable water source. 2-Aminopyridine accounts for approximately 98 percent of the cumulative HI.

2-Aminopyridine in off-site groundwater results in a hazard index greater than 1.0. It was only detected once at 217 Maybrook Rd in September 2003 and the subsequent sample in 2004 was non-detect. All other off-site groundwater sample locations were non-detect for this parameter. This single detection is driving an unreasonable hazard in the HHRA for off-site groundwater.

In addition, the village of Maybrook has not identified any site-related parameters in their quarterly monitoring program to date (end of 2004) in the municipal supply well immediately northeast of the Site.

9.10 HAZARD ASSESSMENT FOR LEAD EXPOSURE

9.10.1 THE IEUBK MODEL LEAD RESULTS

The text results from the IEUBK model run are presented in Appendix Q.5 along with the probability plots for blood lead concentrations. Using model inputs as presented in Table 9.1 and described above in Section 9.5.9, the IEUBK Model predicts that groundwater concentrations will result in approximately 1.13 percent of the children having a blood lead level greater than 10 µg/L. As a result, the children are not considered impacted by the groundwater.

These results indicate that there is no possibility of anticipated health impacts for children based on potential exposure to lead originating from site groundwater. However, as illustrated above, groundwater is the main source of lead exposure to children on this Site.

The lead risk characterization results using the U.S. EPA's IEUBK model indicated that predicted blood lead levels in children will not be above the blood lead level of 10 µg/dL, the U.S. EPA level of concern.

Lead in soils is consistent with background levels in soil, therefore, there is no Site-related source for this constituent. It is only with the outliers removed that lead may be considered statistically higher than background for Lagoon 6 only. However, all lead concentrations in Lagoon 6 are within the 'range' reported for the background samples indicating no significant impact even to Lagoon 6 soils.

Lead was selected as a COPC based on the estimated detection of lead of 109J µg/L at MW-4, an overburden well. The next highest lead groundwater concentration is 10 µg/L at MW-7 and the arithmetic average of the data of 4.36 µg/L, which are both below the action level of 15 µg/L.

9.11 IDENTIFICATION OF UNCERTAINTIES

The purpose of this section is to provide a summary and discussion regarding the uncertainties associated with the HHRA evaluation. The various uncertainties are discussed below in the following sections:

9.11.1 Sampling Procedure Bias;
9.11.2 Use of Maximum Values for EPCs;
9.11.3 Exposure Scenario Assumptions;
9.11.4 Constituents without Toxicity Values;
9.11.5 Evaluation of Tentatively Identified Compounds (TICs);
9.11.6 Dose Response;
9.11.7 The Theoretical Nature of Risk Estimates;
9.11.8 Synergistic Effects;
9.11.9 Security Features;
9.11.10 Northeast Marsh Area;
9.11.11 Southwest Marsh Area;
9.11.12 Otter Kill; and
9.11.13 Modeled Data.

9.11.1 SAMPLING PROCEDURE BIAS

The sampling strategy is a factor that impacts the health evaluation for chemicals at the Site. In the sampling procedures, samples were taken from locations (e.g., test pits) which were visibly contaminated or where contamination would be expected and not from areas of the Site that were apparently uncontaminated. This creates a sampling bias towards worst case (higher) concentrations in the soil samples. The utilization of these samples in the HHRA increases the conservative or health-protective nature of the health risk and hazard assessment.

9.11.2 USE OF MAXIMUM VALUES FOR EPCs

In certain cases, such as data sets with few detected concentrations/high proportions of non-detected values (e.g., one detection in 10 to 15 samples) or the presence of anomalous data (such as the antimony detection in groundwater), the calculation of meaningful 95 percent UCL is prevented by these highly skewed or undetermined data distributions. For these limited or anomalous data sets, the maximum concentration was selected as the RME EPC. In some cases, the use of the maximum detected concentration has resulted in unacceptable hazards and cancer risks.

9.11.3 EXPOSURE SCENARIO ASSUMPTIONS

The purpose of this section is to evaluate the uncertainty associated with the primary exposure scenario assumptions such as land use and frequency of exposure.

Because the assumptions used in the scenarios are often not based on objective test data but are subjective estimates based on judgment and experience applied to the data available, the tendency is to select conservative, health-protective values to guard against under-estimating exposure (and associated risks). This leads to a general over-estimating in all assumptions. When over-estimation of individual assumptions are included in the scenario equations, they are multiplied. This exaggerates the over-estimation of each assumption and over states the total exposure to an even greater degree. The exposure scenarios are, therefore, conservative in nature, thereby providing the necessary factor of safety that is protective of human health.

The intent of this baseline risk assessment was to estimate the potential exposure point intakes for both the "average" (CT) and the "RME" exposure scenarios. In order to accomplish this goal, a series of standardized U.S. EPA exposure assumptions were utilized, where available. In the absence of available U.S. EPA exposure assumptions, professional judgment and U.S. EPA (2000; 2004a) recommended values were used to establish necessary assumptions that are protective of human health.

The CT exposure scenario represents the "average" exposure scenario that may reasonably be expected to occur. The RME exposure scenario represents the reasonable maximum exposure expected to occur. The RME exposure scenario presented in this HHRA was developed in accordance with the U.S. EPA guidance. The exposure scenarios (CT and RME) were developed to represent reasonable exposures, which may occur under both current and future land use conditions. For the CT exposure scenarios, the CT EPC was the same as the RME EPC rather than the average or mean value for all exposure media. This will result in an overestimation of the risks and hazards for the CT exposure scenarios.

The major uncertainties incorporated in the HHRA regarding the physical exposure scenarios are summarized as follows:

- TIC concentrations were assumed to be uniform over the entire exposure area evaluated although most individual TICs were reported in only one sample and one medium. The same can be said for other positively identified chemicals that were detected primarily from one specific area on Site.
- Because of the limited data set, the maximum reported values have been used as exposure point concentrations to estimate exposures to the Northeast Marsh Area sediments and Beaverdam Brook/Otter Kill surface waters. Although the use of maximum values is generally recognized as an appropriate screening approach, it should be recognized that this procedure will likely overestimate the actual exposure by orders of magnitude;
- The actual exposure frequency (EF) for many of the exposure scenarios evaluated is unknown. As a result, professional judgment and U.S. EPA (2000; 2004a) recommendations were used to conservatively estimate exposure frequency and duration, which may result in overestimates of actual risks;
- As presented in Section 9.7.4, the summation of risks and hazards estimated for the individual exposure scenarios for various media and areas of concern without regard for what this combined exposure represents is misleading and results in a very conservative estimate of the combined risk for the future parkland user and the on-Site resident. The straight addition of the individual exposure scenarios assumes that the same individual is exposed to all the various media at the Site and in the Site vicinity over all the exposure periods specified in the individual exposure scenarios. Because the exposure frequency for many of the RME scenarios is high, it can be assumed that the separate individual exposures to the several exposure areas at the Site would have to take place on the same day. Thus, on these frequent multiple exposure days, a parkland user is assumed to spend as much as 6 or 8 hours on Site. In addition, most of the evaluated exposure areas are at a fair distance from one another, and it would not be unreasonable to assume that a parkland user would spend another two hours or more at the Site getting from one place to another in order to receive the exposure represented by the straight summation of risks and hazards. It may be more appropriate to assume that a park user spends at most 2 hours per exposure day at the park with 85 percent of their time on land exposed to Site surface soils, while 5 percent of their time is spent in each of the Southwest Marsh, Beaverdam Brook, and Otter Kill. Therefore, on those days that a parkland user visits several or all areas of the Site, perhaps as many as 78 days/year for an adult, it is reasonable and realistic to assume a lower combined exposure than is represented by the straight summation of all potential exposure scenarios. This more realistic combined exposure for an individual will result in lower risks and hazards;

- The utilization of present exposure point concentrations for future exposure scenarios is conservative due to the fact that source material is not being added and the majority of the Site related constituents will naturally degrade with time under the appropriate conditions. Natural degradation processes include dilution by uncontaminated water, adsorption, volatilization, biodegradation, chemical degradation, and photodegradation. However, degradation/attenuation of the various constituents is dependent on environmental conditions, for example trichloroethylene and benzene typically degrade under very disparate environmental conditions. The use of steady state contaminant concentrations also overestimates the exposure;
- The overburden and bedrock groundwater analytical data was combined into one groundwater data set that was used to estimate the risks and hazards for the on-Site groundwater exposure pathway. Both the overburden and bedrock aquifers are designated by the State as potable supply aquifers. However, a deed notice will restrict the use of the overburden groundwater aquifer and the appropriate institutional controls presented in the Feasibility Study.
- The HHRA assumed 100 percent absorption of chemicals that have been ingested. Actual absorption rates from ingested contaminants may vary from 5 to 100 percent. Therefore, assuming 100 percent absorption of ingested contaminants may overestimate the associated risks; and
- The HHRA estimates a fish consumption rate of 31.9 grams/meal over 365 meals/year for an adult. If it is assumed that 100 or fewer individuals would fish recreationally from Otter Kill, the total fish load would have to be over 1.25 tons per year to satisfy the estimated fish consumption rate for adults. It is very unlikely that the fish population in Otter Kill would support a fish harvest of this magnitude to satisfy the fish consumption rates assumed for this scenario.

9.11.4 EVALUATION OF TENTATIVELY IDENTIFIED COMPOUNDS (TICS)

To maintain a conservative approach in the HHRA, detected TICs in all media were evaluated in the quantitative risk assessment. In cases where toxicity data was available for a given TIC, the data was used to estimate the carcinogenic risk and/or hazard. In cases where toxicity data was not available for a given TIC, a surrogate chemical with published toxicity factors was selected based on structural similarity. Known toxicity data for a surrogate chemical was substituted as the toxicity data for a specific TIC. Due to the similarity in structures, it was assumed that biological activity and toxicity characteristics would also be similar. This fundamental approach is assumed to maintain a conservative approach since the TICs were frequently methyl-substituted isomers (or similar substitutions) of the parent or surrogate chemical and the substitution would be expected to decrease the toxicity.

Unknown TICs were not included in the quantitative risk assessment due to a lack of information for these parameters. As a result, overall risk may be underestimated for these parameters.

Since the toxicity of most of the reported TICs is unknown, their inclusion incorporates uncertainties in the risk assessment of unknown magnitude. Therefore, inclusion of these TICs in the quantitative risk assessment could potentially over- or under-estimate the results of the risk assessment.

The maximum of reported concentrations were used as exposure point concentrations for the CT and the RME, respectively. This is a very conservative approach. The conservative nature of this methodology is exemplified, again, by using the TIC pyridine, 3-ethyl-, reported at a maximum concentration of 180 μg/L in groundwater samples, as an example. The fundamental assumptions in the quantitative evaluation of this TIC in groundwater are as follows:

- Under a future parkland or residential use of the Site, it was assumed that pyridine, 3-ethyl- is uniformly distributed in the groundwater at 180 μg/L;
- Future residents and construction workers spend all their time on Site in direct contact with groundwater having pyridine, 3-ethyl- contamination at the 180 μg/L level; and
- The toxicity of pyridine, 3-ethyl- is assumed to be equivalent to terbutyrn toxicity.

Based on the conservative assumptions described above, the pyridine, 3-ethyl- was estimated to contribute 14.6 percent of the total RME HI of a future residential groundwater exposure.

The overall Site risks, and any associated remedial action decisions, must take into account the magnitude of risk and the uncertainty of the risk that was added to each Lagoon 6 scenario by the inclusion of TICs in the quantitative HHRA.

9.11.5 CONSTITUENTS WITHOUT TOXICITY VALUES

Some constituents required special consideration in the HHRA because there are no published toxicity values. Chemicals without published toxicity values include phenanthrene, alpha-picoline, and 2-aminopyridine.

RfDs for pyrene were utilized for phenanthrene and other non-carcinogenic PAHs without an RfD value. This is believed to be a reasonably conservative assumption. The RfD for 4-aminopyridine was utilized for 2-aminopyridine. This is believed to be a reasonably conservative assumption. The RfD for pyridine was utilized for alpha-picoline. This is believed to be appropriate due to the similarity in structure between these two compounds.

9.11.6 DOSE RESPONSE

One of the major uncertainties in the quantification of risk involves the application of toxicity information. Examples of the uncertainties associated with the toxicity values are presented as follows:

- chemicals may be assumed to be human carcinogens based on animal studies even when there is limited or no available evidence that the chemical is a human carcinogen;
- CSFs are derived from high concentration animal studies and therefore may not be applicable to low concentration exposures;
- CSFs are developed in a conservative manner; and
- RfDs are also established with conservative factors of safety in comparison to actual studies.

9.11.7 THE THEORETICAL NATURE OF RISK ESTIMATES

The results of a health risk assessment assign a numerical value to the probability of a case of cancer developing in a population exposed to a specific amount of chemical which is a known or suspect carcinogen. This numerical value is presented as an upper limit excess cancer risk such as 1×10^{-6}, or one additional cancer case in a million people exposed to the designated chemical and at the designated chemical concentration for their entire lifetime, which is assumed to be 70 years. The model that is applied to calculate this numerical risk value is intentionally biased to give a high value so the true value would not likely be greater and, in fact, may be zero. The Cancer Risk Model and the assumptions used to estimate exposure are protective of the most sensitive populations.

9.11.8 SYNERGISTIC EFFECTS

There is always a concern for synergistic (e.g., combined) effects when a receptor is exposed to several chemicals at the same time. In other words, there is concern that chemicals interact in the receptor to cause an effect significantly greater than the sum of effects of the individual chemicals. For example, volatile organics, semi-volatile organics (e.g., PAHs), and inorganic chemicals are treated as if they all affect the receptor in a similar manner. Their potential carcinogenic risk and hazard ratios are added.

There is no apparent biological reason to suspect that all volatile organic, semi-volatile organic, and inorganic compounds will have synergistic effects. Although the carcinogenic risk probabilities are added, there is no basis to suspect the toxic effects are additive. This suggests that the calculated HI may be higher than the toxicological facts would suggest.

9.11.9 SECURITY FEATURES

The HHRA presented in this report assumes that existing security features such as the Site security fence do not exist. As a result, the exposure assumptions identified for the current exposure scenarios in the HHRA may overestimate risks.

9.11.10 NORTHEAST MARSH AREA

Although chemicals have been identified in the sediments of the Northeast Marsh Area, it cannot be clearly demonstrated that their presence in this area is Site related. In fact, the U.S. EPA and the NYSDEC have agreed that the evaluation of PAHs in sediment from this area is not required as part of the HHRA because the source of these contaminants is likely not related to the Site. However, inclusion in the HHRA of other chemical constituents detected in the Northeast Marsh Area sediment and surface water may also be unwarranted.

9.11.11 SOUTHWEST MARSH AREA

The majority of the estimated lifetime cancer risks for exposure to the Southwest Marsh Area sediments are attributable to benzo(a)pyrene and dibenzo(a,h)anthracene. However, these two PAHs were not detected in the 62 investigative soil samples collected from the Site including the area of the lagoons with the exception of one detection of benzo(a)pyrene at an estimated concentration of 57 J $\mu g/kg$. The presence of PAHs in the sediments of the Southwest Marsh Area is more likely attributable to the former railroad operations and not the Site as indicated by elevated concentrations of PAHs including benzo(a)pyrene and dibenzo(a,h)anthracene at sediment sample location 5 collected from the drainage ditch along the abandoned railroad grade in 1995.

9.11.12 OTTER KILL

The majority of the estimated lifetime cancer risks for exposure to ingestion of fish from Otter Kill are attributable to arsenic. The hazard index for exposure to ingestion of fish from Otter Kill is atributable to arsenic, copper, manganese, and thallium. The fish tissue concentrations were estimated based on the use of literature bioconcentrations factors that may or may not be appropriate for this Site. Arsenic, copper, manganese, and thallium are not Site-related compounds, as the soil concentrations of these metals are consistent with background concentrations. In addition, the risk and hazard estimates are based on a fish consumption rate that is greater than a typical recreational fishing scenario. Also it is very unlikely that the fish population in Otter Kill would support the fish harvest/consumption rates assumed in these exposure scenarios.

9.11.13 MODELED DATA

Cancer risks associated with exposure to arsenic in fish tissue are at the upperbound of the acceptable risk range. However, it should be noted that no other carcinogenic contaminants have been identified in fish tissue, so the cumulative estimated risk is therefore also at the upperbound of the acceptable risk range. In addition to arsenic, copper and thallium were also identified in fish tissue at exposure point concentrations associated with levels of noncancer hazard that exceed the benchmark of a hazard quotient of 1 by less than an order of magnitude. As previously stated in Section 9.4.6, the fish tissue concentrations were estimated from surface water RME concentrations in the Otter Kill using chemical-specific bioconcentration factors. For all three of these contaminants and the other detected contaminants, due to a paucity of data that did not allow for an upperbound of the average concentration to be calculated, the RME surface water concentration therefore defaulted to the maximum detected concentration. This means that the fish tissue concentration was estimated using the maximum detected surface water concentration for all detected constituents including arsenic, copper, and thallium. This presents a very high-end estimate of the fish tissue concentration for all contaminants.

Taking into account the fact that the fish tissue concentrations used to estimate risks and hazard are based on a model that conservatively used the maximum detected surface water concentration and a bioconcentration factor that is likely to present a very high-end estimate of the actual fish tissue concentration, the risks and hazards presented are likely to be biased high and actual exposure to fish tissue is unlikely to result in an unacceptable risk or hazard to human health.

10.0 SUMMARY AND CONCLUSIONS OF RI

10.1 SUMMARY OF RI

10.1.1 RI OBJECTIVES AND FIELD INVESTIGATION

A remedial investigation including a human health risk assessment and ecological risk assessment has been completed to determine the environmental conditions at the Site including the nature and extent of contaminants in the groundwater, surface water, subsurface soils, surface soils, sediment, and former lagoons.

The characterization of the nature and extent of contamination at the Site was achieved by conducting a variety of investigative activities. These investigations included the following:

- Phase I RI Field Investigations – 1991;
- Phase II RI Field Investigations – 1995;
- Treatability Study Investigations – 1996;
- Interim Groundwater Monitoring Program – 1997 to 2004;
- Additional Round of Groundwater Sampling – 2001;
- Natural Attenuation Study Groundwater Sampling– 2001/2002;
- Cultural Resource Survey – 2002;
- Additional Soil Investigation – 2003; and
- Wetland Delineation – 2003.

10.1.2 CHEMICAL DISTRIBUTION

The following Site-related parameters, detected in 5 percent or more of the samples, exceeded the applicable criteria for the following media:

Parameter	*Soil*	*Groundwater*	*Sediment*	*Surface Water*
VOCs				
Acetone	√	√	√	—
Benzene	√	√	—	—
Chlorobenzene	√	√	—	—
1,2-Dichloroethane	—	√	—	—
Ethylbenzene	√	√	—	—

Parameter	*Soil*	*Groundwater*	*Sediment*	*Surface Water*
Toluene	√	√	√	—
Xylene	√	√	—	—
SVOCs				
2-Aminopyridine	√	√	—	—
Alpha-picoline	√	—	—	—
Pyridine	√	—	—	—
Bis(2-Ethylhexyl) phthalate	√	√	√	—

Additional chemical parameters were reported at concentrations exceeding applicable criteria but were detected in less than 5 percent of the samples for a given media and/or were present at similar concentrations in background samples.

Inorganics detected at concentrations exceeding applicable criteria in soil, groundwater, surface water, and sediment were reported at similar concentrations in background samples and/or were determined not to be Site-related (see following discussion on determination of inorganic COCs).

Pesticides were determined not to be Site-related.

Aroclor-1254 was reported at concentrations of 14,000 mg/kg and 15,000 mg/kg for the duplicate for test pit sample RM-24 collected from a depth of approximately 4 feet from Lagoon 2 and at concentrations of 4,800 J mg/kg and 8,900 J mg/kg for the duplicate for surface soil sample SSII-6 collected from Lagoon 6. Of the 65 test pit and borehole samples collected from the Site, these are the only occurrences of an exceedance of the NYSDEC Soil Cleanup Objective of 10 mg/kg for PCBs for subsurface and 1 mg/kg for surface soils at the Site. Based on the results of the remaining soil samples at the Site, elevated concentrations of PCBs are not prevalent. These slight exceedances of the NYSDEC Soil Cleanup Objectives are considered to be an isolated occurrence and not indicative of a widespread PCB issue at the Site.

10.1.3 ECOLOGICAL RISK ASSESSMENT

The results of this ERA indicate that risks or impacts to ecological receptors exposed to Site-related chemicals are negligible. The majority of identified risk to ecological receptors is associated with upper bound concentrations of PCB and mercury in Lagoon 6 surficial soils which comprise 1 percent of the total Site area. The remainder of the Site poses no risk to ecological receptors from exposure to PCBs, mercury, or any other parameter detected in surface soils.

10.1.4 DETERMINATION OF INORGANIC COCs

To determine if an inorganic COPC should be considered as a COC (either Site-related or non-Site-related), the HHRA was conducted to evaluate the risks, or potential risks, posed by the Site.

If an individual inorganic COPC contributed to a carcinogenic risk greater than 10^{-6} or a non-carcinogenic HI greater than 1.0, it will be retained for further consideration as a COC (either Site-related or non-Site-related) in accordance with the evaluation process described in the Additional Soil Sampling Work Plan (Figure 4.1 COPC, COC, AND PRG SELECTION PROCESS). Any inorganic parameters that do not represent an unacceptable human health risk will not be retained as COCs and will not be evaluated further in the FS process.

To summarize the HHRA with respect to the final retention of COCs, the following inorganic parameters (Site and non-Site-related) represented a carcinogenic risk greater than 10^{-6} or a non-carcinogenic HI greater than 1.0 in the HHRA and were further evaluated:

- arsenic, copper, and thallium in Otter Kill Creek surface water to fish for current/future resident/occasional visitor/trespasser;
- manganese in soil for future construction worker;
- arsenic in surface soil in Lagoon 6 for the future resident and maintenance worker;
- iron in surface soil for future resident; and
- manganese, antimony, arsenic, and lead in groundwater for future on-Site resident.

To determine if the above inorganic COCs are Site-related and thus remain as COCs, background screening was conducted for the inorganics using statistical methods to compare the lagoon area inorganic database to the background inorganic database as described in Appendix C of the Additional Soil Sampling Work Plan (CRA, 2003).

Arsenic, iron, lead, manganese, and thallium have been shown to be consistent with background levels in soil, therefore, it is concluded that there is no Site-related source for these constituents.

The background concentrations for copper in surface water are higher than copper concentrations adjacent to the Site in Beaverdam Brook and Otter Kill. Therefore, we have concluded that copper in surface water is being impacted by an upgradient non-Site-related source.

Antimony was not detected in any of the groundwater samples collected in July 2001 with the exception of a detection of 14.4 μg/L for well MW-9D-01. This single detection was driving the risk in the HHRA for groundwater. A confirmatory sample was subsequently collected for TAL inorganic analysis with the Semi-annual program in November 2001. This result was non-detect for antimony at 4.6 μg/L. The July 2001 hit is therefore considered anomalous.

In summary, based on the HHRA and subsequent background level screening, there are no Site-related inorganics that represent an unacceptable human health risk level in any media at the Site under any exposure scenario.

10.1.5 DETERMINATION OF ORGANIC COCs

The following organic compounds, detected in 5 percent or more of the samples, represented a carcinogenic risk greater than 10^{-6} or a non-carcinogenic HI greater than 1.0 in one or more exposure scenarios in the HHRA:

- 2-aminopyridine;
- aniline;
- aroclor-1254;
- benzene;
- bis(2-ethylhexyl)phthalate;
- chlorobenzene;
- toluene; and
- xylene.

Bis(2-ethylhexyl)phthalate, detected at estimated concentrations in three groundwater samples, resulted in a carcinogenic risk of 1.33 x 10^{-5} for the future residential use of groundwater within the range of 10^{-4} to 10^{-6}.

Although the frequency of detection of aniline in soil and groundwater samples is unknown as it has not been specifically analyzed except for samples collected in 1985, aniline will be considered a COC for the Site as it represents a non-carcinogenic HI significantly greater than 1.0 in the HHRA. Two groundwater samples were analyzed for aniline during the 1985 C. A. Rich sampling event. Aniline was detected at concentrations of 48 µg/L at well MW-5 and 140 µg/L at well T-1, exceeding its groundwater standard of 5 µg/L. Aniline was not specifically analyzed in the subsequent groundwater sampling events conducted at the Site as it was not on the parameter list. During the 2001 comprehensive round of groundwater sampling, aniline was detected as a TIC at concentrations of 12NJ at well MW-2 and 16NJ/9.2NJ at well MW-7. These concentrations were qualified with an NJ because they were estimated as the spiked sample recovery was not within control limits. These concentrations exceed the groundwater standard of 5 µg/L for aniline.

The primary COCs for the Site based on exceedances of applicable criteria presented in Section 10.1.2 and the results of the ERA and the HHRA are as follows:

Parameter	*Soil*	*Groundwater*
VOCs		
Acetone	√	√
Aniline	√	√
Benzene	√	√
Chlorobenzene	√	√
1,2-Dichloroethane	—	√
Ethylbenzene	√	√
Toluene	√	√
Xylene	√	√
SVOCs		
2-Aminopyridine	√	√
Alpha-picoline	√	√
Pyridine	√	√

10.2 DEVELOPMENT OF ORGANIC PRGS FOR SOILS AND GROUNDWATER

There are currently no federal or state promulgated standards for contaminant levels in soils. Therefore, To-Be-Considered Soil Cleanup Objectives for the organic COCs at the Site are based on the State of New York soil cleanup criteria published in Technical and

Administrative Guidance Memorandum (TAGM) #4046 – Recommended Soil Cleanup Objectives and Cleanup Levels.

In addition to the known COCs for the Site, a Soil Cleanup Objective 'Guidance' value of 400 mg/kg has been developed for the individual pyridine-based TICs that have been detected at the Site as the non-carcinogenic HI for some of these compounds has been determined to be significantly greater than 1.0 in one or more exposure scenarios in the HHRA. As presented in the May 1, 2006 correspondence from the NYSDEC, for the pyridine-related TICs, the 400 µg/kg value is a cleanup guidance value. Every effort will be made to achieve the guidance value. But as a guidance value, some leeway is permitted. If the Soil Cleanup Objectives are met for the known contaminants, but the pyridine-based TICs are above the guidance value (at a reasonably low concentration), Site cleanup will have been achieved.

The following Soil Cleanup Objectives will apply to the COCs for the Site:

Parameter	***Soil Cleanup Objective (µg/kg)***
VOCs	
Acetone	200
Aniline	1,510 [1]
Benzene	60
Chlorobenzene	1,700
Ethylbenzene	5,500
Toluene	1,500
Xylene	1,200
SVOCs	
2-Aminopyridine	400 [2]
Alpha-picoline	575 [2]
Pyridine	400 [2]

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[1] The soil cleanup objective for aniline has been revised from the value presented in TAGM 4046 based on correspondence from the NYSDEC dated April 7, 2006.

[2] The soil cleanup objectives for the three pyridine compounds were determined by the NYSDEC in correspondence dated August 14, 1996 (see Appendix S).

Parameter	*Soil Cleanup Objective (μg/kg)*
TICs	
Individual Pyridine-Based TICs	400 [3]

Groundwater cleanup levels for organic COCs will be based on the more conservative of the Federal MCLs and the New York Ambient Groundwater standards and guidance values (NYSDEC TOGs 1.1.1, June 1998). The following groundwater cleanup levels are the most stringent of the two sources:

Parameter	*NY State Ambient Groundwater Standards and Guidance (G) Values (μg/L)*
VOCs	
Acetone	50 (G)
Aniline	5
Benzene	1
Chlorobenzene	5
1,2-Dichloroethane	0.6
Ethylbenzene	5
Toluene	5
Xylene	5
SVOCs	
2-Aminopyridine	1 (G)
Alpha-picoline	50 [4] (G)
Pyridine	50 (G)

Final remedial goals for the Site will be based on the remedy selected and the future land use of the Site.

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[3] The soil cleanup objective for the individual pyridine-based TICs was determined by the NYSDEC in correspondence dated May 1, 2006. The soil cleanup objective is a guidance value. If the soil cleanup objectives are met for the known contaminants, but the TICs are above the guidance value (at a reasonably low concentration), the site cleanup objective will have been met.

[4] Alpha-picoline does not have a standard or guidance value. Due to lack of information for this analyte, pyridine substituted as per HHRA.

Following the approval of the Site-related COCs and their PRGs by the U.S. EPA and NYSDEC, the PRGs will then be used to evaluate each remedial alternative during the FS. The retained organic COCs and their respective PRGs will be used in the development of the Preliminary Remedial Action Plan (PRAP) and the Record of Decision (ROD).

10.3 RECOMMENDED REMEDIAL ACTION OBJECTIVES

Remedial action objectives are specific goals to protect human health and the environment. These objectives are based on available information and standards such as applicable or relevant and appropriate requirements (ARARs) and risk-based levels established in the Risk Assessment.

The following objectives have been established for the Maybrook Lagoon Site:

- Treat organic compounds in soil within the Lagoon area of the Site to levels which would allow for residential or recreational use and be protective of impacts to groundwater;
- Eliminate, to the greatest extent practicable, continuing sources of organic soil contamination to the groundwater;
- Remove as much organic contamination as possible within the source area from the shallow aquifer during the treatment of soil;
- Prevent human exposure to contaminated groundwater on Site and off Site;
- Restore groundwater quality to applicable NY State groundwater criteria where technically practicable. For the existing Site groundwater system where a portion of contaminated groundwater is unlikely to be to immediately restored to State groundwater criteria, assure that groundwater monitoring and remedial contingencies be developed and are protective to prevent human exposure to contaminated groundwater off Site; and
- Prevent human exposure to surface soils contaminated with PCB concentrations above 1 part per million.

Appropriate remedial alternatives will be developed for soil and groundwater in the FS to address the above remedial action objectives for the Site. A preferred alternative will be determined based on the evaluation of the various alternatives.

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