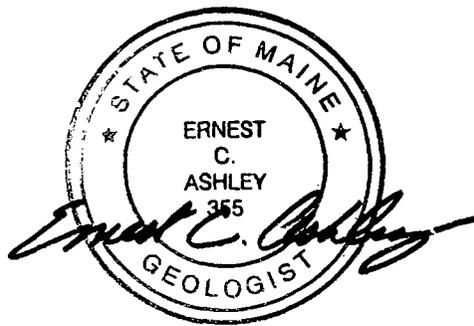


SITE INVESTIGATION REPORT
HOLTRACHEM MANUFACTURING SITE
ORRINGTON, MAINE

VOLUME I

Text

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Section 1 Introduction

1.1 Document Intent

This Site Investigation Report documents the methodology and results of the Site Investigation (SI) of the HoltraChem Manufacturing Site (formerly, LCP Chemicals - Maine) in Orrington, Maine. The SI Report provides quantitative and qualitative information characterizing the nature, rate and extent of releases of hazardous waste and/or hazardous constituents from the site, to groundwater, soil, surface water, sediments, and air. Information provided was collected between June 1994 and October 1998 and includes the results of initial and supplemental field investigations. As part of the SI Report, Human Health and Ecological Risk Assessments provide information regarding the risk posed by the site to humans and the environment. Corrective Action Objectives and Preliminary Media Protection Standards are proposed based on the results of the SI and Risk Assessments.

1.2 Investigation Objectives

As stated in the Work Plan (CDM, 1993), the objectives of this SI are to:

- Characterize and evaluate the landfills as potential contaminant sources;
- Identify other potential sources;
- Evaluate the migration of groundwater and contaminants (hydrogeology);
- Evaluate the contamination of surface soils and their potential to affect groundwater, surface water and air quality;
- Evaluate sediment contamination and sediment transport as a contaminant source;
- Estimate the total loading of contaminants from the site to the Penobscot River and other environmental media; and
- Estimate risks associated with the site specifically regarding soil, groundwater, surface water, sediment and air exposures.

1.3 Regulatory Guidance

The Site Investigation of the HoltraChem Manufacturing Site is being performed as a condition of the Consent Decree in Civil Action No 91-0188-B, United States of America v. Hanlin Group, Inc. The Consent Decree was filed with the U.S. District Court in Maine on December 22, 1993. Hanlin's responsibilities for the site investigation were assumed by the HoltraChem Group as part of their purchase of the Orrington, ME property. The Consent Decree includes requirements for preparation of an SI work plan, SI implementation, SI Report, development of corrective action objectives and media protection standards proposals, and generation of a corrective measures study and report. The sections of the Consent Decree which specify the requirements for the SI report are provided in Appendix A.

CDM developed a draft scope of work for the SI (CDM, February 1993) during the negotiations leading to the Consent Decree. The work plan was developed in accordance with the requirements of the Consent Decree and with the *RCRA Facility Investigation (RFI) Guidance, Development of an RFI Work Plan and General Considerations For RCRA Facility Investigations, Interim Final* (OSWER Directive 9502.00-6D May, 1989). The scope of work and SI work plan were finalized and submitted to EPA on December 28, 1993. The *Work Plan for the Site Investigation of the LCP Chemicals - Maine (LCP) Plant Orrington, Maine*, (Work Plan, CDM, December 1993) was approved by EPA with modifications on June 22, 1994. Comments and modifications were addressed in correspondence between HoltraChem and EPA as well as at a subsequent meeting held at EPA offices in Boston on July 7, 1994. Copies of comments and correspondence which constitute guidance for the SI are provided in Appendix A, and listed in the references provided at the end of this section.

A *Human Health and Ecological Risk Assessment Protocol Document (RAPD, Menzie-Cura Associates and CDM, November 14, 1994)* was developed in accordance with modifications to the Work Plan. The RAPD detailed the specific risk assessment guidance documents used in the development of the protocol and to be used in the execution of the human health and ecological risk assessments. EPA provided comments on the RAPD in a letter received on March 24, 1995. HoltraChem responded in a letter dated April 7, 1995 and the RAPD was approved by EPA on May 25, 1995 and incorporated into the Work Plan. Copies of comments and correspondence which constitute guidance for the SI and risk assessments are provided in Appendix A.

The initial phase of the SI was performed between June 1994 and December 1995 in accordance with the approved Work Plan. A Site Investigation report was submitted to EPA on December 22, 1995. EPA provided comments on the December 1995 SI report on March 14, 1997. The letter and

attached EPA and Maine Department of Environmental Protection (MEDEP) comments indicated the need for additional data to complete the site investigation. HoltraChem proposed supplemental site investigations in a letter to EPA dated June 26, 1997. The work to be performed during supplemental site investigations was to be in general accordance with the existing EPA-approved Work Plan (CDM, 1993). EPA provided an extension to the site investigation schedule to allow for the collection of additional site characterization data. Comments on the proposed supplemental site investigations were received from MEDEP in a letter dated August 21, 1997. Comment responses were provided to MEDEP on September 19, 1997 and October 17, 1997, and the supplemental site investigations were modified to address MEDEP comments. Copies of the June 26 letter outlining the supplemental site investigations and subsequent comments and responses which constitute guidance for the SI are provided in Appendix A.

During the course of the SI, MEDEP requested that HoltraChem perform investigations in response to specific spills or releases. In a letter dated July 27, 1997, MEDEP issued HoltraChem a Consent Agreement and Enforcement Order. The Consent Agreement was signed by the Maine Board of Environmental Protection on December 17, 1997. The Consent Agreement required HoltraChem to perform investigations and to develop and implement a Comprehensive Monitoring Plan. Although the Consent Agreement between HoltraChem and MEDEP is separate from the Consent Decree between HoltraChem and EPA which governs this SI, some of the site characterization data collected to address Consent Agreement requirements is useful to and therefore incorporated in the SI report.

Variances from the Work Plan and the guidance resulting from conditions encountered in the field are documented in the applicable methodology subsections in Section 3.

1.4 Report Organization

The report is organized to provide the necessary background information to evaluate the investigation data, draw conclusions, and to avoid redundancy. The report is divided into five volumes, one volume each of text (Volume 1), tables (Volume 2) and figures (Volume 3), followed by four volumes (Volumes 4, 5, 6, and 7) of appendices. Laboratory data sheets are provided separately.

Background information is provided in Section 2, and includes a site description and a summary of previous and contemporaneous investigations. The SI field program is presented in Section 3. To consolidate related information, Section 3 presents both the SI methodology and results. The subsections are arranged by media sampled or characterized. Each subsection is organized with the following structure: objectives, methodology, variances from the approved work plan, observations,

and analytical results. This SI Report includes all the data presented in the 1995 SI submittal plus the data collected as part of supplemental site investigations.

An evaluation of the data collected during the SI, including the results of data validation, is provided in Section 4, Data Evaluation. The environmental setting, compiling background information and the results of the site investigation, is presented in Section 5, Environmental Setting. Source characterization and waste characteristics are presented in Section 6. The distribution of contamination and evaluation of fate and transport are presented in Section 7. The Human Health and Ecological Risk Assessments are presented in Sections 8 and 9, respectively. However, most of the characterization and sampling used in the risk assessments are discussed in the sections which precede them. The Corrective Actions Objectives and Preliminary Media Protection Standards proposals are provided in Section 10.

Section 2

Background - Summary of Previous and Contemporaneous Investigations

2.1 Site Description

2.1.1 Physical Characteristics

Location

The HoltraChem Manufacturing site (the site) is located approximately 6 miles south of Bangor, Maine on the east side of the Penobscot River in the Town of Orrington, Maine. The plant buildings are located on a relatively flat plateau approximately 500 feet inland from the high tide mark of the Penobscot River and approximately 1,600 feet west of Route 15 (**Figure 2-1**). The plant is bordered to the north and west by the Penobscot River; to the south and east by the Penobscot Energy Recovery Company (PERC), a resource recovery facility which generates electricity by burning of municipal refuse; and to the northeast by undeveloped woodland. Residential and commercial properties are located along Route 15 east of the site and residential properties are located along the banks of the Penobscot River north and south of the site.

In this report, "the site" refers to all the contiguous land owned by HoltraChem Manufacturing Company, which is consistent with the definition of the "Facility" in the Consent Decree. The manufacturing plant (the plant) refers to that portion of the site occupied by structures associated with production and waste streams. The undeveloped portion of the site refers to the wooded portions of the property which are primarily located to the north and east of the plant. The developed portion of the site includes the plant, cleared land and landfills in the vicinity of the plant (Figure 2-1).

Topography

The HoltraChem Manufacturing site encompasses approximately 235 acres, with the manufacturing plant located on 12 acres. The majority of the property is heavily wooded. The land from Route 15 to the plant is uniformly sloped to the west at an average grade of approximately 10 percent. The land surface in the plant area is relatively flat with an average slope of 2 to 3 percent to the west. The land from the plant area to the Penobscot River slopes very steeply toward the river. The plant site is located at an elevation of approximately 65 feet above mean sea level, which is the national geodetic vertical datum (NGVD). A scarp and ridge are present north of the plant. Maximum

elevations of the two hills along the ridge are approximately 110 to 145 feet NGVD, respectively. The saddle between the two hills lies at a elevation of approximately 80 feet NGVD.

Plant Datum

The original plant datum was based on an estimated mean sea level. Surveying performed during the SI determined that the existing plant datum is 1.76 feet below the 1929 NGVD. Therefore, the elevations of all surveyed and measured data points (i.e., monitoring wells and water levels) at the site are actually 1.76 feet higher than indicated using the plant datum. Because all elevations are relative to their respective datum, there is no effect on interpretations caused by the datum discrepancy when comparing elevations onsite. Therefore, to maintain consistency with previous work performed at the site, all elevations surveyed and measured as part of the SI will be reported relative to the established plant datum (PD). However, when comparing water levels at the plant boundaries to elevations offsite, especially relative to the Penobscot River, the discrepancy could be significant. Therefore, corrections have been made and discussions provided, where necessary, to compensate for the difference between the plant datum and offsite elevations which are relative to NGVD.

Site Access

There are two access roads to the facility. The main road is located on the southern side of the property. A secondary road runs from behind the administration building southeasterly through a lockable gate at Route 15. This gate is locked and all vehicular traffic enters the plant through the road on the southern side of the property.

A railroad spur enters the property from the east and then splits where it enters the northeast corner of the plant. Railroad tracks border the north and south edges of the plant. The main chemical loading area is located along the southern plant boundary. A chlorine tank car testing and maintenance building is located along the northern plant boundary. The northern railroad siding extended to the northwest corner of the former sodium chlorate plant but the tracks were removed after sodium chlorate production ceased in 1980.

Surface Water and Stormwater Drainage

The site can be divided into four surface water drainage areas (**Figure 2-2**). The first of these areas receives stormwater runoff from the eastern portion of the plant and surface water flowing to the plant from the southeast. Runoff from the roof of the main production building (referred to as the

“cell building” in this report) and the eastern portion of the plant flows to a series of catch basins and a 36-inch concrete pipe located east of the cell building. Based on a plan drawn by E.C. Jordan Co. Inc. (undated), the invert of the 36-inch concrete pipe ranges from Elevation 60.89 feet PD, east of the electrical substation yard, to Elevation 58.31 feet PD at the outfall to the southerly stream. The southerly stream flows from the southern edge of the plant, and enters the Penobscot River at the head (most landward extent) of what is referred to as the southern cove. The southerly stream is considered “waters of the state” by the MEDEP.

Stormwater in the western and northern portion of the plant flows overland to catch basins and into a 24-inch asbestos-cement pipe which discharges to a paved sump. The paved sump discharges over a weir to the northern stormwater ditch, which flows southwesterly into the Penobscot River. Based on plans by E. C. Jordan dated July 1977, the invert of the 24-pipe ranges from approximately Elevation 56 feet PD at the northwest corner of the former sodium chlorate plant to Elevation 49.56 feet PD at the paved sump. The northern ditch is a permitted outfall under the plant’s general stormwater discharge permit.

A third drainage area is located to the northwest of Landfills 3, 4, and 5. Runoff from this area flows overland to the Penobscot River.

The fourth drainage area lies east of the plant and the landfills. Surface runoff is by overland flow directly to the Penobscot River or is channeled into a northeasterly flowing stream, referred to hereafter as the northerly stream. This stream discharges to the Penobscot River approximately 6,000 feet upstream of where the plant stormwater ditch discharges. The discharge point of the northerly stream is referred to as the northern cove.

2.1.2 Plant Operations

Since its construction in 1967, the plant has been engaged in the manufacture of chlorine and related products, including: sodium hydroxide (caustic soda); sodium hypochlorite (chlorine bleach); hydrochloric acid; and chloropicrin. Information on the processes and waste streams is summarized in **Table 2-1**.

The chlor-alkali process produces chlorine by subjecting a saturated brine to low-voltage, high-amperage direct current between a metal anode and a flowing bed of mercury (the cathode). Chlorine is formed as a gas at the cell anodes. The sodium forms an amalgam with the mercury. The sodium-mercury amalgam flows out of the cell to a decomposer. Purified water is added at the decomposer and the sodium is released from the mercury and bonds with the hydroxide in the water

to form sodium hydroxide (NaOH) and hydrogen gas. Hydrogen gas is used by the plant to produce steam and in the manufacture of hydrochloric acid. The chlorine gas stream is purified by passing it through concentrated sulfuric acid which absorbs the water. The dilute sulfuric acid is transported offsite as a product. Chlorine related products such as hydrochloric acid and sodium hypochlorite bleach are produced with chlorine.

The depleted brine flowing out of the electrolytic cells is treated with hydrochloric acid to liberate and remove additional chlorine gas. The brine stream is then purified by the addition of sodium hydroxide and carbon dioxide to form insoluble precipitates. The precipitates form the K071 brine purification sludges. The brine is then resaturated with salt and returned to the cells. Sodium hydrosulfide (NaSH) is used to facilitate mercury removal in the wastewater treatment system. (Formerly, sodium borohydride was used to facilitate mercury precipitation). The waste product of the wastewater treatment step is the K106 brine sludge. The first wastewater treatment system was installed in February 1972. A new wastewater treatment system was constructed in 1997. Both of the sludges are combined in the brine sludge pits where they are collected prior to disposal. Brine sludges were reportedly discharged to the Penobscot River until approximately June 1970. They were subsequently landfilled onsite until September 1983. Four of the five landfills onsite were constructed and closed prior to being subject to the interim status provisions of the Resource Conservation and Recovery Act (RCRA). Landfill 5 was closed in 1983 and is subject to RCRA monitoring requirements. Brine sludges are currently transported offsite for disposal.

From December 1967 to approximately December 1970, graphite anodes were used in the manufacturing process. Beginning in approximately December 1970 and continuing through 1971, the graphite anodes were replaced with dimensionally stable anodes (DSAs). After conversion to DSAs, potentially explosive concentrations of nitrogen trichloride were produced as a byproduct of the chlorine manufacturing process. Carbon tetrachloride was used to remove the nitrogen trichloride from the chlorine. Carbon tetrachloride was reportedly deposited in Landfills 2, 4 and 5. Onsite disposal of carbon tetrachloride reportedly ceased after an EPA/MEDEP site inspection on February 18, 1982.

Chloropicrin, a tri-chlorinated nitromethane used as a soil fumigant, is manufactured in a separate part of the plant. Chloropicrin manufacture began in the early 1970s and continues today. Calcium chloride is used to dry the chloropicrin. The calcium chloride is dissolved into the brine system when it becomes saturated. Prior to this practice, the initial batches of spent calcium chloride may have been placed into the onsite landfills.

From mid-November 1978 until June 1980, sodium chlorate was produced at the plant. The sodium chlorate process was nearly identical to the chlorine production process but did not use mercury. Sodium dichromate was added to the sodium chlorate process to improve the electrolysis reaction. This process generated brine purification sludges which did not contain mercury but did contain some sodium dichromate. Sludge from the sodium chlorate process was deposited in Landfills 4 and 5. Prior to capping Landfill 4, the sodium chlorate sludge was removed from Landfill 5 and consolidated in Landfill 4.

Utilities

Underground utilities at the site are primarily associated with process (potable, from Bangor Municipal) water, the industrial sewer, the sanitary sewer and the site stormwater drainage systems. Underground brine utilities are primarily gravity flow lines from overflow collection areas to a lined process lagoon located at the western edge of the plant and a single brine pump return line from the lagoon to the brine system in an above ground flexible pipe. A schematic of underground utilities is provided in **Figure 2-3**.

Underground electric and water lines run from the water production wells (RW-1 through RW-7) adjacent to the Penobscot River to the plant. These wells and the utility lines associated with them are no longer in use. There are no underground gas utilities at the site.

The industrial sewer collects industrial wastewaters from the plant and some cooling water which is not recirculated. Wastewater from the cell building is treated at the plant's onsite wastewater treatment system to remove mercury. The system treats wastewater to less than 50 parts per billion ($\mu\text{g/l}$) total mercury. Wastewater discharge from the treatment facility is designated Outfall 002 in the plant's National Pollution Discharge Elimination System (NPDES) permit. Discharge from the wastewater treatment system and the industrial sewer flows through a weir located at the southern edge of the plant. The combined industrial sewer flow is designated Outfall 001 in the plant's NPDES permit.

Based on a plan by E.C. Jordan, dated June 1977, the invert elevations of the industrial sewer range from Elevation 60.40 feet PD at the northeast corner of the Cell Building, to Elevation 54.51 feet PD at a drop manhole located south of the railroad tracks and caustic loading area. Other data related to the industrial sewer such as plans with invert elevations were not located in the onsite archives.

According to Mr. David Tonini, HoltraChem's Technical/Environmental Manager, the majority of pipes related to the brine system are located aboveground. Overflow from the brine system is con-

tained in a lined pond, called the lined process lagoon. Brine from this lagoon is returned to the brine system. A description of the history of the lined process lagoon is provided in Section 6.2.3, Surface Impoundments.

Underdrains beneath the former sodium chlorate plant are connected to the surface water drainage system, which discharges to the paved sump. The underdrains were designed and constructed to lower the water table in this area prior to the construction of the sodium chlorate plant. At several times over the course of the Site Investigation, water was observed flowing in the catch basin near the former sodium chlorate plant and into the paved sump during periods of dry weather. These observations suggest that the underdrains beneath the former sodium chlorate plant are collecting groundwater and discharging to surface water.

Four leachfields have been abandoned in the plant area. Two leachfields are still in use. Onsite leachfields are depicted on Figure 2-3. There are two abandoned leachfields (Nos 2 and 3) located east of the laboratory/change house building which reportedly received sanitary wastewater from the change house. One abandoned leachfield (#8) is west of the utility building and HCl tank farm. This leachfield reportedly received sanitary wastewater from the utility building. An abandoned leachfield located east of the maintenance building (#5) reportedly served a trailer used during construction of the former sodium chlorate plant. An active leachfield (#10) is located east of the office building. Abandoned plant leachfields have been replaced by a newly constructed leachfield (#1) northeast of the office building. Sanitary wastewater is pumped to the distribution box for the new leachfield.

2.1.3 Site History

The plant was constructed in 1967 by International Minerals and Chemical Corporation (IMC) to produce chlorine and related products by the mercury half cell chlor-alkali process. The only known use of the site prior to its development in 1967 was for mining of sand and gravel. The chlor-alkali plant was sold to Sobin Chemical in 1974. At that time, IMC owned 80% of Sobin Chemical. In March 1975, IMC acquired the remaining interest in Sobin Chemical. IMC sold the plant to LCP Chemicals - Maine, Inc. (LCP) a division of Hanlin Group, Inc. on April 30, 1982. HoltraChem Group, Inc. purchased the site from Hanlin Group, Inc. on April 7, 1994.

Brine sludges were landfilled onsite until September of 1983. All landfills were capped with impervious geomembrane liners or clay between 1980 and 1983. Waste sludges are currently transported offsite for disposal. Sodium chlorate was manufactured on site between 1978 and 1980.

The first wastewater treatment system was installed in February of 1972. A summary of the site history can be found in **Table 2-2**. Landfill use is summarized in **Table 2-3**.

2.1.4 Spills and Releases/Areas of Concern

The Consent Decree defines "Area of Concern" as "an area at which hazardous waste or hazardous constituents have or may have been managed or come to be located at the Facility or from which a release or releases of hazardous constituents have or may have occurred at the Facility, including but not limited to each and every landfill and surface impoundment and those areas that have been identified ... as potentially significant spill areas." The Areas of Concern include the five landfills, the lined process lagoon, spill areas within the plant, the stormwater ditches and the sediments in the Penobscot River. As part of the Consent Agreement between HoltraChem and MEDEP, HoltraChem requested that three of the leach fields (#1, #2, and #8) be added to the EPA Consent Decree list of Areas of Concern. These leach fields would then be investigated and the appropriateness of remediations be considered in accordance with the procedures in the corrective action Consent Decree.

A brief description of the areas of concern is provided below. A more detailed discussion is provided under Source Characterization in Section 6.

Landfills

The five landfills are considered potential sources for groundwater contamination. Each of the five landfills received mercury contaminated sludge. Three of the five, Landfills 2, 4 and 5, may also have received carbon tetrachloride. The landfills were open to precipitation while they were active. All of the landfills have been closed and capped for more than 10 years. Because they are capped, they are not considered potential surface exposure pathways. The available data on the history of landfill use is summarized in Table 2-3.

Leach Fields

Leach fields are considered potential sources for groundwater contamination but, like the landfills, do not represent a surface exposure pathway. A December 24, 1997 letter from HoltraChem to the USEPA requested that three leach fields on site be added to the list of Areas of Concern (AOCs), as required by the MEDEP Consent Agreement. Two abandoned leach fields and one active leach field were added to the AOC list in January of 1998. The leach fields which were identified included leach field No. 2, near the laboratory/change house building; leach field No. 8, west of the utility

building and HCl tank farm, and leach field No. 1, located to the northeast of the office building. Leach field No. 1 currently receives sanitary wastewater from the septic tanks located adjacent to the maintenance shop. Leach field Nos. 2 and 8 are abandoned, and received wastewater from the laboratory/change house building and the utility building, respectively.

Surface Impoundments

Surface impoundments are potential sources of groundwater and subsurface soil contamination. Although they may represent a surface exposure, they are part of the plant's process systems and are fenced to prevent inadvertent contact. The lined process lagoon, also known as the lined pond and formerly known as Hickel's Pond, was constructed in July 1970. After it was lined in October 1970, it was used to contain spills in the plant. The systems that have been connected to the lined process lagoon include: the brine saturator tanks, the hydrochloric acid storage tanks, the floor drains from the hydrochloric acid plants, the clarified brine pump, and the filtered brine storage tanks. The lagoon liner has been patched and replaced several times. There are currently four liners beneath the lined process lagoon.

Spills

Spills are considered sources of contamination for surface and subsurface soil, and groundwater. Spills have been reported throughout the production history of the plant. The chlor-alkali process uses large quantities of brine, and the majority of the spills have been of brine. After initial contact with mercury in the electrolytic cells, the brine becomes contaminated with mercury. The average mercury concentration of the brine is reported to be approximately 15 mg/l. Additionally, spills of products produced and used at the facility have occurred. The typical mercury concentrations of these products are: sodium hydroxide (NaOH), 15-40 $\mu\text{g/l}$; hydrochloric acid (HCl), 2-3 $\mu\text{g/l}$, and sodium hypochlorite, 30-80 $\mu\text{g/l}$. A list of recorded spills is presented in **Table 2-4**. Mercury concentrations in brine, sludge and products are summarized in **Table 2-5**.

Sediments and Surface Water in Onsite Surface and Subsurface Drainage Systems

Sediments and surface water in the onsite surface and subsurface drainage systems are considered Areas of Concern as they represent potential onsite human health and ecological exposure points and transport pathways for offsite contaminant migration to the southern cove and Penobscot River. Additionally, the drainage systems are potential sources of groundwater contamination in areas where they may discharge to groundwater.

The north ditch extends from the paved sump at the western edge of the plant to the northern edge of the southern cove of the Penobscot River. The southerly stream passes through the eastern portion of the plant, and flows along the southern plant boundary and enters the Penobscot River in the southern cove. The surface water conveyances are considered Areas of Concern for their classification and permitting issues. The southerly stream is considered "waters of the state" by the MEDEP and the north ditch is a permitted outfall of the HoltraChem plant (003).

Sediments of the Penobscot River

The first brine sludge containment system (known as Mac's Pond) was constructed around 1970 and was located approximately where the paved sump is now. Prior to the construction of Mac's Pond, brine sludge containing mercury was reportedly discharged directly into the Penobscot River via the north ditch and/or outfall 001. Today, plant wastewater, which contains mercury, is discharged to the Penobscot River pursuant to an NPDES permit through Outfall 001. Sediment transported from the plant area via the north ditch and southerly stream enters the Penobscot River at the southern cove. Penobscot River sediments have been identified as an Area of Concern due to present and historical discharges of plant wastewater to the river.

2.2 Previous Investigations

Several investigations have been conducted at the site for geotechnical and environmental purposes. A listing of previous investigations and reports is provided in **Table 2-6**. A description of the various investigations performed by others, and a summary of their usefulness for characterizing environmental media is provided below. A Previous Investigation Location Plan is provided in Appendix B, along with selected figures and data from previous investigations.

2.2.1 Geology/Hydrogeology

Prior to construction of the plant in 1966, E.C. Jordan completed a soil boring exploration program. The purpose of this boring program was to provide geotechnical information for foundation design. Borings in the plant area were generally advanced through till to refusal. Bedrock was not cored to confirm if the refusal was the top of bedrock surface. Additional geotechnical exploration programs were completed by Jordan Gorrill Associates and E. C. Jordan from 1975 through 1987 for specific plant expansion projects. Boring logs from the geotechnical investigations provide data on the stratigraphy beneath the plant and have been used, in part, to develop our understanding of site geology.

In 1975, Lawrence A. Wing and Associates advanced twelve soil borings and installed six piezometers on the ridge. Boring logs and water level measurements from the Wing reports were used to aid in the evaluation of geology and landfill construction on the bedrock ridge. However, ground surface elevations for the borings were not provided on the boring logs, so only approximate elevations can be inferred.

In 1979, Haley & Aldrich investigated two specific areas for additional landfill construction. Four monitoring wells were installed at each site (total eight wells) and 24 test pits were excavated. These investigations provide additional information on soil stratigraphy and site geology in undeveloped portions of the site.

In 1980, P.E. La Moreaux & Associates (PELA) installed eleven monitoring wells which they combined with existing wells to evaluate groundwater flow patterns and to monitor contamination. PELA added three new wells to the groundwater monitoring network in 1983. PELA did not record blow counts, and many soil sample descriptions or boring logs are incomplete. PELA's bedrock monitoring wells were installed by a water well driller and were typically completed with long open hole intervals.

In 1985, PELA compiled and summarized existing site groundwater data. The sampling protocol and decontamination procedures are not well documented, however, so PELA data have been used only on a qualitative basis. Comparison of the relative concentrations of contaminants in groundwater samples collected by PELA with the concentrations detected by Acheron, Inc. (Acheron, 1989) and by CDM (1995, 1997 and 1998) provide some information on temporal trends in groundwater contamination.

In 1985, Cambridge Analytical Associates (CAA) completed a hydrogeologic investigation of the PERC site, which abuts HoltraChem's property to the southeast. CAA advanced four soil borings, installed four monitoring wells, and excavated 54 test pits.

Acheron performed a hydrogeologic investigation of the site in 1989 and 1990. During this investigation, Acheron advanced borings in 27 locations, installed ten monitoring well clusters, installed one additional monitoring well, and retrofitted two existing wells. Acheron also collected soil, sediment and surface water samples. Acheron's investigation is summarized in the *Interim Report to LCP CHEMICALS - MAINE on the Nature, Extent and Fate of Environmental Contaminants at the Orrington, Maine Facility*, dated January 15, 1991. In general, the Acheron data are of good quality and have been used in the SI and Risk Assessments. Specific data is noted in the SI.

In 1995, CDM completed a Site Investigation which included groundwater, soil, and surface water sampling. The results from the 1995 SI are included as part of this report.

2.2.2 Surface Water Sampling

In 1979 and 1980, International Minerals and Chemicals (IMC) performed a preliminary investigation of potential mercury contamination in onsite surface water near Landfills 2 and 4 and in the surface drainage ditches flowing between the plant and the Penobscot River (see Acheron Figure 2-8 in Appendix B). Mercury concentrations ranged from none detected to 70 parts per billion (ppb), with the highest readings found in the transformer yard surface water retention basin. The samples were collected by plant personnel and analyzed in the onsite laboratory. The results were used qualitatively by CDM to focus SI sampling during the 1995 Investigation.

In 1983 and 1985, PELA collected a series of samples from four springs on the banks of the Penobscot River north of and downgradient from Landfills 3, 4, and 5. The samples were analyzed for VOCs and mercury. Except for one report of a trace concentration of carbon tetrachloride, VOCs were not detected in the samples. No mercury was detected at any of these locations. The seeps were resampled by Acheron as described below.

In the spring of 1988, a previously unidentified free flowing seep was observed on the steep embankment between the loading shed and the southern stormwater ditch. The seep was traced to a broken municipal water supply pipe located under the loading area. Water samples from the seep were collected and elevated concentrations of mercury, fecal coliform, specific conductance and pH were detected.

During 1989 and 1990, Acheron completed a sampling program to evaluate the location and extent of contamination in surface waters onsite (see Acheron Figure 4-32 in Appendix B). Three rounds of sampling were completed at four surface water sampling points, four surface seeps along the Penobscot River and at two Penobscot River sampling locations. Sampling rounds were completed in May 1989, August 1989, and April 1990. The Penobscot River locations were sampled at high and low tides for two rounds (August 1989 and April 1990). After statistical analysis, the results of the sampling rounds indicated that two of the surface water sampling locations and one of the seep locations contained elevated levels of mercury. There was no statistical evidence of contamination in the Penobscot River samples taken immediately offshore of the LCP facility. However, no samples were collected downstream of the facility.

In February 1990, an investigation beyond the site boundaries was initiated to characterize mercury concentrations in the Penobscot River. Acheron collected surface water samples from four locations (Orono to Bucksport) in February and March, and from six locations (Orono to Searsport) from April through August 1990. The results of this survey indicated that surface waters of the Penobscot River contained mercury at concentrations ranging from 0.01 ppb (Orono, February 28, 1990) to 0.68 ppb (Bucksport, February 28, 1990). However, the data from this survey were questioned by Acheron because the results were at the detection limits of the analytical technique. A second round of sampling was initiated in January of 1991 utilizing clean sampling techniques to achieve lower detection limits. During this sampling round, duplicate samples were collected and submitted to Maine Environmental, Inc., and to the Brooks Rand Laboratory in Seattle. The Brooks Rand laboratory used cold vapor atomic fluorescence and clean room techniques to achieve very low detection limits. The duplicate sample analyses indicated that the concentrations reported by Maine Environmental during both sampling rounds could be one order of magnitude too high.

The data from previous surface water sampling are relevant only to conditions during the time period sampled, and therefore have been used qualitatively to focus SI sampling efforts.

2.2.3 Surficial Soil Sampling

In 1980, PELA collected 31 surficial soil samples from locations along the northern boundary of the plant and between the plant and the Penobscot River (see Acheron Figure 2-9 in Appendix B). The highest concentrations of mercury were detected in the immediate plant area near the HCl tanks (57 ppm and 90 ppm) and in soils from the north ditch running from the paved sump to the Penobscot River (100 ppm).

Surficial soils in the plant area and offsite were sampled by Acheron in 1988 (see Acheron Figure 2-11 in Appendix B). The data indicate that surficial soils across the site contain concentrations of mercury (up to 110 ppm), with the greater concentrations primarily in the top 6 inches of soil within the plant area. The sample locations with the highest concentrations of mercury were located east of the cell building and former sodium chlorate plant, respectively. The concentrations of mercury detected in surface soil offsite did not appear to be elevated.

On June 20, 1990, Acheron collected soil samples from fifteen locations (see Acheron Figure 4-40 in Appendix B). Mercury concentrations ranged from 0.17 ppm at the background location to 60 ppm in the north ditch.

The results of previous surficial soil sampling were used to focus additional soil sampling around the site, within the plant area, and in site drainage ditches. SI results were compared to those from previous sampling.

2.2.4 Subsurface Soil Sampling

In 1980, PELA collected subsurface soil samples during the installation of monitoring wells P-7, P-8, P-10 and soil boring P-11 which were located within, or in close proximity to, the Landfill Area 1 complex. Monitoring well P-10 and soil boring P-11 were apparently advanced through former Landfill 1a. The mean mercury concentrations reported for soils from P-10 and P-11 were 348 and 53 ppm, respectively. Acheron provided contamination versus depth profiles of the PELA data in their January 1991 report which are also provided in Appendix B. In general, the concentrations of mercury dropped off quickly with depth in the borings for P-7 and P-8, yet remained elevated in borings P-10 and P-11.

In 1989 and 1990, Acheron collected subsurface soil samples from nine monitoring well locations. Samples were collected at 5-foot intervals from the ground surface to the termination of each boring. One subsurface soil sample from each boring was submitted for laboratory analysis. Two samples were submitted from cluster locations B-316 and B-321. Detectable concentrations of mercury were observed only in the borings located south and west of the plant.

Acheron also advanced three soil borings west of the plant. Soil samples were collected for mercury analysis at approximate 5-foot depth intervals. In general, mercury concentrations were high at the surface, and decreased to background levels within 5 to 10 feet of the surface with one exception. A mercury concentration of 8.9 ppm was detected in a soil sample collected from a depth of 20 feet at soil boring SB-A.

Subsurface soil sampling results from previous investigations were used to focus SI sampling efforts. Results from the SI sampling were compared to those from previous investigations.

2.2.5 Sediment Sampling

Between June, 1970 and October, 1971, the EPA, Federal Water Pollution Control Administration, Federal Water Quality Administration, IMC and Sobin collected river sediment samples. The precise locations of these samples are unknown but the analytical results and a general location description are presented on Acheron Tables 5-3 and 5-4 which are provided in Appendix B. These

data are used qualitatively in the SI to evaluate changes in sediment mercury concentrations over time (temporal changes).

In 1972, T. W. Beak Consultants, Limited (Beak) prepared a report for Sobin Chlor-Alkali, Inc. in response to an EPA request to "remove mercury deposits in the cove adjacent to Sobin's plant on the Penobscot River". Beak also evaluated sediment samples obtained from Bangor to Verona Island by EPA in 1971. Sediments collected at that time adjacent to the site contained 49 and 68 ppm mercury on a wet-weight basis. Upstream sediments contained from 0.48 to 0.9 ppm on a wet-weight basis.

On April 26, 1990, representatives of EPA collected a series of river sediment samples near the site. Acheron and the EPA split samples. Mercury was not detected above method detection limits (0.1 ppm) in the sample collected upstream of the site or in the sample collected below the Landfill Area 1 complex. Mercury was detected at concentrations of 0.38 and 1.5 ppm in samples collected from the northern drainage ditch and in the cove, respectively. The Acheron/EPA sample locations and mercury concentrations are presented in Appendix B on Figure 5-2 and Table 5-5, respectively.

Acheron collected river sediment samples from eight locations in June 1990. Samples were reportedly collected from the sediment surface and from a depth of 6 inches in the intertidal zone. However, only one analytical result is reported for each location. The seven samples collected downgradient of the bedrock valley area were characterized as sand and gravel. The sediments collected from the cove area were described as silty sand. Total mercury concentrations in the four samples collected from Landfill Area 1 ranged from 0.01 ppm to 0.04 ppm. Sediments collected from the river near the outlet of the northern drainage channel contained 2.84 to 8.2 ppm total mercury. The two sediment samples collected from the silty sand deposits in the cove contained 0.95 and 61.0 ppm of mercury. The Acheron data used in this report are included in Appendix B, Table 5-6. Sampling locations are shown on Figure 5-3, also provided in Appendix B.

The previous sediment sampling data is limited, dated, and the sampling locations, methodologies and analytical protocols are not well documented. The more comprehensive sediment sampling conducted as part of the SI is considered more representative of current site conditions.

2.3 Contemporaneous Investigations

Due to MEDEP requirements, including the Consent Agreement between HoltraChem and MEDEP, several additional investigations have been conducted on the HoltraChem site during the course of the SI. These contemporaneous investigations have been carried out to characterize air mercury emissions and ambient air quality, soil conditions, and surface and groundwater discharges from

specific areas on site, primarily to address ongoing operational issues. A listing of these contemporaneous investigations is included in **Table 2-6**. This section provides a brief description of contemporaneous investigations. Summary data from investigations pertinent to the SI are provided in Appendix B.

2.3.1 Air Monitoring Investigations

In August, 1997, Earth Tech, Inc. submitted a report to HoltraChem estimating the total mercury emissions from the HoltraChem facility during 1996 production. The study was undertaken to provide HoltraChem with an independent estimate of total mercury emissions into surrounding air. The data used in this estimate were gathered from stack reports, cell room mercury monitoring, and various reports from the EPA, Air and Waste Management Association, and other agencies. Earth Tech estimated total mercury emissions of 285.3 lbs/year; most of this mercury (217 lbs/year) was estimated to come from cell room fugitive emissions. The calculations for and results of this study are included in the report, *HoltraChem Mfg. Co. Mercury Emission Inventory* Earth Tech, August 1997.

A follow-up study was undertaken by Earth Tech late in 1997, and submitted to HoltraChem in March of 1998. This study was designed to model annual, 24-hour, and 1-hour average mercury concentrations surrounding the HoltraChem facility. Isopleths were constructed around the site which showed the maximum spread of mercury from the site occurring in easterly and northerly directions. The mercury concentration isopleths were considered in the Human Health Risk Assessment when assessing the exposure to mercury in ambient air. These isopleths are presented on Figures 1 through 3 of Earth Tech's report which are provided in Appendix B.

In December of 1997, the Maine DEP conducted a short sampling program to monitor ambient air for mercury. During this investigation, sampling locations were selected at the transformer enclosure south of the plant and southerly stream, at a utility pole near the plant entrance, on the north side of Landfill 5 and a two locations within the cell room. Hopcalite sorbent tubes were used to collect air samples. Two samples were collected simultaneously from the transformer enclosure location to evaluate the precision of the air measurements. Mercury concentration detected in the cell room ranged from 10.9 to 44.9 ug/m³. Concentrations detected at the sites surrounding the plant ranged from non-detectable to 4.49 ug/m³. However, the large discrepancies between the two samples collected in the transformer enclosure indicate that the results of this short term monitoring program may not be representative or reliable. The printouts of wind information and mercury concentrations are tabulated in the DEP report, *Ambient Mercury Monitoring at HoltraChem in Orrington* (correspondence from Jeff Emery, MEDEP to Larry Brown HMC, January 13, 1998 Appendix B).

Three off site air monitoring stations and an on site meteorological station were established in August and September 1998 as part of the Comprehensive Monitoring program required by MEDEP as part of HoltraChem's Consent Agreement. The first quarterly monitoring results will be submitted to MEDEP in January 1999. A map of the monitoring locations and the monitoring data which was available at the time this SI report was compiled is provided in Appendix B.

2.3.2 Leach Field Investigations

In July 1997, Acheron, Inc. investigated the mercury concentrations in soil associated with the leach field system on the eastern side of the HoltraChem property. This study was undertaken in response to the Maine DEP "Notice of Violation and Enforcement Intent" issued to HoltraChem in February of 1997 after a brine lab sink was found to discharge to leach field No. 1. A total of 16 samples were collected from beneath the disposal beds of the four leach fields which comprise the system, to a maximum depth of 5 feet below ground surface. Mercury concentrations in soil beneath the leach fields ranged from 0.02 to 13.76 mg/kg. The results of this study are included in Appendix B. The investigation and results are summarized in the report, *Summary Report - Leach Field Investigation*, Acheron, Inc., July 25, 1997.

In September and October 1997, HoltraChem personnel investigated the leach fields and septic tanks at the HoltraChem facility, as required by the Consent Agreement between HoltraChem and MEDEP. The results of this investigation are summarized in Acheron's "*Summary Report, Septic Tanks and Leach fields at the HoltraChem Manufacturing Company in Orrington Maine*". This report was originally submitted to MEDEP on October 17, 1997 and revised on March 27, 1998. The report summarizes the available information on the layout and use of septic systems within the plant area and the results of septic tank and distribution box sampling. Mercury was detected in several septic tanks at concentrations up to 178.6 ppm. The Summary Report is included in Appendix B. As a result of this investigation, leach field Nos. 1, 2 and 8 were added as Areas of Concern to the RCRA Corrective Action Order and to the SI by MEDEP, and agreed to by HoltraChem under the Consent Agreement. The investigations of leach field Nos. 1, 2, and 8 are discussed further in Sections 3.4.5 and 6.2.2.

2.3.3 Surface Water and Sediment Investigations

During April and May of 1997, HoltraChem personnel monitored the mercury concentration and flow rate in the northern stormwater ditch (outfall 003) during dry weather and wet weather conditions. This monitoring program was conducted in response to an EPA letter to HoltraChem

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dated April 9, 1997 regarding the relicensing of HoltraChem's NPDES permit. HoltraChem's response to EPA was dated May 23, 1997. Samples for this study were collected on three days in April and two days in May, to determine the surface water discharges at this outfall. Sampling was conducted during dry weather and wet weather to assess the degree of variability in mercury concentrations encountered during these two conditions. This study found low levels of mercury (0.0084 to 0.0124 mg/l) in surface water during low flow conditions, and slightly higher concentrations (0.0147 to 0.0359 mg/l) during wet conditions. The results of this survey are summarized in Appendix B. HoltraChem's response to EPA was dated May 23, 1997.

On June 3, 1997, Acheron performed surface water and sediment sampling of the Southerly Stream. This sampling was required by MEDEP to assess the impact of a brine release from the brine receiver tanks which was discovered by HoltraChem on February 19, 1997 (discussed in Section 2.3.4 below). Surface water samples from this sampling event contained mercury at concentrations ranging from 0.7 to 1.4 ug/l, and mercury concentrations in sediment ranged from 7.7 to 91 mg/kg. These data are summarized in Appendix B of this report. The complete results are included in the Acheron report, *Surface Water and Sediment Sampling Results - Brine Tank Investigation*, dated June 17, 1997.

As part of comprehensive monitoring required by MEDEP under the Consent Agreement, HoltraChem has collected flow volume and mercury concentration measurements from weirs installed at the paved sump and in the southerly stream since March of 1998. Flow measurements and analytical samples are collected weekly. Analyses are performed at HoltraChem's on site laboratory. Tables summarizing the results are provided in Appendix B. These data were used to estimate the flux of mercury in site surface water and to evaluate exposure concentrations for the risk assessment.

In November 1997, CDM performed a filtration study on the surface waters of the north ditch and southerly stream to support evaluation and design of sediment stabilization measures for the site drainages. The data showed that the majority (75 - 90%) of the mercury load during wet weather/high flow conditions was associated with sediment particles greater than 20 microns (medium silt). During low flow/dry weather conditions, 73% of the 44 ppb of mercury in the northern ditch/paved sump water was in the dissolved form (i.e. was not removed by filtering with a 0.45 micron filter). Although 70% of the mercury in the southerly stream low flow/dry weather sample was also in dissolved form, the total concentrations were much lower (0.82 ppb). The results of the filtration analysis are provided in Appendix B.

2.3.4 Penobscot River Surface Water Investigation

During August and September of 1998, Acheron completed a study of Penobscot River surface water as required by the Consent Agreement between HoltraChem and MEDEP. This study was undertaken to determine the fate of discharges from the site to the river during varying tidal conditions. Dye was injected into surface water and batch discharges from the plant, and was used to model the dispersion of mercury into the southern cove and the plant environs. The sampling methodologies are described in Section 3.7 of this report. Acheron presented the results of the study at a meeting in Waterville, ME on November 17, 1998. The handouts and sample results provided at that meeting are included in Appendix B. These data have been used to evaluate the impact of the site on the Penobscot River. Acheron's report on the Penobscot River surface water investigation will be available in January 1999.

2.3.5 Brine Tank Investigation

Acheron performed a subsurface investigation in the vicinity of the brine receiver tanks following a leak from the north brine tank, discovered by HoltraChem in February of 1997. The scope of work included the advancement and sampling of eight Geoprobe soil borings, installation and sampling of seven piezometers, and assessment of hydrologic conditions in the vicinity of the north brine tank. The results of the brine tank investigation generally indicated elevated mercury concentrations in soil near the brine tanks (up to 13 mg/kg), and trace to low concentrations of mercury in other locations (less than 0.5 mg/kg). Low to moderate levels of mercury (ND to 161 ug/l) were also found in groundwater samples throughout the study area. Acheron identified the southerly stream as the most likely interceptor of contaminated groundwater from this area. Acheron's report is titled *Summary Report - Brine Tank Investigation*, dated May 5, 1997. The results are summarized in Appendix B.

2.3.6 Landfill 5 Monitoring

Semiannual detection and quarterly assessment groundwater monitoring is performed at Landfill 5 as part of post closure care of this RCRA landfill. Reports of this monitoring are summarized annually. The most recent report was submitted to HoltraChem in February of 1998 summarizing the 1997 monitoring program. The results of the Landfill 5 investigations can be found in the *Comprehensive Annual Report - 1997 Groundwater Detection and Assessment Monitoring Programs* Acheron, Inc. February, 1998.

2.4 Interim Measures and Environmental Controls

Since the initiation of the SI, actions have been performed to address or improve environmental conditions at the site. The following list presents a summary of actions which are associated with the remediation or control of potential source material.

Removed approximately 30 cubic yards of mercury contaminated sediment from the paved sump (June, 1995). This was performed as an interim stabilization measure (ISM) to prevent potential discharge of the sediment to the north ditch and southern cove. The material was transported to Stablex for disposal. This ISM was repeated in April 1998 with the removal of approximately 17 cubic yards of sediment;

Removed approximately 28 cubic yards of petroleum containing soil from the locomotive fueling area (December 1997). This action was performed to address a stained soil area and investigate the presence or absence of floating petroleum product. No free product was observed. Soil from this area was temporarily stockpiled onsite and later disposed at Sawyer Environmental Recovery Facility in Hampden, ME.

Submitted a plan for Interim Stabilization of sediment in onsite streams and ditches to MEDEP on November 14, 1997. Modified the plan based on MEDEP February 10, 1998 comments and submitted a revised plan on March 13, 1998.

Cleaned catch basins within the plant of mercury containing sediment (April 21 through 24, 1998) and installed silt control socks (October 20, 1998) to limit discharge of mercury containing sediment to the onsite streams and ditches through stormwater.

Performed pavement repairs and additional paving within the manufacturing plant.

Performed a Pollution Prevention (P2) audit (December 19, 1995) to identify and eliminate potential releases.

Pumped out septic tanks and distribution boxes containing mercury in the solids (July 1996).

Replaced liner in the lined process lagoon (October 1995). Drained lagoon, inspected and repaired liner as part of lined process lagoon subsurface soil investigation (October 1998).

Removed approximately 10 cubic yards of soil associated with the former clarifier (October 1998) and approximately 10 cubic yards of soil adjacent to the cell building (December 1997).

Installed a new wastewater treatment system which reduced mercury discharge through the NPDES outfall 001 (spring and summer 1998).

Reduced the anode repair rate resulting in reduced mercury air emissions (1996-1998).

- Collected and treated groundwater from collection system adjacent to the southerly stream (1992-1998)

In addition to the actions listed above, HoltraChem has performed upgrades to the manufacturing plant (e.g. replaced tanks, cooling towers, pumps, etc.) designed to improve operations and safety at the facility.

2.5 References

Dave Tonini, Technical/Environmental Manager. Various conversations during the SI, 1995-1998

William "Zeke" Miller. Various conversations during the SI, 1992-1995.

Acheron Inc., 1991. "Interim Reports to LCP Chemicals-Maine on the Nature, Extent and Fate of Environmental Contaminants at the Orrington, Maine Facility." (Includes data submitted in previous progress reports and RCRA Quarterly Monitoring Reports pertaining to Landfill 5.)

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Haley and Aldrich, 1979. "Report on Proposed Hazardous Waste Disposal Facility, International Minerals & Chemical Group Plant, Orrington, Maine."

LaMoreaux, P.E. and Associates, 1980. "Hydrogeological Impact of Waste Disposal Sites and Recommendations for Control at the IMC Plant, Orrington, Maine."

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*Section 2
Background - Summary of Previous and
Contemporaneous Investigations*

LaMoreaux, P.E. and Associates, 1985. "Summary of Hydrogeologic Conditions and Compilation of Ground Water Monitoring Data for the LCP Plant Site, Orrington, Maine."

Wing Associates, 1975. "Report on Site Investigations of Waste Disposal Areas of Sobin Chemicals, Inc."

Section 3

Site Investigation

3.1 General

This section presents the Site Investigation field sampling methodology and results. The subsections are arranged by the media sampled or characterized. Each subsection is organized with the following structure: objectives, methodology, variances from approved work plans, observations, and analytical results. Tables providing the analytical results referenced below are bound in Volume II. An evaluation of the data collected during the Site Investigation is provided in Section 4, Data Evaluation. Qualifications which affect the usability of the data has been included on the tables. Figures showing sampling locations are bound in Volume III.

This section is divided into subsections based on media, sampling location, and sampling methodologies. The subsections are as follows:

- Section 3.2 - Surface Geophysics;
- Section 3.3 - Soil Boring and Monitoring Well Installation;
- Section 3.4 - Groundwater Monitoring;
- Section 3.5 - Site Soil Sampling;
- Section 3.6 - Site Surface Water and Sediment Sampling;
- Section 3.7 - Penobscot River Surface Water and Sediment Sampling; and
- Section 3.8 - Ecological Sampling and Characterization.

3.2 Surface Geophysics

Three separate geophysical surveys were performed: seismic refraction, time-domain electromagnetic (TDEM), and very low frequency electromagnetic (VLF). The overall purpose of geophysical surveying was to provide additional characterization of the landfills and/or the surrounding geology. The areas investigated with geophysical surveys are presented on **Figure 3-1**. The geophysist's report is provided as Appendix C. Each geophysical technique is discussed separately below.

3.2.1 *Time Domain Electromagnetic (TDEM) Surveying*

Objectives

TDEM was performed over Landfill Area 1 to identify and locate areas of buried metal which could

represent buried drums or tanks. Landfill Area 1 is the oldest disposal area and its filling history is the least well documented. Identification of areas with buried metal was used to focus groundwater monitoring efforts.

Methodology

The TDEM pulses an electromagnetic field into the earth 75 times per second and is quiescent between pulses. After each pulse, secondary EM fields are briefly induced in moderately conductive earth and for a longer time in buried metal. Between each pulse, the TDEM waits until the response from the conductive earth dissipates and then observes and records during the interval of prolonged buried metal response.

The TDEM survey of Landfill Area 1 was conducted with a GEONICS EM-61 transmitter receiver system. The EM-61 consists of two stacked electric coils separated by approximately one foot. The coils are mounted on a cart which is pulled along the survey line. Data is recorded with a computer which is worn by the surveyor. The survey was conducted along 40 parallel traverses in an approximate north-south direction. Traverses were separated by approximately five feet (1.5 meters). Measurements were made in millivolts, at 0.196 meter intervals automatically triggered by a sensor within the cart wheels.

The survey data was processed for both the upper and lower coils and for the difference between the coils. The data was kriged to produce approximately 1/3 meter grid cells. These data were plotted as three colored contour maps. Interpretation of the color contour maps involved comparison of anomalous areas to background readings and estimation of the volume and depth of metal indicated by the anomalous areas. The geophysist's report including the results of the TDEM survey is provided in Appendix C.

Variances from the Approved Work Plan

The details of the TDEM survey of Landfill Area 1 were not defined in the Work Plan. The survey attempted to cover the entire area under the landfill cap. However, due to physical restrictions and an instrument malfunction, two limited areas below the Landfill Area 1 cap were not covered with the TDEM survey.

The TDEM survey was not extended to the river's edge or to the edge of the bedrock scarp in the western portion of Landfill Area 1. The slope of the Landfill exceeds 30 degrees and surveying

became difficult as the survey cart had a tendency to overturn. However, readings observed as the survey approached this steep section did not indicate buried metal.

Observations

The GEONICS EM-61 provides values that are displayed on a screen and stored in a field computer as the instrument is pulled across the survey grid. The surface of Landfill Area 1 varies from gently to very steeply sloping towards the Penobscot River. Observations of the field crew recording the data were that the area was generally devoid of anomalous readings except in the area near the top of the landfill and partway down the slope close to the bedrock scarp. No anomalous readings were observed on the profile line performed around the lined process lagoon.

Results

Three color contour plots representing the upper and lower coil measurements and the difference between the two are provided in Appendix C. The three plots are similar. The difference plot is designed to minimize the interference of near surface materials.

The color contour plots illustrate two anomalous areas containing concentrations of metal. These areas are outlined on Figure 3-1. One area is a broad band located just down the hill towards the river from the lined process lagoon. This band is approximately 30 feet wide and 85 feet long and trends roughly parallel to topographic contours. One portion of the anomaly shows a denser concentration of metal than the rest of the band. The metal deposit is estimated to be six to fifteen feet below the ground surface.

A second anomaly is a denser concentration of metal located halfway down the landfill slope adjacent to the bedrock scarp. The anomaly covers an area approximately 25 feet by 30 feet. This deposit is estimated to be fifteen feet below ground surface.

The remainder of Landfill Area 1 did not exhibit significant metallic anomalies.

The material causing the metallic anomalies detected with TDEM surveying is not known. However, the anomalies identify where metallic debris, which is different from other disposed material, is concentrated. The location of the metallic anomalies was considered when evaluating differences in contaminants and concentrations observed in the monitoring wells downgradient of Landfill Area 1.

3.2.2 Seismic Refraction Surveying

Objectives

Seismic refraction was performed over the five landfills to acquire more information regarding the underlying geology and the depth to groundwater. The results of seismic refraction surveying were used in conjunction with nearby boring data to further characterize the landfills without penetrating the landfill caps with borings.

Methodology

A total of eight profiles (or geophone spreads) were distributed over five traverses. Two spreads were performed over Landfill Area 1. One spread was performed over and one spread adjacent to Landfill 2. A 1300-foot long traverse consisting of four spreads was performed over Landfills 3, 4, & 5 on the bedrock ridge. Each spread consisted of 12 geophones arranged in a linear alignment. Spacing between geophones was 20 feet in all spreads except on Landfill 2 where the spacing was shortened to 5 feet.

Recordings of seismic events were made with a 12 channel EG&G GEOMETRICS Signal Enhancement Seismograph. Seismic energy was generated by a sixteen pound hammer blow on a surface mounted steel plate located along the alignment of each spread. Seismograms collected in the field were subjected to digital filtering, the data were corrected to a datum, and time-distance plots were generated to associate time data points with subsurface refracting horizons. The velocity and depth characteristics of each refracting horizon were calculated.

Ground surface elevations for the seismic refraction survey were obtained from a two-foot contour interval topographic plan compiled by James Sewall Company. The plan was created from aerial photographs dated October 19, 1980. At that time Landfills 2, 3 and 4 were capped and Landfill 1 was being capped. Landfill 5 was still active. CDM performed a level survey on August 23, 1994 to obtain elevations of the geophone and shot point locations over Landfill 5 from B-303 to the edge of Landfill 4. Topography of Landfill Area 1 was obtained from the Landfill Area 1 construction plans by CDM dated October, 1980.

Variations from Approved Work Plans

The details of the seismic refraction surveys were not defined in the Work Plan. The areas covered by the survey conformed to those described in the Work Plan. Where one line was indicated over

Landfill Area 1 in the Work Plan, two spreads with slightly different orientations were performed to account for the length of the spreads and provide adequate coverage.

Where the Work Plan indicated one line over Landfill 2, one spread was performed over and one spread adjacent to Landfill 2. The spread performed adjacent to Landfill 2 was performed with a geophone spacing of 20 feet and roughly followed a previous seismic refraction survey performed by F. M. Beck. This spread was used to investigate depth to bedrock and to evaluate correlation with the earlier geophysical work. Rather than continue to duplicate Beck's earlier survey down the entire length of Landfill 2, a second alignment with a geophone spacing of five feet was placed directly over Landfill 2 to provide more information on the shallow stratigraphy and depth to groundwater.

The four spreads which comprised the survey alignment over Landfills 3, 4, & 5 corresponded to the survey line indicated in the Work Plan.

Observations

Significant signal noise was anticipated from activities within the operating plant. These noises were observed in the seismic traces of most geophone spreads and especially in spreads conducted over Landfill Area 1 and Landfill 2.

Regular but intermittent noise from a compressor was heard coming from the utility room on the western side of the plant near Landfill Area 1. To minimize the impact of this compressor on the survey, blows with the hammer used to generate seismic energy were timed to occur during quiet periods in the compressor's cycle.

Railroad cars were being moved on the tracks adjacent to Landfill 2 as the survey spreads were being laid out. To reduce potential noise from the railroad, the surveys were conducted when the cars were not being moved and the engine was as far from the survey area as possible.

To further minimize the impact of noise on the survey, multiple blows were performed at each shot point on all spreads. The seismographs from multiple seismic energy events could then be stacked and noises filtered out thereby improving the signal to noise ratio.

Results

The results of the seismic refraction program are presented in Appendix C. The geophysist's report includes tables which present the depth and elevations of identified layers beneath each shot point

and geophone. The associated figures present a graphical representation of the data in the tables and provide illustrations of the seismic data similar to geologic cross sections. Each of the three areas surveyed (bedrock ridge - Landfills 3, 4 and 5, Landfill Area 1 and Landfill 2) are discussed in the paragraphs that follow.

Bedrock Ridge - Spreads A through D

Spreads A-D were performed in a line extending over Landfills 3, 4, & 5 and can be considered a continuous profile. This profile identified three distinct velocity zones. The shallowest zone with a velocity of approximately 1200 feet per second (fps) represents unsaturated, unconsolidated fill and overburden materials. This velocity is similar to the value measured in the fill of Landfill 2. The presence of water in this layer would be expected to increase the velocity to approximately 3600 to 4400 fps. This increase was not observed in any of the spreads performed on the bedrock ridge. The thickness of the fill/overburden deposits in the three landfills was estimated as follows: Landfill 3, 16-20 feet; Landfill 4, 17-20 feet; Landfill 5, 3-16 feet.

Part of the material deposited in Landfill 5 was solidified using the "Solidtex" process which may increase the seismic velocity relative to non-solidified materials. An increased seismic velocity would produce a similar short travel time as would a thinner thickness of fill with the same velocity as was calculated elsewhere. Therefore, the seismic refraction survey may underestimate the thickness of fill in Landfill 5.

Based on the ground surface elevations and the thicknesses of fill and overburden determined from the seismic refraction survey, the elevation of the top of rock beneath the three landfills on the ridge is estimated to range as follows: Landfill 3, 70-77 feet; Landfill 4, 74-78 feet; Landfill 5, 86-104 feet. The thickness of the deposit with the velocity of 1200 fps is representative of fill and unsaturated overburden and therefore it does not necessarily indicate that fill material is in direct contact with bedrock. However, Landfill 4 was reportedly excavated down to the top of rock. Therefore, the thickness of the first layer may be interpreted as the thickness of the fill material and the base of the fill material, the top of bedrock. Because of the higher velocity of the solidified fill material in Landfill 5, the elevation indicated for the top of rock beneath the calculated fill thickness may be higher than the actual elevation in some places.

A zone with an intermediate velocity of approximately 7100 fps is present beneath the low velocity (1200 fps) fill/unsaturated overburden deposit and above a high velocity (15,450 fps) zone. The velocity of the intermediate zone is similar to that measured in till deposits in the vicinity of Landfill 2. However, boring logs near the survey line indicate that the top of bedrock corresponds to the top

of the 7100 fps zone. The thickness of this zone ranges from 15 feet adjacent to B-303 at the northernmost end of the survey to 76 feet near B-304. The average thickness of the unit is approximately 48 feet.

According to the Groundwater Handbook for the State of Maine, fracturing in Maine's bedrock typically decreases as a function of depth. Evaluation of boring logs for wells located on the bedrock ridge indicate there may be a correlation between the degree of fracturing noted and the depth of the higher velocity zone. However, the data to support this observation is limited. Most of the bedrock wells installed by Acheron and prior consultants were advanced with a Barber Air Rotary or traditional rotary equipment and only the notations of fractures made by the drillers are available. Borings for bedrock monitoring wells installed by CDM were cored through the upper 30 feet of bedrock. Only one boring advanced by CDM, MW-410, is located in the vicinity of the survey line and it is approximately 60 feet away. In this boring, a fracture was encountered approximately 36 feet below the top of rock. The rock encountered above this feature was not significantly fractured.

The cause of the lower observed velocity in the upper portion of bedrock is not known. Seismic velocity is an indirect measurement of the density or competence of the rock. Therefore, the observed lower velocity zone indicates that the upper portion of bedrock along the ridge may be less competent, i.e. more fractured or weathered. It is therefore possible that this zone could have a higher hydraulic conductivity than more competent bedrock at greater depths. However, the data available from boring logs and observations made during the SI drilling program do not indicate a highly weathered or fractured zone corresponding to the seismic survey results. One plausible scenario which is corroborated by the VLF survey discussed in Section 3.2.3 is that fractures on the bedrock ridge may be oriented primarily in a north-south direction, essentially perpendicular to the seismic lines. Seismic energy would be slowed down crossing the fractures resulting in the lower than anticipated bedrock velocity. In this case, the depth of the lower velocity bedrock zone represents the approximate effective depth of the north-south trending fractures.

Landfill Area 1 - Spreads E and F

Spreads E and F were performed in two separate orientations across Landfill Area 1. Spread E began adjacent to the lined process lagoon and proceeded westerly toward the Penobscot River. Spread F originated at monitoring well cluster B-326 and proceeded northeasterly toward the bedrock scarp. The two survey lines crossed at approximately the midpoint of Spread F and in the westerly third of Spread E.

Three distinct velocity zones were identified in Spread F while only two of these three zones were distinguishable on Spread E. The deepest of the three zones is characterized by an average velocity of approximately 9066 fps, which is within the range anticipated for moderately dense till. The elevation obtained for this zone corresponds with the dry, olive-grey silty clay encountered 64 feet below ground surface in B-326.

Overlying the 9066 fps zone is a layer with an average velocity of approximately 5844 fps on Spread F and 5514 fps on Spread E. The velocity of this zone is within the range expected for a saturated layer. The top of the middle zone is calculated to be approximately 27.5 feet below ground surface in the vicinity of B-326. A stratigraphic change from silty fine sand to medium to coarse sand with gravel was observed in B-326 at 34 feet below ground surface. The water table in this well cluster ranges from approximately 30 to 33 feet below ground surface.

The top of the 5844 fps layer on Spread F is a smoothly sloping surface with a gradient of 17.5% (or 0.175 ft/ft). A straight line contact between contrasting velocities could be indicative of the top of the water table. The correlation of the top of this layer with the observed depth to water in B-326 and the characteristic straight line slope observed on Spread F indicate that the upper surface of this layer corresponds to the water table in Landfill Area 1. The expression of this surface on Spread E is not a straight line but is a smooth reflection of the surface topography. The fluctuations of the top of the 5514 fps layer in this alignment could reflect true conditions caused by variations of grain size or could be a result of approximations of the ground surface elevation. Based on the elevations calculated for the top of the second layer, the water table beneath Landfill Area 1 ranges from Elevation 47 feet PD to Elevation 7 feet PD along Spread E and from Elevation 44 feet PD to Elevation 0 feet PD along Spread F. These values are consistent with water levels measured in P-7 adjacent to the lined process lagoon and the B-326 cluster at the base of Landfill Area 1. New monitoring wells installed within and around Landfill Area 1 during 1997 provide additional data on groundwater elevations in this area. Measured groundwater elevations in MW-501, MW-512, and MW-513 indicate groundwater elevations of 30.9 feet, 0.1 feet, and 0.69 feet PD, respectively. Although none of these wells lie directly along the paths of the seismic refraction lines, these values are within the range predicted by spreads E and F, lending qualitative support to the geophysical survey results.

The uppermost layer in Landfill Area 1 ranged in velocity from 1008 fps on Spread E to 1298 fps on Spread F. The higher value is similar to the 1225 fps calculated for the uppermost layer in the four spreads across Landfills 3, 4, & 5. The lower value is closer to the 966 fps calculated for the uppermost layer observed on Landfill 2. The seismic refraction survey averages velocities within distinct velocity zones. Therefore, the velocity of the upper zone most likely represents

combinations of fill of varying compositions. The lower velocity observed in the uppermost zone of Spread E relative to Spread F, is indicative of a greater percentage of less dense fill materials.

Landfill 2 - Spreads G and H

Two spreads were performed in the vicinity of Landfill 2. Spread G originated at monitoring well cluster B-301 and extended along the road which runs adjacent to the landfill. Spread H was performed on top of Landfill 2 with the geophone spacing reduced to 5 feet.

Three velocity zones, 1300, 7900 and 17,800 fps, are identified on Spread G. Based on the boring log for B-301-01, these zones correspond to loose overburden or fill, moderately dense glacial till and bedrock, respectively. The depth calculated for the bedrock is shallower than was observed in B-301. However, Spread G does indicate a thick section of glacial till underlies the Landfill 2 area. The thickness of the less dense overburden materials under the road range from 10 to 12 feet.

Spread H was performed directly over Landfill 2 with a reduced geophone spacing of five feet. The shorter geophone spacing was selected to provide greater detail of the shallow stratigraphy and to help detect the water table, if present, within the fill. Three distinct velocity zones are identified on Spread H. The deepest of these with an average velocity of 7937 fps corresponds to the moderately dense glacial till identified as Layer 2 on Spread G. The middle layer on Spread H has an average velocity of 1274 fps which is similar to the uppermost layer observed on the bedrock ridge and in Landfill Area 1. Because this spread was performed directly over the top of Landfill 2, the 1274 fps layer is interpreted to be fill material comprised primarily of sludges from plant operations. The thickness of the fill material ranged from 9.2 to 3.8 feet. If this zone were saturated it would be expected to exhibit a velocity between 3600 and 4400 fps. This higher velocity was not observed. Based on an average ground elevation of 72 feet PD, the base of the fill deposit ranges from 58.6 to 60.5 feet.

The uppermost zone observed on Spread H had an average velocity of 966 fps and ranged in thickness from 1 to 3.8 feet. This is the lowest velocity identified during the seismic refraction program and may represent the landfill cap material.

3.2.3 Very Low Frequency (VLF) Electromagnetic Surveying

Objectives

The VLF survey was conducted over the bedrock ridge to provide information on potential

hydraulically conductive bedrock features, such as fractures or faults. Information regarding potential preferential pathways of groundwater flow in bedrock was used to evaluate proposed monitoring well locations. The results of the VLF survey were used to revise two proposed bedrock monitoring well locations. In conjunction with the results of the drilling and groundwater sampling programs, the VLF survey provides a more comprehensive understanding of the hydrogeology than could be obtained from the drilling program alone.

Methodology

VLF surveying measures changes in magnetic fields created by very low frequency electromagnetic waves which are generated at U.S. Navy submarine communication centers in Cutler, ME, Annapolis, MD, San Juan, Puerto Rico and elsewhere. As the electromagnetic waves propagate through the earth they create magnetic fields with specific orientations. When these electromagnetic waves encounter conductive planar features such as water bearing or mineralized fractures or faults, the orientation of the anticipated magnetic field is shifted. The VLF receiver is capable of measuring the angle of tilt of the magnetic component of the secondary electromagnetic field induced by the VLF transmissions encountering conductive planar subsurface features. An approximation of the direction and dip of the conductive feature can be calculated. When similar features are encountered on successive survey lines, the strike of the conductive feature can also be estimated.

Two days of VLF surveying were conducted at the facility. Measurements were received using an ABEM WADI VLF receiver and transmissions from Aqueda, Puerto Rico, Annapolis, MD and Jim Creek, WA. Based on the results of the first day, a second day was performed extending the survey to further evaluate features.

VLF measurements were processed with a Hjelt filter to derive the components of implied current density with depth. These components are termed "real" and "imaginary". The magnitude of the real component of angle shift provides information on the location, depth and dip of the fracture systems observed. The magnitude of the imaginary component provides information of the degree of conductivity within these fractures. Increased conductivity can result from mineralization or conductive groundwater within the fracture. Computations were consolidated into current density sections for each line. Areas of anomalous current densities were plotted on the base map of the field traverses and the correlations interpreted.

Variations from Approved Work Plans

The VLF survey was not proposed in the Work Plan and was added to the field program as a means

to help characterize the geology and hydrogeology of the bedrock ridge.

Observations

Several anomalous features were detected in the field during surveying. However, interpretation of these features required post-survey data processing and comparison of the anomalies detected across the survey lines. Unacceptable interference was encountered when VLF surveying was attempted across the bedrock scarp in the vicinity of the plant and these lines were not processed or interpreted.

Results

The results of the VLF survey are presented in Appendix B, as curves of the imaginary and real components of the angle shift and as illustrative plots of current density versus depth below each line. An interpretation of the strike and dip of the fracture sets detected with the VLF survey is also included on Figure 3-1.

Line 1 started at monitoring well cluster B-303 and proceeded westerly over Landfills 3, 4, and 5 along the same alignment as the seismic refraction survey. The filtered data suggested a fracture system at 141 feet west, another at 394 feet west, another at 555 feet west and a smaller fracture at 787 feet west. The largest of these was the system encountered at 141 feet west which is beneath Landfill 5. The section consolidating the filtered data suggests that the first system dips to the west to a depth of approximately 98 feet and is truncated by an easterly dipping component which extends to the depth of the survey, approximately 197 feet. The feature at 394 feet west dips to the west and the feature at 558 feet west dips to the east. These two fractures are located beneath Landfill 4. The fracture at 787 feet is located beneath Landfill 3 and is interpreted to be shallow. The imaginary component indicated mineralization or conductive water within all of the systems.

Line 2 started at the proposed location of monitoring well MW-406-B1 and extended to a point approximately 79 feet past TP-103. It was subsequently extended an additional 400 feet. A fracture was indicated at the 470 feet west station which is located north of Landfill 4. This system is a deep feature and appears to correlate with a merging of the two fractures located beneath Landfill 4 on Line 1. Anomalous readings were detected north of Landfill 5. However, only a small indication of a deep fracture was observed.

Line 3 was performed as close as practical to the break in slope from the bedrock ridge to the Penobscot River. The line was extended to the east to investigate the fracture observed beneath Landfill 5. A fracture was detected north of Landfill 4 just south of the drainage swale which leads

to the Penobscot River. This was interpreted to be the northernmost expression of the fractures observed beneath Landfill 4. Another fracture was observed north of the drainage swale which was interpreted to be the northernmost expression of the fracture observed beneath Landfill 5.

Line 4 was performed in a north-south direction between Landfills 4 and 5 in an attempt to locate conductive features with an east-west orientation. This survey did not identify any conductive east-west trending fractures in the bedrock of the ridge.

Line 5 was performed west of Landfill Area 1. A fracture was identified dipping to the north which unlike the fractures detected on the bedrock ridge, was not conductive. Either this feature was not mineralized or did not contain conductive groundwater.

3.3 Soil Boring and Monitoring Well Installation

Soil borings were advanced and monitoring wells were installed during both the 1995 site investigation and the 1997 supplemental site investigation. Boring and monitoring well locations are presented on **Figure 3-2**.

3.3.1 Soil Borings

Objectives

Soil borings were advanced to collect soil samples for geologic and chemical analysis and to install monitoring wells for the collection and analysis of groundwater samples.

Methodology

During the 1995 site investigation, soil borings were advanced by augering or wet rotary drive and wash techniques. Typical deep boring advancement began with 4-1/4 inch inside diameter (ID) hollow stem augers which were advanced to the top of the water table. After the water table was reached, the hollow stem augers were withdrawn and 5-inch ID casing was advanced until till or casing refusal was encountered. The 5-inch casing was seated into the till and the hole advanced to the top of rock with a 4-1/2 or 4-7/8 inch diameter roller bit. Bentonite was added to the drilling fluid only when drilling through the overburden strata, and only in boreholes advanced to bedrock. Approximately five feet of bedrock was then drilled with the roller bit to create a socket into which 4-inch ID casing was installed. Grout was emplaced in the hole and 4-inch ID casing was lowered

into the rock socket. The grout was allowed to cure for a minimum of 24 hours prior to advancing the boring into bedrock using an HW core barrel. If the thickness of till was greater than 50 feet, the 4-inch steel casing was temporarily set and used as a guide for the core barrel. Upon completion of the boring and construction of the well, the temporary casing was removed and the well grouted back to the ground surface. This procedure was used at MW-409-B1 and B-404-B1.

The core hole was advanced in five-foot runs, typically 30 feet into competent bedrock. Penetration into bedrock varied depending on the number and depth of potential water bearing fractures encountered at each location. Adjacent shallower borings were advanced using similar techniques to install intermediate depth monitoring wells. Borings for shallow monitoring wells were typically advanced using only hollow stem augers.

Advancement of soil borings during the 1997 supplemental site investigation was similar to the previous site investigation with few exceptions. All 1997 soil borings were advanced using wet rotary drive and wash methods. Six, five and four-inch ID casing were used to advance soil borings through overburden material. At locations where bedrock monitoring wells were installed, five foot rock sockets were created to install either permanent or temporary casing.

Borings through bedrock were advanced by coring with a 2.1 inch ID double barrel diamond-bit core, run at five-foot increments. Wire line coring methods were employed during the installation of the MW-505 and MW-511 clusters. Permanent casing was grouted five feet into the top of bedrock at MW-505-B1, MW-506-B1, and MW-509-B1. Temporary casing was used at MW-503-B1, MW-504-B1, and the MW-511 cluster. Temporary 4-inch ID casing was set five-feet into competent rock and volclay grout was placed from the bottom of the hole to the ground surface. Grout was allowed to set for a minimum of 24 hours and then further advancement into rock took place through the center of the temporary casing. Upon completion of the boring and construction of the well, the temporary casing was removed and the well grouted back to the ground surface.

Variations from Approved Work Plans

Modifications were made to the soil boring and monitoring well installation program to account for additional data generated by the geophysical program and for conditions encountered in the field. Modifications were discussed with EPA prior to implementing the changes. A summary of the modifications is provided below.

Variations during the initial site investigation - 1995

- The proposed CDM-7 cluster was relocated after the soil boring at the initial location did not encounter the fill material which was the target of continuous sampling for mercury analysis. The boring which was not converted to a monitoring well is number B-412.
- Modifications were made to proposed locations CDM-7, CDM-8 and CDM-9 based on the results of the VLF survey or physical restrictions at the site.

Using VLF surveying, a significantly conductive feature was identified beneath Landfill 5. Existing and proposed monitoring wells were not likely to have intercepted this potential bedrock fracture. Therefore, CDM recommended that monitoring well cluster CDM-8 be relocated to evaluate this feature.

Two less conductive features were also identified beneath Landfill 4. Existing and proposed wells would not encounter these features either. Therefore, CDM recommended that well cluster CDM-7 be relocated to the northeast and drilled deeper to potentially encounter these features.

The location of well cluster CDM-9 was moved approximately 50 feet to the west to account for physical restrictions encountered in the field. The proposed location was in a drainage swale which reportedly flooded due to ice dams in a nearby culvert. The location was moved slightly to place the well on higher ground.

- Permanent casing was not grouted into bedrock prior to coring at clusters CDM-4 and CDM-9 as described in the Work Plan Section 6.2 and EPA modifications.

Approximately 100 feet of till was encountered at monitoring well cluster CDM-4 (MW-404-O1), located near the salt storage pad. This depth to bedrock was greater than anticipated based on existing boring logs and interpretations. CDM's rationale for clusters CDM-4 and CDM-9 was that after surface casing was installed through any potentially contaminated shallow zones, temporary casing set through thick (>50 feet), dense till would be adequate to protect the bedrock aquifer from potential cross-contamination.

Variations during the Supplemental Site Investigation - 1997

- Two bedrock monitoring well clusters (MW-505 and MW-511) were added to further define groundwater conditions and flow between the site and Ferry Road residences. Each bedrock monitoring well cluster was located on adjacent PERC property and included a deep bedrock monitoring well and an intermediate bedrock monitoring well. Screen intervals were selected based on comparative elevations and size and presence of water bearing fractures. Screen intervals were discussed with MEDEP's Project Geologist before monitoring well installation.
- Three additional overburden monitoring wells (MW-510-O1, MW-512-O1 and MW-513-O1) were installed. The locations of these three wells were discussed with MEDEP's Project Geologist before boring advancement and well installation. Monitoring well MW-510-O1 was installed on the slope between the salt storage pad and the paved sump to further delineate the chloropicrin plume in the area. Monitoring well MW-512-O1 was installed between the MW-401 cluster and the B-326 cluster. This well was installed to assess the degree of contamination between the two existing clusters. Monitoring well MW-513-O1 was installed between the MW-326 cluster and monitoring well MW-402-O1. This monitoring well was installed to assess the degree of contamination between the B-326 cluster and MW-402-O1. Screen intervals were selected to monitor the zones of potentially highest contamination overlying the till.
- Borings were advanced through two landfills on the property, as described in the Supplemental Site Investigation Proposal (CDM, June 1997). Monitoring well MW-501-O1 was advanced through the cap of Landfill No. 1, and monitoring well MW-506-B1 was advanced through the cap of Landfill No. 4. The initial SSI proposal did not describe the method used to penetrate and reseal the landfill cap. Prior to advancement at each of these locations, a backhoe was used to dig away the cap material above the landfill liner. Once exposed, the liner was cut in a star pattern to allow access and advancement of the drilling casing. Upon monitoring well completion, the liner was repaired and the monitoring well surface completion constructed to prevent infiltration.

Repairs began by placing bentonite powder to a depth of eight inches below the hypalon liner and to a distance of 12 inches surrounding the protective casing. The liner was laid down over the initial layer of bentonite powder and an additional eight inches of bentonite powder was then placed over the liner. Bentonite was placed extending to a minimum of eight inches past any cut in the liner. A hole was cut in the center of a piece of bentonite-

filled geocomposite clay liner (GCL) allowing it to be placed down over the protective casing and over the bentonite. The GCL was attached to the steel surface casing with stainless steel clamps. An additional 12 inches of bentonite powder was then placed above the GCL. Fill material used in the original cap was then augmented with additional bentonite powder and compacted in six inch lifts. The bentonite patch and GCL formed a cone to deflect percolating water away from the sealed area. The common fill cap material was then replaced and reseeded.

Observations

Observations made during soil borings are noted on the boring logs provided in Appendix C.

Results

Stratigraphic data from soil borings are presented in the soil boring and monitoring well installation logs provided in Appendix D. Cross-sections presenting stratigraphic depiction of the HoltraChem property are presented in Section 5. Chemical analyses of soils and groundwater are provided in the following sections.

3.3.2 Subsurface Soil Sampling

Objectives

Soil samples were collected during soil boring advancement to characterize the subsurface stratigraphy, and for laboratory testing. Selected soil samples representative of the characteristic soil types encountered at the site were analyzed for grain size distribution.

Methodology

Soil sampling was performed on the deepest soil boring in each cluster. Split-spoon samples were collected using ASTM standard methods for soil penetration tests (140 pound hammer, 30-inch drop). Continuous split-spoon soil samples for mercury analysis were collected from 400 series soil borings B-403-B1 (CDM-3) and B-407-B1 (CDM-7) where contamination versus depth profiles were established. Continuous split-spoon soil samples for mercury analysis were collected from all 500 series soil borings on the HoltraChem property, with the exception of MW-508-O1, MW-509-O1, MW-512-O1, and MW-513-O1. Borings MW-508-O1 and MW-509-O1 were installed adjacent to existing borings; data from the existing borings were used to characterize subsurface conditions at

these locations. MW-512-O1 and MW-513-O1 were installed for groundwater monitoring at the base of Landfill Area 1, and sampling at these locations was conducted at 5-foot intervals. Monitoring well clusters MW-505 and MW-511 encountered minimal amounts of overburden material, and split spoons were not collected at these soil boring locations. In soil borings where thick sections of dense till or thick gravel layers were encountered, split-spoon sampling was conducted at 5-foot intervals.

Mercury analysis was conducted on all 500 series soil boring soil samples at the on-site HoltraChem laboratory. Split samples collected from the 500 series soil borings were sent to CDM laboratory for mercury analysis as part of the QA/QC program. Volatile organic compounds (VOCs) were collected and submitted to CDM laboratory when elevated levels were detected in headspace readings.

In addition to split-samples collected as part of the QA/QC samples, trip blanks, rinsate blanks and soil duplicate samples were collected. Trip blanks were submitted for VOCs with samples submitted to CDM Laboratory. Rinsate blanks were collected by running deionized water over decontaminated split-spoons. Rinsate blanks were analyzed for mercury at HoltraChem laboratory.

Rock cores were collected to confirm and characterize bedrock at each 400 series cluster location except MW-402 (CDM-2), which penetrated overburden deposits only. Rock cores were collected at all 500 series monitoring well clusters, and at monitoring wells MW-506-B1 and MW-509-B1. Rock coring was performed using a 5-foot long, 2.1 inch I.D., double barrel diamond-bit corer. Wireline coring methods were employed during the advancement of MW-505 and MW-511 well clusters. The competency or degree of fracturing of the rock was established by calculating the rock quality designation (RQD) which is defined as the total length of pieces greater than 4-inches long divided by the total run length times 100. The size and frequency of water bearing fractures were noted.

Variations from Approved Work Plans - 1995

Selected soil samples representative of the characteristic soil types encountered at the site were analyzed for grain size distribution. These additional analyses were performed to provide the data necessary for soil characterization required by the Consent Decree.

Variances from Approved Work Plans - 1997

A shelby-tube was collected from the soil boring for monitoring well MW-504-O1 to characterize a confining layer at a depth of 10 to 12 feet. The laboratory was unable to extract enough of the sample to run a permeability test; however, both grain size and Atterberg limit analyses were performed.

Wire-line coring methods were employed in bedrock monitoring well clusters off HoltraChem property. This method was used to facilitate advancement for deep and intermediate bedrock monitoring wells.

Observations

Observations made during subsurface soil sampling are provided on the soil boring and monitoring well installation logs, provided in Appendix D.

Results

Grain size distributions and Atterberg limits are provided in Appendix D.

The analytical results of subsurface soil sampling are summarized in **Table 3-1**.

3.3.3 Monitoring Well Installation

Objectives

Monitoring wells and monitoring well clusters were installed to evaluate water level elevations, establish groundwater flow directions, determine the relative permeability of the screened horizons, and monitor groundwater quality.

Methodology

Overburden Monitoring Wells

Overburden monitoring wells were constructed of 2-inch inner diameter (ID) Schedule 40 PVC casing and well screens with non-threaded caps. The well screen was typically 10 feet in length with a slot size opening of 0.010 inch. Continuously wound "V" slotted PVC screens were used to

facilitate well development and improve sample quality. Shorter screens were installed where necessary to isolate the formation of interest. Overburden monitoring well installations were backfilled with 00 Morie sand around the screen, a bentonite seal and grout. The filtering sand pack typically extends from approximately two feet below the screen to two feet above it. A minimum of two feet of bentonite seal was placed above the sand. The annular space from the top of the bentonite seal to the ground surface was filled with a cement-bentonite grout. Monitoring wells were completed at the ground surface with a locking protective steel casing, which was a minimum of seven feet long and installed a minimum of four feet into the ground. A flush mounted roadway box was installed at MW-507-O1 to allow access for vehicular traffic. A cement collar was emplaced around the wells after the grout had set up. The collars were sloped away from the wells to permit drainage. The monitoring well designation number was marked on the outside of the protective casing.

Bedrock Monitoring Wells

Bedrock monitoring wells were constructed of 2-inch ID Schedule 40 PVC casing and well screens with a non-threaded cap. The well screens have a slot size opening of 0.020 inch and were typically 20 feet in length. Standard machine slotted PVC screens were used for bedrock well completions as turbidity problems in bedrock were not anticipated. Screen intervals were typically set between 10 to 30 feet below the top of competent rock. Bedrock monitoring well installations were backfilled with filtering sand around the screen, a bentonite seal and grout. A filter pack of #2 Morie sand was placed around the screen to a minimum level of 1 foot above the top of the well screen. A 1-foot layer of 00 Morie sand was placed at the top of the sand pack to prevent slurry from penetrating the sand pack. A bentonite slurry seal was emplaced by tremie method, typically extending from the top of the filter sand to 2 feet above the top of bedrock.

The remainder of the annulus was backfilled with a cement-bentonite grout emplaced using the tremie pipe method which introduces grout from the bottom of the hole upward thereby reducing the possibility of bridging. A locking cap was welded onto the 4-inch casing for wells where permanent casing was left in place. In all other respects surface completions of bedrock monitoring well installations are identical to those of overburden monitoring wells.

Variances from Approved Work Plans - 1995

- The bedrock interval at well cluster CDM-4 was abandoned.

Approximately 100 feet of till was encountered at monitoring well cluster CDM-4 located near the salt storage pad. After penetrating the till, confirming bedrock and setting temporary casing, fifteen feet of rock core was recovered. The core barrel became stuck in the rock on the fourth five-foot core run. Repeated attempts to free the stuck core barrel resulted in twisting-off the three-inch and four-inch casings approximately 40 feet below ground surface. The four-inch and a portion of the three-inch casing were recovered but the core barrel was never freed. The boring was plugged and abandoned by pressure grouting it to the surface. Based on the difficulties encountered at this location and the number and location of other bedrock wells in the area, CDM decided not to attempt another bedrock installation at this location.

- Overburden monitoring well construction on the bedrock ridge was revised to account for site specific conditions.

Modifications were made to shallow overburden well construction to account for very thin saturated thicknesses in soil above rock. At two locations, MW-407-O1 and MW-408-O1, two and four feet, respectively, of saturated overburden were encountered above bedrock. To permit a greater amount of groundwater to accumulate in the overburden monitoring well, a five-foot sump was installed in a socket drilled into rock. The annular space between the five-foot PVC sump and the rock was filled with bentonite. The base of the screen and sand interval was located at the top of rock.

- At the request of EPA, an additional well, MW-411-B1, was installed at the original location of CDM-8. This well was intended to be completed as an overburden well. Groundwater was not encountered in overburden but was encountered at the bedrock-overburden interface. Therefore, this well was completed in the upper five feet of bedrock.

Variations from Approved Work Plans - 1997

- Repairs to hypalon landfill liners were required around monitoring well MW-501-O1 (Landfill No. 1) and monitoring well MW-506-B1 (Landfill No. 4). A combination bentonite and geocomposite clay liner (GCL) was placed and secured around the surface casing of each of these monitoring wells, as described in Section 3.3.1, Soil Borings.

Observations

A larger than anticipated volume of filter pack sand was required to complete MW-401-B1. The interval that required extra sand pack was from 43 to 41 feet below ground surface. This interval corresponded to observations of highly fractured rock during drilling.

Chloropicrin odor and tearing (lacrimation) were noted by the drilling crew when pulling the augers and installing monitoring well MW-402-O1. The presence of chloropicrin was later confirmed with groundwater sampling.

No other significant observations were noted.

Results

Monitoring well construction is summarized in **Table 3-2**.

3.3.4 Monitoring Well Development

Objectives

Monitoring wells were developed after completion to:

- remove drilling induced water and solids;
- establish communication between the borehole and surrounding formation;
- permit measurement of the hydraulic conductivity of the formation; and
- minimize the turbidity of the groundwater samples.

Methodology

Upon completion of monitoring well installation, each well was developed using a WaTerra inertial pump and a surge block. Development proceeded until field measurements of pH, temperature and specific conductance stabilized and the turbidity of the development water was less than 50 nephelometric turbidity units (NTUs) or up to a maximum of 3 hours. Development water was discharged through the plant's wastewater discharge system.

Variances from Approved Work Plans - 1995 and 1997

Due to slow recharge, some wells pumped dry and required repeated pumping to develop. For these wells, development proceeded for more than three hours. Additional well development was performed on some wells prior to sampling to further reduce turbidity.

Observations

Observations and measurements of turbidity were made during monitoring well development. Development continued until the purge water measured less than 50 NTUs or further improvement appeared unlikely.

Results

In general, monitoring wells installed by CDM produced groundwater samples with low turbidity. Due to the very fine grained nature of some screened materials, however, some groundwater samples remained above 50 NTUs despite repeated development efforts.

3.4 Groundwater Monitoring

Groundwater monitoring was performed to further characterize directions of groundwater flow, and the concentrations and distribution of groundwater contamination present at the site. Groundwater monitoring comprised water level measurements, tidal cycle monitoring, monitoring well sampling and analysis, and seep sampling and analysis. Based on the initial 1995 sampling results, selected wells were resampled as part of the original site investigation. In addition to the 1995 sampling round of onsite monitoring wells, four residential wells on Ferry Road were sampled as part of the site investigation.

During the 1997 supplemental site investigation, all newly installed monitoring wells and selected existing monitoring wells were sampled. In addition to onsite monitoring well sampling, eight residential drinking water wells located on Ferry Road were also sampled as part of the supplemental site investigation.

In January 1998, the first in a series of quarterly groundwater sampling rounds was conducted as part of the Comprehensive Monitoring plan required as by HoltraChem's Consent Agreement with MEDEP. This sampling incorporated residential wells located on Ferry Road, and six monitoring

wells located between the plant and Ferry Road. Additional groundwater detection monitoring rounds were conducted in April and July of 1998.

3.4.1 Water Level Measurement

Objectives

Monitoring of water levels was performed in the fall, winter and spring of 1995 and in the fall of 1997 to characterize groundwater flow directions and velocities during seasonal high and low groundwater levels. Groundwater sampling was performed in the spring of 1995 to coincide with seasonal high water levels and in the early winter of 1997 to coincide with seasonal low water levels.

Methodology

The depth to water in monitoring wells was measured using a electronic water level indicator with calibrated tape (0.01 foot increments).

Comprehensive monitoring of water levels was performed on October 5, 1994, January 30, 1995, May 1, 1995, September 29, 1997 and December 3, 1997. Water levels were collected during 1998 Comprehensive Monitoring Plan groundwater sampling. Because this was not a comprehensive round of all monitoring wells, the data have not been tabulated for the SI. However, water level summaries will be presented with the CMP data.

Variances from Approved Work Plans - 1995

The round of synoptic groundwater level elevations collected in January 1995 was not anticipated in the December 1993 Work Plan. The January round was added to acquire data on winter hydrogeologic conditions and provide water level information on an approximate quarterly basis throughout the project.

Variances from Approved Work Plans - 1997

A comprehensive round of water levels, not anticipated in the original work plan, was collected in December 1997 after installation of the SSI wells was complete.

The water supply well 8-8, which historically provided process water to the plant, was sampled in January 1998. The water level in well 8-8 was measured to evaluate the groundwater elevations east of Landfill 2. The water level in the nearby flooded gravel pit was also measured.

Observations

Comparison of water levels in existing wells with previously collected data (Acheron, 1989) indicates that the fall of 1997 was drier and water levels were lower than in previous years. Based on precipitation data, the winter and spring of 1995 were also unusually dry, and water levels were correspondingly low compared to earlier years.

Results

A summary of water level elevations collected during the site investigation is provided in **Table 3-3**. Vertical gradient calculations are provided in **Table 3-4**.

3.4.2 Tidal Cycle Monitoring

Objectives

In 1995, tidal cycle monitoring was performed to evaluate the effects of the 12 to 14 foot tidal range on monitoring well clusters adjacent to the Penobscot River. Specifically, tidal cycle monitoring was performed simultaneously in all wells of a tidally influenced cluster to evaluate the average hydraulic heads and the net vertical hydraulic gradients at those locations.

Acheron previously evaluated the tidal efficiency and delay of selected wells on the site. With the exception of two bedrock wells on the ridge, the wells affected by tides were located in the lower valley near the Penobscot River. Acheron did not measure all wells in a cluster simultaneously, however, and therefore could not determine the vertical gradients due to differing delay times in the individual wells.

During the 1997 supplemental site investigation, CDM monitored the tidal impact on the groundwater elevations in newly installed monitoring well clusters MW-505 and MW-511. These wells are located near the Penobscot River on the PERC property between the plant and the residences on Ferry Road. These data were collected to evaluate the hydraulic connection between the bedrock in this area and the Penobscot River.

In July, 1998, CDM performed tidal cycle monitoring of wells screened in the sand and gravel deposits at the base of Landfill Area 1, and simultaneously collected water level elevations from the Penobscot River. These data were collected to evaluate the hydraulic gradients between the sand and gravel deposits and the river.

Tidal cycle monitoring was repeated in October of 1998, following the installation of piezometers into the upper end of the beach face. Water levels were monitored simultaneously in wells beneath the landfill, and in the piezometers. Tidal monitoring was conducted over two tidal cycles.

Methodology

All tidal cycle monitoring was performed with the same equipment used to record hydraulic conductivity tests, the Hermit 2000 data logger and pressure transducers. The tests were set up to run over several days, measuring the water levels at five minute intervals.

Data was downloaded to a personal computer, corrected for the elevation difference between plant datum and NGVD and any discrepancies between the data logger's internal clock and the start time of the test as documented in the field book. The high and low water elevations were calculated as well as the average water level for the well. The efficiency of the well was determined by dividing the water level fluctuation by the tidal range and multiplying by 100. The delay was calculated by measuring the time elapsed between high tide in the river and the corresponding next high water level in the well. The net vertical gradient was calculated by comparing the average water level elevations over several tidal cycles between the adjacent screened intervals within the well cluster.

Tidal cycle monitoring along the base of Landfill Area 1 was conducted over three days, between July 20 and July 22, 1998. Pressure transducers and data loggers were installed in B-326-O2, MW-401-O1, MW-402-O1, MW-512-O1, and MW-513-O1. A transducer was also installed in a stilling rod in the Penobscot River, to measure tidal fluctuations in the river. The elevation of the stilling rod was surveyed from the top of PVC on MW-401-B1. The tidal test was initiated at low tide, to ensure that the transducer in the river remained underwater throughout the test.

To process the tidal cycle monitoring data, a survey of the beachface below Landfill Area 1 was conducted. This survey included measurements of the average beachface slope, and the length of the exposed shoreline below the landfill. Data from the survey were used in analyses to calculate the elevation of the seepage face exposed at low tide, and the average gradient between sand and gravel deposits and the river.

An additional round of tidal cycle monitoring was performed in October of 1998. During this monitoring round, transducers were installed in wells B-326-O2, MW-401-O1, MW-512-O1, and MW-513-O1, and in two piezometers along the beachface. The data from this tidal monitoring round were used to refine the initial estimates of head gradients below the landfill.

Variances From Approved Work Plans

Tidal cycle monitoring of the wells installed on the PERC property was not anticipated in the Work Plan but was added to evaluate the hydraulic connection of bedrock in that area to the Penobscot River.

Tidal cycle monitoring and beachface surveying to evaluate the hydraulic gradients below Landfill Area 1 were not anticipated in the Work Plan. These field measurements were added to provide the data necessary to evaluate the groundwater flux between the sand and gravel deposits and the Penobscot River.

Observations

Graphs of water level measurements and text files of associated data were generated. In some wells, the rising portion of the tidal curve was steeper than the receding curve. This shoulder in the sinusoidal curve most likely results from groundwater backing up during high tides and taking longer to drain than it takes for the wells to fill on the incoming tide. The time when this shoulder is observed most likely corresponds to the period of maximal groundwater discharge to surface water.

Results

The water level elevation curves and data from the tidal cycle monitoring are provided in Appendix E. The efficiencies and delays of the measured wells are summarized in **Table 3-5**. The net vertical hydraulic gradients are presented in **Table 3-6**.

3.4.3 Hydraulic Conductivity Testing

Objectives

In-situ hydraulic conductivity tests were performed on each newly installed monitoring well to estimate the hydraulic properties of the intervals in which they are screened. Pump tests were

performed to evaluate the connectivity of wells screened in similar intervals, or at adjacent locations.

Methodology

Hydraulic conductivity testing was performed in the fall of 1994 after monitoring well development. After initial data interpretation, selected tests were re-run in May 1995. Monitoring wells installed as part of the 1997 supplemental site investigation were tested in the fall of 1997. Typically, a falling head and a rising head slug test were performed. Falling head tests were not performed on any well where the screen interval straddles the water table.

Tests were performed using a Hermit 2000 automated data logger, down hole pressure transducer and an approximately six-foot long solid PVC slug. Four and five foot slugs were used during the 1997 testing of wells with limited water column height. Initial water levels were measured prior to installing the pressure transducer. The falling head test was started when the slug was rapidly lowered into the well, displacing water and causing an instantaneous rise of the water level in the well. The falling water levels were recorded at logarithmic time intervals by the data logger. Rising head tests were performed by withdrawing the slug and recording the rising water levels. In wells where the screen interval straddles the water table, the slug was lowered into the well and the water level allowed to equilibrate prior to withdrawing the slug and performing the rising head test.

In addition to hydraulic conductivity testing on existing monitoring wells, a pump test was performed to evaluate the degree of connectivity among monitoring well clusters MW-505, MW-511, and B-321. Pressure transducers and data loggers were used in two clusters to monitor the effects of extended pumping on a third cluster. Data from these tests were plotted to evaluate the degree of interconnectedness among these wells.

Hydraulic conductivity test data were downloaded to a personal computer and processed using spreadsheet software. Data were evaluated using the Hvorslev method for slug test analyses. Hydraulic conductivity calculations are provided in Appendix E.

Variances from Approved Work Plans - 1995

- Bedrock wells were not tested for yield during drilling. Potential hydraulically conductive zones in bedrock were identified during drilling by evaluating the bedrock cores. In-situ hydraulic conductivity tests were performed on bedrock wells after completion. In-situ testing of the completed well was selected as the best means to evaluate bedrock hydraulic conductivity because the data would be collected identically to that obtained from

overburden wells and the initial water level elevations could be accurately measured prior to testing.

Variances from Approved Work Plans - 1997

- A pumping/recovery test was conducted on monitoring well MW-508-O1 due to the very slow recharge rate. This monitoring well was pumped down approximately 20 feet and allowed to recharge while a pressure transducer and data logger recorded the rebound.
- Hydraulic conductivity testing was not conducted on monitoring wells MW-501-O1 and MW-503-O1 due to minimal water column height. However, monitoring well MW-503-O1 was continuously pumped at a rate of 1.5 gallons per minute during development, suggesting a high hydraulic conductivity unit in this well.
- Due to the oscillation of water levels observed in the hydraulic conductivity test of MW-513-O1, the hydraulic conductivity of this well was approximated using Uffink's method for oscillation tests.
- Pumping tests and tidal influence tests were conducted using pressure transducers and data loggers on both of the new bedrock PERC property monitoring well clusters and the existing B-321 cluster.

Observations

The rate of recovery was observed during testing and compared to the expected rate based on the geologist's observations of soil or rock in the screen interval. Test data was observed during recovery to ensure that the test was run long enough to provide adequate data for interpretation. In general, wells screened across coarse grained material, such as MW-402-O1, recovered quickly from the induced water level change. Wells screened in fine grained material such as MW-404-O1 and MW-508-O1 took longer to recover. Bedrock wells where few fractures were observed also had long recovery times.

Observations of the test data were made after the data was plotted. When plotted on semi-logarithmic axes, hydraulic conductivity test results should exhibit a straight line section as the well recovers to the initial static water level. In several tests, this straight line section was not clearly defined and the test results were determined to be inconclusive. These tests were re-run in the spring prior to groundwater sampling.

An oscillation response was observed in the hydraulic conductivity test for MW-513-O1. The slug test was re-run in July of 1998, with the same oscillatory response. This condition is occasionally observed in highly conductive wells, and the conductivity of this well was approximated using Uffink's method.

Results

Hydraulic conductivity values calculated from slug tests are summarized in **Table 3-7**. Graphs of the test data and calculations are provided in Appendix E.

3.4.4 Monitoring Well Sampling

Objectives

Groundwater samples were collected from new and existing monitoring wells in accordance with the Work Plan. CDM collected groundwater samples during two comprehensive rounds, one in 1995 and one in 1997. In addition, samples were collected from 26 monitoring wells during July of 1998 as part of the Comprehensive Monitoring Plan. The three most contaminated wells based on historical sampling (B-316-O1, B-326-O2, and P-2A) were sampled in 1995 for Appendix IX parameters. The remainder of the wells were analyzed for compounds and elements that have been detected at the site in previous sampling rounds.

Methodology

Groundwater samples were collected from all newly installed wells and selected existing monitoring wells during both 1995 and 1997. Series 500 monitoring wells were installed after 1995 groundwater sampling was conducted; therefore, these wells were included only in the 1997 comprehensive groundwater round. Groundwater monitoring was performed in accordance with the Work Plan, as described below.

Prior to sampling, CDM inspected existing monitoring wells to determine their suitability for sampling. All newly installed and most existing wells were then purged and sampled using a WaTerra well pump and dedicated tubing. Purging and sampling of four of the existing six-inch diameter open-hole wells installed by PELA (P-3, P-4 and P-13) were accomplished using the previously installed submersible pumps. Groundwater samples were transferred from the dedicated pump tubing directly to the sample bottles, beginning with the VOC samples. All measurements and observations were recorded. Samples were delivered to the laboratory by CDM personnel or shipped

by Federal Express. Prior to shipment, CDM's chain-of-custody (COC) form was completed according to standard procedures.

The first comprehensive groundwater sampling round was performed in May, 1995. All groundwater samples were analyzed for VOCs, dissolved metals (mercury, iron, manganese, magnesium and sodium), chloride, sulfide, sulfate, and total organic carbon (TOC). Parameters measured in the field included pH, temperature, specific conductance and dissolved oxygen. Groundwater from the three most contaminated wells, based on historical sampling and field screening (B-316-O1, B-326-O2, and P-2A), were analyzed for Appendix IX parameters.

The second comprehensive groundwater sampling round occurred during the 1997 supplemental site investigation. Groundwater sampling protocols were similar to those in 1995, with few exceptions. Dissolved mercury was added to the list of parameters to determine if mercury was attributable to sediment found in the well or if the mercury was in solution. Dissolved mercury sampling was conducted using a 0.45 micron in-line filter. A dedicated filter was attached to dedicated WaTerra tubing for each monitoring well, and the sample was filtered in the field.

During the 1997 comprehensive round, groundwater samples from four monitoring well clusters (B-303, B-304, B-306 and B-307) surrounding Landfill No. 5 were collected as split samples by CDM during Acheron's quarterly monitoring of Landfill 5.

Variations from Approved Work Plans - 1995

A method for chloropicrin analysis in water (EPA Method 551) was developed and approved after the Work Plan was written. In 1995, a separate sample, consisting of two 40-ml VOA vials, was collected from each well sampled. These samples were analyzed by the CDM laboratory for chloropicrin using EPA Method 551. Because chloropicrin is not an Appendix IX analyte, separate chloropicrin samples from the wells sampled for Appendix IX parameters were analyzed by CDM rather than by the contract laboratory which performed the remainder of the Appendix IX analyses.

Water quality field measurements were collected with a Horiba combination meter. This meter measures turbidity, salinity, dissolved oxygen, pH, temperature and specific conductance. Measurements of oxidation/reduction potential (ORP) of groundwater samples were attempted at several locations using a platinum electrode ORP probe, but the ORP probe produced erratic results. The directions included with the ORP measurement package indicated that at high levels of dissolved oxygen (>0.1 mg/l), the ORP probe is unreliable and that direct measurement of dissolved

oxygen is more useful for calculating the ORP. Therefore, no further attempts at measuring ORP with the platinum electrode were made.

Due to the very large volumes of water contained in the six-inch diameter P-series bedrock wells and their typically slow recovery rates, these wells were not purged of three times their standing well volumes. These wells were purged and sampled using their dedicated submersible pumps and were purged of at least one full well volume. When the well had recovered to approximately 75% of pre-purging water level, the well was pumped until the field measured parameters had stabilized, and the well was then sampled.

Variances from Approved Work Plans - 1997

Evaluation of the chromatograms from the 1995 groundwater sampling round indicated that chloropicrin could be detected using EPA Method 8260. Prior to the 1997 sampling round, the CDM laboratory developed the required calibrations and method detection limits and the chloropicrin analyses for onsite groundwater samples were performed using EPA Method 8260. Chloropicrin in residential well samples continued to be analyzed by the drinking water method 551 which provides lower detection limits.

In January 1998, the first of a series of quarterly detection monitoring rounds was conducted at the HoltraChem Plant. This quarterly sampling event was conducted to monitor groundwater conditions between the plant and the Ferry Road residences as required by the Consent Agreement Comprehensive Monitoring Plan. Monitoring wells B-321-B1, B-321-B2, MW-505-B1, MW-505-B2 and MW-511-B2, all located between the HoltraChem Plant and the Ferry Road residences, were sampled as part of this first round. Monitoring well MW-511-B1 was frozen, and was not sampled in the January 1998 groundwater sampling round. Two additional quarterly sampling events were conducted during April and July of 1998. These sampling events, conducted in conjunction with residential sampling, included the above list in addition to MW-511-B1.

Water supply well 8-8, which once served as a supply well for plant process water, was sampled during January 1998. This well was selected for sampling because its groundwater elevation is slightly below that in the vicinity of Landfill No. 2 (MW-409-O1).

In July of 1998, 26 monitoring wells were sampled as part of the Comprehensive Monitoring Plan (Acheron, April 1998) required as part of the Consent Agreement with MEDEP. The wells included in this groundwater monitoring were selected to characterize quarterly trends in groundwater contamination, and to provide data to delineate the extent of contamination around specific

operational areas of the plant. CDM performed the first round of sampling for this Comprehensive Monitoring Plan. Additional quarterly sampling rounds will be conducted by Acheron.

Observations

Due to the very fine grained composition of the soil in some screened intervals, some wells did not "clean up" to below 50 NTUs. CDM installed wells were developed and purged until further development would not produce lower turbidity levels, and then sampled. Additional development was not performed on the Acheron or PELA installed wells. Turbidity measurements are provided with the field measured analytical parameter results.

Results

1995 analytical results for the standard inorganic and conventional parameter list are summarized in **Table 3-8**. A summary of detected volatile organic compounds for this sampling round is presented in **Table 3-9**. A compilation of all VOC results is provided with the analytical data sheets in Appendix G. A summary of the detected Appendix IX parameters is presented in **Table 3-10**. A compilation of all Appendix IX analyses is provided in Appendix G. Results of 1995 confirmation sampling, which are discussed in the following section, are found in **Table 3-11**.

1997 analytical data are presented in **Tables 3-12** and **3-13**, listing inorganic and conventional parameters, and volatile organic compounds, respectively.

Results from the 1998 Comprehensive Monitoring Plan sampling, listing inorganic and conventional parameters and VOCs, respectively, are included in **Tables 3-14** and **3-15**.

3.4.5 Confirmation Sampling

Objectives

Based on the results of the round of groundwater samples collected in May 1995, selected wells were resampled to confirm initial sampling results, to obtain lower turbidity samples, or to provide for better comparisons with previous sampling rounds. The specific objectives and rationale for resampling each well are provided below:

B-321-B2 Samples were collected from B-321-B2 to confirm the reported mercury concentration of 3 $\mu\text{g/l}$, which is above the MCL of 2.0 $\mu\text{g/l}$. Samples were collected

for filtered (dissolved) and non-filtered (total) mercury analyses. A second non-filtered confirmation sample was collected using low flow purge and sample techniques.

During September of 1998, a sample was collected from B-321-B2 to evaluate the influence of the southerly stream on groundwater quality in this well. The well was sampled after four days of dye releases into the stream, which had loaded this surface water body with visible concentrations of dye. The well was purged using standard methods outlined in the Work Plan, and a sample was collected for fluorometer analysis.

- B-321-B1 This well was sampled as part of the B-321 cluster, but only during the initial confirmation round.
- B-316-O1 This well was sampled using the low flow purge and sample technique because elevated concentrations of several metals were detected in the groundwater sample submitted for Appendix IX analysis. The turbidity of the initial sample was 999 NTUs, indicating that the value was above the measurement capacity of the instrument. This well is completed in till and sampling is difficult due to its slow recharge.
- B-326-O2 Until the 1997 sampling round, this well historically had the highest mercury concentrations of any well on the HoltraChem property. Both filtered and non-filtered samples were collected for mercury analysis during confirmation sampling to evaluate whether the mercury present was in a dissolved form and to permit comparison of the 1995 data to the filtered samples collected by Acheron in 1990.
- MW-410-B1 This well was resampled to confirm the initial analytical results because it had higher organic and inorganic concentrations than the other newly installed and existing wells on the bedrock ridge.

Methodology

The confirmation round samples collected from B-321-B1, B-321-B2, B-326-O2, and MW-410-B1 were collected using the previously installed dedicated tubing and WaTerra inertial pumps. Groundwater sampling proceeded identically to the initial SI sampling round except that at B-321-B1, B-321-B2 and B-326-O2 both filtered and non-filtered samples were collected for mercury

analysis. Filtering was accomplished using a 0.45 micron in-line filter and discharging the filtered water directly to the pre-preserved sample bottles.

The second confirmation sample from B-321-B2 and the confirmation sample collected from B-316-O1 were collected using the low flow purge and sample techniques. Because the internal casing of both of these wells is 1.5-inch ID PVC, a submersible pump with a sufficiently small OD was not available. Purging and sampling were performed using a 1.0-inch QED pneumatic bladder pump.

The water level in the well was measured prior to pulling the existing WaTerra tubing and again prior to sampling. Tubing was pulled approximately 12 hours prior to purging and sampling each well. Tubing removed from the well was not allowed to touch the ground, stored in plastic bags and later returned to the well. The pump was installed a minimum of 5 feet above the bottom of the well to prevent disturbing any accumulated sediment.

Bladder pumps operate by intermittently inflating a bladder which forces well water past a ball valve up the discharge tubing. The rate of pumping is determined by the frequency of the bladder inflations and the time require to refill the sample chamber. The flow rate was adjusted to obtain the maximum pumping rate while maintaining a constant water level elevation in the well, and never pumping greater than one liter per minute. Due to the very slow recharge observed at B-316-O1, the pump would occasionally be turned off to allow sufficient water to refill the well and permit purging and sampling.

Each well was purged until the field measured parameters stabilized. Slightly more than three well volumes were removed from B-321-B2. Approximately two well volumes were removed from B-316-O1 due to very slow recharge.

In 1998, sampling of B-321-B2 for fluorometer analysis was conducted using the procedures outlined in the Work Plan. Three well volumes were purged from the well before the sample was collected.

Variations from Approved Work Plans

Confirmation sampling was not proposed in the Work Plan, but was added to provide additional data with which to interpret the SI groundwater analytical results. Low flow purging and sampling was discussed with EPA prior to its selective use at the site.

Observations

The bedrock groundwater samples collected from B-321-B2 were turbid, which is unusual for a properly constructed bedrock monitoring well. This observation, and the difference between the filtered and non-filtered sample analytical results (dissolved and total, respectively) indicate that the mercury detected in the samples obtained from B-321-B2 is associated with suspended sediment. These observations prompted the collection of the second confirmation sample from this well using low flow purge and sample techniques. The sample collected with this technique was not turbid and the non-detectable mercury result confirmed the observation that the mercury detected in this well is not in a soluble form.

An approximate 6-inch accumulation of sediment was observed above the stainless steel inertial check valve when the tubing was removed from B-316-O1. Due to the very slow recharge rate of this well, it was not possible to maintain a constant water level during pumping. Instead, the pump would be operated at its lowest rate until no more water was discharging. Then the pump would be turned off and the well allowed to recharge. Even using the low flow purge and sample technique, the turbidity of the water from this well remained above 100 NTUs after the standing water had been removed. The turbidity at the time of confirmation sampling was 101 NTUs.

Results

A summary of 1995 confirmation sampling results, including comparisons to the initial SI sample analytical results, is presented in Table 3-11.

The initial SI sampling round was the first time mercury was detected in B-321-B2. During the second round, both filtered and non-filtered samples were collected to evaluate the effect of elevated turbidity observed in this bedrock well and to provide for direct comparison with Acheron data which was field filtered. When the filtered sample contained non-detectable levels of mercury and the non-filtered sample contained 4 $\mu\text{g/l}$, a decision was made to collect one more sample using low flow purge and sample techniques. This technique is considered to provide samples which are most representative of what may migrate with groundwater. The low flow purge and sample technique resulted in a non-detectable concentration of mercury in a non-filtered sample from this well.

Fluorometer analysis of well B-321-B2 indicated that dye was present in groundwater from this well after four days of dye loading. Dye was found at a concentration of 65 $\mu\text{g/l}$, compared to 6 $\mu\text{g/l}$ in a background sample collected from MW-505-B2. This result indicates that surface water is recharging to shallow bedrock groundwater at this location.

Mercury was not detected in either the filtered or non-filtered sample collected from B-321-B1.

Low flow purge and sampling of B-316-O1 showed all of the previously detected metal concentrations below detection limits with the exception of mercury and zinc. Low flow purge and sampling technique obtains values considered more representative of groundwater conditions because metals sorbed to or present in the sediment do not go into solution in the presence of the nitric acid preservative.

The 1600 $\mu\text{g/l}$ total v. 1500 $\mu\text{g/l}$ dissolved concentrations from B-326-O2 indicate that the mercury is in a soluble form and that the concentrations are lower than they were in 1990.

The second round of analyses from MW-410-B1 confirmed the results of the first for most of the analytes detected.

3.4.6 Residential Well Sampling

Objectives

During the 1995 initial site investigation, four residential wells were sampled to provide current data on the water quality of the residences of Ferry Road. During the 1997 supplemental site investigation, the number of Ferry Road residential wells sampled was increased to eight as additional residents expressed interest in having their wells sampled. Residential well sampling was added to a new detection monitoring program, which included between six and eight residential wells and three well clusters (MW-505, MW-511, and B-321) on the PERC property. Quarterly sampling of the residential wells began in January of 1998 as part of this detection monitoring program required by the Consent Agreement Comprehensive Monitoring Plan.

Methodology

Prior to the 1995 investigation, four Ferry Road residences were sampled in 1989 by Acheron and EPA. The 1989 and 1995 rounds of residential well sampling included the DeSanctis and Hazeltine wells, which are the two residences closest to the plant, and the Danforth and Judd properties, which have the deepest and shallowest wells, respectively, of the residences along Ferry Road.

During the 1995 residential well sampling, where possible, the depth to water from the ground surface was measured prior to running the kitchen tap. Water samples were collected after the tap was allowed to run for approximately 10 minutes. 1995 drinking water samples were analyzed for

total mercury. Field parameters were measured and samples for mercury analyses were collected in one liter plastic bottles pre-preserved with nitric acid. A second sample was collected for mercury analysis by the HoltraChem plant. The samples were not filtered.

During the 1997 supplemental site investigation, a total of eight drinking water wells were sampled. Four drinking water wells, Scribner, Jeffery, Staples and Healey, were sampled in addition to the four drinking water wells sampled in 1995. Techlaw (formerly A.T. Kearney), representing EPA, was present and collected split samples from selected drinking water wells.

Analyses conducted on residential well samples in 1997 included the full parameter list conducted on-site at the HoltraChem Plant. Field parameters were recorded and samples were collected for total metals including mercury, VOCs, chloropicrin, and major cations and anions.

The first of a series of quarterly groundwater sampling rounds was conducted in January 1998, which included seven drinking water wells. The wells sampled included five previously sampled wells (Danforth, Judd, Hazeltine, DeSanctis and Jeffery) and two additional wells (Bennett and Norlen's Water Treatment). Wells sampled as part of the January 1998 quarterly round were sampled for the same parameters as in 1997.

Variances from Approved Work Plans

Residential sampling was not included in the Work Plan. It was discussed with EPA prior to sampling and noted in Bimonthly Progress Report No. 8.

The quarterly sampling rounds conducted in January, April and July of 1998 were not included in the initial Work Plan. These sampling rounds were added to monitor groundwater conditions in the Ferry Road residential wells, in conjunction with the detection monitoring program, which is part of HoltraChem's Comprehensive Monitoring Program. Two wells were added to those that had been sampled in the past. One resident (Bennett) asked to have their well tested after an offer from HoltraChem. A sample was collected from Norlen's Water Treatment as a background sample.

Observations

In 1995, the inside of several surface casings were noted to be rusted. A splash was noted when the well cap was replaced on the Judd well. Approximately seven minutes after the tap was turned on, slightly iron stained water and traces of rust were observed in the water.

Mr. Fred Phillips met the samplers at the DeSanctis property in 1995. He had already removed the well cap and reported that a rock may have dropped into the well when he did. He also mentioned that he had hit the top of the well with a snow plow during the winter dislodging the top. Mr. Phillips mentioned that the pump in the DeSanctis well is set very near the bottom of the well and several times during the year when they use more than their usual amount of water, the well will produce slightly red, somewhat turbid water. During sampling, the water produced from the DeSanctis well was observed to be quite turbid and red. Mr. Phillips noted that the water sample collected was more turbid than he had observed before.

Results

The results of residential well sampling are presented in **Table 3-16**. Mercury was not detected in any of the residential samples during any of the sampling rounds. A trace of methyl tertiary-butyl ether (MTBE), a common gasoline additive that is highly mobile in groundwater, was detected in a residential well near Rte 15. Otherwise, VOCs and chloropicrin have not been detected in the residential or background wells sampled. The presence of MTBE in a well upgradient and cross-gradient from the site is not considered to be site related.

3.4.7 Seep Sampling

Objectives

Seeps often occur at the surface or discharge point of preferential bedrock flow pathways. Because seeps represent a surface expression of groundwater flow, they can be a potential exposure point to contaminants in groundwater and a particularly relevant place to sample. Seep sampling was performed in accordance with the Work Plan. Seep sample analytical results have been incorporated into the human health and ecological risk assessments. Seep samples were collected by Acheron in 1998 to assess the potential for groundwater discharges to impact the Penobscot River and the dye study surface water sampling results.

Methodology

Seeps were located in 1995 during a site reconnaissance of the shoulders and base of the bedrock ridge. No seeps were observed along the southern side of the bedrock ridge, facing the plant area or Landfill 2. Five seeps were observed near the shoreline of the Penobscot River on the north side of the ridge. A sixth seep was observed in the southern cove during river sediment sampling. This seep

was designated SSP-007 due to a blind duplicate being labeled SSP-006 during initial seep sampling. Seep sample locations are presented on **Figure 3-3**.

Seeps were sampled using one-to-two foot lengths of polyethylene tubing which was placed into the water flow to cause the water to flow off the rock or ground surface and into the sample glassware. Seep SSP-007 flowed onto a sand and gravel bank, and was sampled by dipping laboratory cleaned glassware into the flow until sufficient sample volume was obtained.

Each seep except SSP-007 was sampled for total mercury and VOCs. In 1995, sample SSP-007 was analyzed for mercury only. One duplicate and one field equipment blank were also collected. The equipment blank was obtained by pouring DI water through a one-to-two foot length of polyethylene tubing into the sample glassware. A sample of SSP-007 was also collected during the 1997 SSI sampling. This sample was analyzed for the same parameters as the groundwater samples.

During the surface water and dye sampling conducted by Acheron in September of 1998, seep SSP-007 was sampled to evaluate the source of water discharging at this location. After four days of dye discharges to the southerly stream, a sample was collected from this seep to determine whether groundwater here was being recharged by the southerly stream.

During September of 1998, three surface seeps were also sampled along the exposed beach between the HoltraChem outfall and the bedrock ridge. These seeps were selected for sampling based on a greater observed discharge of bank storage at these locations in comparison with other beach face discharge in the vicinity. Water from these seeps was impounded behind wooden dikes to allow sediments to settle out. Water was then extracted from the impounded water using an acid-washed sampling tube and clean sampling techniques. The samples were sent to Frontier Geosciences for analysis.

Variations from Approved Work Plans

The exact location of seeps could not be determined until the field reconnaissance was performed at the time of sampling. A seep which had been noted by PELA above Landfill 2 and was identified for sampling (SSP-005) in the Work Plan was not observed during field sampling; therefore no sample was obtained. An additional seep observed at a bedrock outcrop in the southern cove during sediment sampling was later sampled for mercury in 1995, for the full suite of groundwater parameters in 1997, and for dye in 1998. Dye testing was required as part of HoltraChem's Consent Agreement Comprehensive Monitoring Plan, therefore sampling of seeps identified during the 1998 dye testing was not anticipated in the original workplan or in the proposed Penobscot River studies.

These samples were collected to assess the potential for groundwater discharge to impact the Penobscot River and dye study surface water sampling results.

Observations

Seeps were characterized as wet areas on the rock face where water was observed to be dripping or trickling. Typically an increase in hydrophilic vegetation was observed. Some portion of the water discharging at the seeps may have come from overburden deposits.

Seep SSP-001 is located approximately 100 yards up river from the drainage swale which extends from Landfills 4 and 5 to the Penobscot River. This location is downgradient from monitoring wells MW-408-O1 and MW-408-B1. The seep was discharging just above the high tide line over a bed of sand and gravel and was surrounded by moss and other hydrophilic vegetation.

Seep SSP-002 is located approximately 20 feet up river from the drainage swale which extends from between Landfills 4 and 5 to the Penobscot River. This seep was also discharging just above the high tide line over a bed of sand and gravel and was surrounded by moss and other hydrophilic vegetation.

Seep SSP-003 is located within the drainage swale which extends from between Landfills 4 and 5 to the Penobscot River approximately 40 feet above mean sea level. This location is down gradient of monitoring well MW-411. It is located beneath an overturned tree stump and is surrounded by hydrophilic vegetation. This seep was flowing over a bed of sand and gravel.

Seep SSP-004 is located approximately 300 feet down river of the drainage swale which extends from between Landfills 4 and 5 to the Penobscot River. This location is approximately down gradient of soil boring B-412 and from monitoring wells B-309-O1, B-309-B1 and P-13. This seep appears to discharge primarily through foliations in the bedrock although some water may trickle down from the overlying soil. Based on the location of this seep and the lack of other observable seeps in the vicinity, seep SSP-004 is interpreted to be the seep where measurable concentrations of mercury and VOCs were previously detected by Acheron.

Seep SSP-005 is located approximately 400 feet down river from seep SSP-004. Seep SSP-005 is located approximately 200 feet down river of a perpendicular from the river to monitoring wells MW-406-O1 and MW-406-B1. Water was observed flowing from the rock surface and from very thin overburden soils present above the seep. Seep sample SSP-006 is a duplicate of SSP-004.

Seep SSP-007 is located in a bedrock outcrop near where the southern drainage ditch enters the southern cove. This seep is better characterized as a spring, and was first observed during sediment sampling. It was later sampled for mercury only because VOCs had never been detected in the nearest monitoring well. The spring is covered by the river at high tide but was observed to flow during all phases of the falling and rising tide.

Seep SSP-008 is located along the beach exposed at low tide, below and slightly downriver of Landfill Area 1. This seep was one of the three areas of groundwater discharge observed during the reconnaissance of the beachface in late August of 1998.

Seep SSP-009 is located downriver from SSP-008, below the middle pumphouse from the former supply wells below the plant. Groundwater discharge at this seep was not as much as at SSP-008, but sufficient water was impounded behind the wooden dam to collect a relatively clear sample.

Seep SSP-010 is located directly adjacent to the HoltraChem outfall pipe. Groundwater at this seep was observed to be flowing at a rate comparable to that of the southerly stream; because it could not be fully impounded, however, a quantitative measurement of groundwater discharge was not possible.

Results

The results of seep sampling are presented on **Table 3-17**. Mercury was not detected in any of the seep samples during 1995. Mercury was detected at a concentration of 0.22 $\mu\text{g/l}$ from seep SSP-007 during the 1997 sampling round. Carbon tetrachloride was detected at a concentration of 2.1 $\mu\text{g/l}$ in seep SSP-003, which flows in the gully north of Landfills 4 and 5.

In an effort to establish the source of mercury present in monitoring well B321-B2 and in seep SSP-007, a sample was collected from this seep following four days of dye loading to the southerly stream. Water collected from this seep had a dye concentration of 7,000 $\mu\text{g/l}$ and a visibly pink tint, indicating significant surface water recharge to this seep.

3.4.8 Lysimeter Sampling

Objectives

The purpose of the soil lysimeters was to assess the migration potential of soil mercury, and to evaluate the potential impacts of percolate on groundwater quality. Surficial soil data was used to

target three locations for lysimeter installation. Lysimeters were installed in the sidewalls of test pits adjacent to soil samples SSS-024, SSS-053, and SSS-035 (Figure 3-5), representing high, medium, and low detected soil mercury concentrations, respectively. Percolate was collected from the lysimeters and analyzed for total and dissolved mercury content.

Methodology

Test pits were excavated within 3 feet of associated soil sampling locations. Pits were dug to a maximum depth of approximately 3 feet, and soil lithologies were described from the undisturbed soil column. After soil descriptions were completed, a shallow pan lysimeter was driven into the side of each excavation. In two of the three test pits (SSS-035 and SSS-053), the soil was too hard to allow full penetration of the lysimeter into the soil. In these cases, plastic sheeting was used to cover the exposed portion of the lysimeter to ensure that rainwater did not enter the sample bottles directly. The soil at location SSS-024 was loose enough to allow the lysimeter to be driven fully into the wall, ensuring that all water entering the sample bottle came directly from infiltration through the soil.

The lysimeters used for this study have a capture area of approximately 1.5 sq ft., and each unit has a metal screen to support the soil above the lysimeter. These lysimeters are 2 inches deep at one end, and tapered toward the front to provide a sharp edge for installation. A metal sampling port protrudes from one corner of the lysimeter and the units were installed to induce flow to this corner. Teflon lined tubing was extended from the sample port to an empty sample bottle. All connections were sealed with Teflon tape and secured with duct tape.

Sample bottles were initially left in place to allow natural infiltration from rainfall. After approximately three weeks, CDM returned to the lysimeters to inspect the bottles and found that no natural infiltration had entered the sample bottles. The test pit excavated at SSS-024, however, had completely filled with water during this period, making this sample bottle inaccessible and the sample unrepresentative. Two 5-gallon buckets of natural rainwater had been collected from the site between the time of lysimeter installation and the first inspection. Water from these buckets was analyzed at HoltraChem's onsite laboratory to ensure that no mercury was present in the rainwater. This rainwater was impounded above the lysimeters at sampling locations SSS-035 and SSS-053 in an effort to induce infiltration. Two weeks later, CDM returned to the lysimeters and found one full sample bottle at SSS-053. Following the spring thaw, the bottle at SSS-035 had also been filled. These samples were collected and sent to CDM lab for analysis.

To ensure that mercury was not introduced into the water sample from associated soil particles, samples were decanted in the laboratory before being preserved.

Variances from Approved Work Plans - 1997

Installation of soil lysimeters was not proposed in the Work Plan but was added in a letter from HoltraChem to MEDEP dated October 17, 1997. This letter outlined the standards to be followed and the general sampling locations. Sampling was performed in accordance with ASTM standard #D4696-92, "Standard Guide for Pore-Liquid Sampling from the Vadose Zone."

Observations

The test pit installed at soil sampling location SSS-024 became flooded during the first two weeks of sample collection. Because the sample bottle became inundated by the rising water table, this sample was deemed unrepresentative of flow-through pore water.

Results

The water sample from the SSS-053 lysimeter had a total mercury concentration of 2.0 $\mu\text{g/l}$. The dissolved mercury concentration for this sample was below the detection limit of 0.2 $\mu\text{g/l}$. The mercury concentration in the soil water sample from the SSS-035 lysimeter was below the detection limit of 0.2 $\mu\text{g/l}$. Because the total mercury concentration was below the detection limit, a filtered sample was not analyzed at this location.

3.5 Site Soil Sampling

Soil sampling was performed to characterize the distribution of mercury and VOCs, where applicable, in surficial and near-surface soils. Surficial soil sampling during 1995 included twelve in-plant soil samples and six samples from outside the plant. During 1997, samples were collected from 50 locations sitewide. Three soil sampling locations were revisited during 1998 to collect samples for desorption testing.

In addition to surficial soil samples, near-surface soil samples were collected from test pits onsite and from geoprobe locations within and surrounding leach field Nos. 1, 2, and 8.

3.5.1 Sitewide Soil Sampling

Objectives

The purpose of soil sampling was to evaluate the concentration and distribution of mercury and, in

selected areas, VOCs or cyanide in the site soils. The effects of past surface spills and releases, and deposition of contaminants were investigated by soil sampling near known spill locations, and at selected locations throughout the plant operations area. Sitewide soil samples were collected to assess the effects of air deposition of mercury. Soil sampling locations complemented the previous work completed by Acheron and EPA. Data from soil sampling have been used to evaluate human health and ecological risks.

Methodology

Soil samples were collected within the site plant operations area, sitewide and in background locations in 1995 and 1997. In 1995, soil samples were collected from twelve areas within the plant (SSS-001 through SSS-012) and from six areas outside of the plant (SSS-013 through SSS-019). In 1997, soil samples were collected from 50 locations sitewide (SSS-021 through SSS-070). Sampling locations SSS-025, SSS-053, and SSS-065 were revisited in 1998 for desorption testing. Sitewide soil sample locations from the 1995 sampling are shown on **Figures 3-3 and 3-4**. Soil sampling locations from the 1997 sampling are shown on **Figures 3-5 through 3-9**.

1995 In-Plant Soil Sampling

Soil sample locations within the plant are shown on Figure 3-4. These locations were selected based on their proximity to manufacturing operations which handle hazardous materials, areas of known spills, and areas in which previous soil sampling detected contamination. Soil sample locations were located, and relocated as required, based on field observations of the proposed sampling area and discussions with plant personnel. At each of these areas, between one and three individual soil sample locations were identified. Soil samples were collected from two depth intervals 0 to 6 inches, and 12 to 18 inches. All samples were analyzed for total mercury, and soil samples collected at areas SSS-009, SSS-010, and SSS-011 from the 12 to 18-inch interval were analyzed for VOCs and chloropicrin. The locations adjacent to the salt storage pad were revisited to collect samples for cyanide analysis. Cyanide sampling was performed due to Maine DEP's concern that cyanide may have been used as an anti-caking agent in salt.

Based on the results of initial soil sampling, selected areas were revisited to further delineate the extent of contamination as required by the Work Plan. CDM collected additional samples from the areas where greater than 20 mg/kg of mercury was detected in the initial round. CDM sampled 20 feet away from the target initial locations in up to four directions. However, it was often not possible to move in four directions due to overlap of areas and/or the physical restrictions at the site (buildings, tanks, etc.).

Decontamination (decon) of sampling equipment occurred at the beginning of each day, and during the day, as sampling equipment (stainless steel mixing bowls, trowels, and augers, asphalt chisel spades, and auger handles, etc.) were used. Decon procedures occurred on separate plastic sheeting at the decon pad constructed for the well installation program. The decon procedures for the sampling equipment were as follows: scrub with detergent (Alconox) and water, water rinse, 1% nitric rinse, distilled (DI) water rinse, methanol rinse, final DI water rinse, and wrap in aluminum foil.

When the sample location was located in an area covered by asphalt, the rotary hammer drill was used to cut and remove the asphalt layer. The rotary hammer chisel spades were decontaminated in the same manner as other sampling equipment to prevent cross-contamination between sample locations. To ensure no cross-contamination of soils and sampling augers between the 0 to 6 inch and 12 to 18-inch sample intervals at each location, the sample augers were also changed between depth intervals. Since CDM suspected that mercury concentrations would decrease at depth, changing the augers reduced the possibility of transferring mercury contamination from the shallow sample interval to the lower sample interval. These depth in the auger hole to the sample interval was measured from existing grade at each sample location. At locations where asphalt was present (generally 2"-3"), or where there was gravel surface covering (generally 1"-2"), these non-soil materials were not included in the 0 to 6-inch sample interval. In these instances, where surficial non-soil materials were present, the depth of the 0 to 6-inch sample interval was extended up to 8 inches below grade, as necessary, to gather enough sample media.

Mercury and VOCs samples were collected in 125 ml. glass jars with Teflon-lined caps. Samples for each were tightly packed and filled to the lip of the bottle to prevent volatilization of the contaminants. A description of the soil sample was recorded at each location and depth interval. Soil samples were collected over the specified depth interval and were thoroughly mixed in a bowl with a trowel. The soil composite was then transferred in the sample bottle, sealed, and placed in a zip-lock bag. Soil samples collected for VOC analyses at locations SSS-009, SSS-010, and SSS-011 from the 12 to 18-inch depth interval were collected directly from the flights and barrel of the auger to limit the potential for volatilization.

Following collection of the soil sample, the auger hole was backfilled with the remaining excavated material, and the sample location was measured by steel tape from known locations within the plant yard. Finally, a photograph was taken of each soil sample location.

A total of 6 duplicates were collected (five for mercury, one for VOCs), and a total of five field (equipment) blanks were collected during initial soil sampling. A trip blank for VOCs and for

mercury was also collected. Aqueous trip blanks for mercury and VOCs analyses were collected on August 15, 1994, prior to any soil sample collection. These trip blanks remained in the sample coolers and field vehicle over the program duration, and delivered to the laboratory with the site soil samples. A total of three duplicates and three equipment blanks were collected during the second round of soil sampling. Equipment blanks were collected at the sample area prior to coring the soil locations and compositing the soil sample in the mixing bowl. For each equipment blank, the aluminum foil was removed from the soil auger, hand trowel and mixing bowl; U.S.P. distilled water was poured over the auger and trowel, into the mixing bowl, and transferred from the mixing bowl into a 1 liter sample bottle for aqueous mercury analysis.

1997 Sitewide Soil Sampling

A total of 50 soil samples (SSS-021 through SSS-070) were collected then analyzed for mercury by the CDM laboratory during the initial round of soil sampling in August and October 1997. The 1997 soil sample locations are shown in Figure 3-5. Samples were collected from 43 locations on the HoltraChem property (4 within the plant operations area and 39 samples sitewide outside of the operations area), 4 locations on PERC property (SSS-060 through SSS-063), and 3 background locations in Dorothea Dix Park (SSS-054 through SSS-056).

The samples were collected from surficial soils (0.0-0.5 feet) using a shovel and trowel. To obtain representative samples and reduce the potential for cross-contamination at sampling locations, holes were dug with a shovel and the sample was obtained from soil which did not contact the shovel with a cleaned stainless steel trowel. The samples were composited from five grab samples in a box formation approximately 10 feet on a side. The grab samples from the corners and the center of the box were composited in a stainless steel bowl with a stainless steel trowel or spoon and transferred to the laboratory glassware.

Three grab soil samples (SSS-071 through SSS-073) were collected in the vicinity of the fiberglass repair building then analyzed for VOCs by the CDM laboratory. The samples were obtained from 1.0 to 1.5 feet with a hand auger and transferred to the laboratory glassware with a stainless steel trowel. The hand auger holes were screened for VOCs with a photoionization detector immediately after sample collection.

Four grab samples (SSS-064 through SSS-067) were collected from the sand blast grit area on site and analyzed for the RCRA 8 regulated metals: arsenic, barium, cadmium, chromium, lead, mercury, selenium, and silver. These samples were collected from the surface to depths ranging

from 0.3 to 0.5 feet, due to the soil compaction and presence of cobbles. The samples were removed with a shovel and the soil transferred to laboratory glassware with stainless steel trowels.

A total of six duplicate soil samples and two equipment rinsate blanks were collected for mercury analysis during the soil sampling.

Based on the results of initial sitewide soil sampling, locations SSS-024-01 (19 mg/kg) and SSS-025-01 (52 mg/kg) were revisited to further delineate the extent of elevated mercury concentrations in these areas. Grab soil samples were collected at these supplemental locations. The samples were analyzed for mercury by immunoassay and by HoltraChem's on-site laboratory. Delineation samples SSS-024-02 through SSS-024-05, and SSS-025-02 through SSS-025-05 were collected 12 feet away from the initial sampling locations in four directions, at a depth of 0.0 to 0.5 feet. Samples SSS-024-06, SSS-024-07, SSS-025-06, and SSS-025-07 were collected 3 feet away from the initial sampling locations, at a depth of 1.0 to 1.5 feet.

The results of the first round of delineation sampling suggested the need for additional soil sampling at both SSS-024 and SSS-025. Samples for this round were collected based on CDM's interpretation of potential source areas for mercury contamination, with the intent of determining the maximum extent of contaminated soils in these areas. Samples SSS-024-12, SSS-024-14 through SSS-024-18, and SSS-025-11 through SSS-025-15 were collected from a depth of 0.0 to 0.5 feet. Samples labeled SSS-024-11 and SSS-024-13 were collected from a depth of 1.0 to 1.5 feet. The locations of the SSS-024 and SSS-025 delineation samples are shown in Figures 3-6 and 3-7, respectively.

Soil sampling locations SSS-065 and SSS-067 in the sand blast grit area were also revisited in an attempt to delineate mercury contamination in these areas. Mercury was detected in soil samples SSS-065-01 and SSS-067-01 at concentrations of 29 mg/kg and 310 mg/kg, respectively. At each of these locations, grab samples were collected from the top 1 to 2 inches of soil, on a 12 foot grid from the initial sampling location. Frozen soil throughout the sandblasting area precluded the collection of deeper samples at these locations. The locations of the SSS-065 and SSS-067 delineation samples are shown on Figures 3-8 and 3-9, respectively.

Variations from Approved Work Plans - 1995

Refusal was encountered at a few locations prior to reaching the second sample interval and a second sample was therefore not collected. Due to the physical restrictions of the site, it was not always possible to move out in four locations from each initial sampling point where greater than 20 mg/kg was detected. In addition, no additional sampling was performed after the second round. Differentiation of each spill within the brine handling areas of the plant would not be feasible.

Variances from Approved Work Plans - 1997

Due to the generally low concentrations of mercury observed and the lack of observable strata, samples of discrete vertical soil intervals were not collected. Based on comments from MEDEP, the leachability analysis was not conducted and was replaced with the collection of percolating water from the vadose zone using soil lysimeters (see Section 3.4.7).

Observations

The sampling performed in 1995 adequately identified the areas of the plant where elevated concentrations are likely and defined the general limits of those areas. Generally, the soils were described as a light brown to brown, loose coarse sandy fill with small/medium cobbles. Occasionally till was encountered. An increase of fines was often observed from the surface (0.0-0.5') to the near surface (1.0-1.5') interval.

No VOC readings (or "hits") were detected in jar headspace, auger holes or in the breathing zone during the 1995 sampling program. No VOC readings were detected from the three auger holes advanced in the vicinity of the fiberglass repair building in 1997.

Background mercury concentrations in air were generally between non-detect and 0.003 mg/m³ during the 1995 sampling program. One background concentration of 0.018 mg/m³ was measured during the collection of sample SSS-06A. However, it was still below the 0.040 mg/m³ action level established by the Health and Safety Plan.

The sitewide sampling in 1997 identified areas of elevated mercury concentrations due to spills and plant activities and due to air deposition. Soil types varied sitewide. Most sampling locations consisted of a layer of topsoil overlying brown to light brown sand and silt with varying amounts of gravel and organics. The top several inches of soils within wooded areas contained larger amounts of organic material. Bedrock was encountered at several sampling locations collected on the north side of the ridge near the bank of the Penobscot River at depths of 2 to 5 inches below ground surface.

Results

A summary of 1995 soil sample analytical results for mercury, pH and cyanide is presented in **Table 3-18**. A summary of the mercury concentrations detected in the sitewide soil samples collected in 1997 is presented in **Table 3-19**. Results of the volatile organic analysis of soil samples are

presented in **Table 3-20**. Delineation sample results collected at samples SSS-024, 025, 065, and 067 are summarized in **Table 3-21**. The results of the four samples analyzed for RCRA 8 metals are summarized in **Table 3-22**.

Chloropicrin is not a standard RCRA target compound; therefore chloropicrin was analyzed as a volatile analyte by a modification to EPA Method 8240. Calibration was performed at four levels. Chloropicrin data should be considered estimated values. The reporting limit for chloropicrin was generally 13 $\mu\text{g}/\text{kg}$ dry. Other than chloropicrin at two locations, no VOCs were detected in site soils.

No VOCs were detected during headspace screening or laboratory analysis of the three hand auger soil borings adjacent to the fiberglass repair building.

3.5.2 Desorption Testing

Objectives

Batch desorption tests were conducted on surficial soil samples collected from the HoltraChem Manufacturing site to evaluate the leachability of mercury in site soils. The desorption tests were designed to develop a desorption isotherm for mercury in the soils. Desorption isotherms are a graphical representation of the partitioning of a contaminant between soil and water at equilibrium. Data on the leachability of mercury in site soils was used to evaluate proposed preliminary media protection standards for soil.

Methodology

Data for the desorption isotherm for mercury was obtained by adding varying amounts of soil containing mercury to water and allowing the mixture to equilibrate. The liquid phase was then analyzed for mercury. The results indicate the amount of mercury liberated from the soil at various dilutions, and thus the potential for mercury to be released to the groundwater from the soil and conversely the amount of mercury retaining capacity of that soil.

Soil samples were collected on July 29, 1998 from areas of the HoltraChem site where elevated mercury concentrations had previously been detected. Each of the samples was analyzed for total mercury by the CDM Laboratory. Soil samples SSS-071 and SSS-072 were also analyzed for total organic carbon. Sample SSS-071, collected from the vicinity of the weir and the pumphouse, contained mercury at 57 mg/kg. Sample SSS-072 was collected from the landfill ridge and

contained mercury at 13 mg/kg. The locations of samples SSS-071 and SSS-072 were chosen because the mercury contamination in these areas is attributed to a spill, and air deposition/mercury contaminated fill, respectively. Samples SSS-071 and SSS-072 were collected in approximately the same location as samples SSS-025 and SSS-053, respectively (see Figure 3-5). The samples were collected from approximately 0 to 2 inches below ground surface. Samples were homogenized prior to sampling using stainless steel bowls and trowels.

Rainwater collected from the Frontier Cablevision building located in the city of Bangor, Maine was used as the extraction fluid. To begin the desorption tests, 100 ml of rainwater was distributed into 250 ml plastic bottles. Five different weights (0.1, 1, 5, 10, and 25 grams) of the 13 mg/kg (SSS-72) and 57 mg/kg (SSS-71) mercury soils were added to the bottles. Duplicate bottles were prepared for each of the dilutions and saved for future analysis, if required. Two triplicate samples were made. One blank bottle (no soil added) was prepared. The 1:11 dilution (plus 2 triplicate) bottles were shaken for 26 hours, then removed from the shaker and filtered under a pressure of 15 to 25 inches Hg through a 0.45 micron filter. The filtrate was collected and preserved with nitric acid. The aqueous samples were submitted to the CDM Laboratory for mercury analysis, beginning with the sample containing 25 grams of 57 mg/kg soil and proceeding to the lesser soil weight and concentration dilutions if mercury was detected. The blank sample was analyzed for Hg, pH, and total dissolved solids (TDS).

Variances from Approved Work Plans

The soil desorption testing was not included in the work plans but was described in the letter to EPA and MEDEP dated July 2, 1998 stating that batch desorption tests would be performed on three samples: one from the landfill ridge, one from the plant, and one from the lower meadow. The three samples were collected and analyzed for mercury. The sample from the plant (SSS-073) and the landfill ridge (SSS-072) contained similar mercury concentrations. Therefore, batch desorption testing was performed on the samples from the ridge and the lower meadow only.

Observations

The sample dilutions with greater than 1 gram of soil were difficult to filter. Centrifuging the samples was attempted with little success. Filtering the sample with one gram of soil took over one hour. The samples had to be filtered under pressure.

Results

The results of the batch desorption tests are summarized in **Table 3-23**. Percentages of mercury desorbed ranged from 0.0 to 0.35% indicating very little mercury leaches from soils tested. Calculations and the desorption isotherms are presented in Appendix E.

3.5.3 Test Pits

Objectives

Test pits were excavated for lysimeter installation. The soils encountered were described to document the constituents of fill, if present, and native material in each pit, and to characterize the material through which infiltrating water passes. Test pits were advanced in three areas: below the plant adjacent to the northern ditch (SSS-024); along the bedrock ridge near landfill 3 (SSS-034); and on the north side of the bedrock ridge (SSS-035 and SSS-053).

Methodology

Test pits were excavated to a depth of 2 to 2.5 feet using a rubber tire mounted backhoe. After each test pit was completed, a description of the soil column was logged and lysimeters were installed into the side of the pit (see Section 3.4.7). Due to the shallow depth of bedrock, the soil section at location SSS-034 was unsuitable for the installation of a lysimeter. This test pit was described, however, to characterize the surficial soil along the bedrock ridge. Test pit logs are included in Appendix D.

Variations from Approved Work Plans

The excavation of test pits was not proposed in the Work Plan, but was included in the proposal for lysimeter installation dated October 17, 1997.

Observations

The soil encountered at SSS-034 and SSS-035 appeared to be native material. Soil at SSS-034 was shallow, with weathered bedrock fragments encountered at a depth of approximately 2 feet, and fresh bedrock at approximately 2.5 feet.

The soil at SSS-024 and SSS-053 was loose, and fragments of anthropogenic materials were encountered at each location. Broken fragments of asphalt were encountered at SSS-053, and approximately ten graphite anodes were observed at SSS-024. These observations indicate that filling or reworking of material has occurred at these locations.

3.5.4 Leach Field Investigations

Objectives

Investigations of former leach fields Nos. 2 and 8 at HoltraChem Manufacturing were conducted on August 17 and 18, 1998, respectively. Soil samples were collected and analyzed for mercury to evaluate their potential to be sources for groundwater contamination.

Methodology

Leach Field No. 2 Investigation

The former leach field No. 2 consists of four rows of perforated pipe which were used to distribute wastewater prior to 1975. In 1975, a series of concrete chambers were installed at the north side of the leachfield. According to as built plans, the chambers can be accessed via inspection manholes. The leach field systems are located approximately 6 feet below grade.

The investigation of leach field No. 2 was performed using a Geoprobe sampler. Four geoprobe borings were advanced. Three of the locations (LF2-1, LF2-2, and LF2-3) were advanced along the former perforated leaching pipes. The fourth geoprobe boring was advanced through a manhole into a leachfield chamber. The fifth proposed geoprobe boring location was abandoned due to the presence of underground electrical lines. The locations of the geoprobes are shown on **Figure 3-10**.

Samples were obtained by driving geoprobe rods and plastic sampling sleeves at two foot intervals. Each two foot sampling sleeve was cut open and the soil description logged. Samples were composited over two foot intervals except when a significant change in strata was observed. The samples were composited in stainless steel bowls with stainless steel trowels. The two foot composited samples were submitted to the onsite HoltraChem laboratory for mercury analysis. Ten percent of the samples (3 samples) were split and submitted to the CDM laboratory in Cambridge, Massachusetts for confirmation and quality control/quality assurance purposes.

Leach Field No. 8 Investigation

Leach field No. 8 was located between the HCl tank farm and the lined process lagoon in the main plant. Existing plans of the leach field indicated a series of distribution lines extending from a distribution box. However, according to Gene Bridges of H.E. Bridges Construction, who has performed much of the excavation work on site, the leach field consisted of hemlock planks which were located parallel to the bedrock scarp west of the HCl tank farm and extending toward the lined process lagoon. Mr. Bridges indicated that the planks were likely removed during subsequent work in the area.

Leach field No. 8 was investigated by excavating a series of test pits where the hemlock planks were thought to have been located. Test pit LF8-TP1 was excavated in the location of the distribution lines shown on the existing plan. Test pits LF8-TP2 and LF8-TP3 were excavated perpendicular to the bedrock scarp, between LF8-TP1 and the lined process lagoon. The locations of the test pits are shown on **Figure 3-11**.

Samples were collected from each of the three pits. Three samples each were collected from test pits LF8-TP1 and LF8-TP3, and two samples were collected from LF8-TP2. The samples were submitted to the onsite HoltraChem laboratory for mercury analysis. One sample (ten percent) was split and submitted to the CDM Laboratory for confirmation and quality control/quality assurance purposes.

Variations from Approved Work Plans

Leach field investigations were not anticipated in the original work plan. As a requirement of a Consent Agreement between HoltraChem and the Maine Department of Environmental Protection (MEDEP), leach fields No. 1, 2, and 8 were added to the Areas of Concern. A plan for leach field investigations was transmitted to EPA and MEDEP in a letter dated July 2, 1998. A copy of this letter is provided in Appendix A.

One of the proposed geoprobe locations for the investigation of leach field No.2 was not sampled due to uncertainty regarding the location of underground electric lines. Due to the uncertainty of the location of leach field No. 8, the investigation was performed with test pits rather than with geoprobes as initially proposed. This change was communicated in advance in a letter to EPA and MEDEP dated August 12, 1998. A copy of this letter is provided in Appendix A.

Observations

At each of the geoprobe locations, a layer of fill was encountered from the ground surface to depths ranging from 4 to 6 feet below ground surface. The fill consisted of orange brown or grey brown sand and silt with varying amounts of gravel. Underlying the fill, and above the wet black gravel leachfield layer, a clay, or silt with clay layer was observed. Underlying the leachfield layer, a layer of mottled grey and orange brown clay with varying amounts of silt and fine sand was observed to a depth of 12 to 14 feet. The borings were terminated in a layer of grey or brownish grey clay with varying amounts of silt and gravel. The maximum depth of penetration was 16 feet in each of the borings except LF2-4, which was completed at 14 feet. No evidence of leach field distribution lines or hemlock planks was observed in the test pit trenches for the leach field No. 8 investigation. Boring logs and test pit logs from the leach field investigations are included in Appendix D.

Results

The analytical results of soil sampling from leach field Nos. 2 and 8 are summarized in **Table 3-24**.

3.5.5 Lined Process Lagoon Investigation

Objectives

An investigation of the soil beneath the lined process lagoon was performed as a result of groundwater sampling at the base of Landfill Area 1 which indicated that the lined process lagoon may be a former and possibly continuing source of the mercury concentrations detected downgradient.

Five Geoprobe borings were advanced beneath the lined process lagoon during October of 1998 to evaluate the vertical distribution of mercury in soils beneath the liner. One boring was advanced in each corner of this surface impoundment, and one in the center. Borings were advanced through sand and gravel material and up to eight feet into the underlying till.

Methodology

Before each soil boring, the liner was cut in an "x" shape, and the liners were pulled back to expose underlying soil. Four liners were encountered at each location. Soil borings were advanced to a total depth of 11 to 18 feet beneath the liner. Borings were terminated a minimum of four feet into the till, except at location LPL-004, where refusal was encountered approximately six inches into the till.

Borings were advanced in two foot intervals, using a 1.5 inch Geoprobe sampler. Soil was recovered from each interval in polycarbonate sleeves, which were removed from the Geoprobe sampling spoon and examined. The sleeves were then cut and soil was transferred to stainless steel bowls. Soil was homogenized over two foot intervals, except in sample LPL-005-07, where two distinct lithologies were encountered within the sample interval and two separate samples were collected. The locations of the lagoon borings are shown on Figure 3-11. Upon completion, borings were backfilled with bentonite clay and the top liner was patched.

Variances from Approved Work Plans

The procedures for the lined process lagoon investigation were not included in the original Work Plan, but were described in a letter from HoltraChem to EPA and MEDEP dated July 2, 1998. There were no variances from the investigation methodologies outlined in this letter.

Observations

Before the Geoprobe sampling, the lagoon liner was inspected for signs of damage. The liner appeared intact, except for a hole along the eastern wall approximately one inch by two inches in size. Below this hole, an abraded area approximately five inches in diameter was observed. Both of these flaws were located where the brine return line had rested on the liner, and they appeared to have been caused by abrasion from this PVC pipe. In addition, approximately 75 small salt piles were observed across the base of the liner. These salt piles appeared to be a result of brine passing up through pinholes in the liner and leaving salt accumulations after cleaning had been completed. The salt piles were between one half and one inch in diameter, and approximately one-half inch in height.

Pools of water were found between the liners; these pools moved when the liner was stepped on. Water or moisture was observed between each of the four liners when the liners were cut. At location LPL-005, sand and a piece of gravel were also found between the liners. A number of angular bumps, approximately one to two inches in size, were also observed in this corner of the lagoon.

The lithologies encountered during subsurface investigation were consistent with previous descriptions of the sand and gravel unit beneath Landfill Area 1, and the brown and grey tills. Soil directly beneath the liner was generally dry to moist, with dampness increasing with depth. A chlorine odor was noted in soils directly beneath the liner at locations LPL-001-03 and LPL-002-01. OVM readings for health and safety monitoring were consistently non-detect.

Results

Dry weight mercury concentrations in soils beneath the lined process lagoon ranged from 0.33 to 30.4 mg/kg. The highest concentrations of mercury were detected in samples collected from the first two depth intervals beneath the center of the lagoon, with visibly oxidized samples, and in samples collected from directly above the till. In general, mercury concentrations dropped sharply across the till interface. The results of mercury analyses for samples collected beneath the lined process lagoon are tabulated in **Table 3-25**.

3.6 Site Surface Water and Sediment Sampling

Objectives

The stormwater drainage ditches were sampled to evaluate concentrations of mercury, TOC, and at selected locations volatiles, in the surface water and sediment. The analytical results have been used to assess the potential for the surface water and sediment to contribute to contamination both at the site and to the Penobscot River. The analytical results have also been used to assess current surface water quality and the impact of former waste disposal practices at the site. Additional data on site surface water and sediment was collected by Acheron and HoltraChem, as described in Section 2.3.3. Pertinent data from previous and contemporaneous investigations is provided in Appendix B.

Methodology

Surface water samples were collected from the sample locations at the 12 areas shown on Figure 3-3. Sediment was collected from the 0 to 6 inch depth interval, and all sediment samples were analyzed for total mercury and TOC. Surface water was analyzed for mercury by EPA Method 6010/7000 series, and the method reporting limit was 0.50 $\mu\text{g/l}$. Sample SSW-010 was additionally analyzed for VOCs and chloropicrin by EPA Method 8240. Chloropicrin is not a standard RCRA target compound, therefore chloropicrin was analyzed as a volatile analyte as a modification to EPA Method 8240. Chloropicrin was treated as any other analyte, except the calibration was only performed at four levels. Chloropicrin data should be considered estimated values. The detection limit for chloropicrin was 5 $\mu\text{g/l}$. Sediments were analyzed for mercury by EPA Method 6010/7000 series, and the method detection limit was 0.30 mg/kg, dry weight. No sediment samples were analyzed for VOCs. Sediment samples, were analyzed for total organic carbon (TOC) by the Lloyd Kahn Method, and the reporting limit was 0.1% by weight.

Decontamination (decon) of sampling equipment occurred at the beginning of each day, and during the day, as sampling equipment (stainless steel mixing bowls, slotted spoons, screen strainers, and funnels, etc.) were used. The decon procedures for the sediment sampling equipment were same followed for the site soil sampling program described above. To ensure no cross-contamination of the surface water sample by sediment which was suspended in the water during sediment sampling, all surface water samples were collected first before sediment collection at each location.

Surface water samples were collected from the ditch or stream bank. The sample bottles were immersed below the water level, but not deep enough to disturb sediments. Following collection of the surface water sample, water quality parameters (pH, temperature, conductivity, dissolved oxygen and turbidity) were measured at each sample location. The sampling plan proposed that sediment samples would be collected from the top 6 inches at each sample location. However, there was generally not more than 4 inches of sediment, and often there was only 1 to 2 inches of sediment at each location. Therefore, samples were collected from within the general sampling area, where fine sediment had accumulated. Only site sediment sample location SSD-002 was collected during a time of no flow. Mercury sediment samples were collected in 125 ml glass jars with Teflon-lined caps. Samples for each were tightly packed and filled to the lip of bottle. Mercury and VOC surface water samples were collected in 1 liter plastic bottles and 40 ml amber vials, respectively.

Sediment samples were collected using a stainless steel slotted spoon from the surficial layer of organic material, to avoid collection of native material underlying the sediment. Leaves, sticks, and pebbles were excluded from the sample. The sample was then placed into a stainless steel screen strainer to assist in removing excess free water from the sediment. When it appeared that the sample was sufficiently drained, it was placed into a stainless steel bowl, and mixed thoroughly. The sediment sample was then placed in the appropriate sample container for mercury and TOC analyses. A description of the sediment sample collected at each location was recorded. The areas in which sediment samples were taken were generally subject to flow velocities which would tend to scour any lighter sediments from the bottom of the ditch. The sediment and surface locations were photographed, and their locations were recorded on the site map.

At sediment sample location SSD-005, the highly organic appearance of the sediment required the sample team to proceed with additional procedures to ensure adequate percent solids. The sampling relied on the standard operating procedure (SOP) developed by CDM for sediment filtration at the Davis GSR superfund site (SOP # 7710-016-03, CDM 1994). The process uses filtration by gravity, and was based on methodology similar to EPA's Paint Filter Liquids Test Method 9095. For sediment sample SSD-005, after the free water was drained from the sediment using the screen strainer, the sample was filtered by placing it in a large stainless steel funnel lined with filter paper.

The filter used was Whatman 541 Filters, which are suitable for critical analytical filtrations. The funnel was covered with aluminum foil, supported by a ring stand, and allowed to drain for 30 minutes. The filtered sample was then transferred to a mixing bowl, thoroughly mixed and put into the sample container.

A total of 10 surface water and 10 sediment samples were collected during this program. One duplicate surface water sample was collected for mercury analysis, one duplicate sediment sample was collected for mercury and TOC analyses, and one field (equipment) blank was collected during sediment sampling. An aqueous trip blank for mercury analysis was collected on September 21, 1994, prior to any surface water and sediment sample collection. The trip blank remained in the sample cooler and field vehicle over the program duration, and was delivered to the laboratory with the samples. One aqueous equipment blank was collected during the site surface water and site sediment sampling (SSW/SSD) program. The equipment blank was collected at the sample area prior to sediment collection and compositing in the mixing bowl. For the equipment blank, the aluminum foil was removed from the slotted spoon, screen strainer, and mixing bowl; U.S.P. distilled water was trickled over the spoon and strainer into the mixing bowl, and then poured from the mixing bowl into a 1 liter sample bottle for aqueous analysis of mercury. No equipment (field) blank for aqueous analysis of VOCs was collected.

Variations from Approved Work Plans

There was no surface water/standing water sample available for collection at location SSW-002, and there was no sediment sample available for collection at location SSD-010, due to scouring of the manhole bottom by the underdrain water flow.

Observations

Generally, the sediments were described as coarse to medium-coarse sand, with some gravel, and some silty gray organic material. The sediment samples collected at SSD-005 (paved sump) and SSD-006 (retention pond) were very silty due to the low flow velocity in these areas.

Field measured turbidity values for surface water samples ranged from 1 to 18 NTUs.

Results

The site surface water (SSW) analytical results for mercury and field measured parameters are presented in **Table 3-26**. VOCs were analyzed in surface water only at sample SSW-010, and these results are shown in **Table 3-27**. Site sediment (SSD) sampling results are presented in **Table 3-28**.

3.7 Penobscot River Surface Water and Sediment Sampling

Characterization of the Penobscot River included a survey of the bottom type and conditions, water quality and salinity measurements, surface water and sediment sampling. The methodology and results of river characterization are presented in the Ecological Risk Assessment, Section 9. The methodology and results of surface water and sediment sampling are provided below.

3.7.1 River Surface Water Sampling

Objectives

Sampling in 1995 was performed to evaluate background conditions and the distribution of mercury and VOCs in river surface water adjacent to the site. Data from Penobscot River surface water sampling have been used to assess surface water quality near and adjacent to the site, and to determine the impact of the site on the river.

River water sampling in August 1997 was performed to evaluate the distribution and concentration of mercury in the southern cove and the river during low, high, ebb, and flood conditions and to identify contaminant sources to the Penobscot River and the relative impact of these sources.

River water sampling was also performed in conjunction with dye testing, as part of the Penobscot River studies conducted by HoltraChem during August and September of 1998 as part of the Consent Agreement Comprehensive Monitoring Plan. The general objective of this study was to evaluate the distribution, fate and transport of mercury discharged by HoltraChem to the Penobscot River. The specific objectives of this surface water sampling task were (1) to determine background concentrations of mercury in the fresh water flow of the Penobscot River and the natural saline water from Penobscot Bay, (2) to attempt to validate the distribution of mercury as predicted from the dye study, and (3) to define to what extent, if any, the mercury concentration in the Penobscot River is influenced by groundwater flow and the resuspension of existing sediment from the HoltraChem site.

The surface water data were compared to data previously collected by Acheron and others. Mercury and VOC analytical results were used in the Risk Assessment to determine the health and ecological risks associated with the Penobscot River.

Methodology

1995 River Surface Water Sampling

Prior to collecting river surface water and sediment samples in 1995, selection of appropriate background locations was required. Several factors complicated selection of the background locations. The salinity of the Penobscot River in the vicinity of the site is transitional with brackish water below the site and fresh water upriver. The cove located at the southern portion of the site is larger than other coves in the vicinity and contains several environments not observed elsewhere. A greater amount of residential, commercial and industrial development of the river banks is present upriver of the site closer to Bangor and Brewer.

Potential background locations were identified based on field reconnaissance, calculations of the tidal influences and discussions with Maine DEP personnel. River reconnaissance identified several potential locations, however none of them contained the two distinct vegetated environments observed in the southern cove near the site. The following selection criteria were developed for evaluation of potential background locations:

- Location upriver of the site;
- Location away from direct discharge from dense residential, commercial or industrial development;
- Sufficient size and slope to maintain a tidal mud flat;

The increased development of the river bank in the vicinity of Bangor and Brewer limited the area considered to approximately three miles upriver of the plant. The number of coves, especially ones with tidal mud flats in this section, was very limited. Two coves were identified which had the largest tidal mud flats. The largest of these was located at the discharge point of the stream that drains undeveloped portions of the site. This location is referred to as the northern cove. The second tidal flat cove was observed approximately 1400 feet further upriver behind a gravel spit. Because of the potential that the stream discharging from the undeveloped portion of the site could impact the northern cove, both of these coves were selected and sampled as background locations. Comparison

of the surface water and sediment results between the two upriver coves would serve to verify whether the northern cove was an appropriate background location.

1995 Penobscot River surface water sampling locations are presented on **Figure 3-12**. Water samples from the Penobscot River were collected for analysis at the following stations:

- Offshore of the cove approximately 1400 feet upriver of the northern end of the site (RSW-001);
- Offshore of the cove at the northern end of the site (RSW-002);
- Immediately offshore of Landfill Area 1 (RSW-003);
- In the cove at the southern end of the site (RSW-004); and
- Immediately downstream of the cove at the southern end of the site (RSW-005).

Penobscot River surface water samples were collected proceeding from downriver of the plant toward upriver to prevent contaminating downstream samples by upstream sampling activities.

All river surface water samples except RSW-004 were collected at low tide. The sample collected in the cove, RSW-004, was collected toward the end of a falling tide. Surface water samples were collected approximately above the corresponding low tide sediment sampling locations.

Penobscot River water samples were sampled using "clean" techniques and analyzed using cold vapor atomic fluorescence to obtain ultra-low detection limits. Clean techniques refer to strict adherence to field procedures required to minimize the potential for trace levels of contamination to be introduced to the samples from the surrounding environment. The sampling techniques used for Penobscot River Water sampling were as follows:

- Two samplers put on fresh clean-room gloves;
- The sampler designated "Dirty Hands" retrieved the double bagged sample bottle from the cooler and opened the outer bag;
- The sampler designated "Clean Hands" opened the inner bag and pulled out the sample bottle;

- Without touching anything but the sample bottle, the "clean" hands sampler leaned over the up wind and up current side of the boat, unscrewed the bottle cap and poured out the acidified, analyte-free water which was transported from the lab with the sample bottles;
- The sample bottle was then immersed, mouth down, approximately one foot below the water surface and inverted;
- The sample bottle was rinsed once with river water and then filled from approximately one foot below the water surface as described above;
- The bottle cap which was held below the water surface during rinsing and sampling was screwed onto the sample bottle prior to removal from the water;
- After tightening the bottle cap, the "Clean Hands" sampler replaced the bottle in the inner bag and sealed it;
- The "Dirty Hands" sampler then dropped the completed bottle label into the outer bag and sealed it;
- The mercury sample was placed back in the cooler on ice and the remaining samples for hardness and VOCs were collected.

All Penobscot River surface water samples were analyzed for hardness, and total mercury. Samples RSW-001, RSW-003, RSW-004, and RSW-005 were also analyzed for VOCs.

1997 River Surface Water Sampling

Penobscot River water samples collected in 1997 were obtained from locations selected to evaluate the effects of groundwater, surface water, and NPDES permitted outfalls on river water quality during high, low, ebb, and flood conditions. Samples RSW-007 and RSW-020 were collected upstream and downstream of the plant, respectively. Sample RSW-019 was collected across river from the southern cove. Samples RSW-010, -011, -012, -013, -014, -015, -016, -017, and -018 were collected within the southern cove. Sample RSW-018 was taken in the vicinity of the PERC outfall pipe, and RSW-011 was collected in the vicinity of the HoltraChem outfall. Samples RSW-008 and RSW-009 were collected at the base of landfill Area 1. Sampling stations were located by the use of GPS and measurements from site features and landmarks when possible.

Sampling locations RSW-012, -013, -014, -015, and -016 were not covered with water at low tide and therefore were not sampled at that time. A water sample was not collected from the upstream sampling location, RSW-007, due to assumed minimal impact from site activities.

River water samples collected in August of 1997 were collected by the “clean” techniques sampling methodology as described for the 1995 sampling event. All river water samples were submitted to Brooks-Rand laboratory in Seattle, Washington for total mercury analysis using cold vapor atomic fluorescence. River water samples were not analyzed for VOCs or hardness in the 1997 sampling round.

1997 Penobscot River surface water sampling locations are shown in **Figure 3-13**.

1998 Dye Testing and Surface Water Sampling

During August and September of 1998, Acheron, Inc. collected surface water samples as part of their dye study to evaluate the fate and transport of surface water discharges from the HoltraChem site. This study was required as part of the Consent Agreement with MEDEP. CDM observed and participated in three days of this surface water sampling, to understand the approach and evaluate the methodologies employed. Four split samples were sent to Brooks Rand Laboratories to provide independent corroboration of the data obtained from this study.

Dye was injected into the southerly stream, north ditch, and outfall 001 sequentially and then simultaneously during the dye study. Initial dye dumps were added in high concentrations, to allow the plume to be monitored before the sampling portion of the study was performed. Following the initial tracking of the plume, dye was added to each discharge point in concentrations proportional to the concentration of mercury in each source. A flow-through fluorometer was then used to locate and quantify the dispersed dye in the river, and corresponding water samples were collected to assess the mercury concentrations surrounding the outfall pipe.

All samples from the dye testing study were collected using “clean” techniques as described for the 1995 sampling event. Because the samples were generally collected at depth, however, the sampling methodologies included a few modifications:

- Acid-washed sample tubing was removed from its packaging and extended to the desired length, according to the river depth indicated by the depth finder.

- The fluorometer input tube and sampling tube were attached to a weight, and the flow-through fluorometer pump was turned on. Once the desired dye concentration was identified, the sampling pump was turned on to purge water from that sampling depth.
- The sampling tube was purged for 30 seconds, to allow sample water to fully flush through the tubing. The “dirty hands” sampler handled the tubing, while the “clean hands” sampler removed sample bottles from their bags and placed them in a sampling rack.
- Sample bottles were filled, capped, and returned to their bags by the “clean hands” sampler.
- The outer bags were labeled and samples were placed on ice.

Following the simultaneous discharge phase of the dye study, additional dye was injected into the southerly stream over a period of five days, to evaluate the distribution of southerly stream discharge into the southern cove. After the first four days of the dye dump into the southern stream, CDM collected groundwater and seep samples from monitoring well B-321-B2 and seep SSP-007. These water samples were analyzed in the fluorometer to assess whether surface water is influencing groundwater quality in this area (see Sections 3.4.5 and 3.4.6).

In addition to the dye study, tidal bore monitoring was conducted during September of 1998 to evaluate the maximum extent of surface water transport upriver during flooding tide conditions. This monitoring was conducted by placing drogues (apples and oranges) in the river midchannel of Outfall 001 at low tide. The position of the drogues at the end of the flooding tide was noted as an indication of tidal influence upriver from the plant.

Variations from Approved Work Plans - 1995

Hardness was added to the river surface water analytical program to evaluate the availability of metals in the surface water environment. The regulatory action levels for metals are typically hardness dependent. There were no other variations in sampling locations, rationale or techniques.

Variations from Approved Work Plans - 1997

During the 1997 surface water sampling, there were no variations from the approved Supplemental Site Investigation Work Plan. A plan for additional surface water sampling to be performed with

additional sediment toxicity sampling in December 1997 was abandoned due to ice blanketing the sediment of the cove and river bank. It was observed that this ice remained attached to the bottom during the incoming and high tides, thereby preventing potential resuspension of sediment. Therefore, any surface water samples which could be collected during the December sampling trip would not be representative of conditions observed and measured in the summer.

Variations to Approved Work Plans - 1998

Variations to the proposed Penobscot River studies were made based on river conditions, initial dye testing results, and discussions with MEDEP and EPA personnel. The dye test was performed later in the summer than originally proposed due to high flows in the river remaining longer than expected. After initial dye tracking results were available, a meeting was held at the HoltraChem facility on August 26, 1998 to review the results and adjust the study to acquire the desired information. Variations included adjustment of dye concentrations and sampling transects to permit tracking of the batch discharges, and the addition of sampling locations in the vicinity of the HoltraChem outfall independent of dye concentrations. Additional adjustments (additions) were made to the surface water sampling program after discussions with EPA at the HoltraChem facility on September 21, 1998. These discussions occurred while observing surface water sampling of a batch discharge and after noting sediment resuspension occurring in the southern cove and along the banks of the Penobscot River. Sampling of turbid and non-turbid water in the cove, off of the HoltraChem outfall and up river of the HoltraChem site were discussed and added to the program. Also discussed and subsequently collected were samples of the groundwater seepage beneath Landfill Area 1 and near the HoltraChem outfall.

Observations

The RSW-004 location was collected approximately two thirds of the way through the falling tide. The sample location was approximately 75 feet down river of the Penobscot Energy Recovery Company's (PERC's) NPDES outfall. Bubbles were observed floating downriver of PERC's outfall and the sampling location was located in shore of the trail of bubbles. Field observations during 1997 river surface water sampling noted turbidity.

Results

Analytical results for mercury, hardness and field measured parameters in Penobscot River surface water samples collected in 1995 are presented in **Table 3-29**. VOC analytical results are provided in **Table 3-30**. No VOCs were detected. Although chloropicrin was not a target compound of the VOC analysis, the presence or absence of chloropicrin was evaluated by examination of the

chromatograms. No chloropicrin was detected. Mercury analytical results for river water samples collected during the four tides in 1997 are summarized in **Table 3-31**. The results of Acheron's Penobscot River studies are included in Appendix B

3.7.2 River Sediment Sampling

Objectives

Site sediments were collected in 1995, 1997, and 1998, each time with slightly different objectives.

The general objective of river sediment sampling was to evaluate background conditions and the distribution of mercury and VOCs in river sediments adjacent to the site. In 1995, sediment samples were collected from two background areas upriver from the site, from the bank adjacent to Landfill Area 1, and from distinct environments within the southern cove. Each of these areas provides distinctly different biological and physical intertidal environments. The quantity and quality of biota, as well as the potential for mercury accumulation, varies with each location.

Additional samples were collected in August and October of 1997 to further delineate the extent of mercury contamination in the southern cove sediments. The August 1997 sampling round attempted to evaluate mercury concentrations in the vicinity of the north ditch and the southerly stream, to characterize the effects of plant runoff on sediments in the southern cove. Sampling in October of 1997 was performed following the August sampling event, to delineate the lateral and vertical extent of mercury.

In December 1997, sediment samples were collected from six locations in the southern cove to represent high organic carbon ("organic") sediment and low organic carbon ("granular") sediment. Chemical and biological analyses (sediment toxicity tests) were conducted on the whole sediment, and the interstitial (pore) water was extracted using a combination of centrifugation and filtration to generate dissolved and total fractions of the pore water. Chemical analysis of pore water produced concentration data for both total mercury and methylmercury. Site-specific partition coefficients for organic and granular sediments were calculated.

In July 1998, whole sediment was collected from two locations in the southern cove. The objective of this sampling was to obtain samples from the most contaminated areas. Chemical analysis and sediment toxicity tests were conducted on the whole sediment samples.

All of the data from Penobscot River sediment were used to assess sediment quality near the site, and to evaluate the impact of the site on the river. The sediment data were compared to data previously collected by Acheron and others to evaluate temporal trends in river sediment contamination. Mercury and VOC analytical results were used in the Risk Assessment to determine the health and ecological risks associated with Penobscot River sediments.

Methodology

1995 River Sediment Sampling

1995 river sediment sample locations are presented on **Figures 3-12** (greater river) and **3-14** (southern cove). During the 1995 sampling round, samples were collected from the respective areas as follows:

- Background area 1 - one intertidal and one subtidal location (RSD-01A and B);
- Background area 2 - one intertidal and one subtidal location (RSD-02A and B);
- Off Landfill Area 1 - one intertidal and one subtidal location (RSD-03A and B);
- Organic mat/reed bed - three locations (RSD-04A, B and C);
- Mid-intertidal/sedge bed - three locations (RSD-05A, B and C);
- Sand and gravel area- five locations (RSD-06A through E);
- Lower intertidal/fine silt - five locations (RSD-07A through E); and
- Subtidal cove area - two locations (RSD-08A and B).

Intertidal sampling locations were located approximately halfway between high and low tide levels. Subtidal sampling locations were collected approximately 3 to 5 feet off shore of the low tide water line. Samples were collected from two depths at each location. Surface grab samples were collected from the first 0.0 to 0.1 foot interval of sediment. A second sample was collected from approximately 0.8 to 1.0 foot below the sediment surface. Surface grab samples were collected with stainless steel trowels or spoons. Most second depth interval sediment samples were collected using dedicated three-inch diameter lexan plastic tubes. Second depth interval samples from sand and gravel locations were collected using stainless steel hand augers.

Lexan tubes were pushed or pounded into the sediment until the leading edge was approximately one foot below the top of the sediment layer or until refusal which ever came first. The tubes were then filled with river water and capped to maintain the suction required to retain the sediments in the tube during tube withdrawal. Sediment from the bottom of the tubes was transferred to stainless steel bowls, and homogenized before being placed into laboratory glassware.

For more coarse grained sediments, hand augers were advanced to the sampling interval and retrieved, and sediment was transferred to stainless steel bowls and homogenized before being transferred to laboratory glassware. It was often necessary to repeatedly advance the tubes or hand augers to the target interval to obtain sufficient sample volume. Repeated advances were made within an approximately two-foot radius of the initial sampling location and care was taken not to encounter sediment which may have been disturbed by contamination from shallower depths. Sampling locations were staked and distances between locations and site features were measured upon completion of sampling.

Sediment samples were analyzed for:

- Total mercury;
- Total organic carbon (TOC);
- Sediment grain size; and
- Acid Volatile Sulfide/Simultaneously Extracted Metals (AVS/SEM) for some locations

AVS/SEM was analyzed from the first 0.1 foot of sediment collected from one randomly chosen station in each sediment type within the intertidal and subtidal areas of the cove. AVS/SEM was analyzed from only the first 0.1 foot section of the core because this represents the most biologically active depth. The AVS/SEM ratio provides an estimate of bioavailability of mercury in sediments.

1997 River Sediment Sampling

1997 river sediment sample locations are presented on **Figure 3-15**.

Sediment sampling conducted during August 1997 was conducted in a similar fashion to the sampling in 1995. Two sediment samples were collected per location. The shallow sampling interval was collected from a depth of 0.0 to 0.1 foot (samples designated -01) and the second interval was collected from 0.8 to 1.0 foot (samples designated -02) below the top of the sediment. In a few locations, additional samples were collected at depths of 1.3 to 1.5 feet (samples designated -03) and 1.8 to 2.0 feet (samples designated -04). The shallow samples were collected directly from the surface with stainless steel spoons or trowels. The deeper samples were collected with lexan tubes within softer sediments and with hand augers in coarse grained sediments. Samples were transferred from the tubes or augers to stainless steel bowls and the sediment descriptions recorded in the log book. The samples were homogenized with stainless steel trowels and placed into the laboratory glassware.

The samples were analyzed for mercury and total organic carbon (TOC) by the CDM laboratory. A select number of samples were placed in zip lock bags and submitted to CDM's geotechnical laboratory for grain size analysis. Selected samples were analyzed by the on-site HoltraChem laboratory and the preliminary results were used to determine subsequent sampling locations.

Samples were collected in the following locations in August of 1997:

- Thirteen samples (RSD-09A and B, 10A through C, and 11A through H) were collected outward from the northern stormwater outfall.
- Fourteen samples (RSD-12A through E, 13A through E, and 14A through D) were collected in the inner southern cove, directly downstream of the southerly stream's outfall.
- Ten samples (RSD-15A through H, and 16A and B) were collected in the outer southern cove, downstream of the southern stream's outfall. These samples were collected with lexan tubes from a boat, as the locations were covered by the incoming tide.

Sampling locations were staked and distances from the shore and other site features were measured.

Twenty-six sediment locations were sampled in October 1997 using a vibrating sediment coring device mounted on a boat. The sediment cores were collected by TG&B Marine Services of Bourne, Massachusetts. The coring device was used to obtain sediment samples at depths greater than were obtainable with hand augers or lexan tubes and at locations that were covered with water at low tide. The locations for sediment coring were chosen to supplement existing sediment data, to evaluate the extent of elevated mercury concentrations detected in previous sampling rounds south of the PERC outfall, at the mouth and near the north ditch (Outfall 003), at depth at the mouth of and in the vicinity of the HoltraChem outfall (Outfall 001), and at depth near the southerly stream outfall.

Sediment cores were collected in aluminum core barrels. The cores were cut into 1 foot intervals, the recovery and sediment description recorded, and the sample was extruded to a stainless steel bowl, homogenized with a stainless steel trowel and transferred to the laboratory glassware. The sample intervals collected were: 0.0 to 0.1 (-01), 0.8 to 1.0 (-02), 1.0 to 1.5 (-03), 1.5 to 2.0 (-04), 2.0 to 2.5 (-05), and 2.5 to 3.0 (-06) feet. At each location, an attempt was made to core to 5 feet within the sediment. However, due to the coarse grain size of the sediment underlying the cove at various depths, only 8 out of 26 locations could be cored to depths greater than 1 foot and samples greater than 2 feet were obtained at only two locations. The sediment sampling locations were recorded using a differential global positioning system (GPS).

Selected samples collected during first two days of coring were analyzed using immunoassay field test kits to obtain a rapid estimate of the mercury concentration. These values were used to identify subsequent sampling locations and confirm adequate characterization of cove sediments. Samples were also sent to the on-site HoltraChem laboratory to provide quantitative results (wet weight) to confirm immunoassay results. The samples were submitted to the CDM laboratory for final quantitative values of mercury, as well as total organic carbon analyses. Three sediment samples from the coring program were submitted to CDM's geotechnical laboratory for grain size analysis.

Eight duplicate samples and three rinsate blanks were collected and analyzed during the August 1997 sampling round. During the sediment coring in October 1997, seven duplicates and two rinsate blanks were collected and analyzed.

1997 Sediment Toxicity Sampling

Twelve large volume (bulk) sediment samples were collected in December 1997 for sediment toxicity testing, total and methylmercury chemical analysis, pore water extraction and pore water total mercury and methylmercury chemical analyses. The bulk sediment samples were labeled RSD-021 through RSD-032, and were collected from locations in which hot spots were identified during the Fall 1997 sediment sampling.

Bulk samples were collected by shovel, or in the case of the submerged organic muck, the oar of the boat. The objective of the sampling was to obtain large volumes of sediment with a range of mercury concentrations, and to collect organic and granular sediment with the highest possible concentrations of mercury. Sampling equipment was decontaminated between locations by rinsing in river water.

At each sample location, sediment was placed into a cooler lined with heavy duty plastic garbage bags. The garbage bags were then lifted from the cooler and packed with snow until the samples were homogenized. Bulk sediments were homogenized by thoroughly stirring the sediment within each bag with a decontaminated stainless steel spoon. Within three hours, the homogenized samples were transferred to two two-gallon large-mouth plastic jars for transport to the analytical laboratories and one ziplock bag for transfer to the geotechnical laboratory. Samples for sediment toxicity testing and analytical chemistry were packed on ice (~4 degrees C) and transferred under chain-of-custody to Aquatec Laboratories of Burlington, Vermont.

In the laboratory, interstitial (pore) water was extracted from whole sediment using a combination of centrifugation and filtration (see Appendix F - Aquatec Laboratory Report dated December 1997).

New polycarbonate centrifuge bottles (acid rinsed) were assigned and dedicated to each sample. For high organic carbon content sediments (RSD-029, 030 and 031), pore water was extracted in two steps. Pore water from low organic carbon content sediment (RSD-022, 023 and 025) was extracted in the initial step, then transferred to sample bottles supplied by Brooks Rand Ltd. The first centrifugation step was performed using an IEC PR-7000M centrifuge at 7,400xG at 4°C for 15 minutes. The second centrifugation step was performed by transferring first step supernatants to 500 ml bottles using a 125 ml disposable polystyrene serological pipet. Bottles were spun using an IEC B-22 centrifuge at 10,000xG at 4°C for 30 minutes and supernatants were transferred by clean, disposable pipettes to sample bottles provided by Brooks Rand Ltd. Rinse blanks were collected following these procedures and analytical results indicated no cross contamination. The laboratory procedures are detailed further in Appendix F.

1998 Sediment Toxicity and Speciation Sampling

The December 1997 toxicity sampling yielded bulk sediment samples with lower than anticipated total mercury concentrations. Because sediment toxicity testing on these samples did not identify a concentration at which the sediment was toxic, one additional attempt was made during July of 1998 to collect large volume samples for biotoxicity analysis. During this sampling event, five additional sediment samples were also collected to evaluate the prevalent forms of mercury present in southern cove sediments. These mercury speciation samples were collected from granular, organic, and transitional sediments to assess the effects of sediment type on mercury speciation in the southern cove.

Large volume sampling for biotoxicity analysis was conducted in a similar fashion to the December 1997 sampling. Sampling locations were targeted based on observed mercury concentrations (hot spots) in the southern cove during previous sampling events. Multiple samples were collected along the low tide level, near the location of the previously exposed HoltraChem outfall. To ensure that sediments with high levels of mercury were submitted for toxicity analysis, samples were first screened at the HoltraChem lab. After the initial screening, the sample which had been collected near the location of sample RSD-10A was selected to be sent to Aquatec Laboratories. An additional sample with low levels of mercury contamination was also sent to Aquatec to be used in sample dilutions, if needed.

Mercury speciation samples were collected from two locations within the southern cove (organic and transitional sediment), and two locations along the low tide line near the formerly exposed HoltraChem outfall (granular sediment). The organic samples were collected by pushing clean 2" diameter lexan tubes into the organic mat, capping the tubes, and pulling out an intact core.

Transitional samples were collected from the base of these tubes, where organic material graded into granular sandy materials. Granular samples were collected from the low tide line with stainless steel trowels. All samples were then transferred to stainless steel bowls, homogenized, and placed in sample jars for shipment to the laboratory. As in the biotoxicity sampling, these samples were first screened at the HoltraChem lab to ensure adequate levels of mercury for the selected analyses.

Speciation analysis was performed on five sediment samples to evaluate the type of mercury present in southern cove sediments and to assess its bioavailability. Speciation was performed by sequential extraction using extraction fluids which were either compound specific (e.g. methylene chloride extract for methylmercury analysis) or of increasing acidic strength to evaluate how tightly bound the mercury was to the sediment. The extraction fluids were aqua regia to evaluate total mercury, methylene chloride to evaluate methylmercury, 1 N (normal) hydrochloric acid (HCl) to evaluate the amount of weakly associated inorganic forms [Hg(II)] and 12 N Nitric acid (HNO₃) followed by an aqua regia digestion to evaluate the amount of mercuric sulfide (HgS) present. The amount of mercury in strongly bound forms other than HgS was obtained by subtracting the amounts of the methylmercury, weakly associated and HgS forms from the total mercury number.

Variances from Approved Work Plans-1995

A sediment sampling program was initially proposed in the Work Plan. The 1995 sampling program was revised based on comments received from EPA, NOAA, and MEDEP and the results of a site and river reconnaissance performed in September 1994. A revised sediment sampling program was proposed in the Risk Assessment Protocol Document (RAPD). The sediment sampling performed followed the general procedures described in the Work Plan while the locations sampled and the analyses conformed to the program detailed in the RAPD.

The extent of the various environments within the southern cove differed slightly from the sketch provided as Figure 6 in the RAPD. The sampling locations were adjusted accordingly to collect samples representative of the distinct environments present and provide sufficient coverage across the cove and in the areas of potential discharge to the cove.

The variability of the substrate and the task of obtaining representative samples from several different depositional environments required field decisions to tailor sampling efforts to the specific conditions encountered. As described above, the type of sampler used depended on the composition of the substrate sampled. In addition, field decisions were made to ensure the representativeness of the samples collected.

In the central portions of southern cove and at the background locations, the intertidal silt often exceeded one foot in thickness. This deposit was typically the consistency of chocolate mousse and appeared homogeneous from top to bottom. The field decision was made to collect the surface sample from the top one inch of this silt and the second depth interval from the granular material that lay directly below the silt at locations where the silt thickness exceeded one foot.

At background location RSW-02A, the organic silt was approximately 5 inches thick and directly underlain by a medium sand. However, the interval from 0.8 to 1.0 foot below the top of sediment surface was comprised of coarse gravel which could not be recovered with the hand auger. A field decision was made to collect the second depth interval sample from the sand at approximately 0.5 to 0.75 foot below the sediment surface.

At background location RSW-01A, the granular substrate beneath the organic silt was also too coarse to permit sample collection. The sample location was, therefore, moved inshore to the pinch out of the intertidal silt where the granular material contained a larger percentage of sand.

Variances from Approved Work Plans-1997

The locations of samples collected in August 1997 varied from the locations proposed in the Supplemental Site Investigation Work Plan (CDM, June 1998). Five transects of sample locations were sampled in the southern section of the cove. The locations were chosen in the field based on sediment and cove conditions and initial data obtained from samples submitted to the on-site HoltraChem laboratory. Additional sediment intervals were sampled (1.3 to 1.5 and 1.8 to 2.0 feet) beyond the intervals proposed in the Work Plan.

The methodology for the sediment coring was proposed in correspondence from CDM to EPA dated October 22, 1997. Sampling intervals were altered based on verbal comments from the MEDEP to correspond to the previous -01 and -02 depth intervals and to provide for 6-inch interval samples below one foot. Each foot of sediment obtained deeper than 1 foot was split into 2 separate samples. Obtaining samples deeper than 3 feet was not possible due to the granular and cobbly nature of the underlying sediments. Selected samples were submitted to the CDM Geotechnical laboratory for grain size analysis. Samples were not passed through a #200 sieve in the field.

Biotoxicity and speciation analyses during December of 1997 and July of 1998 were not in the initial Work Plan but were added to assess the bioavailability and toxicity of mercury in Penobscot River sediments.

Variations from Approved Work Plans-1998

The additional sediment sampling performed in 1998 was not anticipated in the original Work Plan but was added to provide additional data on the species of mercury present in southern cove sediments and the toxicity of the sediments in the area of the HoltraChem and north ditch outfall where the highest sediment mercury levels were detected. The 1998 sediment sampling and analysis program was described in correspondence to MEDEP and EPA dated July 2, 1998. No variations to the proposed sampling were necessary. Dilution of the sample analyzed for sediment toxicity was not necessary as no toxicity was observed in the most contaminated sample collected.

Observations

During the sampling event in 1995, the intertidal silt was often observed to exceed one foot in thickness in the central portions of the southern cove and at the background locations. This silt appeared homogeneous from top to bottom, fibrous, organic and is best described as being the consistency of chocolate mousse. It did not exhibit a sulfide odor indicative of anaerobic conditions. The silt did not extend up the steeper banks above the approximate mid-tide line. The members of the sampling team questioned whether this deposit was permanent or transient.

Additional sampling throughout the southern cove in 1997 better defined the cove sediment stratigraphy. The organic silt layer was observed mainly in the southern section of the cove (south of the PERC outfall) at thicknesses ranging from 6 inches to 2 feet. Sand and gravel deposits were encountered underlying the organic silt in the majority of the sampling locations. Organic silt was not observed beyond the low tide line where the faster river current was observed.

Bulk sediment samples (approximately 2 gallons each) for whole sediment toxicity analyses were collected from areas where some of the highest sediment mercury concentrations were detected in the August and October 1997 grab sampling programs. Samples were homogenized similarly to the grab samples except that larger volumes of sediment were involved. Although specific hot spots were targeted for bulk sampling, similarly elevated concentrations were never observed in the larger volume bulk samples. It appears that the most elevated concentrations of mercury are in discrete layers in the sediment and that when a larger sample volume is homogenized the resulting average concentration is less than was represented by the grab samples.

Results

The Penobscot River sediment sampling analytical results for mercury, TOC and grain size (as % fines passing a #200 sieve) are provided in **Table 3-32**. The results of VOC analyses are provided in **Table 3-**

33. The results of SEM/AVS analyses are presented in **Table 3-34**. The results of dissolved and total mercury, and dissolved methylmercury chemistry on sediment interstitial (pore) water are presented in **Table 3-35**. Results of whole sediment toxicity testing are presented in **Table 3-36**. The results of speciation analyses are presented in **Table 3-37**.

3.8 Ecological Sampling and Characterization

The methodology and results of ecological characterizations are provided in the Ecological Risk Assessment presented in Section 9. The methodology and results of biota sampling are provided below.

3.8.1 Soil Invertebrate Sampling

Objectives

Soil invertebrate samples (earthworms, designated SIS) were collected to analyze mercury concentrations in their tissues during the 1995 site investigation and the 1997 supplemental site investigation. The results of the analyses were used to assess exposure to small mammals and birds from bioaccumulation. At each station, a composite soil sample (SSS) was also collected to evaluate the soil mercury concentration to which the worms were exposed. Sampling locations were selected to provide analytical results representative of the typical environments found at the site outside of the immediate plant area. Invertebrate sampling locations are presented in **Figure 3-16**.

Methodology - 1995

During 1995, composite worm and soil samples were collected from five randomly chosen stations in the upland and near shore area of the site, and from the background area. These locations are identical to the soil sampling locations SSS-015 through SSS-020.

The four onsite locations were selected based on the results of an initial site walkover and are as follows:

- Upper Woodland east of Landfill 5 (SIS-001 & SSS-015)
- Upper Meadow north of Landfill 3 (SIS-002 & SSS-016)
- Upper Woodland west of Landfill 3 (SIS-003 & SSS-017)
- Lower Meadow Adjacent to the Penobscot River (SIS-005 [Duplicate SIS-006] & SSS-019 [Duplicate SSS-020])

Dorthea Dix Park, south of Hampden (west side of Penobscot River) was selected as the background location for earthworm and soil sampling (SIS-004 & SSS-018).

At each sampling area except the lower meadow adjacent to the Penobscot River, earthworms and soil were collected within an approximately 25-foot radius of the staked sampling location. To collect sufficient sample for the duplicate, earthworms and soil from the lower meadow were collected from an area approximately 20 feet wide extending approximately 50 feet upriver and downriver from the staked location along the tree lined portion of the riverbank.

Within each sampling area, a small soil sample was collected at each location where earthworms were found. Using a stainless steel scoop, up to five soil samples were collected from each worm sampling area. These subsamples were given an alphabetical suffix after the sample location. The set of soil samples from each area were submitted to the laboratory with instructions to composite the subsamples into one composite soil sample for each sampling area.

Worms were placed in zip-locked plastic bags in the field and stored on ice until delivery to the laboratory. Prior to delivery, worms were placed on moistened filter paper and allowed to depurate so that the analyses would represent the mercury concentration of the worm tissue only and not include the soil within the worms.

Methodology - 1997

During 1997, five additional invertebrate samples were collected, four from the ridge area and one from the meadow below the HoltraChem plant. These invertebrate sampling locations were co-located with soil samples SSS-024, SSS-032, SSS-034, SSS-050, and SSS-042, and were designated SIS-007 through SIS-010. Invertebrate sampling during the 1997 investigation was conducted after sitewide soil sampling was complete; sampling locations were selected to represent high, medium, and low concentrations of mercury in associated soils.

Invertebrate samples were collected within a 25 foot radius of associated soil sampling locations. Earthworms were collected in jars and allowed to depurate for two days before shipment to the laboratory. Samples were rinsed with deionized water between the first and second day and between the second and third days of storage, to clean them of soil material. Samples were transferred to clean jars before shipment.

Variations from Approved Work Plans

The specific procedures for earthworm and soil sampling were not described in the RAPD or the approved Work Plan. The field program was tailored to the environments to be sampled and the conditions encountered in the field.

Observations

During the 1995 investigation, the weather had been dry preceding the earthworm and soil sampling, and earthworms were not readily found. To locate earthworms, rocks and logs were overturned until sufficient sample volume was acquired.

In 1997, rainfall before and during sampling allowed earthworms to be readily found in surficial soils. Difficulties were encountered in finding earthworms at sampling locations in the northern portion of the plant (SSS-041, SSS-042, SSS-043). Samples were collected at SSS-042, after finding no worms at the other two locations.

Results

The analytical results of earthworm and associated soil sampling are provided in **Table 3-38**.

3.8.2 Small Mammal Sampling

Objectives

A trapping program was conducted in the wooded portions of the site and at the background location to collect the small mammals present for analysis of their body burdens of mercury.

The data will be used in the ecological risk assessment to characterize the amount of mercury uptake in the small mammals on the site and the potential exposure to predators which feed on those species.

Methodology

Small mammal samples (SMS) were collected using box and spring traps. Lines of traps were set in the wooded sections of the site and in the background area, Dorthea Dix Park. The locations where specific samples were caught are indicated on Figure 3-16.

After collection, the small mammals were placed in zip-lock bags and stored on ice until delivery to the laboratory. Prior to shipment, the samples were washed to remove any soil which might be on their fur.

Variations from Approved Work Plans

The specific procedures for small mammal sampling was not described in the RAPD or the approved Work Plan. The field program was tailored to the environments to be sampled and the conditions encountered in the field. A three-day effort was necessary to collect the required samples.

Results

Results from the small mammal sampling are included in **Table 3-39**.

3.8.3 Fish Tissue Sampling

Objectives

Samples of fish were collected to evaluate their body burdens of mercury. Results of fish tissue analysis were used to evaluate the potential for mercury to bioaccumulate in fish and, if possible, evaluate the impact of the site on the fish in the Penobscot river. Two types of fish samples were collected: eels and killifish.

Methodology

Eels were collected off the southern cove and offshore of the background area upriver. Sampling locations are presented on **Figure 3-17**. The eels were obtained from traps set and harvested by a commercial fisherman. The fisherman pulled the traps but personnel conducting the SI removed the eels from the traps and placed them in zip-lock bags. Nine eels were collected off the southern cove and six eels from traps set immediately upriver of the northern cove background area. The eels were stored on ice, weighed, measured, tagged with an identifying number, placed in separate zip-lock bags and transported on ice to Battelle Ocean Sciences Laboratory in Duxbury, Massachusetts.

An attempt to collect killifish was made in June. However, no killifish were observed in the southern cove at that time. A second effort was made to collect small fish in the southern cove and at the two background locations on August 7, 1995. Specimens which are considered to be resident were collected in the southern cove. However, only migratory species (herring) were caught in the two background

locations. Samples of the fish caught in the southern cove were submitted to Battelle Ocean Sciences laboratory for analysis.

Variances from Approved Work Plans

Fish tissue sampling was not proposed in the Work Plan but was added to the sampling program in the Risk Assessment Protocol Document. The RAPD specified several potential methods for catching fish. In its comments on the RAPD, the MEDEP suggested obtaining fish from local fishermen. Traps were set by SI sampling personnel, however the local fisherman's traps were much more successful in obtaining larger eels which were of greater interest to the SI. Therefore, samples were collected directly from the fisherman's traps while still at the trap site to ensure that the fish were representative of the areas of interest to the SI.

The RAPD called for samples of killifish to be collected at both the southern cove and the background areas. However, after inspection of the small fish collected in the background areas, it was determined that only migratory species were caught, and therefore no samples of resident small fish from the background areas were obtained. Killifish were obtained from the southern cove and forwarded to the laboratory for analysis.

Observations

The southern cove is a unique environment in the stretch of the river from Bangor to Orrington and possibly down to Bucksport. It has a greater amount of sedge and contains the only organic mat and reed growth observed. This additional vegetation could be the reason that killifish were caught in the southern cove but not at the background locations.

Results

The results of fish tissue sampling are summarized in Table 3-39.

Section 4 Data Evaluation

4.1 Introduction

The data usability assessment presented in this section identifies qualifications to the analytical data based on review of quality control information. In summary, almost all the data collected is usable for all site characterization and risk assessment purposes. In the rare instances where data was rejected, these changes have been made to and noted on the tables in Section 3.

The data usability assessment is divided into five subsections as follows:

- 1994 soil, surface water and sediment sample data (4.2);
- 1995 soil and sediment sample data (4.3);
- 1995 groundwater and surface water sample data (4.4);
- 1997 soil, sediment and Groundwater, and (4.5);and
- 1998 water and soil data (4.6).

4.2 1994 Soil, Surface Water and Sediment Sample Data

Laboratory quality control data associated with all samples submitted from the HoltraChem Manufacturing site in Orrington, Maine for the sampling conducted from the first of August to the twenty-second of September, 1994 was reviewed and is summarized in this report. Associated sample delivery groups (SDGs) include CDM Laboratory task #: 940808-02, 940819-01, 940906-00, and 940923-08. The samples were analyzed by the CDM Laboratory of Cambridge, Massachusetts; and by ITS-Aquatec Laboratories, of Colchester, Vermont. The samples were analyzed for mercury, volatiles organic compounds, chloropicrin, and TOC. The samples were analyzed according to the analytical methods described in Test Methods for Evaluating Solid Waste Physical/Chemical Methods, SW-846 November 1990. This data usability assessment is based on the review of the following quality control information:

- Laboratory Method Blanks
- Matrix Spike/Matrix Spike Duplicates
- Surrogate Spike Recovery
- Laboratory Duplicates
- Laboratory Quality Control Samples
- Sample Holding Time Data

Project-generated quality control data was also reviewed. Two trip blank and six equipment blank samples were included in these sample delivery groups. In addition, seven field duplicates were also included in these sample delivery groups.

All applicable quality control data were evaluated on a per analysis basis and are summarized below by analytical parameter. Hold time and field blank data are summarized in the data validation report, included with the laboratory analytical results. Approximately 20% of analyses were also validated by Gradient Corporation, of Cambridge, MA. This review encompasses all of Gradient's comments. Gradient's data validation reports are included with the laboratory data. This review was conducted in accordance with SW-846 data quality guidelines, USEPA HLP data validation criteria, and the Guidance for Data Useability in Risk Assessment, April 1992.

4.2.1 Mercury Analysis

Laboratory, Field, and Equipment Blanks

Mercury was not detected in any laboratory method or calibration blanks. No significant contamination was noted in any of the field or equipment blanks.

Matrix Spike/Matrix Spike Duplicate Recoveries

There were four aqueous matrix spike analyses associated with the samples in these tasks numbers: SSW-008-01, SSW-001-01, SSW-010-001, and EB-8/16A. Matrix spike recoveries were within the recommended criteria.

There were ten soil matrix spike/matrix spike duplicate (MS/MSD) analyses associated with the soil samples: SBS-403-02, SSS-01A-01, SSS-03B-01, SSS-04C-03, SSS-08C-02, SSS-11A-01, SSS-12A-01, SSB-007-00, SSB-007-22, and 94-04620 (not an HoltraChem Manufacturing sample). The sample result for samples SSS-03B-01, SSS-04C-03, and SSB-007-00 exceeded the spike concentration by a factor of four or more, no action is taken.

The following table lists the analytes that did not meet the 75-125% recovery criteria.

Analyte	Client ID	% Recovery (MS/MSD)
Mercury	SSS-01A-01	43/45
Mercury	SSS-08C-02	87/66
Mercury	SSS-12A-01	358/155

The MS/MSD results from samples SSS-01A-01, SSS-08C-02, and SSS-12A-01 were applied to the associated "SSS- soil boring" samples. It is the reviewer's opinion that the MS was inadvertently double spiked for the MS analysis of SSS-12A-01. The usability of the mercury data obtained from the analysis of these samples is qualified in accordance with the following requirements:

- All positive results and associated detection limits are considered estimated values (J).

Laboratory Duplicate Precision

There were three aqueous laboratory duplicate analyses associated with the samples in tasks numbers: SSW-008-01, SSW-001-01, and EB-8/16A. All duplicate results were within the recommended criteria.

There were eleven soil laboratory duplicate analyses associated with the samples in these tasks numbers: SBS-403-02, SSS-01A-01, SSS-03B-01, SSS-04C-03, SSS-08C-02, SSS-11A-01, SSS-12A-01, SSB-007-00, SSB-007-22, SSD-006-01, and 94-04620 (not an project related sample). The following table lists the analytes that did not meet the 35% relative percent difference (RPD) criteria.

Analyte	Client ID	RPD
Mercury	SSS-04C-03	38
Mercury	SSS-12A-01	50

The laboratory mercury duplicate analysis of sample SSS-04C-03 reported values of 89 mg/kg dry for the parent sample and 61 mg/kg dry for the duplicate analysis, with an RPD of 38. The laboratory mercury duplicate analysis of sample SSS-12A-01 reported values of 18 mg/kg dry for the parent sample and 30 mg/kg dry for the duplicate analysis, with an RPD of 50. The usability of the mercury data obtained from the analysis of samples SSS-04C-03 and SSS-12A-01 is qualified according to the following criteria:

- The reported results are considered estimated values (J) at maximum concentrations of 89 mg/kg dry and 30 mg/kg dry, respectively.

Laboratory Triplicate Precision

The thirteen soil samples associated with task number 940808-02 were analyzed in triplicate to determine the precision of the method in the sample matrix. The results displayed excellent correlation between the replicates. The recoveries obtained from the matrix spike/matrix spike duplicate, as well as the laboratory duplicate; associated with the quality control batch, were within the recommended control criteria. The results obtained were used to determine that triplicate analysis was not necessary for soil samples analyzed for mercury by EPA SW-846 Method 7471.

Field Duplicate Precision

There were six field duplicate samples associated with the samples in these task numbers: SSS-04C-03/SSS-04C-01, SSS-09B-03/SSS-09B-01, SSS-08C-03/SSS-08C-02, SSS-07A-03/SSS-07A-01, SSS-06C-03/SSS-06C-01, and SSW-005-02/SSW-005-01. The following table lists the analytes that did not meet the 50% RPD criteria for soils.

Analyte	Client ID	RPD
Mercury	SSS-04C-03/SSS-04C-01	51
Mercury	SSS-06C-03/SSS-06C-001	160

The field duplicate results from field duplicate samples SSS-04C-01/SSS-04C-03 and SSS-06C-03/SSS-06C-01 were applied to the validated "SSS- site soil" samples. The positive and nondetect mercury results in all the 1994 "SSS" samples are considered estimated (J and UJ,

respectively). The results of these two sets of duplicates are usable as estimated values at the maximum concentrations reported.

Laboratory Control Sample Recoveries

The laboratory control sample analyzed for mercury reported recoveries that were within the recommended criteria. All samples were analyzed within the method specified criteria.

Laboratory Mercury Performance Study

The laboratory Buffalo River Sediment sample, associated with the preparation batches; reported recoveries that were within the acceptable criteria. Ten BRS replicates were analyzed on the sixth of October, 1994 for a precision and accuracy study.

Holding Times

All mercury analyses associated with these task numbers were conducted within the specified holding time.

Overall Assessment

The overall precision and accuracy, as determined by QC checks including laboratory duplicates, laboratory control samples, and matrix spike samples, were acceptable for these data with the following exceptions.

The reported mercury results for the "SSS- site soil" samples are estimated based on individual matrix spike and duplicate analysis data.

4.2.2 Volatile Organic Analysis

No target compounds were detected in any of the laboratory method blanks or in the trip blank TB815A. Chloropicrin was detected in two of the environmental samples SSS-09B-02 and SSS-11B-02. The other six environmental samples were reported as not detected for all other target compounds. It should be noted that chloropicrin was included as a volatile organic target analyte. Chloropicrin results should be considered estimated values, as a detailed method validation study was not performed. Because an EPA-approved analytical method for chloropicrin in soils is not available.

All VOC surrogate spike percent recoveries were within the recommended criteria. MS/MSD data was reviewed. All MS/MSD recoveries were within the acceptable percent recovery and relative percent difference criteria.

There was one field duplicate sample associated with the samples in these task numbers: SSS-09A-03/SSS-09A-02. The field duplicate results were within the recommended criteria.

The method specifies a holding time of fourteen days beyond the collection time for sample analysis. All samples were analyzed within specified hold time, with the exception of sample SSW-010-01 which was analyzed one day out of hold time. The positive chloroform and carbon tetrachloride results are considered estimated (J). The results are usable as estimated values, but may be biased low due to possible sample degradation.

4.2.3 Total Organic Carbon Analysis

The Total Organic Carbon (TOC) analyses were performed by Aquatec Laboratories. No significant concentrations were detected in the laboratory method blank.

The reported laboratory control sample percent recovery was all acceptable.

All total organic carbon analyses were conducted within the holding times specified in the method.

4.3 1995 Soil and Sediment Data

Laboratory quality control data associated with all samples submitted from the HoltraChem site in Orrington, Maine for the sampling conducted from the third of May to the fifth of May, the sixth and the seventh of May, 1995 is summarized in this section. Associated SDGs include CDM Laboratory task #: 950505-03, 950608-02, and 950608-03. The samples were analyzed by the CDM Laboratory of Cambridge, Massachusetts; GeoTesting Express of Acton, Massachusetts; and by ITS-Aquatec Laboratories, of Colchester, Vermont. The samples were analyzed for mercury, volatiles organic compounds, cyanide, grain size and TOC. The samples were analyzed according to the analytical methods described in Test Methods for Evaluating Solid Waste Physical/Chemical Methods, SW-846 November 1990. This data usability assessment is based on the review of the following quality control information:

- Laboratory Method Blanks

- Matrix Spike/Matrix Spike Duplicates
- Surrogate Spike Recovery
- Laboratory Duplicates
- Laboratory Quality Control Samples
- Sample Holding Time Data

Three field blank samples, three equipment blank samples and nine field duplicates were included in these sample delivery groups.

All applicable quality control data were evaluated on a per analysis basis and are summarized below by analytical parameter. Hold time and field blank data are summarized in the data validation report. Approximately 20% of the sample analyses were also validated by Gradient Corporation, of Cambridge, MA. This review encompasses all of Gradient's comments. Gradient's data validation reports are also provided in the laboratory data package. This review was conducted in accordance with SW-846 data quality guidelines, USEPA CLP data validation criteria, and the Guidance for Data Useability in Risk Assessment, April 1992.

4.3.1 Mercury Analysis

Laboratory, Field, and Equipment Blanks

Mercury was not detected in any laboratory method or calibration blanks. No significant contamination was noted in any of the associated equipment blanks. Low levels of mercury, less than two times the detection limit, were detected in two of the field blanks (SSS-RFB-01 and SSS-RFB-02) associated with the soil samples collected the third to the fifth of May 1995.

Matrix Spike/Matrix Spike Duplicate Recoveries

There were eleven soil matrix spike/matrix spike duplicate (MS/MSD) analyses associated with the samples in these task numbers: SSS-03D-01, SSS-05E-02, SSS-07H-01, SSS-05F-03, SSS-06H-02, RSD-01A-01, RSD-07A-02, RSD-03A-03, RSD-07A-03, SSS-020, and RSD-06E-01.

The sample result for samples SSS-07H-01, SSS-05F-03, SSS-06H-02, and SSS-020 exceeded the spike concentration by a factor of four or more, no action is taken.

The following table lists the analytes that did not meet the 75-125% recovery criteria.

Analyte	Client ID	% Recovery (MS/MSD)
Mercury	SSS-03D-01	130/140
Mercury	SSS-05E-02	82/21
Mercury	RSD-07A-02	248/296

The MS/MSD results from samples SSS-03D-01 and SSS-05E-02 were applied to the associated "SSS- site soil" samples. The MS/MSD results from sample RSD-07A-02 were applied to the associated "RSD- river sediment" samples. The usability of the mercury data obtained from the analysis of these samples is qualified in accordance with the following requirements:

- All positive results and non-detected results are considered estimated values (J).

Laboratory Duplicate Precision

There were ten soil laboratory duplicate analyses associated with the samples in these task numbers: SSS-03D-02, SSS-07H-02, SSS-07I-01, SSS-06H-01, RSD-01A-02, RSD-01B-01, RSD-03B-02, RSD-07A-03, RSD-08B-02, and SSS-018. All duplicate results are within the recommended criteria.

Field Duplicate Precision

There were nine field duplicate samples associated with the samples in these task numbers: SSS-20A/SSS-19A, SSS-05F-03/SSS-05F-01, SSS-03F-03/SSS-03F-01, SSS-06E-03/SSS-06E-02, RSD-03A-03/RSD-03A-02, RSD-06B-03/RSD-06B-01, RSD-07A-03/RSD-07A-01, RSD-07D-03/RSD-07D-02, and RSD-08A-03/RSD-08A-01. The following table lists the analytes that did not meet the 50% RPD criteria.

Analyte	Client ID	RPD
Mercury	SSS-05F-03/SSS-05F-01	138
Mercury	RSD-06B-03/RSD-06B-01	52

Mercury	RSD-07A-03/RSD-07A-01	68
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The results from field duplicate samples SSS-05F-03/SSS-05F-01 were applied to the validated "SSS- site soil" samples. The field duplicate results from field duplicate sample RSD-07A-03/RSD-07A-01 and RSD-06B-03/RSD-06B-01 were applied to the validated "RSD- river sediment" samples. The usability of the mercury data obtained from the analysis of these samples is qualified in accordance with the following requirements:

- All positive results and non-detected results are considered estimated values (J).

Laboratory Control Sample Recoveries

The laboratory control samples analyzed for mercury reported recoveries that were within the recommended criteria. All samples were analyzed within the method specified criteria.

Holding Times

A holding time of 28 days from the time of sample collection to sample analysis was used for data usability purposes. All soil mercury analyses associated with these task numbers were conducted within the specified holding time.

The holding time was exceeded by 1 day for three equipment blank samples: EQB-01, EQB-02, and EQB-03 associated with these tasks. The results are usable as estimated values.

Overall Assessment

The overall precision and accuracy, as determined by QC checks including laboratory duplicates, laboratory control samples, and matrix spike samples, were acceptable for these data with the following exceptions.

The reported mercury results for the "SSS- site soil" and "RSD- river sediment" samples are estimated, based on individual matrix spike and duplicate results. All positive results and non-detected results are considered estimated values (J).

4.3.2 Volatile Organic Analysis

No target compounds were detected in any of the laboratory method blanks. Acetone was detected in environmental sample RSD-07B-02. The other four environmental samples were reported as not detected for all target compounds.

All VOC surrogate spike percent recoveries were within the recommended criteria. All MS/MSD recoveries were within the acceptable percent recovery and relative percent difference criteria.

The method specifies a holding time of fourteen days beyond the collection time for sample analysis. All samples were analyzed within specified hold time.

4.3.3 Total Organic Carbon Analysis

Associated Total Organic Carbon (TOC) quality control data was reviewed. The analyses were performed by Aquatec Laboratories.

There were three soil laboratory duplicate analyses associated with the samples in these task numbers: RSD-02A-01, RSD-05B-01, and RSD-07C-01. All duplicate results are within the recommended criteria.

There were five field duplicate samples associated with the samples in these task numbers: RSD-03A-03/RSD-03A-02, RSD-06B-03/RSD-06B-01, RSD-07A-03/RSD-07A-01, RSD-07D-03/RSD-07D-02, and RSD-08A-03/RSD-08A-01. The following table lists the analytes that did not meet the 50% RPD criteria.

Analyte	Client ID	RPD
TOC	RSD-03A-03/RSD-03A-02	~160+ (*)
TOC	RSD-08A-03/RSD-08A-01	66
TOC	RSD-06B-03/RSD-06B-01	115
TOC	RSD-07A-03/RSD-07A-01	~192+ (*)

(*) Estimated RPD

The field duplicate results from field duplicate sample RSD-03A-03/RSD-03A-02, RSD-07A-03/RSD-07A-01, RSD-08A-03/RSD-08A-01, and RSD-06B-03/RSD-06B-01 were applied to the validated "RSD- river sediment" samples. The positive and nondetect mercury results in all the "RSD" samples are considered estimated (J and UJ, respectively). The results are usable as estimated values.

All total organic carbon analyses were conducted within the holding times specified in the method.

4.4 1995 Groundwater and Surface Water Data

Laboratory quality control data associated with all aqueous samples submitted from the HoltraChem site in Orrington, Maine for the sampling conducted from the second to the eleventh of May, 1995; the sixth of June, 1995; and the seventh of August, 1995 is summarized in this report. Associated SDGs include CDM Laboratory task #: 950504-00, 950505-04, 950505-05, 950511-01, 950512-07, 950607-03 and 950808-01. The samples were analyzed by the CDM Laboratory of Cambridge, Massachusetts. The samples were analyzed for metals (iron, manganese, calcium, sodium, potassium, magnesium, and mercury), volatiles organic compounds, chloropicrin, sulfide, chloride, sulfate, total dissolved solids, and TOC. The samples were analyzed according to the analytical methods described in Test Methods for Evaluating Solid Waste Physical/Chemical Methods, SW-846 November 1990; and Methods for the determination of Organic Compounds in Drinking Water, Supplement 1, July 1990. This data usability assessment is based on the review of the following quality control information:

- Laboratory Method Blanks
- Matrix Spike/Matrix Spike Duplicates
- Surrogate Spike Recovery
- Laboratory Duplicates
- Laboratory Quality Control Samples
- Sample Holding Time Data

Project-generated quality control data was also reviewed. Two field blank, six trip blank, and one equipment blank samples were included in these sample delivery groups. In addition, six field duplicates were also included in these sample delivery groups.

All applicable quality control data were evaluated on a per analysis basis and are summarized below by analytical parameter. Hold time and field blank data are summarized. Approximately

20% of the sample analyses were also validated by Gradient Corporation, of Cambridge, MA. This review encompasses all of Gradient's comments. This review was conducted in accordance with SW-846 data quality guidelines, USEPA CLP data validation criteria, and the Guidance for Data Useability in Risk Assessment, April 1992. Gradient's data validation reports are provided with the laboratory data.

4.4.1 Mercury Analysis - Standard

Laboratory, Field, and Equipment Blanks

Mercury was not detected in any laboratory method or calibration blanks. No significant contamination was noted in any of the associated field or equipment blanks.

Matrix Spike/Matrix Spike Duplicate Recoveries

There were five aqueous matrix spike analyses associated with the samples in these tasks numbers: MW-404-01, SSS-RFB-001, B-320-02, MW-106-B1, and MW-409-01. Matrix spike recoveries were within the recommended criteria.

Laboratory Duplicate Precision

There were five aqueous laboratory duplicate analyses associated with the samples in these tasks numbers: MW-403-01, SSP-01, B-320-03, B-321-B2, and P-7. All duplicate results are within the recommended criteria.

Field Duplicate Precision

There were six field duplicate samples associated with the samples in these task numbers: MW-101-02/B-326-02, MW-103-01/MW-407-01, MW-104-B1/MW-407-B1, MW-105-B1/B-309-B1, MW-106-B1/MW-410-B1, and SSP-006/SSP-005. All field duplicate results are within the recommended criteria.

Laboratory Control Sample Recoveries

The laboratory control sample analyzed for mercury reported recoveries that were within the recommended criteria. All samples were analyzed within the method specified criteria.

Holding Times

All soil mercury analyses associated with these task numbers were conducted within the specified holding time, with the exceptions listed below. The holding times were exceeded by 1 to 7 days for mercury samples B-301-B1, B-301-01, B-301-02, B-303-B1, B-303-B2, B-303-B3, B-303-01, B-304-B1, B-306-B1, B-306-B2, B-307-B1, B-307-B2, B-309-B1, B-309-01, B-315-B1, B-315-01, B-315-02, B-321-B1, B-321-B2, B-327, MW-408-B1, MW-408-01, MW-409-B1, MW-409-01, MW-410-B1, MW-411-B1, P-3, P-4, P-7, P-11, P-13, FB-1, FB-2, MW-105-B1, and MW-106-B1. All associated mercury results are considered estimated (J).

Overall Assessment

The overall precision and accuracy, as determined by QC checks including laboratory duplicates, laboratory control samples, and matrix spike samples, were acceptable for these data. The following samples are qualified as estimated due to hold time violations: B-301-B1, B-301-01, B-301-02, B-303-B1, B-303-B2, B-303-B3, B-303-01, B-304-B1, B-306-B1, B-306-B2, B-307-B1, B-307-B2, B-309-B1, B-309-01, B-315-B1, B-315-01, B-315-02, B-321-B1, B-321-B2, B-327, MW-408-B1, MW-408-01, MW-409-B1, MW-409-01, MW-410-B1, MW-411-B1, P-3, P-4, P-7, P-11, P-13, FB-1, FB-2, MW-105-B1, and MW-106-B1.

4.4.2 Mercury Analysis - Low Detection Limit

This data usability assessment is based on the review of the laboratory quality control data associated with all samples submitted from the HoltraChem Manufacturing site in Orrington, Maine; for the sampling conducted on the sixth of June 1995. The associated SDG is referenced as BRL Tracking #: 95BR128. The samples were analyzed by the Brooks Rand Ltd. of Seattle, Washington; for mercury by Cold Vapor Atomic Fluorescence Spectroscopy (CVAFS).

All associated quality control information reported by the laboratory was reviewed. In general, all reported quality control information was acceptable, with laboratory duplicates and the analysis of certified reference material, NIST 1641C, all within control limits. The laboratory reported that method blank concentrations were about twice the normal concentration, but it is suspected that it is due to slight reagent contamination. This slight contamination should not affect the sample results, as the method blank concentrations are below the detected sample concentrations. The sample concentrations are reported as blank corrected concentrations. This data is accepted as reported by BRL without further qualification.

4.4.3 Metals Analysis: Iron, Manganese, Calcium, Magnesium, Potassium, and Sodium

Laboratory, Field, and Equipment Blanks

Iron, manganese, magnesium, and potassium were not detected in any laboratory method or calibration blanks. Low concentrations of calcium and magnesium, less than two times the detection limit, were detected in the laboratory method blanks. No significant contamination was noted in any of the associated field or equipment blanks for manganese and potassium. Traces of iron, calcium, magnesium, and sodium were detected, at less than three times the detection limit. No formal action is recommended for the associated samples.

Matrix Spike/Matrix Spike Duplicate Recoveries

There were four aqueous matrix spike analyses associated with the samples in these task numbers: MW-404-01, MW-409-01, P-4, and B-303-B1. Matrix spike recoveries were within the recommended criteria with the following exceptions: iron results for sample P-4; calcium, magnesium and sodium results for sample MW-409-01; and, calcium, magnesium, sodium, potassium, iron, and manganese results for sample MW-404-01. The native concentration detected for these elements, in samples MW-409-01, P-4, and MW-404-01 exceeded the spike concentration by a factor of four or more, no further action is taken.

Laboratory Duplicate Precision

There were five aqueous laboratory duplicate analyses associated with the samples in these task numbers: MW-403-B1, P-7, P-11, B-303-B2, and MW-102-01. All duplicate results are within the recommended criteria.

Field Duplicate Precision

There were four field duplicate samples associated with the samples in these task numbers: MW-103-01/MW-407-01, MW-104-B1/MW-407-B1, MW-105-B1/B-309-B1, and MW-106-B1/MW-410-B1. All field duplicate results are within the recommended criteria with the following exceptions: iron results for samples MW-103-01/MW-407-01, MW-104-B1/MW-407-B1, and MW-105-B1/B-309-B1.

The field duplicate results for iron from field duplicate samples MW-103-01/MW-407-01, MW-104-B1/MW-407-B1, and MW-105-B1/B-309-B1 were applied to the validated groundwater

samples. All associated iron results in the groundwater samples are considered estimated (J). The results are usable as estimated values.

Laboratory Control Sample Recoveries

The laboratory control sample analyzed for iron, manganese, calcium, magnesium, sodium, and potassium reported recoveries that were within the recommended criteria. All samples were analyzed within the method specified criteria.

Holding Times

The method specifies a holding time of six months beyond the collection time for sample analysis. All samples were analyzed within specified hold time.

4.4.4 Volatile Organic Analysis

Associated volatile organic quality control data was reviewed. No target compounds were detected in any of the laboratory method blanks, field blanks, or in the trip blanks.

All VOC surrogate spike percent recoveries are acceptable for useability criteria. All MS/MSD recoveries were within the acceptable percent recovery and relative percent difference criteria.

There were five field duplicate samples associated with the samples in these task numbers: MW-103-01/MW-407-01, MW-104-B1/MW-407-B1, MW-105-B1/B-309-B1, MW-106-B1/MW-410-B1, and SSP-006/SSP-005. The field duplicate results were within the recommended criteria.

The method specifies a holding time of fourteen days beyond the collection time for sample analysis. All samples were analyzed within specified hold time.

4.4.5 Chloropicrin Analysis

Chloropicrin analysis was performed by a modified Method 551. Associated chloropicrin quality control data was reviewed. No target compounds were detected in any of the laboratory method blanks or field blanks.

Chloropicrin surrogate spike percent recovery data was reviewed. All spike recoveries were within the recommended criteria. Laboratory duplicate samples analyzed with the environmental samples were within the recommended criteria.

There were four field duplicate samples associated with the samples in these task numbers: MW-103-01/MW-407-01, MW-104-B1/MW-407-B1, MW-105-B1/B-309-B1, and MW-106-B1/MW-410-B1. The field duplicate results were within the recommended criteria..

The laboratory control samples analyzed for chloropicrin reported recoveries that were within the recommended criteria. All samples were analyzed within the method specified criteria.

All chloropicrin analyses associated with these task numbers were conducted within the specified holding time, with the exceptions listed below. The following samples were extracted outside the recommended holding time: B-316-B1, B-320-B1, B-320-01, MW-401-B1, MW-401-01, MW-402-01, MW-404-01, MW-406-B1, P-7, P-13, MW-102-01, and MW-103-01. Due to high native concentrations of chloropicrin in these samples, they were extracted multiple times to obtain an adequate dilution that would be within the calibration range. This process prevented all the samples to be re-extracted within the recommended holding time. No further action is recommended for these samples.

4.4.6 Total Organic Carbon Analysis

No significant TOC concentrations were detected in the laboratory method blank. Low level amounts of TOC were detected in the two field blanks, less than two times the detection limit. No formal action is recommended for the associated samples.

The reported laboratory control samples percent recovery were within the recommended criteria.

There were four field duplicate samples associated with the samples in these task numbers: MW-103-01/MW-407-01, MW-104-B1/MW-407-B1, MW-105-B1/B-309-B1, and MW-106-B1/MW-410-B1. All field duplicate results are within the recommended criteria. It should be noted that although trace amounts of TOC were detected in some of these samples, the levels were close to the detection limit. All samples were diluted 10X to minimize matrix interferences.

All total organic carbon analyses were conducted within the holding times specified in the method.

4.4.7 Conventional Wet Chemistry Analyses

Laboratory, Field, and Equipment Blanks

Associated quality control data was reviewed for sulfide, chloride, sulfate, and total dissolved solids (TDS) analyses. Sulfide, chloride, and sulfate were not detected in any laboratory method or calibration blanks. Low concentrations of TDS were detected in the laboratory method blanks and field blanks. No significant contamination was noted in any of the associated field or equipment blanks for these parameters. No formal action is recommended for the associated samples.

Matrix Spike/Matrix Spike Duplicate Recoveries

Matrix spike recovery data was reviewed for chloride and sulfate. Matrix spike recoveries were within the recommended criteria.

Laboratory Duplicate Precision

Laboratory duplicate data was reviewed for chloride, sulfide, total dissolved solids, and sulfate. All duplicate results are within the recommended criteria.

Field Duplicate Precision

There were four field duplicate samples associated with the samples in these task numbers: MW-103-01/MW-407-01, MW-104-B1/MW-407-B1, MW-105-B1/B-309-B1, and MW-106-B1/MW-410-B1. All field duplicate results are within the recommended criteria.

Laboratory Control Sample Recoveries

The laboratory control samples analyzed reported recoveries that were within the recommended criteria. All samples were analyzed within the method specified criteria.

Holding Times

All samples were analyzed within specified hold time.

4.5 1997 Soil, Sediment, Surface, and Groundwater Data

This subsection assess all sample analyses conducted in 1997. The CDM Laboratory, Cambridge, MA was the primary laboratory utilized. The Brooks Rand Ltd. laboratory provided analysis of biota samples as well as aqueous, low level mercury analyses by EPA Method 1631. All data was reviewed by Gradient Corporation of Cambridge, MA. A combination of a Modified Tier I and selected Tier III data validations were performed. Criteria for acceptability of data were based on the Region I Tiered Inorganic and Organic Data Validation Guidelines, July 1, 1993; the Region I Data Validation Functional Guidelines for Evaluating Inorganic Analyses, November 1988, and Gradient's professional judgement.

The following table provides a list of the laboratory reports reviewed. The remainder of this section provides a summary of data usability issues as determined by Gradient Corporation on a report by report basis.

1997 Soil, Sediment, Surface Water and Groundwater Data

SDG #	Validation Type	Sample Types	Parameters Analyzed
970815-26	Modified Tier I	1 Soil	Volatiles/Mercury
970815-27	Modified Tier I	13 Surface Soils	Mercury
970815-28	Modified Tier I Tier III	18 Sediment 70 Sediment/3 Aqueous	Mercury Mercury
970822-62	Modified Tier I Tier III	25 Surface Soil/2 Aqueous 6 Surface Soil	Mercury Mercury
970908-146	Modified Tier I	10 Soils	Mercury
971003-235	Modified Tier I Modified Tier I	33 Monitoring Wells 33 Monitoring Wells/1 trip blk	Mercury Volatiles plus chloropicrin
971009-248	Modified Tier I Tier III	26 Mon. wells/surface waters 7 Monitoring Wells	Mercury Mercury
971009-248	Modified Tier I Tier III	24 MW,3 SW, 1 trip blank 8 Monitoring Wells/1 trip blk	Volatiles plus chloropicrin Volatiles plus chloropicrin

971009-249	Tier III Tier III	9 Residential Wells/3 Sur. Soil 9 Residential Wells	VOC, Metals, WetChem, Bromide
971009-250	Tier III	8 Surface Soils/1 Aqueous 4 Surface Soils	Mercury RCRA Metals
971120-404	Modified Tier I Modified Tier I	1 Soil 1 Soil	Volatiles Mercury
97BR158	Modified Tier I Tier III	43 River Surface Waters 12 River Surface Waters	Mercury Mercury
97BR217	Modified Tier I	5 Biota(earthworm) Samples	Mercury

SDG: 970815-26
Sampling Date: 8/14/97
Soil Matrix: Hg, VOC & chloropicrin

All data met site specific DQOs and are usable for all intended purposes without qualification.

SDG: 970815-27
Sampling Date: 8/14/97
Soil Matrix: Hg

All data met site specific DQOs and are usable for all intended purposes without qualification.

SDG: 970815-28
Sampling Dates: 8/12/97, 8/13/97, 8/14/97
Sediment Matrix: Hg

All data met site specific DQOs and are usable for all intended purposes with qualification. Mercury results are considered estimated (J) due to sample percent solids less than 30% and poor field duplicate precision. The qualified mercury results are usable as estimated values. All sediment samples with the exception of the following field duplicate pair samples are considered estimated (J): RSD-09B-01/RSD-09B-10, RSD-10C-01/RSD-10C-10, RSD-11E-02/RSD-11E-10, RSD-12D-02/RSD-12D-10, RSD-13C-02/RSD-13C-10, RSD-14A-01/RSD-14A-10, RSD-15B-01/RSD-15B-10.

SDG: 970822-62

Sampling Dates: 8/15/97, 8/18/97, 8/20/97

Soil Matrix: Hg

All data met site specific DQOs and are usable for all intended purposes without qualification with the exception of the mercury results for samples SSS-052-10 and SSS-052-01, which are considered estimated (J) due to the reported concentration exceeding the instrument calibration range. These results are usable for all intended purposes as estimated values.

SDG: 970908-146

Sampling Dates: 8/20/97, 8/28/97, 9/4/97

Soil Matrix: Hg

All data met site specific DQOs and are usable for all intended purposes without qualification.

SDG: 971009-235

Sampling Dates: 9/30/97, 10/1/97, 10/2/97

Aqueous Matrix: Hg, VOCs & chloropicrin

All data met site specific DQOs and are usable for all intended purposes without qualification with the exception of the mercury results for sample B-315-01, which is considered estimated due to the reported concentration exceeding the instrument calibration range. These results are usable for all intended purposes as an estimated value. Gradient Corporation also qualified as estimated the mercury results for B-306-B2 and MW-403-02. The results were qualified due to their reported mercury concentrations being greater than the reporting limit but less than 2X the reporting limit, which increases the uncertainty of the reported concentrations. The qualified results are usable as estimated values.

Gradient Corporation also qualified the VOC and chloropicrin results as estimated (J) for sample B-315-B1 due to a potential bias due to high surrogate recovery. These results are usable for all intended purposes as estimated values.

SDG: 971009-248

Sampling Dates: 10/6/97, 10/7/97, & 10/9/97

Aqueous Matrix: Hg, VOCs & chloropicrin

All data met site specific DQOs and are usable for all intended purposes with qualification. Total and dissolved mercury results are considered estimated (J) due to poor field duplicate

precision which is an indication of potential sample heterogeneity. The qualified mercury results are usable as estimated values. All positive total mercury results with the exception of the following samples are considered estimated: B-320-B1, B-326-01, and B-326-01-50. All positive dissolved mercury results with the exception of the following samples are considered estimated: B-320-B1-50 & MW-503-01-50.

The nondetect VOC and chloropicrin results reported for sample B-327-01 are considered to be unusable due to low surrogate recovery and were flagged with a R. This data is not usable for any project decisions. The positive VOC and chloropicrin results for sample B-327-01 are qualified as estimated. In addition, the VOC and chloropicrin results reported for sample MW-509-B1 are qualified as estimated values and are flagged with a J. Also, the positive acetone results reported for samples B-327-01 and MW-509-B1 are considered to be non-detected at the reported concentrations due to blank contamination.

SDG: 971009-249

Sampling Dates: 10/7/97

Aqueous Matrix: Hg, VOCs & chloropicrin

All data met site specific DQOs and are usable for all intended purposes without qualification. It should be noted that the reanalysis results for the VOC and chloropicrin analyses for samples RWS-004-02, RWS-006-01, RWS-007-01, and RWS-008-01 are to be used for all project decision making.

SDG: 971009-250

Sampling Dates: 10/7/97

Soil Matrix: metals including mercury

All data met site specific DQOs and are usable for all intended purposes with qualification. Total mercury results for the sediment samples were estimated (J) due to a continuing calibration outside of acceptable limits.

SDG: 971120-404

Sampling Date: 11/19/97

Soil Matrix: metals including mercury

All data met site specific DQOs and are usable for all intended purposes without qualification.

SDG: 97BR158

Sampling Dates: 8/18/97, 8/19/97

Aqueous Matrix: low level mercury

All data met site specific DQOs and are usable for all intended purposes without qualification.

SDG: 97BR217

Sampling Date: 10/27/97

Biota Matrix: low level mercury

All data met site specific DQOs and are usable for all intended purposes without qualification.

4.6 1998 Water and Soil Sample Data

Laboratory quality control data associated with all sample submittals from the HoltraChem site in Orrington Maine for sampling conducted from January through October 1998 was reviewed and is summarized in this subsection. A tabulation of associated samples is presented below.

Summary of samples included in this data quality assessment:

SDG #	Sampling Dates	Sample Types	Parameters Analyzed
980202-81	1/27 - 1/28/98	8 Waters (RWS)	Metals TOC Bromide WetChem
980202-82	1/29/98	2 Waters (MW)	VOC Metals TOC Bromide WetChem Sulfide
980202-83	1/28-1/28/98	5 Waters (MW)	Metals TOC Bromide WetChem
980430-472	4/27/98	13 Waters (B,MW,RWS)	Metals TOC WetChem
980501-485	4/28/98	1 Water (SSS)	Metals (Hg)

980729-1051	7/20 - 7/21/98	13 Waters (B,MW,RWS)	Metals TOC Bromide WetChem
980730-1053	7/21 - 7/23/98	9 Waters (B,MW,TB)	VOC Metals Bromide WetChem
980730-1056	7/21 - 7/23/98	13 Waters (B, MW)	Metals (Hg,Fe,Mn)
980731-1062	7/27 -7/28/98	7 Waters (B, MW)	VOC, Metals, WetChem, Bromide
980804-1071	7/29/98	3 Soils (SSS)	Metals (Hg)
980804-1073	7/28 - 7/29/98	2 Soils (RSD)	Metals (Hg)
980821-1169	8/17 -8/18/98	4 Soils (LF)	Metals (Hg)
980911-1263	7/29/98	2 Soils (SSS)	Metals (Hg)
980916-1295	9/15 - 9/16/98	9 Waters (SSS)	Metals , WetChem
981001-1369	9/15/98	4 Waters (SSS)	Metals (Hg)
980914-1275	9/14/98	2 Soils (SSS)	TOC
981005-1392	7/29/98	1 Soil	TOC
981009-1422	10/7/98	19 Soils (LPL)	Metals (Hg)

Notes:

- Metals= Ca, Fe, Mg, Mn, K, Na, Hg; unless denoted otherwise.
- Bromide analysis was conducted by the University of Maine. Analytical results were reported with insufficient QC data to perform an assessment. One group of water samples was analyzed by CDM; in this instance all QC data was within acceptance limits.

The following data assessment is based on review of the analytical reports and the QC data contained therein; specifically:

- Laboratory Blank Results
- Matrix Spike Recovery
- Surrogate Spike Recovery (for volatiles)

CDM Camp Dresser & McKee

December 22, 1998
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- Laboratory Duplicate Precision
- Laboratory Quality Control Samples
- Sample Holding Time Data

All applicable quality control data were evaluated and are summarized below by analytical parameter.

4.6.1 Trace Metals (ICP)

Samples were analyzed either for a series of metals (Fe, Ca, Mg, Mn, K, Na, and Hg), a reduced series of metals (Fe, Mn), or just Hg alone. For some SDGs dissolved as well as total metal concentrations were determined. QC data was reviewed for all of the analytical batches. The QC test parameters encompassed the same parameter set for which the associated samples were being tested. Any excursions will be presented along with what samples and what specific metal analytes are affected.

Blanks

No target analyte was detected at or above the method reporting limit (MRL)

Matrix Spike /Spike Duplicates

Laboratory #	ID	Comments
98-00363	Not a project related sample	Fe,Mg,Mn,K,Na excursions discussed below
98-02378	B-321-B1	Ca, Mg, K, Na excursions discussed below
98-06086	MW-511-B1	Ca, Mg, K, Na excursions discussed below

Recoveries for matrix spiked samples all with acceptance limits (80 - 120 %R) with the following exception.

SDG 980202-81 Sample RWS-011-01(98-00363); Iron, Manganese Magnesium, Potassium were all over- recovered and Sodium under- recovered. It was noted that these excursions may be

attributed to high sample concentrations relative to the spike amount, sample concentration greatly exceeded the spike amount. No action is recommended.

Analyte	Sample Conc. ug/L	%R
Iron	28000	750
Manganese	18000	500
Magnesium	22000	165
Potassium	2000	127
Sodium	140000	0

SDG 980430-472 Sample B-321-B1; Calcium, Potassium and Sodium were all over-recovered and Magnesium under-recovered. It was noted in the report narrative that these excursions may be attributed to high sample concentrations relative to the spike amount. No action is recommended.

Analyte	Sample Conc. ug/L	%R
Calcium	290000	450
Magnesium	36000	40
Potassium	7200	136
Sodium	510000	810

SDG 980729-1051 Sample MW-511-B1; Calcium, Potassium, Sodium and Magnesium were under-recovered. It was noted that these excursions may be attributed to high sample concentrations relative to the spike amount. No action is recommended.

Control Samples (LCS or QCS)

At least one LCS or QCS was processed with each analytical batch. All results were within acceptance limits (80-120%R).

Laboratory Duplicates

Laboratory #	ID	Comments
98-00363	RWS-011-01	All RPD < 20%
98-00371	8-8-01	All RPD < 20%
98-02378	B-321-B1	All RPD < 20%
98-06086	MW-511-B1	All RPD < 20%
98-06102	MW-506-B1	All RPD < 20%
98-06153	P-13	Na: RPD <20%
98-06159	MW-503-01	Fe, Mn RPD < 20%

All within limit of 0 -20 % Relative Percent Difference (RPD)

Hold times

All sample ICP metals analyses were conducted within the EPA maximum allowed holding time of 180 days.

Dilutions

Initial sodium analyses for samples MW-501-O1, B-326-O2 and MW-101 (duplicate of B-326-O2) were estimated because the concentrations were outside the instrument quantification limits. Subsequent diluted analyses were performed, however, the reported results of these analyses at the recorded dilutions are inconsistent with the initial analyses and with other samples collected from these wells. Therefore, the results of the initial analyses should be used for all project work and should be considered estimated concentrations.

4.6.2 Trace Metals (Hg)

Samples were analyzed either for a series of metals (Fe, Ca, Mg, Mn, K, Na, and Hg) or just Hg alone. For some SDGs dissolved as well as total metal concentrations were determined. QC data was reviewed for all of the analytical batches. Any excursions will be presented along with what samples are affected.

Blanks

No target analyte was detected at or above the Method reporting limit (MRL)

Matrix Spike /Spike Duplicates

Laboratory #	ID	Comments
98-00364	RWS-001-03	OK 80 -120 %Recovered
98-00371	8-8-01	OK
98-02378	B-321-B1	OK
98-02379	B-321-B2	Diss.Hg: OK
98-06086	MW-511-B1	OK
98-06089	B-321-B2	Diss Hg OK
98-06098	RWS-011-03	OK
98-06101	MW-410-B1	OK
98-06119	MW-405-B1	OK
98-06126	MW-508-01	OK
98-06153	P-13	OK
98-06605	LF8-TP2-01 (soil)	OK
98-07755	LPL-005-06	OK

Control Samples (LCS or QCS)

All analyzed control samples were within acceptance limits (80-120%R)

Laboratory Duplicates

Laboratory #	ID	Comments
98-00363	RWS-011-01	RPD < 20%
98-00371	8-8-01	RPD < 20%
98-00372	MW-505-B1	RPD < 20%
98-00376	B-321-B2	RPD < 20%
98-02378	B-321-B1	RPD < 20%
98-02379	B-321-B2(4/27/98)	Diss. Hg : RPD < 20%
98-02171	not a project related sample	RPD < 20%
98-06086	MW-511-B1	RPD <20%
98-06089	B-321-B2	Diss.Hg RPD <20%
98-06098	RWS-011-03	RPD <20%
98-06101	MW-410-B1	RPD < 20%
98-06266	not a project related sample	RPD < 20%
98-06119	MW-405-B1	RPD < 20%
98-06126	MW-508-01	RPD < 20%
98-06153	P-13	RPD < 20%
98-06159	MW-503-01	RPD < 20%
98-06184	SSS-71 (soil)	RPD < 20%
98-06605	LF8-TP2-01	RPD < 20%
98-06841	not a project related sample	RPD < 20%
98-07117	not a project related sample	RPD < 20%
98-07755	LPL-005-06	RPD < 20%

All laboratory duplicate analyses were within limit of 0 -20 % Relative Percent Difference (RPD)

Hold times

All sample analyses were conducted within the EPA recommended holding time of 28 days, with the following exception.

Samples SSS-071 and SSS-072 were analyzed under SDG 980911-1263 beyond the holding time relative to sample collection to get a baseline value for the desorption study. These samples were originally analyzed under SDG 980804-1071 within the holding time specification and no further action is recommended.

4.6.3 Volatiles

Blank

No target analyte was detected at or above the method reporting limit (MRL) with the following exception:

Two blanks were analyzed as part of batch I980801/8260A_AQU/306 analyzed on 7/31/98 the first blank was ND, however the second blank B98-04715 analyzed 7/31/98 contained acetone at 130 ug/l, trichloroethene at 3.3 ug/l and cis-1,2-dischloroethene at 4.9 ug/l. Two samples were associated with this blank : MW-506-B1, MW-409-01. The positive detect for TCE in MW-506-B1 should be qualified as Not Detected at that concentration (UJ @ 1.2 ug/l); all other of these compounds were not detected in these samples and no qualification is needed.

Two blanks were analyzed as part of batch I980810/8260A_AQU/313 analyzed on 8/7/98; the first blank was ND, however the second blank B98-04927, analyzed with diluted samples, contained acetone at 110 ug/l. No acetone was detected in the affected samples at or above the given reporting limit: No action taken.

MS/MSD

Laboratory #	ID	Comments
98-00371	8-8-01	OK 80 - 120 % Recovery

98-06015	not a project related sample	OK
98-06106	MW-401-01	OK
98-06108	MW-512-01	OK
98-06153	P-13	OK except Chloropicrin see below
98-06158	8-8	OK

Matrix spike recovery of chloropicrin in sample P-13 was 140% with acceptance limits of 80 - 120%; the positive detect for chloropicrin in this sample is considered estimated (J).

Laboratory Duplicate Precision

Laboratory #	ID	Comments
98-06102	MW-506-B1	ALL RPD < 20%
98-06107	MW-101	ALL RPD < 20% except excursions noted below
98-06159	MW-503-01	ALL RPD < 20%

Duplicate D98-06107 had RPD in excess of 20% for Bromodichloromethane (39%), Bromoform (34%), Carbon Disulfide (65%), Chloroform (37%), Chloropicrin (72%), Dibromochloromethane (40%); Results for these compounds in this sample are considered estimated values.

Laboratory Control Samples

All laboratory control samples analyzed as part of this investigation met acceptance limits.

Holding Times

All volatiles analyses were conducted within the EPA maximum allowed holding time of 14 days from time of sample collection.

4.6.4 Wet Chemistry

Wet Chemistry testing encompasses TDS, Chloride, Sulfate, Carbonate-Alkalinity and Bicarbonate-Alkalinity. Data quality assessment of TOC and Sulfide results is also contained in this section. Analysis of soils for TOC was performed by Hudson Environmental Services; insufficient QC data was reported to assess the data.

Blank

No target analyte was detected at or above the method reporting limit (MRL) with the following exception:

The TDS blank associated with batch I980805/2540C_AQU/45 yielded 5.0 mg/L TDS at an MRL of 5.0mg/L. The associated samples

MW-511-B1 MW-511-B2 B-321-B1 B-321-B2 MW-505-B1 MW-505-B2
 RWS-004-04 RWS-0005-05 RWS-003-05 RWS-007-04 RWS-006-03 RWS-001-05
 RWS-011-03 MW-410-B1 MW-506-B1 MW-409-01 MW-501-01 B-326-02
 MW-401-01 MW-101

all contained TDS at greater than 10 x MW-512-01 the blank concentration. No action taken.

MS/MSD

Laboratory #	ID	Comments
98-00364	RWS-001-03	Cl, SO4: OK
98-00363	RWS-011-01	TOC: OK
98-02382	MW-505-B1	Cl : 0%R, SO4: OK
98-02455	not a project related sample	SO4: OK
98-06086	MW-511-B1	SO4, Cl: OK
98-06153	P-13	Cl : OK
98-06094	RWS- 003-05	TOC : OK

Chloride spike of sample MW-505-B1 diluted out due to high native concentration. No action recommended.

Laboratory Duplicate Precision

Laboratory #	ID	Comments
98-00363	RWS-011-01	TOC, TDS: RPD < 20%
98-00364	RWS-001-03	Cl, SO4: RPD < 20%
98-00376	B-321-B2	TOC, Alkalinity: RPD <20%
98-00371	8-8-01	Sulfide: RPD <20%
98-02378	B-321-B1	TDS, Alk: RPD < 20%
98-02382	MW-505-B1	Cl, SO4 : RPD < 20%
98-02390	RWS-005-04	TOC: RPD < 20%
98-02455	not a project related sample	SO4: RPD < 20%
98-04112	not a project related sample	TOC: RPD < 20%
98-06086	MW-511-B1	TDS,SO4, Cl <20%
98-06101	MW-410-B1	TDS: RPD <20%
98-06153	P-13	Cl: RPD <20%
98-06998	not a project related sample	TDS: RPD < 20%

All analyzed laboratory duplicates met the acceptance limits of less than 20 % RPD.

Laboratory Control Samples

All control samples analyzed as part of this investigation met acceptance limits with the following exception:

Alkalinity carbonate 141%R at 12 mg/L with MRL of 5mg/L; Bicarbonate alkalinity and total alkalinity were both within 10% of the expected values (120 mg/L and 140 mg/L respectively).

Affected samples : RWS-011-01, RWS-001-03, RWS-010-01, RWS-002-03, RWS-003-03, RWS-004-03, RWS-00-03, RWS-007-07, 8-8-01, MW-505-B1, MW-505-B2, MW-511-B2, B-321-B1, B321-B2. Because the QCS concentration was just over 2x the MRL and the other alkalinity QCS values met the accuracy criteria - No action is taken regarding these data.

Holding Times

All sample analyses were conducted within the EPA Maximum Allowable Holding Times with the following exception:

pH analysis of water samples collected 4/27/98 and 4/28/98 was conducted on 4/30/98; these samples were received at the CDM laboratory on 4/29/98. Affected samples: B-321-B1, B321-B2, MW-511-B1, MW-511-B2, MW-505-B1, MW-505-B2, RWS-001-04, RWS-006, RWS-007-03, RWS-004-04, RWS-011-02, RWS-003-04, RWS-005-05. These laboratory data are considered estimated values. pH is also measured in the field at the time the samples are collected.

TDS analyses of samples received under SDG 980729-1051 were analyzed two or three days beyond the EPA maximum allowed holding time of seven days from sample collection. Affected samples : MW-511-B1, MW-511-B2, B-321-B1, B-321-B2, MW-505-B1, MW-505-B2, RWS-004-04, RWS-0005-05, RWS-003-05, RWS-007-04, RWS-006-03, RWS-001-05, RWS-011-03. These data are considered estimated values.

The analyses of samples received under SDG 980729-1051 for pH was conducted on the date that the samples were received at the laboratory, but this was three and four days after the samples were collected. Affected samples: MW-511-B1, MW-511-B2, B-321-B1, B-321-B2, MW-505-B1, MW-505B2, RWS-004-04, RWS-005-05, RWS-003-05, RWS-007-04, RWS-006-03, RWS-001-05, RWS-011-03. These laboratory data are considered estimated values. pH is also measured in the field at the time the samples are collected.

TDS analyses of four samples received under SDG 980730-1053 were analyzed one to two days beyond the EPA maximum allowed holding time of seven days from sample collection. Affected samples: MW-410-B1, MW-506-B1, MW-409-01, MW-501-01. These laboratory data and are considered estimated values.

Section 5

Environmental Setting

5.1 Introduction

This section provides a description of the environmental setting of the site including the regional and site specific geology, hydrology and hydrogeology. The environmental setting provides the framework for evaluating the distribution of contamination and the fate and transport of contaminants.

5.1.1 Geologic History

The regional geology of the Orrington area is a product of geologic processes dominated by plate tectonics and glaciation.

The sediments which formed the rocks of the Orrington area were deposited in a sea called Iapetus which existed in the area approximately 400 million years ago (MY). The North American and European (Avalonian) continental plates were converging during Ordovician and Silurian periods (500-430 MY and 430-390 MY respectively). The sediments deposited between the two converging continental plates were then compressed and altered (metamorphosed) as the two plates collided during the Devonian period (390-345 MY). Granitic rock was formed at greater depths and the rising molten rock created the intrusive bodies which form some of the nearby mountains. The collision of the two plates thrust older rock over younger rock south of the Norumbega fault, which is located approximately two miles south of the HoltraChem Site. When the two plates split at what is now the Atlantic Ocean, part of the Avalonian plate (located between the Norumbega fault and the present coast) was left behind.

The present topography of the site and the overburden deposits are primarily a result of relatively recent glacial activity. The last major glacial event was the Wisconsin-age continental glaciation which is believed to have reached its maximum extent approximately 40,000 years ago. At this point the ice was more than 10,000 feet thick and extended to George's Bank, approximately 180 miles to the southeast.

As the glaciers melted, the load of rock and soil they transported was deposited on land as glacial tills and ice contact deposits. The material ground beneath the glacial ice resulted in a very dense layer known as lodgement till or hardpan. The material carried along on and in the ice which was

deposited as the ice melted is known as ablation till. Glacial tills ground beneath or dropped by glacial ice typically form hummocky terrain.

The amount of water bound in glacial ice lowered sea level to almost 400 feet below its current position. At the same time, the weight of the ice depressed the earth's crust beneath the glaciers. The ice began to retreat approximately 20,000 years ago. By about 14,000 years ago, the edge of glacial ice was just offshore of the Maine coastline. As the glaciers melted, sea level rose and the earth's crust rebounded. However, isostatic rebound was slower than sea level rise, and the sea transgressed far inland of the current coastline. Glacially derived sediments were deposited in the near shore sea. The glacio-marine sediments have been found as far inland as Medway (north of Bangor) on the Penobscot River and as high as 400 feet above current sea level. The Orrington area was inundated after the glacial ice melted, and marine sediments are found locally on top of glacial tills.

As the earth's crust continued to rebound, the sea receded and present day coastal Maine emerged from the water. During the emergence, the sea reworked some of the marine sediments and glacial tills. Wave action winnowed fine grained material (silt and clay) from glacio-marine deposits and glacial till and redistributed sand and gravel. Some of the onsite and local sand and gravel deposits may be a result of these processes. After the sea retreated, the processes of erosion from freezing and thawing, wind and rain continued to work on the landscape to bring it to its present day appearance.

5.1.2 Bedrock Units

Rock Type

The bedrock geology of the Orrington region is characterized by a northeasterly trending suite of Ordovician/Silurian-age metasediments (metamorphosed sedimentary rock) referred to as the Vassalboro Formation. The composition of the sedimentary rock from which they were formed was typically calcareous sandstone, and interbedded sandstone and impure limestone. The Vassalboro Formation is bordered to the south by the Devonian-age metasediments of the Bucksport Formation. Devonian intrusive rocks that are common to the Winterport and Mt. Waldo granitic plutons are found to the south and east of the site area.

The metasedimentary strata in the Orrington region are believed to have been subjected to two deformational events: Acadian folding and late stage Acadian orogenic (mountain building) movement and plutonic emplacement. The present mineralogic assemblages and dominant structural trends of the metasediments are interpreted to be the result of the orogenic events. The degree of metamorphism is moderate, and is characterized as the greenschist facies due to the predominance of the green mineral chlorite.

Topography

The bedrock topography of the Orrington area is a result of faulting and folding, uplifting and downwarping of the bedrock by tectonic forces and intrusion of the granitic rock. The forces of glacial ice, freeze-thaw, wind and water preferentially eroded softer rock and had less effect on more resistant rock. The Maine coastal region south of Bangor is typified by relief of less than 300 feet except for the more resistant granitic intrusions which form the mountains along the coast south of the site.

Structure

Upward folding of local rock resulted in the Liberty-Orrington antiform. This antiform and the rocks at the site have been cut by the Norumbega fault and its associated splays. The main zone of the fault is located approximately two miles south of the site area, but a splay cuts across the site.

Fracture trace analysis performed by Acheron identified three prominent lineament directions: N72-85W, N45-52W, and N28E (see Figure 4-6, Appendix B). These lineament directions are in general agreement with the regional lineaments depicted on the regional photolinear map (Figure 4-5, Appendix B).

5.1.3 Overburden Units

The surficial geology of the Orrington region reflects the region's glacial and post-glacial depositional history.

Glacial deposits include lodgement and ablation till, and ice-contact deposits including eskers and kames. Glacial tills typically contain a wide variety of grain sizes from clay and silt up to boulders. Tills generally exhibit low hydraulic conductivity (permeability) due to their high content of fine grained material (silt and clay). Typical hydraulic conductivities range from $10 \text{ E-}7$ to $10 \text{ E-}5$ cm/sec. Lodgement till typically contains more clay and is usually denser than ablation till due to its compaction beneath the great weight of the ice. Because of its density and high percentage of fine sized particles, lodgement till typically exhibits very low hydraulic conductivity. Ablation till typically contains more silt and sand and occasionally contains pockets of water-washed sand and gravel.

Glacial till at the HoltraChem site was laid directly atop bedrock by the glacier itself, causing a thin veneer in the uplands and thicker deposits in the lowland areas. As the glacier melted during the deglaciation period, ice-contact and glacio-fluvial deposits, most notably sand and gravel, were laid down by pro-glacial and subglacial streams. These deposits often parallel modern drainage systems, a phenomenon is observed along the length of the Penobscot River. Hydraulic conductivities of the

sand and gravel deposits typically range from 10 E-4 to 10 E-2 cm/sec. Some gravel deposits exhibit hydraulic conductivities as high as 10 E-1 cm/sec. Hydraulic conductivities of these deposits are determined by grain size and sorting.

Marine transgression capped the glacio-fluvial deposits with glacio-marine silt and clay deposits, which display distinctly poor permeability (typically 10 E-6 cm/sec or lower). These deposits are known throughout coastal Maine as the Presumpscot formation. Glacial till is usually observed at the surface in the uplands of the Orrington area. In the lowlands, the till deposits thicken and are often overlain by late-glacial sand and gravel or marine silt and clay deposits. Occasionally the tills or marine sediments, especially on the ridges, were reworked by the sea as the land rebounded and emerged from the water. These deposits resemble beach sands and have relatively high hydraulic conductivities.

5.2 Regional Hydrology and Hydrogeology

5.2.1 Climate - Precipitation

The climate in the Orrington area is temperate, with cold winters and mild summers. Temperatures range from sub-zero (Fahrenheit) in mid-winter to the mid-to-upper eighties in the summer months. Precipitation averages approximately 40.25 inches per year, 3.35 inches/month. The months with the highest precipitation are typically November and December (average 4.92 and 3.88 inches, respectively). The driest months of the year are typically August (average 2.85), and March and May (both average 2.97 inches). The primary growing season occurs after snow melt in April and extends through August. Evapotranspiration, the combined processes of evaporation and transpiration from plants, is greatest during the months of May and June and may account for the return of much of the water which infiltrates the root zone to the atmosphere. An average of 15% of the available annual precipitation (approximately 6 inches) is anticipated to infiltrate to groundwater.

The prevailing wind direction at the Orrington plant is from the south and south west. During the spring and summer months, thermal energy created by warmer temperatures on the land relative to the cold coastal water create updrafts which pull cooler air up the Penobscot River Valley. During the winter months, wind directions are likely to be more variable with storms coming in from the south and west and occasionally from the northeast. Weather data compiled at Bangor Dow field, including a wind rose, is provided in Appendix E.

5.2.2 Surface Water Bodies

The HoltraChem plant is located on the east bank of the Penobscot River, approximately 5 miles south of Bangor. Below Bangor, the river is tidal with an average range measured at Hampden of

12.8 feet. In the plant area the river is approximately 30 feet deep and is essentially fresh water. An occasional salt water wedge is present along the bottom during periods of high tides and low river flows. Salinity soundings were performed during the SI and no salt water wedge was observed. However, brackish water was reportedly produced from onsite wells adjacent to the river during high tides.

The drainage area of the Penobscot River above the HoltraChem site is approximately 8,570 square miles. The maximum flow recorded at the Eddington gage was 153,000 cubic feet per second (cfs) in April 1982. The Eddington gage is located approximately 10 miles upstream of the HoltraChem Site, but provides a good indication of the river conditions at Orrington. The average flow of the Penobscot River measured at the Eddington gage is approximately 16,400 cfs. The 7 day -10 year low flow (7Q10 flow) at the Eddington gage is approximately 4,000 cfs. During the 1995 SI river sampling, the average flow was 10,000 cubic feet per second. During the latter part of that summer, the flow was approximately 3,000 cubic feet per second due to an extended drought. Surface water flows during the dye study performed in 1998 ranged between 4000 and 7000 cfs.

The river channel in the vicinity of the HoltraChem site is characterized by three distinct environments. To the north of the plant, the river flows southeasterly along a bedrock outcrop. In this stretch, the bedrock outcrop forms the shoreline and there are no deposits of sand or gravel except for a small bank at the mouth of a gully north of Landfill 4. After rounding the western end of the bedrock ridge, the river turns south and a steeply sloping sand and gravel bank is present extending from the base of Landfill Area 1 to a cove located at the southern boundary of the plant property.

A regional stream, the Souadabscook, enters the Penobscot River in Hampden, opposite the HoltraChem plant. The Souadabscook drains primarily agricultural land west of Bangor.

5.2.3 Water Bearing Units

Groundwater is present in both the overburden and bedrock. However, with the exception of local deposits of sand and gravel, the majority of the overburden deposits are till and are not typically permeable enough to be developed for groundwater supplies.

Because the crystalline bedrock of the area has very low primary porosity (typically < 2-3%), groundwater in bedrock is present primarily in fractures. Residential wells are typically installed into bedrock with surface casings sealing off the overburden deposits. Wells in the vicinity of the HoltraChem site range from 90 to 365 feet deep. The depths of the wells are often determined by the number of water bearing fractures encountered and the storage capacity required of each well.

5.2.4 Groundwater Flow

The Penobscot River is the regional discharge point for groundwater flow. Recharge areas potentially extend to the local drainage divides, located approximately 10 miles to the east. The amount of recharge to overburden and bedrock is primarily dependent on the permeability and slope of the surface deposit. Steeply sloping surfaces of low permeability will limit infiltration while permeable, relatively level deposits will enhance infiltration. Infiltration rates in glaciated terrain typically range from 35% of available runoff (precipitation - evapotranspiration) over till deposits to up to 90% over sand and gravel deposits. Based on the average annual rainfall and evapotranspiration percentages calculated for the Bangor/Orrington area, infiltration to groundwater ranges from approximately 5 inches per year over tills to approximately 12 inches per year over sand and gravel. The majority of the ground surface in the Orrington area is covered by till. A downward vertical gradient is typical in recharge areas.

Along the eastern bank of the river, groundwater flows from east to west toward the river. As the groundwater approaches the discharge point, an upward vertical gradient will predominate. This upward vertical gradient has been observed in bedrock and overburden well clusters on the HoltraChem site.

5.2.5 Water Quality

Groundwater

The bedrock groundwater in the Orrington area is used as a drinking water source. In general, the water quality is adequate for residential use. The bedrock groundwater is often characterized as hard, typically with 1 to 2 grains per gallon (17 to 34 mg/l) hardness (as CaCO₃) and sometimes measuring up to 10 grains/gallon (171 mg/l). Typical water problems are related to high concentrations of iron (2-5 mg/l) and to a lesser extent manganese (≤ 1 mg/l). Some wells yield very high levels of iron and manganese (e.g. the Norlens Water Treatment Service well which contained 28 mg/l iron and 18 mg/l manganese when sampled in January 1998). These wells are typically treated with water softeners. Elevated salt concentrations, either due to saltwater intrusion or road salting, are occasionally a problem (Tom Demaso, Norlen's Water Treatment Service, Orrington, Maine, November 1998). During sampling of the residential wells on Ferry Road, measurements were obtained for salinity and specific conductance. Salinity values ranged from 0.00% to 0.06%, indicating that some wells yield slightly saline water.

Little information is available regarding the use of overburden groundwater as a drinking water supply. Prior to obtaining Bangor City water, the Town of Hampden operated a gravel packed well near the bank of the Souadabscook Stream. This well developed iron and manganese problems which would have required Hampden to treat the water. Rather than building a treatment plant,

Hampden began purchasing Bangor water. They still operate the gravel packed well to sell water for commercial uses. The drilling water for the SI was obtained from this well to prevent the possibility of introducing chloroform, which is often present in chlorinated municipal water supplies, into the subsurface.

Surface Water

The Penobscot River in Orrington is classified as Class C in Maine's Surface Water Classification System. Class C is the third highest classification, and the waters "shall be of such quality that they are suitable for a drinking water supply after treatment, recreation in and on the water, fishing, industrial process and cooling water supply, hydroelectric power generation, navigation, and as a habitat for fish and other aquatic life. "

Just below the southern cove, along the southwest border of the HoltraChem site, the Penobscot River is classified as SC due to the increasingly brackish nature of the river. Estuarine waters are classified SA through SC. Class SC is the fourth highest classification, and the waters "shall be of such quality that they are suitable for recreation in and on the water, fishing, aquaculture, propagation and restricted harvesting of shellfish, industrial process and cooling water supply, hydroelectric power generation and navigation and as a habitat for fish and other estuarine marine life.

"Discharges to Class C waters may cause some changes to aquatic life, provided that the receiving waters shall be of sufficient quality to support all species of fish indigenous to the receiving waters and maintain the structures and function of the resident biological community." (Maine Water Pollution Control Law, Section 465.4C)

A dioxin-based fish advisory for the Penobscot River has been in effect since 1989. It recommends no more than two meals of fish a month from Penobscot River waters and pertains to the reach from Lincoln (north of Bangor) to the mouth of the river (Bucksport).

A mercury based advisory currently exists for limiting the consumption of fish from all lakes and ponds in Maine. This advisory recommends that pregnant women and young children not eat fish and that others limit their intake to between 6 and 22 meals a year depending on a number of factors such as fish size, specific lake, etc.

On March 28, 1997, the MEDEP issued a fish advisory for all inland rivers and streams in the state due to elevated levels of mercury. Among its recommendations, the advisory recommends limiting fish consumption to between 1 and 2 fish meals per month for fish from the Penobscot River; pregnant women, nursing mothers and children under the age of 8 are advised not to eat any fish from lakes, ponds or rivers in the state (Maine Department of Human Services, March 1997).

5.2.6 Water Uses

The Penobscot River is used recreationally and for transportation. Pleasure craft and barges were regularly observed on the river during the course of the SI. Recreational uses include fishing and may include swimming, although the current is strong and the water temperature is cold most of the year. There is a commercial eel fishery in the reach of the river south of Bangor.

5.3 Site Geology

5.3.1 Introduction

The geology of the site is the most significant factor influencing groundwater flow and potential subsurface contaminant transport. Subsurface stratigraphy at the site was investigated by advancing twenty-two soil borings during the 1995 site investigation, and an additional 17 soil borings during the 1997 Supplemental Site Investigation. Soil samples were collected for geologic description from twelve of the borings in 1995, and ten of the borings during the fall of 1997. Monitoring wells were installed in twenty-one of the borings in 1995, and all 17 borings during 1997. Figure 3-2 shows locations of soil borings and monitoring wells. Geologic logs of soil borings advanced during the 1995 and 1997 site investigations are presented in Appendix D. To develop a geologic model of the site, information obtained during the supplemental site investigation has been combined with results of previous geologic and geophysical investigations. Since 1965, at least ten geotechnical and environmental drilling programs have been performed at the site. These programs were summarized in Section 2 (see Table 2-4). A Previous Investigation Location Plan is provided in Appendix B.

5.3.2 General Stratigraphy

Subsurface stratigraphy encountered during the recent investigation is generally consistent with stratigraphy described by CDM (1995), Acheron (1991), and with the regional stratigraphy described in Section 5.1. Geologic materials encountered during drilling included fine-to medium-grained phyllite bedrock, and overburden materials consisting of glacial till, unconsolidated glacial outwash deposits and marine sand, silt and clay. Various man-made fill deposits were also encountered. The thickness and distribution of geologic materials is controlled by the bedrock surface topography beneath the site. On the prominent bedrock ridge located north of the plant area, overburden units are relatively thin and discontinuous, with laterally variable thicknesses. Beneath the plant area, south of the bedrock ridge, greater and more consistent thicknesses of overburden deposits are present. The sediments observed adjacent to the Penobscot River are significantly different from those observed beneath the plant area or on the bedrock ridge.

The three settings outlined above allow the site to be segregated for descriptive purposes into three geologically distinct areas, termed the "ridge", "plant" and "river" areas. The boundary between the ridge and the plant and river areas is defined by the bedrock scarp running approximately west to east across the site from the Penobscot River to the flooded gravel pit east of Landfill 2. Landfills 3, 4, and 5 are situated within the ridge area. The plant area extends from the flooded gravel pit east of Landfill 2 through the manufacturing facility to the lined process lagoon. The river area extends from the lined process lagoon to the southern cove.

Bedrock

Bedrock encountered during drilling and exposed in outcrops at the site consists of light gray to gray, fine- to medium-grained micaceous phyllite, a low-grade metamorphic rock. The rock encountered at the site is consistent with the descriptions of the Vassalboro Formation in regional geologic texts and maps.

The great relief in the bedrock surface is due to the expression of a fault interpreted to exist between a deep bedrock valley beneath the plant and the bedrock ridge. The topography of the bedrock surface is presented in **Figure 5-1**. The elevation of the top of the bedrock surface ranges from Elevation -90 PD (B-320-B1) to Elevation 96.8 PD (B-303-B1). Bedrock crops out in three general areas of the site: along the Penobscot River north of the bedrock ridge, along the bedrock scarp between the ridge and the plant, and in the southern cove near the outfall of the southerly stream.

The phyllite bedrock is hard, well indurated, and slightly weathered in outcrop. During the installation of bedrock monitoring wells, the upper 5 feet of bedrock was penetrated with a roller bit, to allow permanent casing to be set. In the boring for MW-411-B1, the upper portion of the bedrock was soft enough to allow augers to penetrate approximately 4 to 6 feet into the rock. In the boring for MW-511-B1, there appeared to be a zone of fragmented bedrock in the upper 5 feet of core. With the exception of these two locations, the rock encountered during the initial 5 feet of bedrock drilling appeared to be very resistant.

Metamorphic alteration has imparted a well-developed planar foliation on the rock, resulting in anisotropic fracture patterns and cleavage. During the 1997 field season, the orientations of prominent foliation planes and fractures were measured at six outcrops on site. The average strike and dip of the foliation as measured at these outcrops are approximately N86W to S16W, and 58 to 78 degrees to the northeast, respectively. A polar projection and a map of fracture orientations are presented in Appendix E. Virtually all of the porosity of the local bedrock is assumed to be due to metamorphic foliation structures and secondary jointing within the rock. Bedrock cores obtained during drilling, and inspection of local outcrops indicate that significant fracturing occurs along foliation planes; secondary joint sets in outcrop are oriented primarily in a northeasterly direction (N15E to N60E), but these joint orientations are less consistent than the orientation of foliation planes.

5.3.3 Overburden Units

Subsurface investigations identified four general overburden units underlying the HoltraChem site: glacial till, glacio-fluvial outwash, fine-grained marine sediments, and man-made fill. The following paragraphs describe the physical characteristics and distribution of overburden units in the ridge, plant, and river areas.

Glacial Till

Overlying bedrock in most locations is a dense, gray or brown glacial till. In some locations, this till crops out at the ground surface. At other locations, the till is overlain by light brown to brown, unconsolidated mixtures of sand, silt and gravel with minor portions of clay. The stratigraphically lower "gray till" is found only in the plant area. The stratigraphically higher "brown till" is found in both the plant and ridge areas, and occasionally overlies the gray till. A description of each of these units follows.

The gray till consists of gray, medium to very dense mixtures of silt and clay, with minor amounts of coarse- to fine-grained sand and gravel. In several new and existing borings, the upper portion of the gray till contains a higher sand and gravel content. Boulders up to 6 feet in diameter were encountered in two borings. During recent or previous investigations, geologic sampling did not encounter interbedded sand or gravel intervals within the till that could provide a preferential pathway for groundwater flow. Because of the tight, compact nature of this unit as well as the lack of coarse interbedded layers, this unit forms a low permeability impediment to vertical groundwater flow into bedrock, wherever encountered. A grain size distribution curve of a gray till sample obtained from MW-404 is provided in Appendix E.

The brown till, in contrast to the gray till, has greater vertical variability and a wider distribution of grain sizes. It consists of dense to very dense, light to dark brown mixtures of silt, sand, and gravel. Grain size curves for the brown till encountered in B-402-O1 and B-406-O1 are provided in Appendix E.

Because the till layers typically have low hydraulic conductivities, the depth and thickness of till deposits help to delineate the pathways for overburden groundwater flow. The extent and thickness of glacial till are illustrated in **Figure 5-2**. Till deposits are thickest in the bedrock valley beneath the manufacturing plant (101' at MW-404) and in the vicinity of Landfill 2 (81.5' at B-301). Till deposits are thin on the bedrock ridge and are very thin to absent beneath and northeast of Landfill 4. A boring performed by L.A Wing through Landfill 4 in 1975 did not encounter till deposits above refusal which, based on other borings, was interpreted to be bedrock. Till was not encountered above bedrock in the boring for MW-506-B1, which was installed through Landfill 4.

The thick deposits of glacial till observed beneath the plant area thin abruptly to the south and toward the Penobscot River. On the southern side of the plant, the till is very thin at B-327 and MW-504. Till is not present at MW-503, located south of the railroad spur terminus. Till is also not present in B-321-B1, nor at MW-511 or MW-505. In the western and northern portions of the plant, the till is absent at B-320-B1, adjacent to the river at the southern cove. Till is also very thin to absent adjacent to the bedrock scarp at the northern boundary of the manufacturing plant (B-315), and in MW-401-B1, at the base of Landfill Area 1.

Glacio-fluvial Outwash Deposits

Outwash deposits encountered in both the ridge and plant areas consist of interbedded, moderately stratified, heterogeneous mixtures of medium to very dense, light to dark brown, orange brown, and light gray to gray sand, with little to no silt and minor amounts of gravel and clay. Occasionally, silt or gravel content may approach 50 percent. The extent and thickness of sand, gravel, and fill deposits are shown on **Figure 5-3**.

As indicated on Figure 5-3, a narrow channel of sand and gravel is located adjacent to the bedrock scarp at the northern edge of the plant. Grain size distribution curves for two samples obtained from MW-405-B1 are provided in Appendix E. Based on geotechnical borings performed in the plant area, these deposits do not extend beneath the plant area.

Another sand and gravel deposit is present south of the plant in MW-503 and MW-504. This deposit is characterized by a medium-dense, gray-brown sand overlying a gray-brown gravel. At MW-503 the gravel directly overlies bedrock, while at MW-504 the gravel and bedrock are separated by approximately two and a half feet of dense silty till. The top of this sand and gravel deposit ranges from 31' PD (approx. 33' NGVD) in MW-504 to 59' PD in MW-503. The bottom of this deposit is approximately Elevation 26' PD in both MW-503 and MW-504.

The thickest deposits of sand and gravel occur adjacent to the Penobscot River. The maximum thickness is observed in B-320-O1, where it is approximately 94 feet thick and extends from 10 feet below ground surface to a depth of approximately 104 feet (-90 feet NGVD). In B-326-O1 this deposit extends from 34 to 64 feet below ground surface (elevation 0 to elevation -30 NGVD). A grain size distribution for the sand and gravel deposit observed in MW-402-O1 is provided in Appendix E.

The sand and gravel deposits at the base of Landfill Area 1 are interpreted to be fluvially deposited river sand and gravel. These sand and gravel deposits thin towards the plant, corresponding to a thickening of the glacial till. Cooling water and firewater supply wells installed in the sand and gravel near the Penobscot River were initially capable of flows in excess of 1000 gallons per minute. Based on the limited size of the aquifer and the large volume of water these wells produced, the

majority of their yield was probably the result of induced infiltration from Penobscot River water, indicating a good connection between groundwater and surface water in this area.

Marine Deposits

Two types of marine deposits were observed on the site. In the lower elevations of the plant and river areas, fine grained silt and clay deposits are observed near the ground surface. These are interpreted to be Presumpscot Formation: fine grained sediments carried by glacial streams and deposited in near shore marine environments. These deposits are primarily silt and clay sized particles, although they are composed primarily of rock flour, rather than clay minerals. The silt and clay deposits are observed in the top 10 feet of B-320-B1 and B-321-B1. This deposit was also observed in geoprobe borings advanced through leach field No. 2, east of the plant laboratory/changehouse building.

Another deposit of suspected marine origin is present on the bedrock ridge. Fine to medium well sorted sand is visible outcropping on the northeastern (upriver) side of the gully north of Landfill 4. This deposit strongly resembles beach sand and it is suspected that during emergence of coastal Maine, the fine grained marine sediments were winnowed by wave and wind action. This deposit is noted on boring logs for MW-408-O1 and MW-408-B1.

Anthropogenic Fill Material

The plant area was graded prior to construction. Fill material was used to replace soil of poor load bearing capacity. Based on boring logs and hand augured soil samples, the fill material within the plant area includes reworked sand and gravel, and till deposits. Fill was also placed as bedding for plant roads and railroads.

The five landfills also represent fill deposits. Mercury contaminated sludge was reportedly mixed with sand prior to or during its emplacement in the landfills to improve its load bearing capacity and to facilitate handling. Each of the landfills is capped. In addition to the Hypalon placed over Landfill Area 1 and Landfills 3, 4 and 5, each landfill is covered with soil.

An area north of Landfill 4 was formerly used as a sand and gravel borrow area. This area was refilled with material removed from the plant area during construction of the former sodium chlorate plant. Continuous soil samples were collected during advancement of the soil boring for monitoring well MW-407-B1, which is in the filled borrow area. The sample descriptions indicate that the fill material is approximately 17 feet thick. The arcuate shape of the gully indicates that the northeast, upriver side represents the extent of sand and gravel excavation and that the southwest, downriver, side of the gully represents the extent of the fill. Based on a ground surface elevation of approximately 82 feet and the lowest surface expression of the gully (an indentation in the 50 foot

topographic contour), the maximum depth of the fill material here is anticipated to be approximately 30 feet.

The graded area below the paved sump, along the southern side of the northern stormwater outfall, also appears to comprise fill to a depth of approximately 2 feet. During the installation of soil lysimeters in this area, fragments of man-made materials, including approximately 10 graphite anodes, were observed.

Surface Soils

The Soil Conservation Service map of Penobscot County, Maine, sheet Number 256, indicates the presence of 14 soil types on the site. Examples of these descriptions range from "Adams loamy sand, 0 to 8 percent slopes (AaB)" to "Thorndike very stony silt loam, 15 to 35 percent slopes (TvD)". A copy of the SCS map and soil legend is provided in Appendix E.

The near surface soil at the site varies in composition among the ridge, plant, and river areas. The distribution of surface soil types has been further influenced by excavation, construction and filling. The following descriptions of soil distribution and soil profile have been compiled from observations made during soil sampling and soil borings during the SI, and from test pit logs of previous investigations. The soil profile observed in most test pits is generally consistent with the descriptions of soil stratigraphy developed from borings; however, observation of mottling and other indications of seasonally high groundwater levels are included.

Surface soil on the ridge is typified by an approximately 0.5 to 1.0 foot thick layer of loam with a high percentage of organic material (leaves, leaf mold, roots, etc.). On the ridge north of Landfill 5, mottling was observed at 2.5 feet below ground surface and water seeped into a pit (TP-103) at 3.5 feet below ground surface. The soil was described as till which became firm and platy at approximately 2 feet below ground surface. North of Landfill 3, till was observed close to the ground surface to the total depth of excavation at 3.8 feet (TP-106). In another excavation between Landfill 4 and the river (TP-107), the soils were described as fine sand and gravel. Mottling or other signs of seasonally high groundwater table were not observed.

The ground surface within the plant area is primarily paved. Where soil is exposed it is typically a sand fill of varying grain sizes. In the western portion of the plant, in the vicinity of the brine handling areas, it is capped with crushed stone. Test pits were excavated by previous consultants within the plant area, primarily to investigate soil types in the vicinity of proposed plant construction for the sodium chlorate plant and the warehouse building. Test pits in this area encountered a variety of materials, including silty gravelly sand, organic silt, peat, wood, clayey silt and fill. In general, the test pit logs indicate groundwater within approximately 3 feet of ground surface in the vicinity of the former sodium chlorate plant. Acheron advanced one test pit into the bedrock scarp. They encountered bedrock at one foot below ground surface at the top of the pit and 4.5 feet below ground

surface at the bottom of the pit. They described the soil as till. Acheron also excavated a pit near the road and stormwater ditch in the low lying area east of the plant. The material encountered was a clayey silt/silty clay, consistent with the descriptions of the Presumpscot Formation. Water was observed at a depth of 2 feet, which corresponded to the depth to water in the adjacent stormwater ditch.

In the river area, the surface soil is typically dark brown gravelly sand with a thick vegetated layer of grasses. Closer to the plant area near B-316, the sand was observed to be finer grained and contained a little clay. The log of a test pit (TP-111) excavated by Acheron near B-316 indicates fill to a depth of 8 feet with a sharp contact between a one foot thick layer of blue-green silty clay and stratified sand and silt. Test pits advanced by CDM during the soil lysimeter investigation found fill to a depth of more than 2 feet -in the area below the paved sump. Groundwater in this area rose to approximately 8 inches below ground surface following the installation of the lysimeter.

The saturated hydraulic conductivities of the near surface soils are assumed to be similar to those tested with in-situ hydraulic conductivity (slug) tests in monitoring wells. By definition, the unsaturated hydraulic conductivities of these materials must be lower. An attempt to directly measure the unsaturated hydraulic conductivity at the site using a Guelph permeameter was unsuccessful.

Selected soil samples were collected from soil borings and analyzed for grain size distribution. The samples were selected to be representative of the various deposits observed at the site. A sample of the till which underlies much of the plant area was submitted from MW-404-B1. Two samples of the sandy soil were submitted from MW-405-B1, adjacent to the bedrock ridge near the former sodium chlorate plant. A sample of the till observed in the ridge area was submitted from MW-406-B1. Grain size curves and tabulations are presented in Appendix E.

The total organic carbon (TOC) content of selected site soils were measured by Acheron (Table 3-23, Appendix B). TOC ranged from 290 mg/kg to 3100 mg/kg, and averaged approximately 1000 mg/kg.

The storage capacity of site soils was evaluated by measuring the soil moisture at varying pressures and establishing a soil moisture retention curve. Two site soils were tested, including a well sorted (narrowly graded) medium sand from the ridge area, and till collected from MW-404-O1. These two soils are considered to represent the most and least permeable soils at the site and therefore the soils with the two extremes of soil moisture retention capacity. The calculations and curves are presented in Appendix E. The higher fines content of the till retained significantly more moisture than did the relatively clean sand deposit.

Infiltration was estimated based on a formula provided in Water Resources Inventory of Connecticut, USGS and Connecticut Water Resources Bulletin No. 28, 1978. The formula uses the percentage of

surface area covered by glacial drift (sand and gravel) and empirical data shows good correlation with river systems throughout the northeast. Attempts to directly measure infiltration using a dual ring infiltrometer were unsuccessful due to the presence of gravel or due to leakage in sandy soils.

5.4 Site Hydrogeology

The hydrogeology of the site can be broadly divided into bedrock and overburden groundwater flow. In many locations on site, the till layer acts as an aquitard, maintaining the separation between these two water-bearing units. Stratigraphic cross sections were generated to illustrate site geology, extent and continuity of high permeability units, preferential pathways, and the relative thicknesses of saturated and unsaturated materials. The locations of cross sections were chosen to provide both cross-gradient and along-gradient expressions of subsurface soil conditions. In conjunction with the preceding descriptions of site geology, these cross sections help to form the conceptual hydrogeologic model. Cross sections can be found in the following figures:

- Figure 5-4** Stratigraphic Cross Section Location Plan
- Figure 5-5** Stratigraphic Cross Section A-A'
- Figure 5-6** Stratigraphic Cross Section B-B'
- Figure 5-7** Stratigraphic Cross Sections C-C' and D-D'
- Figure 5-8** Stratigraphic Cross Section E-E'
- Figure 5-9** Stratigraphic Cross Section F-F'

5.4.1 Bedrock

The phyllite bedrock observed at the site has very low primary porosity (typically 2-3 % or less). Therefore, the hydraulic properties are dominated by the degree and orientation of fractures. The highest hydraulic conductivities observed during the 1997 field season were found in MW-505-B1 (3.06E-2 cm/sec), MW-506-B1 (1.04 E-3 cm/sec), and MW-505-B2 (9.06 E-4 cm/sec). The highest conductivities observed during the 1994 Site Investigation were found in MW-401-B1 (1.36 E-3 cm/sec), MW-408-B1 (3.86 E-4 cm/sec), and MW-410-B1 (8.76 E-4 cm/sec).

Groundwater flow in bedrock will generally trend from areas of high to low hydraulic head. However, the actual flow pathways are determined by interconnected fractures capable of transmitting water. The specific path taken by groundwater may therefore be less direct than would be indicated by flow strictly perpendicular to groundwater contours. The majority of fractures observed in bedrock cores corresponded with bedding planes which have been mapped in outcrops as trending N86W to S16W and dipping 58 to 78 degrees north toward the Penobscot River. Additional fracture patterns were noted in outcrops and indicated by the VLF survey. These fracture sets are oriented north and northwest from the plant area toward the Penobscot River. The presence

of fracture sets oriented toward the Penobscot River will likely serve to conduct groundwater flow to the river.

In addition to natural fractures, additional fracture sets may have been created during plant expansion to the edge of the bedrock scarp. When the railroad spur was constructed to serve the former sodium chlorate plant, blasting of bedrock was conducted in the vicinity of monitoring well P-2A. The extent of fractures induced by blasting is expected to be limited.

In general, most of the groundwater on the bedrock ridge flows from the highest points located northeast of Landfill 5 toward the low point of the ridge near Landfill 4 and northwest toward the Penobscot River. A lesser amount flows south toward the plant, where it discharges to overburden deposits or flows beneath the till to the Penobscot River.

5.4.2 Overburden

Groundwater flow in overburden deposits is controlled by the relative hydraulic conductivities of the sand and gravel deposits. Hydraulic conductivities measured in overburden deposits during the SI ranged through five orders of magnitude; from $3.93 \text{ E-}7$ (MW-508-O1) to $2.56\text{E-}1 \text{ cm/sec}$ (MW-405-O1). The lowest hydraulic conductivities were measured in till deposits beneath the manufacturing plant. The highest hydraulic conductivities were measured adjacent to the bedrock scarp north of the former sodium chlorate plant (MW-405-O1) and in the sand and gravel deposits adjacent to the Penobscot River (MW-512-O1, MW-401-O1, MW-402-O1, and MW-513-O1).

Groundwater will seek preferential pathways through material with the highest hydraulic conductivity. The dense till material, where present, will limit groundwater flow. Till is observed overlying bedrock across portions of the ridge, beneath the plant area and beneath most of the area adjacent to the Penobscot River (see Figure 5-2). Where less dense, more granular material overlies glacial till, as is the case across most of the site, till will limit downward migration of groundwater and the predominant flow direction will be horizontal or along the top of the till surface toward the river. In some cases, fractures within the till may provide pathways for groundwater to migrate through this low permeability layer; in general, though, the till is expected to provide a surface for predominately lateral migration.

Granular fill or former stream channel beds will often form preferential flow pathways. Prior to the construction of the former sodium chlorate plant, the area adjacent to the bedrock scarp was wet and soil removal was conducted prior to the plant expansion into that area. This area was backfilled to support anticipated construction activities. The screened interval in MW-405-O1 may be in granular backfill related to construction of the former sodium chlorate plant. In general, however, hydraulic conductivities in wells located adjacent to the bedrock ridge were higher than elsewhere in the plant or river areas, with the exception of the very permeable sand and gravel deposits adjacent to the river (e.g. MW-512-O1).

5.4.3 Anthropogenic Features

In addition to naturally occurring preferential flow pathways, anthropogenic features may significantly influence groundwater flow. The plant is constructed above a thick sequence of dense till of low hydraulic conductivity. Based on drillers logs of borings advanced prior to plant construction, between two and ten feet of silty sand overlie the till in some locations. Subsurface utilities which are located below the water table may act as preferential flow pathways. Flow may also occur along the bedding or disturbed soil in utility trenches. The majority of the underground utilities within the plant are related to process water and the industrial sewer, and are installed below the frost line, approximately five feet below ground surface. Two abandoned water lines and an abandoned electrical conduit lead from the plant to the former river well pump houses. One of these water lines is an abandoned pipeline from the Hampden water supply which crosses beneath the Penobscot River. The approximate locations of underground utilities are indicated on Figure 2-3. Where applicable, underground utilities are also indicated on the stratigraphic cross sections (Figures 5-5 through 5-8).

The industrial sewer collects discharge from various buildings around the plant and flows through the neutralization tank prior to flowing to the NPDES discharge at the Penobscot River. The discharge pipe from the neutralization tank traverses in a straight line from the treatment wier to outfall 001 at the Penobscot River. The elevation of the industrial sewer ranges from Elevation 60.40 PD to Elevation 54.51 PD within the plant area. Stormwater is collected in two stormwater sewers which range in elevation from Elevation 56 PD to Elevation 49.56 PD.

Potentially the most significant anthropogenic feature affecting groundwater flow is the underdrain system constructed beneath the former sodium chlorate plant. The underdrains discharge to the first manhole of the western stormwater sewer located at the northwest corner of the former sodium chlorate plant. These underdrains flow constantly, even during extended periods of dry weather, suggesting that they receive discharge from groundwater in addition to surface water runoff. The invert elevation of the manhole near the former sodium chlorate plant is approximately Elevation 54 PD.

Groundwater is also occasionally collected and discharged from the basement of the office building. The basement of this building is approximately Elevation 60 PD. The building foundation had a perimeter drain which connected to a culvert leading to the southern drainage. During 1994 and 1995, seepage was observed through the foundation and in the spring of 1995 water flowed into the basement which required the use of a sump pump located in an electrical utility chamber. Plant personnel believe that the perimeter drain or its discharge line have become clogged.

5.4.4 Groundwater Flow

Groundwater contour maps for overburden and bedrock are provided as **Figures 5-10** and **5-11**, respectively. Synoptic groundwater level measurements for these maps were collected in December of 1997.

A groundwater divide is present approximately down the center of the bedrock ridge. The top of till or rock may locally affect the location of the divide in overburden. Specific fractures may locally alter the location of the groundwater divide in bedrock. Groundwater north of the divide flows directly to the Penobscot River. Groundwater south of the divide flows into the overburden deposits beneath the plant. Overburden groundwater will seek preferential pathways through deposits or anthropogenic features of relatively higher hydraulic conductivity.

Hydraulic conductivity values from available monitoring wells and descriptions from boring logs throughout the plant indicate that the soils adjacent to the bedrock scarp have higher hydraulic conductivities than those beneath the plant. These overburden deposits along the base of the landfill ridge are likely to form a preferential pathway for groundwater flow. Underdrains below the former sodium chlorate plant capture much of the groundwater between the bedrock ridge and the plant. This water is then discharged to the Penobscot River via the paved sump and the north ditch.

Varying degrees of groundwater seepage have been observed along the beach face adjacent to the Penobscot River. Areas of highest groundwater/beachface discharge appear to correspond to the alignment of abandoned Hampden water line which crosses the river, the water lines and electrical conduits leading from the plant to the pumphouse buildings and around the HoltraChem Outfall 001. These observations indicate that the utilities in this area of the site may represent preferential pathways for groundwater flow.

Shallow groundwater contours east of the plant are very close to the ground surface elevation. The stormwater ditch that extends from the railroad tracks west of Landfill 2 to just northeast of the transformer yard is approximately six feet deep. More water was observed flowing out of this ditch near the transformer yard than was flowing in where it crosses beneath the railroad tracks west of Landfill 2. Based on the shallow groundwater contours and the observed increased flow in the stormwater ditch, it appears that shallow groundwater is discharging to this ditch east of the plant.

The water table is also very close to the ground surface where the ditch passes between the HoltraChem and PERC plants. Based on the groundwater contours in this area and the wetland vegetation observed, this stormwater ditch also appears to be receiving recharge from groundwater.

Groundwater discharging along the southern side of the plant near the loading shed is collected in an eight-inch perforated subsurface drainage pipe installed for this purpose. The groundwater collection pipe was installed after water was observed flowing into the stormwater ditch from the bank.

Groundwater contours trend in a north-south direction as they pass the B-321, MW-505, and MW-511 clusters, indicating that the wells on the PERC property and the residences on Ferry Road are cross gradient from the HoltraChem site. Groundwater flow in this area is approximately perpendicular to and towards the Penobscot River, which is a regional discharge point for groundwater in the Orrington area. Monitoring well MW-511-B1 is artesian, with a static water level more than 3 feet above ground surface. A spring was noted on the bank of the southern cove at the base of the southerly stream. This spring is a discharge point for groundwater in this area.

Hydraulic Conductivities

Hydraulic conductivities were calculated for three general stratigraphic units: bedrock, till, and sand and gravel deposits. The geometric mean of the available hydraulic conductivity data from Acheron (Table 2-4, Appendix B) and slug tests performed for the SI (Table 3-3) were calculated for the three units. Calculations are provided in Appendix E.

Hydraulic conductivities in bedrock ranged from 7.1 E-7 cm/sec (2.0 E-3 ft/d) to 1.8 E-3 cm/sec (5.1 ft/d).

Hydraulic conductivities in till ranged from 1.8 E-6 cm/sec (5.1 E-3 ft/d) to 4.0 E-5 cm/sec (0.11 ft/d).

Hydraulic conductivities in sand and gravel deposits ranged from 4.6 E-4 cm/sec (1.3 ft/d) to 2.6 E-1 cm/sec (725 ft/day).

The hydraulic conductivity values for till deposits were generally more homogeneous than those for drift and bedrock deposits. Hydraulic conductivities obtained from wells screened in the till were all within one order of magnitude, while values for the bedrock and overburden deposits varied by as much as three orders of magnitude.

Horizontal Hydraulic Gradients

Horizontal hydraulic gradients on the site are controlled by the surface, till surface and bedrock topography, and the horizontal hydraulic conductivities of site media. In general, lower conductivity materials are able to support steeper hydraulic gradients; water elevations in higher conductivity materials are more easily equilibrated by flow through these media.

Water elevations in bedrock are 60 to 70 feet above the river level within 200 feet of its bank, a horizontal hydraulic gradient of 0.30. This large hydraulic gradient suggests that the low hydraulic conductivity of the bedrock limits groundwater flow to the Penobscot River.

A similar steep hydraulic gradient (approximately 0.27) is observed from the plant area to the Penobscot River. In this area, a relatively thin veneer of sand and gravel deposits is underlain by steeply dipping glacial till, which acts as an aquitard limiting the downward percolation of groundwater. The apparently steep hydraulic gradient in this region is indicative of groundwater flow along the surface of the till. Although flow in this area largely occurs in sand and gravel deposits, the volumetric flow rates from the plant to the river are not as high as this gradient would imply, due to a shallow saturated thickness of overburden deposits.

Sand deposits adjacent to the bedrock scarp along the northern edge of the plant support a very small (0.002) hydraulic gradient. These sand deposits have hydraulic conductivities two to three orders of magnitude greater than the till deposits or bedrock. The higher hydraulic conductivity of this deposit facilitates the equilibration of hydraulic heads.

Hydraulic gradients between Landfill Area 1 and the river are strongly influenced by tidal variations in the river. The response of the aquifer to the tides, however, is nonlinear and involves a significant time lag, because water can enter the aquifer vertically during the flooding tide, but must seep horizontally out from the beach during low tide (Turner et al., 1996). The net response of the water levels in the Landfill Area 1 wells, then, is a time-dependent hydraulic gradient. During low tide, a large hydraulic gradient is present from the landfill to the river, but this gradient is significantly reduced by the development of a seepage face along the shoreline. During high tide, the gradient reverses, pushing water back toward the plant from the river. The time-dependent hydraulic gradient was modeled using the sloping beach face solution to the Boussinesq equation (Nielsen, 1990; Turner et al., 1996). The analytical solution, based on the physical characteristics of the beachface and the observed tidal variations in five wells below Landfill Area 1, indicated an average seepage face elevation of 1.865 ft NGVD, during a maximum tidal range of 14 feet. Tidal cycle monitoring and modeling of the water table super-elevation relative to the river indicate that the time-averaged gradient below Landfill Area 1 ranges from 0.001 to 0.005.

Seepage Velocities

Seepage velocities were calculated for areas of the site where significant discharge to the river occurs. The seepage velocity, also known as the mean linear velocity, was calculated from the hydraulic conductivity, the hydraulic gradient, and an assumed effective porosity. For flow below Landfill Area 1, seepage velocities were calculated from gradients and conductivities where applicable, and from flux estimates and saturated cross sectional areas for methods where gradients were not calculated. Calculated seepage velocities provide an estimate of how quickly groundwater moves in various deposits. The assumed effective porosities were 25% for sand and gravel, 5% for till, and 2.5% for bedrock. The assumed saturated cross sectional area below Landfill Area 1 was 6,750 ft.² (see Appendix E).

The calculated seepage velocities were as follows:

- For flow in bedrock from the bedrock ridge to the Penobscot River, 5 ft/day, or 1800 ft/yr.
- For flow in overburden sand and gravel along the bedrock scarp, 0.10 ft/day, 36.5 ft/yr.
- For flow in overburden sand and gravel below Landfill Area 1, 0.24 to 0.42 ft/day, or 86 to 155 ft/yr.

These seepage velocity calculations should be considered estimates as our evaluation of porosity for materials on the site is limited to measurements on similar deposits at other sites. Differences in porosity can lead to significant overestimates or underestimates of seepage velocities. For example, if the estimate of porosity for any of the materials were halved, (e.g. bedrock porosity equals 1.25% vs. 2.5%) seepage velocities for that deposit would be doubled. Nonetheless, the calculated seepage velocities above provide at least an order of magnitude approximation of how quickly groundwater is migrating through site media. Seepage velocity calculations are provided in Appendix E.

Vertical Hydraulic Gradients

Vertical hydraulic gradients were calculated from data collected at monitoring well clusters. A summary of the vertical hydraulic gradients was provided in Table 3-4.

The observed general trends in vertical hydraulic gradients are divided into regions. Upward hydraulic gradients were observed along the river and in the eastern half of the plant area. Downward gradients were observed along the bedrock ridge, where thin overburden deposits are discharging to underlying bedrock, and along the western edge of the plant area, where overburden deposits are underlain by steeply dipping glacial till.

A strong upward gradient within the bedrock is observed in monitoring well clusters B-303 on the bedrock ridge and in the B-321, MW-505, and MW-511 clusters on the PERC property south of the plant. These clusters with multiple screens in bedrock indicate that regional bedrock groundwater flow is discharging to the Penobscot River. Upward gradients are also apparent between bedrock and overburden in the vicinity of the Penobscot River below the plant. On the bedrock ridge, a downward gradient is typically observed between the overburden and the bedrock wells in locations where the overburden is saturated.

Three well clusters are located adjacent to the bedrock scarp in the vicinity of the plant. A fourth well cluster is located along the bedrock scarp adjacent to the Penobscot River. An upward gradient is present at the MW-405 cluster. There is almost no vertical gradient present at the B-315 cluster. A downward vertical gradient is present at the MW-403 cluster, and an upward vertical gradient is present at the MW-401 cluster. This pattern indicates that groundwater from the bedrock ridge is discharging to overburden deposits adjacent to the scarp upgradient of cluster B-315.

An area of downward hydraulic gradients is present across the center of the site, from north of the lined process lagoon to the HoltraChem/PERC property line near MW-504. Downward hydraulic gradients were observed for clusters MW-403, MW-503, B-316, and MW-504.

Due to differing tidal efficiencies and delay times, simultaneous multiple well water level monitoring was required to evaluate the net vertical hydraulic gradients in tidally influenced well clusters adjacent to the Penobscot River. These net gradients are summarized in Table 3-6. In general, an upward gradient was observed between the bedrock and the deepest overburden screen. A downward gradient was typically observed between the shallowest and the next deepest overburden screened interval. These converging vertical gradients indicate discharge is occurring to the Penobscot River most readily through the intermediate depth interval, which corresponds to the high conductivity sand and gravel deposits.

Infiltration

The amount of water recharging groundwater aquifers is governed by precipitation, surface runoff, evaporation and transpiration (evapotranspiration), and by characteristics of the surface such as slope, vegetation and the infiltration capacity of surficial deposits. Till, clay, or pavement at the ground surface will limit infiltration. Conversely, sand and gravel permit rapid infiltration. Steep slopes, such as the bedrock scarp and the slopes from the ridge to the Penobscot River, typically have high surface water runoff and low infiltration. The landfill caps may have differing impacts based on their specific design and construction. Landfill 4 has a 2.5-foot thick clay cover over a granular drainage layer and a Hypalon membrane. The water which runs off of Landfill 4 to the north will most likely infiltrate the surficial sand deposits present there. The presence of landfills and other man-made surfaces were addressed through conservative assumptions of infiltration rates.

Infiltration was calculated for three types of surficial deposits: till, sand and gravel, and pavement. Estimates of recharge and runoff were made considering the percentage of the surface area covered with sand and gravel. Although little water is expected to penetrate pavement, a value one half the infiltration expected for drift was used to account for potential infiltration through small unpaved areas and cracks.

Based on the extent of till, sand and gravel or fill as shown on Figures 5-2 and 5-3, surficial material types were assigned to recharge areas to calculate a water balance.

5.4.5 Water Balance

The water balance was calculated in two ways. First, the hydraulic gradients and hydraulic conductivities measured at the site were used to estimate the amount of groundwater discharging to the Penobscot River. Second, the volume of water recharging the groundwater at the site was calculated based on the infiltration assumptions described above.

The regional and site specific groundwater flow to the river was calculated using the hydraulic gradient between the plant and the river and an average (geometric mean) of the bedrock and till hydraulic conductivities. The individual geometric means for bedrock and till hydraulic conductivities were generally similar and the bedrock and till can be considered to act as one relatively low permeability unit. For fractured rock, an aquifer is often considered to be 500 feet thick with the hydraulic conductivity decreasing with depth. To account for the decreasing hydraulic conductivity with depth and the fact that all our measurements were of the upper portion of rock, an aquifer thickness of 300 feet was used. The length of shoreline from downgradient of Landfill 1 to the B-320 cluster was assumed to be the area of discharge. The quantity of groundwater estimated to be discharging to the Penobscot River by this method was 35 gpm.

For the aquifer recharge calculation, groundwater was assumed to discharge to the Penobscot River through two main routes: from the ridge north to the river, and from the ridge, the valley and the plant to the river. The areas where recharge would contribute to groundwater flowing through these areas was digitized and the amount of infiltration was estimated based on its assigned surface type. The amount of water estimated to be discharging to the Penobscot River by this method was 37 gpm.

The close agreement of the numbers generated by these two different approaches provides some confidence that the assumptions used for factors such as hydraulic conductivity, infiltration and recharge areas are reasonable. However, a water balance is only an estimate and there are many assumptions included which introduce uncertainties. For instance, an average hydraulic conductivity was used which may be lower than the actual hydraulic conductivity of localized deposits. An assumption was made of the extent of the area contributing to recharge of groundwater. Because the Penobscot River is a regional groundwater discharge point, the area of contribution to groundwater flow could be significantly greater than the area assumed for the water balance calculation. This would result in more groundwater passing beneath the site and discharging to the river than was estimated. However, as this water infiltrated in the upper reaches of the watershed it would be expected to pass below water which infiltrated closer to the site. This deeper water is referred to as underflow. Because underflow would most likely pass beneath the site in bedrock (below till) it is not expected to come in contact with contamination and, therefore, does not impact contaminant flux from the site.

Water balance calculations for this investigation were performed to estimate the flux of contaminated groundwater off site. Because the highest concentrations of mercury and chloropicrin observed in groundwater are primarily in the wells below Landfill Area 1, additional calculations quantifying the groundwater flux off site in this region were performed. The water flow through this area was estimated using four different methods.

The first two methods used assumed infiltration or a combination of hydraulic gradients, conductivities and infiltration to estimate the amount of water which would discharge from the plant area beneath Landfill Area 1 to the Penobscot River. The second two methods used hydraulic gradients modeled for or measured in the sand and gravel deposits at the base of Landfill Area 1. Each of these methods contain assumptions with associated uncertainties. The four methods were

performed to evaluate the range of groundwater fluxes, identify similar results in the predicted fluxes with the various approaches, and to provide for evaluation of the ranges and degree of uncertainty in the estimates. A description of the various methods for estimating groundwater flow from the base of Landfill Area 1 follows.

The first method estimated the volume of water recharging the area which discharges between the bedrock scarp and the north ditch based on the infiltration assumptions described in Section 5.4.4. The volume of water available for groundwater recharge was computed based on estimated evapotranspiration rates at varying times throughout the year. A percentage of available water was then assumed to infiltrate to groundwater based on an empirical relationship between observed infiltration and percentage of stratified drift in surficial materials (USGS, 1979). Establishment of a recharge area for the river area was complex. An area of infiltration was established which began on the southern side of the ridge and included the plant and the area between the plant and the river. Because of cracks in the pavement and the lack of evapotranspiration from plants, the paved plant area (from the salt storage pad to the office) was conservatively assumed to have one half the infiltration rate of sand and gravel (stratified drift). The area from the plant to the Penobscot River was assumed to be sand and gravel. Water percolating through the areas of infiltration described above was assumed to pass through and be contaminated with the concentrations detected in the wells adjacent to the river.

The theoretical recharge area extends at least to the nearest surface water divide to the east. The shallow groundwater flowing toward the plant from the east is expected to discharge to the southern drainage ditch and its tributaries. However, some groundwater, at greater depths, will likely flow through the overburden and bedrock deposits beneath the plant. Groundwater at depth is not expected to become contaminated except where it passes through the areas of high chloropicrin and mercury concentrations. To account for underflow which will pass through contaminated areas, the volume of groundwater seeping through the till at the western side of the plant (near MW-404-O1 and P-7) was calculated. This portion of the calculated underflow was added to the volume calculated to pass through contaminated zones.

The volume of water calculated to infiltrate the plant and the south half of the bedrock ridge (which discharges to the plant) was approximately 1020 ft³/day or the equivalent of 5.3 gpm. However, during the course of the SI, groundwater was observed to be discharging to the underdrains at the former sodium chlorate plant and potentially along the storm sewer leading from the underdrains to the paved sump. Flows at the paved sump were measured during periods of no observed overland flow at rates ranging from 2.19 gpm in August to 21.7 gpm in April. Water which infiltrates the plant but is discharged at the paved sump is not available for flow beneath Landfill Area 1 where the greatest levels of contamination in groundwater have been documented. Conservatively assuming a base flow in underdrains and underground utilities of approximately 3 gpm, the remaining flux of groundwater available to discharge beneath Landfill Area 1 was estimated to be 2.3 gpm. Limited infiltration through the Landfill Area 1 cap, and discharge from the till layer, were then added to this flux to estimate a discharge below the landfill.

In the second method, the cross sectional area of the stratified drift and the hydraulic gradients and hydraulic conductivities measured at the site were used to estimate the groundwater flux above Landfill Area 1. The hydraulic gradients between monitoring wells B-315-O1 and MW-403-O1, located along the bedrock scarp where the stratified drift deposits are thickest, were used to calculate the flux of groundwater from the site toward Landfill Area 1. Using a cross sectional area of approximately 6000 ft² and an average hydraulic conductivity of 1.8 E-3 cm/sec (5.1 ft/day) yields an estimate of 0.8 gpm. Because the hydraulic gradients measured in the plant are reflective of any influence the underdrains may have, it is not necessary or appropriate to subtract the flow of water observed at the paved sump during dry conditions. Infiltration through the landfill cap and discharge from the till were added to the estimated flux above the landfill to estimate the total discharge of water to the river.

These two methods provide a range of volume estimates for groundwater moving from the plant area to the river area. The contributions from seepage through the underlying till (underflow) and infiltration in the area between the plant and the river are estimated at 0.12 gpm and 1.14 gpm, respectively. Therefore, the estimated discharge of groundwater along the base of Landfill Area 1 ranges from 2.1 gpm (397 ft³/day) to 3.6 gpm (685 ft³/day). [see Appendix E]

Two additional methods of estimating groundwater flux from the base of Landfill Area 1 were performed which utilize the net hydraulic gradients and hydraulic conductivities of the sand and gravel deposits adjacent to the river. As discussed in Section 5.4.4, water level and horizontal hydraulic gradients in this area are strongly influenced by the large tides (12.8 foot mean) in the Penobscot River. Measurement of average water levels and net hydraulic gradients in the aquifer required monitoring of water levels throughout the tidal cycles. In the first of the hydraulic gradient methods, the elevation of the water table just inland of the river bank (beach) was modeled using equations presented in Turner, et. al., 1996. The groundwater flux estimated through this method was 5.1 gpm. However, due to the observed development of a seepage face which the authors acknowledged was not adequately addressed by the available calculations, and the sensitivity of the flux calculations to minor changes in hydraulic gradients, the model gradients were considered potentially inaccurate, and therefore required site specific verification.

To provide for direct measurement of hydraulic gradients between monitoring wells at the base of Landfill Area 1 and the Penobscot River, two piezometers were installed at the top of the river bank and water levels were again collected over the entire tidal cycle. The piezometers were surveyed when installed and again after data inconsistencies (negative gradients indicating flow from the river toward the plant) were observed. Groundwater flux estimates generated from the well and piezometer data range from 3.6 to 7.1 gpm. It is important to note that due to the high hydraulic conductivities at the base of Landfill Area 1, differences in water level measurement and survey elevations of less than one tenth of a foot can change calculated flux estimates by a factor of 2. Therefore, the groundwater flux estimates from the modeled and monitored gradient approaches should be compared to and used with the fluxes estimated from the other two methods. This comparison indicates a minimum predicted flux of 2.1 gpm, a maximum of 7.1 gpm, and a mean of 4.8 gpm.

5.5 References

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Section 6

Source Characterization

6.1 General

This section provides descriptions of the Areas of Concern investigated during the SI and provides information on the physical characteristics of the waste material present in those areas. The purpose of this section is to provide the background information used to evaluate contaminant distribution (presented in Section 7).

6.2 Areas of Concern

Areas of Concern and potential source areas that were identified in the Work Plan include:

- the five landfills;
- the lined process lagoon;
- plant area releases as listed on Work Plan Table 3-1 entitled "Summary of On-site Reported Spills and Releases," and Work Plan Table 3-2 entitled "Potentially Significant Spill Areas" (also Attachment "C" of the Consent Order);
- sediments in the Penobscot River; and
- onsite sediments in the northern stormwater ditch and the Southerly Stream.

As a requirement of a Consent Agreement between HoltraChem and the Maine Department of Environmental Protection (MEDEP), leach fields No. 1, 2, and 8 were added to the Areas of Concern.

Sampling locations for the SI were selected to evaluate the potential source and spill areas identified in the Current Assessment Summary (Section 3) of the Work Plan. Sampling locations for the SSI were selected to further delineate contamination identified during the initial Site Investigation. The following subsections provide information on the Areas of Concern. Areas of Concern are identified on **Figure 6-1**.

6.2.1 Landfills

Five landfills were constructed at the plant site and subsequently closed. All of the landfills received mercury contaminated sludges from the brine treatment system and/or the wastewater treatment system. Three of the five, Landfills 2, 4 and 5, also reportedly received carbon tetrachloride. The landfills were open to precipitation while they were active and are all presently capped. Data on landfill use is provided in Table 2-2. This table was compiled from data contained in the 1991 Acheron report and recollections of plant personnel. Therefore, the volumes and quantities located are considered approximations and are not based on specific records.

Landfill Area 1

Landfill Area 1 encompasses about 2.0 acres on a hillside sloping to the Penobscot River. It consists of several discrete disposal sites, including the former Hickel's Pond, and landfill cells 1a and 1b. The current lined process lagoon (which is in the same area as the former Hickel's Pond) is located within Landfill Area 1. The history of disposal practices in this area is not well documented.

Landfill cells 1a and 1b were filled between 1970 and 1972, with approximately 1000 tons of sludge. The sludges deposited in landfills cells 1a and 1b were generated prior to the construction of the wastewater treatment plant. The wastewater treatment system is designed to precipitate mercury in an insoluble form. Therefore, brine sludges generated prior to construction of the wastewater treatment plant are likely to contain mercury in a more soluble form such as mercuric chloride. According to Mr. William "Zeke" Miller, former Technical Assistant to the Plant Manager, the mercury concentrations in the sludge prior to wastewater treatment were higher than after treatment. The sludge material was pumped as a slurry from the sludge pits (see Figure 2-2) to landfill cells 1a and 1b. Sand was mixed with the sludge/slurry prior to closure.

Mercury contaminated graphite anodes were also deposited on this hillside during the early years of plant operation. The anodes were reportedly removed by a recycling company. However, small pieces of anodes and anode stems were reportedly observed during the construction of the raw water well pump house structures. Construction debris was reportedly deposited in Landfill Area 1, on the hillside just north of the lined process lagoon and east of landfill cell 1b. Drums formerly stored adjacent to Landfill Area 1 were reported to have been removed. There are no direct reports of drums being placed in Landfill Area 1.

In 1972, landfill cells 1a and 1b were covered with soil. In 1980, Landfill Area 1 was capped by grading the existing fill to provide a uniform surface for installing a 30 mil impervious Hypalon cap.

The cap was covered with 12 inches of common fill, 6 inches of loam, and grass. Landfill Area 1 has a fairly uniform slope of approximately 30 percent towards the Penobscot River. Runoff from the landfill flows to two drainage swales on each side of the landfill which ultimately discharge to the Penobscot River. The landfill also has an erosion control berm and several gas vents. The landfill cover appears to be in good condition. There are no signs that the landfill has been modified since closure, and the vegetative cover is well established with no observable vegetative stress.

Some erosion has occurred at the bank of the Penobscot River below Landfill Area 1. To prevent a possible slope failure, HoltraChem has developed a plan to armor the toe of the landfill with a rip rap blanket. The plan was communicated to EPA in a letter dated July 2, 1998, and is scheduled to be implemented in the spring or summer of 1999.

In 1985, PELA advanced two soil borings through waste material of landfill cell 1a. Soil samples were collected and analyzed for mercury. Depth profiles provided in Acheron's Interim Report (January 1991) indicate that P-10 and P-11 terminated 28 and 20 feet above mean sea level, respectively. Mercury concentrations in soil from P-10 and P-11 averaged 348 mg/kg and 53 mg/kg, respectively. The boring log of P-10 indicates that it was still in fill material at its termination. A monitoring well was installed in P-10 which has never contained water. The total depth of the well is approximately 15 feet. The soil boring log of P-11 indicates that clay underlies 18 feet of sand fill from 18 to 30 feet below ground surface. However, the sample descriptions include roots, indicating that the clay could be fill other than brine sludge.

The TDEM survey performed during the SI detected two metallic anomalies within Landfill Area 1. One area is a broad band located just down the hill toward the river from the lined process lagoon. This band is approximately 30 feet wide and 85 feet long, and trends roughly parallel to topographic contours. One portion of the anomaly shows a denser concentration of metal than the rest of the band. The metal deposit is estimated to be six to fifteen feet below the ground surface. A second anomaly is a denser concentration of metal located halfway down the landfill slope adjacent to the bedrock scarp. The anomaly covers an area approximately 25 feet by 30 feet. This deposit is estimated to be fifteen feet below ground surface. The remainder of Landfill Area 1 did not exhibit significant metallic anomalies.

The seismic refraction surveys conducted over Landfill Area 1 identified a low velocity layer indicating that the unsaturated fill is approximately 25 feet thick. The layer underlying the unsaturated fill had a velocity which could be interpreted to represent saturated overburden such as fill or sand and gravel deposits. The line between the two differing velocities is relatively straight on one of the two lines, further indicating that it likely represents the top of the saturated zone. Based

on this interpretation, the water table under Landfill Area 1 on the date of the seismic surveys ranged from approximately mean sea level near monitoring well cluster B-326 to approximately Elevation 40 feet PD near the lined process lagoon.

Groundwater recharge to this area is limited by the Hypalon cap and low permeability till soils encountered in P-7 and MW-404. Therefore, the groundwater table is anticipated to be close to the top of the low permeability till deposit where this deposit is above sea level. The till is present in P-7 at approximately Elevation 44 feet PD and drops to below sea level before reaching the B-326 cluster. Because of its higher hydraulic conductivity, the water level in the sand and gravel deposits is expected to be close to sea level.

The exact depth of sand/sludge and other fill within Landfill Area 1 is not known. The reported locations of landfill cells 1a and 1b are near the bedrock scarp above the 40' ground surface elevation contour. Based on the MW-501 boring, the PELA borings and the seismic and TDEM work performed for the SI, the depth of fill is estimated to range from 6 to 25 feet below ground surface.

Monitoring well MW-501-O1 was installed through Landfill Area 1, directly downgradient from the lined process lagoon. The highest soil mercury concentration detected in this boring was 17.84 mg/kg, collected from 4 to 6 feet below the liner. Soil concentrations below this depth were less than or equal to 0.05 mg/kg, until detections of 1.1 mg/kg from 24 to 26 feet and 10.19 mg/kg from 26 to 28 feet below the liner. Based on the low concentrations of mercury detected below 6 feet, it appears that contaminated fill at this location is limited to the top 6 feet. The water level in MW-501-O1 was measured at approximately 28 feet below ground surface (Elevation 30 PD). The higher concentrations detected below the uncontaminated soil zone are therefore most likely attributable to transport of mercury in groundwater. Although some percolation typically occurs through landfill caps, contaminated soil located below the cap and above the groundwater level is not likely to be a significant continuing source of groundwater contamination. Soil containing mercury as a result of groundwater transport is likely to be at or near the adsorption capacity of that soil and therefore would not represent a significant continuing source.

Groundwater quality within and downgradient from Landfill Area 1 was monitored at MW-501, MW-512, MW-513, and the MW-401 and B-326 clusters. Prior to the 1997 groundwater monitoring round, the highest concentrations of mercury in overburden deposits were detected in B-326-02. The concentrations detected by CDM in 1995 were 2630 $\mu\text{g/l}$ in the parent sample and 1660 $\mu\text{g/l}$ in the duplicate. Because of the disparity between the parent and the duplicate samples in B-326-02, and the need to evaluate temporal changes in concentrations, a second round of samples was collected and analyzed for mercury. Both filtered and non-filtered samples were collected to allow

comparison with earlier samples collected by Acheron which were filtered. In the confirmation round, the concentrations detected were 1600 $\mu\text{g/l}$ total and 1500 $\mu\text{g/l}$ dissolved in both the parent sample and in the duplicate. The similarity of the total and dissolved concentrations indicate that the mercury in B-326-O2 is in a soluble form.

Monitoring well MW-501-O1, located in the center of Landfill Area 1 just downgradient of the lined process lagoon, was installed and sampled during the 1997 investigation. Mercury was detected in this well at a concentration of 4600 $\mu\text{g/l}$ (total) and 3500 $\mu\text{g/l}$ (dissolved). Monitoring well MW-512-O1 located at the base of the landfill area had mercury concentrations of 810 $\mu\text{g/l}$ (total, duplicate sample) and 660 $\mu\text{g/l}$ (dissolved, parent sample). Monitoring well MW-513-O1 had mercury concentrations of 1000 $\mu\text{g/l}$ (total) and 710 $\mu\text{g/l}$ (dissolved) during the 1997 monitoring event. These data again indicate that most of the mercury is in a soluble form in groundwater in the landfill area. The data also suggest a source area near the eastern edge of Landfill Area 1, as concentrations of mercury are decreasing to the west.

MW-401, located directly downgradient from landfill cells 1a and 1b, had lower concentrations of mercury and much lower concentrations (greater than order of magnitude) of chloride than MW-501-O1 and B-326-O2, which are more directly downgradient from the lined process lagoon and the location of the former Hickel's Pond. These data suggest that groundwater contamination in Landfill Area 1 is not due to groundwater coming in contact with fill material in landfill cells 1a and 1b. Instead, the data indicate that the lined process lagoon is a likely source area for groundwater contamination at the base of Landfill Area 1.

Landfill 2

Landfill 2 is located in a valley area northeast of the plant and received waste sludge from 1972 through 1973. Landfill 2 dimensions are approximately 200 x 60 feet or 12,000 square feet. It was operated from 1971 through 1973 and reportedly received approximately 1500 tons of brine sludge. The sludge in Landfill 2 was generated prior to the installation of wastewater treatment which reduced the solubility of mercury in the sludge. One load estimated to be approximately four gallons of carbon tetrachloride, was reportedly placed in Landfill 2. No further disposal of carbon tetrachloride are reported to have occurred.

Landfill 2 was covered with soil in 1973 and capped with clay in 1980. PELA's 1980 report, "Hydrogeological Impact of Waste Disposal Sites and Recommendations for Control at the IMC Orrington Plant, Orrington Maine," provides the following description of the Landfill 2 closure:

Closure included the excavation [of fill] and placement of 15 to 18 inches of clay along the southern portion, replacement of the fill material, and capping the entire site with 18 to 24 inches of clay and 6 inches of loam.

The landfill cover appears to be in good condition. There are no signs that the landfill has been modified since closure. There are no signs of erosion and the vegetative cover is well established with no observable signs of stress.

The geology beneath Landfill 2 can be inferred from the soil borings and monitoring wells located nearby and from the results of the two seismic refraction lines performed in the area. The boring logs of monitoring well B-301-B1 indicated a dense clayey glacial till underlies that portion of the valley down to a depth of approximately 94 feet below ground surface. The boring log of 8-8, located approximately 100 feet east of Landfill 2, indicates that glacial outwash sand and gravel is present at that location but is overlain by 17 feet of clay.

One seismic line was performed along the side of Landfill 2 with a spacing of 20 feet between geophones. This line indicated the presence of 10 to 12 feet of unsaturated fill material (velocity 1300 fps) overlying a thick sequence of glacial till (see Section 3). The glacial outwash deposit present at 8-8 is not observed on the seismic line performed adjacent to Landfill 2.

To more accurately evaluate the depth of fill material and the elevation of the groundwater table, a second seismic refraction line was performed over Landfill 2. The geophone spacing used was 5 feet to allow better definition of the shallow refractions. This line indicated a 1 to 4.5 foot thick layer with very low velocity (966 fps) material which probably represents the clay cap. This is underlain by an 8 to 12 foot thick layer of unsaturated fill material which is underlain by till. The five foot spacing between geophones should have identified the presence of a saturated layer within the fill if one was present. Based on the seismic refraction survey, the elevation of the base of the fill and the top of the till ranged from Elevation 58.5 to Elevation 61 feet PD which is below the water levels observed in B-303-01 and MW-409-01.

Based on the low permeability deposits underlying Landfill 2, the absence of a detectable saturated fill layer on the seismic survey, and the lack of detectable levels of contamination in the shallow wells in the vicinity, it appears that the mercury bearing sludges in Landfill 2 are not a continuing source of groundwater contamination.

Landfill 3

Landfill 3 was constructed in 1972 and filled with waste excavated from a former disposal area called Mac's Pond. Mac's Pond was located south of the current lined process lagoon, in the approximate location of the paved sump. Mac's Pond operated from before June 1970 through 1971, and its dimensions were reportedly 50 ft x 60 ft (3,000 square feet). The pond reportedly received approximately 1000 tons of brine sludge generated prior to wastewater treatment. In 1972, Mac's Pond was excavated and the materials were deposited in Landfill 3. The dimensions of Landfill 3 are approximately 190 ft x 200 ft, or 38,000 square feet.

In 1972, Landfill 3 was covered with soil. In 1980, Landfill 3 was capped by regrading and installing a Hypalon membrane. The membrane was covered with 12 inches of common fill and 6 inches of loam. The landfill cover currently appears to be in good condition. There are no signs that the landfill has been modified. There are no signs of erosion and the vegetative cover is well established with no observable signs of stress.

The base of Landfill 3 is estimated to be at approximately Elevation 82 ft PD, based on boring CA-5 advanced by L.A. Wing in 1975. The ground surface elevation of this boring is unknown but is assumed to be approximately 95 feet based on the elevation of the current top of Landfill 3. The L. A. Wing boring was advanced through 0.5 feet of topsoil and 13.5 feet of fill. After passing through fill material, the boring log notes 7.3 feet of till above refusal which is interpreted to be bedrock. The recent seismic refraction survey, (see Section 3) estimated the thickness of the fill and overburden deposits at Landfill 3 to range from 16 to 20 feet which is consistent with the L. A. Wing borings.

The monitoring well installed by L. A. Wing (CA-5) at a depth of 20.5 feet contained 0.5 feet of water in it when it was measured after installation. PELA monitored the water level in CA-5 four times in April 1980. PELA reported the water at Elevation 74.75 (assumed PD) in the April 3, 1980 round and the well was dry during the other three rounds. Based on this data, PELA concluded that the highest seasonal water levels were approximately seven feet below the base of the fill.

Monitoring wells located downgradient of Landfill 3 include: P-13, MW-405-B1, MW-405-O1, MW-406-B1, MW-406-O1, MW-410-B1, and P-2A. Due to the proximity of Landfill 3 to Landfill 4, it is difficult to distinguish groundwater quality characteristics which are specifically representative of Landfill 3. Nevertheless, based on the approximate elevation of the base of the fill and the available water level elevations from CA-5 and MW-410-B1 (highest 76.78 PD, January 1995), it appears that the fill material in Landfill 3 is above seasonally high groundwater levels. The

cap on Landfill 3 is likely to prevent infiltration through landfill materials and into underlying groundwater. Based on these data, Landfill 3 is not a likely source of groundwater contamination.

Landfill 4

Landfill 4 is located on the ridge between Landfills 3 and 5. The dimensions of Landfill 4 are 150 x 300 feet, or about 45,000 square feet. As shown on Figure 2-2, Surface Drainage Area 3, Surface Drainage Area 2 and part of Surface Drainage Area 1 receive stormwater runoff from Landfill 4. The landfill received sludge from 1972 through 1980, and it was capped in 1980. During its operation, it reportedly received approximately 8,000 tons of brine sludge and 100 tons of chlorate plant waste. Landfill 4 also reportedly received carbon tetrachloride. During the early years of chloropicrin production, the calcium chloride used to dry the chloropicrin may also have been deposited in Landfill 4.

According to closure plans for Landfill 4, this landfill was regraded, and covered with 6 inches of gravel, a 30 mil. reinforced Hypalon liner, a 6 inch gravel drainage layer, 2.5 feet of clay and 6 inches of topsoil. The landfill cover appears to be in good condition. There are no signs that the landfill has been modified since closure, there are no signs of erosion, and the vegetative cover is well established with no observable vegetative stress.

In 1975, L.A. Wing advanced six borings (4, 5, 6, 7, 7A and 8) and installed three monitoring wells (4, 5 and 7A) in and around Landfill 4. Refusal was reached in well 4 at a depth of 9.5 feet and water was observed in the piezometer at a depth of 8.3 feet immediately after installation. Well 5 encountered refusal at 8 feet and the sample description at that depth reads "Brown, Silty Sand, Coal, Ashes". A piezometer was installed at 7.5 feet but no water level reading was recorded. A third well, 7A, was installed through the berm of the landfill. This well, installed to a depth of 15 feet, contained water at 11 feet immediately after installation. Again, no surface elevations are available for the L.A. Wing wells. PELA reported water level elevations of approximately 75 ft PD for four water level measurements of CA-7, which is thought to be in the same location as L.A. Wing's 7A.

Monitoring well MW-506 was installed through Landfill 4 to evaluate the depth to the base of the fill and the top of bedrock, and to monitor water level elevations and groundwater quality directly beneath the landfill. The highest concentration of mercury detected in soil samples was 239 mg/kg, in a sample collected from 2-4 feet below the Hypalon cap. Soil mercury concentrations below that depth ranged from 1.26 to 95.51 mg/kg and the last soil sample, which was collected between 14 and 16 feet below the Hypalon cap, contained 19.64 mg/kg. Split-spoon refusal was encountered at 16 feet. Based on a ground surface elevation of 97.3 feet and the soil sample data discussed above, the

base of fill/top of bedrock is at 79.3 feet at this location. The water levels measured in MW-506-B1 have ranged from Elevation 72.02 to 73.73 PD, approximately six feet below the base of fill.

The highest groundwater elevations in monitoring wells surrounding Landfill 4 are as follows: P-4, 68.88; B-309-01, 74.03; MW-410-B1, 76.78; P-3, 80.50, B-304-B1, 81.51; B-306-B3, 81.76 PD. All of these elevations were measured during January of 1995, and represent the maximum water levels observed over five monitoring rounds.

Monitoring wells potentially downgradient of Landfill 4 include: P-2A, P-3, P-4, P-13, MW-405-B1, MW-405-O1, MW-406-B1, MW-406-O1, and MW-410-B1. Mercury, carbon tetrachloride, chloroform, and chloropicrin have been detected in groundwater downgradient of Landfill 4. Monitoring well MW-410-B1 was installed downgradient of Landfill 4 in 1995. In that year, mercury was detected at a concentration of 50 $\mu\text{g/l}$, and carbon tetrachloride and chloroform were detected at concentrations of 210 and 570 $\mu\text{g/l}$, respectively. The mercury concentrations for this well have fluctuated from 9 $\mu\text{g/l}$ in 1997 to 62 $\mu\text{g/l}$ total and 52 $\mu\text{g/l}$ dissolved in 1998. VOC concentrations have remained lower than first detected. The available historical data indicate that mercury concentrations in other wells located downgradient of Landfill 4 have been decreasing over time.

During the 1997 and 1998 sampling, mercury, carbon tetrachloride and chloroform were also detected in MW-506-B1. Mercury was not detected during the 1997 sampling round, but was detected at concentrations of 19 $\mu\text{g/l}$ (total) and 12 $\mu\text{g/l}$ (dissolved) during the 1998 sampling. Carbon tetrachloride (54 $\mu\text{g/l}$) and chloroform (20 $\mu\text{g/l}$) were each detected in MW-506-B1 during the 1997 groundwater monitoring. The concentrations of each of these volatile compounds remained relatively stable between 1997 and 1998. Chloropicrin was detected only in MW-406-B1, and only during the 1995 sampling round.

The water level elevations relative to the base of fill/top of rock in MW-506-B1 indicate that groundwater is not currently coming in contact with waste material in Landfill 4. Although the fluctuations in concentration and limited number of monitoring rounds do not conclusively indicate whether Landfill 4 is a source of mercury and/or VOC contamination, the volume of water contained in the bedrock fractures beneath the landfill is limited. Therefore, the concentrations detected in groundwater to date do not indicate a significant release of contaminant mass. Monitoring wells MW-410-B1, MW-506-B1, and the wells surrounding Landfill 4 will continue to be monitored as part of the Comprehensive Monitoring Plan.

Landfill 5

Landfill 5 was constructed in 1978 and used until 1983. The Landfill 5 dimensions are approximately 200 x 140 feet or 28,000 square feet based on surface mapping. Acheron reported the dimensions of Landfill 5 as 80 x 140 feet, or 14,800 square feet (Acheron, 1991). Landfill 5 operated from 1978 through 1983. During that period it reportedly received approximately 3,300 tons of brine sludge, 100 tons of chlorate plant waste, and 360 gallons of carbon tetrachloride.

According to PELA's 1980 report "Hydrogeological Impact of Waste Disposal Sites and Recommendations for Control at the IMC Orrington Plant, Orrington Maine," Landfill 5 was originally used to hold sodium chlorate plant waste. It was reportedly excavated to approximately six feet below ground surface and was unlined. In the spring of 1980 the chlorate waste was removed and transferred to Landfill 4 just before that landfill was capped. The western portion of Landfill 5 was then lined with clay prior to disposal of mercury contaminated brine sludge. Approximately one half of the sludge deposited in Landfill 5 was solidified by the "Solidtex" process.

Landfill 5 was closed after RCRA regulations took effect and is considered the only "RCRA landfill" on the HoltraChem property. It was closed by grading the fill, and capping it with 6- inches of sand, a 30-mil HDPE liner, 6-inches of sand, 6-inches of common fill and 6-inches of topsoil. The landfill cover appears to be in good condition. There are no signs that the landfill has been modified. There are no signs of erosion and the vegetative cover is well established with no observable stress.

Four clusters of monitoring wells surround Landfill 5. The well clusters have been monitored quarterly since their installation in mid-1989, except for an approximately three year period from June 1991 to October 1994. Sampling results from each of the sampling events in these monitoring well clusters indicate that groundwater in the vicinity of Landfill 5 contains only low levels of VOCs (carbon tetrachloride and chloroform levels are below 20 $\mu\text{g/l}$ in B-306-B1 and B2).

Based on a 1980 base map compiled by James Sewall Company, the ground surface of the Landfill 5 area at that time ranged from Elevation 92 feet PD to Elevation 96 feet PD. The center of a depression in the northwest corner of the landfill is indicated by a closed 90 foot contour. Seismic data collected during the SI indicate the top of bedrock to be at approximately Elevation 91 feet PD in the western portion of the landfill.

Shallow water levels in the Landfill 5 area during the spring of 1995 ranged from 78.42 feet in P-3 to 99.79 feet in B-303-O1. The highest water levels observed in B-303-O1 and B-304-O1 were 100.65 and 81.69 feet PD, respectively. Both of these measurements were made in March of 1990. A theoretical straight line water table drawn between the highest water levels observed in B-303-O1 and B-304-O1 indicates that the water level beneath Landfill 5 could approach elevation 90 in the center of the landfill. Water levels measured during the SI, however, have not been as high as these historical values. The cap is expected to prevent infiltration of rainwater into the landfill area, and the water levels beneath the landfill cap are therefore not expected to reach this theoretical maximum elevation. Throughout most of the monitoring rounds, the overburden monitoring wells in the vicinity of Landfill 5 have been dry, indicating that groundwater has not historically been in contact with the fill material in Landfill 5. The presence of clay and/or till beneath the sludge fill will further reduce the possibility of groundwater contacting contaminated fill.

6.2.2 Leach Fields

Six leach fields are present on the HoltraChem site. Waste material in the tanks feeding the leach fields was first investigated in the summer of 1996, and most recently during the spring of 1998. These preliminary investigations identified leach field Nos. 1, 2, and 8 as Areas of Concern. A discussion of leach fields No 1, 2, and 8 follows.

Leach Field 1

Leach field No. 1 is located east of the plant and the railroad tracks, and north of the road leading to the nitromethane storage building. Leach field No. 1 consists of four rectangular leach fields, each with dimensions of 20 x 100 feet. One of these beds was constructed in or near 1980; the other three were installed in 1989. The leach field receives waste water from the septic tanks located adjacent to the maintenance shop (referred to as System 4), and indirectly from System 2 (the laboratory/change house) and System 6 (the chlorate building). Sampling of the waste material in System 4 in July, 1996, found elevated levels of mercury in the septic tank solids (102 mg/kg) and distribution box solids (71 mg/kg).

An investigation of leach field No. 1 was conducted by Acheron in the spring of 1997 (Acheron, Inc., July 1997). Sixteen soil samples were collected from beneath the disposal beds. Soil mercury concentrations ranged from 0.25 mg/kg to 13.67 mg/kg. The highest soil mercury concentration was detected from a depth of 3-4 feet below ground surface, in the boring advanced at the upstream end

of the oldest disposal field. The average of the four samples collected from the older disposal bed was 6.24 mg/kg. Concentrations in the more recently installed disposal fields averaged 0.68 mg/kg.

Groundwater in the vicinity of leach field No. 1 flows westward toward the HoltraChem plant. There are no monitoring wells located directly downgradient of leach field No. 1 to assess whether it is a continuing source of groundwater contamination. However, based on the soil concentrations reported from Acheron's investigations and the high organic carbon content typically associated with leach fields, leach field No. 1 is not expected to be a continuing source of groundwater contamination.

Leach Field No. 2

Leach field No. 2 comprises two leach field systems which are located beneath the grassy area between the laboratory/change house building and the plant offices. A concrete chambered leach field was installed in 1975 over an existing leach field. According to available plans, the existing leach field consisted of four 50-foot leach lines. Leach field No. 2 was abandoned in 1978 when the primary plant system, leach field No. 1, was expanded. When operational, water flowed from the change house building to the north tank and from the north tank to the south tank, then into a pump station. From here, water was pumped to a third tank which is located beneath the concrete chambers. Water then flowed up from the third tank and across the leaching area created by the concrete chambers. During its operation, leach field No. 2 received the output from floor drains, shower drains, sinks, and toilets in the change house building. All available records indicate that the laboratory sinks are and always have been connected to the industrial sewer.

Solids collected from the System 2 north solids tank contained mercury at concentrations of 171.42 and 178.59 mg/kg (9/23/97 and 3/20/98, respectively). Solids in the south tank contained 17.4 mg/kg when sampled on 9/23/97. Contamination in leach field No. 2 is attributed to accumulation of mercury from floor and shower drains, originating on boots and clothing of site workers, and potentially from excretion of mercury into sanitary waste.

An investigation of leach field No. 2 was conducted in August 1998, using a Geoprobe sampler. During this investigation, CDM collected soil samples for mercury analysis. Samples were collected along perforated leaching pipes, and through one of the leach field chambers. Mercury concentrations along the perforated pipes ranged from 0.001 to 14.4 mg/kg, dry weight; samples collected from the leach field chamber had dry weight mercury concentrations ranging from 0.036 to 17.9 mg/kg.

Groundwater in the vicinity of leach field No. 1 flows westward toward the HoltraChem plant. There are no monitoring wells located directly downgradient of leach field No. 2 to assess whether it is a continuing source of groundwater contamination. However, based on the soil concentrations reported from Acheron's investigations and the high organic carbon content typically associated with leach fields, leach field No. 2 is not expected to be a continuing source of groundwater contamination.

Leach Field No. 8

Leach field No. 8 was located along the base of the bedrock scarp, northwest of the hydrochloric acid tank farm. This leach field was constructed from hemlock planks, and has been out of operation since the period 1978-1980, when the new leach field system was constructed. During its operation, leach field No. 8 received wastewater from a holding tank in the location currently occupied by the HCl tank farm. This tank received drainage discharge from the old maintenance building.

The leach field No. 8 area was most recently investigated during August of 1998. Three test pits were excavated in the vicinity of the former leach fields, and soil samples were collected from each of the three pits. The test pits were excavated along 30 foot transects in the vicinity of the former leach field, but no traces of the hemlock planks or other infrastructure were identified. Dry weight mercury concentrations in soil samples collected from these test pits ranged from 0.04 to 2.5 mg/kg.

According to Gene Bridges of H.E. Bridges Construction, who has performed much of the excavation work on site, the leach field consisted of hemlock planks which were located parallel to the bedrock scarp west of the HCl tank farm and extending toward the lined process lagoon. Mr. Bridges indicated that the planks were likely removed during subsequent work in the area.

Based on visual characterization of the three test pits and the results of soil sample analysis, it appears that leach field No. 8 was removed during construction of plant utilities in this area. Therefore, leach field No. 8 is not considered a continuing source of groundwater contamination.

6.2.3 Surface Impoundments

Two surface impoundments are located at the western side of the plant. The salt storage pad is an asphalt pad with an approximately 6-inch high asphalt berm. The pad was constructed in 1967 when the plant was built. Salt, which is typically shipped to the plant by truck, is stored on the pad in piles. This salt is brought to the salt pile area and transferred to the brine saturators by a front end loader.

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Salt may be released to the ground surface through incidental spillage during transfer to or from the salt pile with the front end loader. Rain falling on the salt pile will become saturated with salt and if the quantity of runoff is sufficient to top the berm, salt saturated water could flow to the unpaved ground surface beside the asphalt pad and eventually to the paved sump. Salt crystals were observed on the ground surface adjacent to the salt storage pad during the SI, indicating that some overflowing does occur. Because sodium ferrocyanide was reportedly used as an anti-caking agent in some loads of salt, three soil samples were collected around the salt storage pad and analyzed for cyanide in the spring of 1995 (see Figure 3-4 and Table 3-16). Cyanide was not detected in any of the samples. Based on that data collected for the SI, the salt storage pile is not a source of mercury or cyanide contamination but is a source for the high conductivities observed in the north ditch.

The second surface impoundment is the lined process lagoon. The lined process lagoon is located within Landfill Area 1 at the western boundary of the plant. Overflow from the brine system is contained in the lined pond and is then returned to the brine system. After coming in contact with mercury in the cell building, the circulating brine typically contains an average concentration of approximately 15 mg/l of mercury. The brine in circulation is hot from the reaction in the cells. Some precipitation of salts, carbonates and sulfates is likely to occur when saturated brine cools in the lined process lagoon.

The lined process lagoon was formerly referred to as Hickel's Pond. During this time, it was used for temporary storage of the materials that were used in the cell room and other plant locations. Materials that were stored in the pond include: cell end box water (which covered the mercury in the cells), wash water from cleaning floors, and liquids from miscellaneous spills. The brine system later became a closed-loop system.

Hickel's Pond was excavated and the lined process lagoon installed at the same location in October, 1970. The lagoon was lined in 1976 with a 45 mil Hypalon Liner, and again in July 1979 with an identical liner. In 1981, it was lined again due to a leak. In 1983, LCP planned to close the lined process lagoon by cleaning the liner, placing sand and topsoil into the pond and installing a Hypalon cap over the fill. However, this closure never took place. In 1983, sludge was removed, the liner was again cleaned and patched, and the lagoon returned to service. The lined process lagoon was last relined in September 1995. Four liners were observed during the investigation of the lined process lagoon in October of 1998. Mercury concentrations in soils recovered from beneath the lined process lagoon ranged from 0.02 mg/kg to 30.4 mg/kg. The highest concentrations of mercury were detected in samples collected from the first two depth intervals beneath the center of the lagoon, from lenses of oxidized sand and gravel deposits, and from soils close to the sand/till interface. The maximum concentration detected at the till interface was 20 mg/kg. The lowest

concentrations of mercury were found in soils collected from within the till. The till appears to significantly limit further downward migration of mercury.

The lined process lagoon is considered a potential source for the groundwater contamination detected in monitoring well MW-501-O1 and the wells screened in the sand and gravel deposits at the base of Landfill Area 1.

6.2.4 Spill Areas

Spills have been reported at the site throughout the production history of the plant; the majority of the spills have been of brine. After initial contact with mercury in the electrolytic cells, the brine becomes contaminated with mercury at an average concentration of approximately 15 mg/l. Spills are reported to occur principally from overflows, sometimes caused by pump failures. Various spills of other products, including sodium hydroxide, have also occurred. Sodium hydroxide, as a product of the cell reaction, also contains mercury. The concentration of mercury in sodium hydroxide typically ranges from 15 to 40 $\mu\text{g/l}$ (see Table 2-5). A list of Reported Spills and Releases is presented in Table 2-4.

Shortly after the plant was constructed in 1967, the major roadways were paved. The area from the chlorate plant building to Landfill Area 1 was paved at the time the chlorate plant was constructed in 1978. Additional paving adjacent to the railroad loading area, and between the sludge pits and the chlorine storage area, occurred in 1988.

Two separate chloropicrin releases were reported to have occurred at the end of the railroad loading area. A spill of approximately 1,000 gallons occurred on the ground surface in 1985. This spill reportedly flowed into a catch basin and subsequently into the paved sump and the northern stormwater ditch which flows to the Penobscot River.

A second spill of chloropicrin was detected during an excavation around a raw water line in 1987. The raw water line was constructed in a gravel lined trench which extended from the pump building of well R-1 to the plant area at approximately the north end of the railroad loading area. Chloropicrin was detected while excavating north of the railroad tracks, approximately 70 feet from the end of the tracks. This location is approximately 10 feet from where the underground chloropicrin line crossed the former raw water line. The former raw water line is reportedly at a depth of approximately 5 feet below ground surface and the chloropicrin line crossed above it. After this spill was detected, the underground chloropicrin line was abandoned and replaced with the

current aboveground line. The duration of the leak and the amount of chloropicrin released are not known. These releases are considered sources for the contamination detected in monitoring wells B-316-O1, B-316-B1, and MW-402-O1.

The spill areas within the plant are likely to be source areas for mercury to overland surface water runoff, and in locations where the spills exceed the adsorption capacity of the soil, leaching to groundwater could occur. The areas where chloropicrin spills have occurred may contain residual contamination. However, based on its relatively high solubility and low affinity for soil, the chloropicrin product is expected to have passed through soil to groundwater. The residual chloropicrin in soils is not expected to be a significant continuing source of groundwater contamination.

6.2.5 Stormwater Drainages

The north ditch and southerly stream flow from the plant area to the southern cove and the Penobscot River. These drainages are part of the plant stormwater collection system and are included in the plant's general stormwater discharge permit. Surface impoundments formed by concrete weirs are located below the plant in each drainage. The impoundment on the southerly stream is not paved while the impoundment on the northern ditch is paved, forming what has been referred to as the paved sump. These impoundments were constructed to function as emergency spill control structures.

Surface water and suspended sediment flowing from the plant have transported mercury to the impoundments and to a lesser extent to the sediment beds of the stormwater drainages themselves. Based on groundwater contouring presented in Section 5, the north ditch discharges to groundwater along its upper reach below the paved sump, and may receive recharge from groundwater closer to the Penobscot River. The opposite appears to occur in the southerly stream: groundwater contours indicate that the southerly stream receives groundwater recharge in its upper reaches until it passes beneath a culvert near the PERC property line. Beyond this point, it is above the groundwater table until it falls to the Penobscot River at the southern cove.

The stormwater drainages represent transport pathways, not source areas in themselves. As discussed above, surface water discharging to groundwater may represent a source of groundwater contamination removed from the primary source (e.g. plant area spills).

6.2.6 Penobscot River Sediments

Prior to the construction of Mac's Pond, brine sludge containing mercury was reportedly discharged to the Penobscot River. Transport of mercury may also have occurred from surface water and associated sediment discharge, and from the permitted discharges from the plant through its NPDES outfall. The amount of mercury which may have reached the cove sediments through various routes is unknown.

Sediment near the north ditch outfall was sampled by the Federal Water Pollution Control Administration and the Federal Water Quality Administration in 1970 and 1971. Additional sediment samples were collected by Acheron and EPA in 1990. The available historical data is provided in Appendix B. The samples collected in 1990 and 1995 contained significantly lower concentrations than the samples collected in 1970 and 1971. Samples collected in 1997 indicate that mercury is still present in cove sediments.

6.3 Waste Characteristics

6.3.1 Properties of Mercury

Mercury, the primary inorganic contaminant of concern, has a number of important chemical properties that determine how this substance cycles through the different media in the environment. Mercury exists in the environment in three main forms: elemental mercury, ionic mercury, and organometallic mercury.

Elemental Mercury, Hg(0)

Hg(0) is a silver liquid that is considerably heavier than water, with a specific gravity of 13.594, at 20°C. Because of its liquidity at room temperature, uniform volume expansion, and high surface tension, mercury has found widespread use in measuring devices, such as thermometers and barometers. It has a solubility of 60 µg/L in water at 25°C (Stumm and Morgan, 1981), and a vapor pressure of 2.58×10^{-6} atm (or 1.96×10^{-3} mm Hg) at 25°C (Lide, 1994). This vapor pressure is large when compared with the vapor pressure of other metals at 25°C, as shown in **Table 6-1**. The higher the vapor pressure, the more likely it will volatilize, or evaporate, into the atmosphere. Conversely, the lower the vapor pressure, the more stable the element, and the less likely it will undergo this change of state. Elemental mercury therefore has the potential to volatilize when exposed to air, especially at higher temperatures.

Because elemental mercury has a high vapor pressure and very low water solubility, its Henry's constant (K_h), a constant that relates the metal's equilibrium concentration in air compared to its equilibrium concentration in water, is large for a metal. K_h is approximately 0.35 at 25°C (Stumm and Morgan, 1981). At normal temperatures, this form of mercury is readily lost by volatilization from water solutions.

Ionic (Oxidized) Mercury, Hg(I) or Hg(II)

Ionic mercury is the dominant form of mercury in natural waters, and is present in a variety of chemical species: Hg^{2+} , $HgCl_2$, $HgCl_3^-$, $HgCl_4^{2-}$, $HgOH^+$, $Hg(OH)_2$, and other inorganic complexes (Morel and Mason, 1994).

Some important mercury salts are listed below (Lide, 1994):

- Mercuric chloride ($HgCl_2$) - corrosive sublimate, a poison
- Mercurous chloride (Hg_2Cl_2) - calomel, occasionally still used in medicine
- Mercury fulminate ($Hg(ONC)_2$) - a detonator widely used in explosives
- Mercuric sulfide (HgS) - vermilion, a paint pigment

Mercury salts are fairly soluble in water. Mercuric chloride has a solubility of 69,000 mg/l at 20°C, one million times the solubility of elemental mercury. According to Stumm and Morgan (1981) the major Hg(II) species found in solution are $Hg(OH)_2$ in fresh water and $HgCl_4^{2-}$ in seawater.

Mercury(II) forms strong complexes with organics. The literature indicates that it has a strong affinity for sulfhydryl groups and amino groups in proteins, amino acids, and other organics (Bodek *et al.*, 1988). Mercury can also bind to inorganic materials in soil and sediment. The partitioning of mercury with soil and sediment is discussed in section 6.3.3.

Organometallic Mercury

Organometallic mercury occurs predominantly as methylmercury (CH_3Hg^+) and dimethyl mercury ($(CH_3)_2Hg$). These forms of mercury are usually produced from a chemical conversion of Hg(II), by bacteria capable of this transformation. From the standpoint of human health, this transformation is undesirable because methylmercury and dimethyl mercury may be taken up by benthic organisms which live in sediments.

Once methylmercury is assimilated into benthic organisms, it can bioaccumulate in aquatic species. Bioaccumulation is the process by which a toxic chemical builds up in the fatty tissue of an organism over time. Because methylmercury is more soluble in lipids than in water, it bioaccumulates in lipids. Typically, small doses over long periods of time may result in elevated contaminant levels as the organism ages. Therefore, fish may concentrate methylmercury as it travels up the food chain, and elevated concentrations may be encountered in the flesh of the higher organisms. Bioaccumulation of methylmercury is enhanced because this compound is relatively unreactive, and resistant to decomposition (Stumm and Morgan, 1981). Methylmercury compounds, once formed, are not readily demethylated.

6.3.2 Atmospheric Cycling of Mercury

As mercury cycles through the air, water, and soil/sediment, it may undergo many chemical and biological transformations. Its transportation is complicated by a large number of potential pathways. However, the understanding of mercury's behavior has improved measurably over the past 10 years due to improved sampling and analysis techniques.

Atmospheric deposition and input from the watershed are the major sources of mercury to lakes lacking point source pollution. The majority of mercury entering lakes from these sources is inorganic mercury, Hg(II). Methylmercury is rarely directly deposited into lakes. Instead, it is formed within the lake by the methylation of Hg(II) (Winfrey and Rudd, 1990).

Hg(II) may be reduced to Hg(0) in lake water by microbial activity. This results in a supersaturation of Hg(0) in the water compared to the atmosphere, and due to its high vapor pressure, Hg(0) can easily volatilize (Driscoll *et al.*, 1994). In the atmosphere, direct or indirect photochemical processes reoxidize Hg(0) to Hg(II) which is then scavenged by rain and redeposited on land or water (Morel and Mason, 1994).

With the presence of hydrogen sulfide in lakes, Hg(II) may not be reduced, but instead may precipitate as insoluble HgS. This is generally assumed to render the mercury unavailable for methylation (Winfrey and Rudd, 1990). Otherwise, Hg(II) can be scavenged by dead and live particles. A fraction of this sinks to the bottom, and may be methylated by microorganisms in the sediment. Much of the methylmercury progresses up the food chain. It is therefore methylmercury and its precursor, Hg(II) (not Hg(0)), that are of concern with respect to effects on biota (Morel and Mason, 1994).

Once synthesized, however, methylmercury may not remain in that form. It may either be further methylated to dimethyl mercury ($\text{CH}_3\text{-Hg-CH}_3$), or, although resistant to decomposition, maybe demethylated by microbial processes. The vapor pressure of dimethyl mercury is 8.3 Kda at 25°C (Lide, 1994). This value is high, which implies that the compound is readily released to the atmosphere. However, the formation of dimethyl mercury primarily occurs at high pH (>7), and it is not expected to be significant at low pH (Winfrey and Rudd, 1990). Demethylation is a microbial process mediated by methylmercury-resistant bacteria. The transformation results in the formation of elemental mercury (with ionic mercury as an intermediate) and methane (Winfrey and Rudd, 1990).

6.3.3 Mercury Partitioning

When mercury is present in the water column, it may bind to solid particles or to sediment. A partition coefficient is a unitless constant that describes the ratio of a chemical's equilibrium concentration on the solid substrate to its equilibrium concentration in the aqueous phase. Although partition coefficient (K_d) values for mercury are very sensitive to geochemical parameters such as pH, sorption capacity of soils, organic activity, and redox conditions, a number of studies have documented K_d values for various environments. The literature indicates that K_d for mercury in sediments is on the order of $1.34\text{-}1.88 \times 10^5$ (Moore and Ramamoorthy, 1984). K_d values for mercury in soils are tabulated in the EPA soil screening guidance documentation, and range from 10 to 5,280 (EPA, 1996). These values indicate that mercury greatly favors sedimentation by adhering to solid particles, rather than remaining dissolved in the water column. Therefore, the mobility of mercury in soils and sediments is limited, and can result in greater accumulations in these media. Mercury sorption favors highly organic, fine sediments due to complexation equilibria associated with organics, and increased surface area and cation exchange associated with clay minerals.

The strong complexation of mercury with organic material in sediments is evidenced at the HoltraChem site by the results of mercury speciation and pore water extraction analyses. Speciation analysis of sediments from the southern cove by sequential extraction indicated that an average of 99.5% of the mercury in site sediments is strongly bound (i.e., was not liberated by digestion in 1 N HCL, a weak acid).

Site-specific mercury partitioning coefficients for sediments were calculated using centrifugation and pore water analyses of sediments from the Penobscot River. The mean partitioning coefficient, K_d , for mercury in sediments described as "organic" (>10% TOC) was approximately 640,000, while the mean K_d for sediments described as "granular" (<1% TOC) was approximately 54,000. These site-

specific values compare with or are higher than published values ($1.34 - 1.88 \times 10^{-5}$, Moore and Ramamoorthy, 1984). These analyses indicate a strong affinity of mercury for the solid phase, particularly solids with high organic carbon content.

Site-specific mercury partitioning coefficients were also calculated for site soils, using the results of batch desorption analyses. Although the desorption results did not allow a representative Freundlich isotherm to be established, point estimates of K_d were calculated using individual water and soil mercury concentrations from six batch desorption results. Site-specific desorption results indicated a range of K_d values from 2,107 to 8,647, with an average of 5,493. The mean of the site specific results is in good agreement with the upper range of values tabulated in the EPA soil screening guidance documentation (5,280, EPA, 1996).

6.3.4 Fate and Transport of Organic Contaminants

The organic contaminants cycle differently from mercury due to their distinct chemical makeup. Several factors which become important in characterizing the migration of organic contaminants include the compound's solubility, volatility, and biodegradation potential. These items play a major role in determining whether or not high concentrations are expected in soil and/or groundwater. The organic chemicals detected at this site include: acetone, carbon disulfide, carbon tetrachloride, chloroform, chloropicrin, m-cresol, and p-cresol. A brief description of the fate and transport characteristics of these chemicals is given below.

Acetone

Acetone is a highly volatile and highly miscible organic contaminant (vapor pressure = 2.7×10^2 mm Hg; water solubility = 100%). When released to the environment, acetone will volatilize and may leach into the groundwater. Acetone's miscibility in water would suggest that it does not adsorb appreciably to soil. The organic partition coefficient of acetone is 2.2 ml/g. There is evidence suggesting that acetone biodegrades fairly rapidly in soils.

If released into water, acetone will likely both biodegrade and volatilize to the atmosphere. Adsorption to sediment is not expected to be significant, and the potential for acetone bioconcentration in fish is negligible (Howard, 1990b).

Carbon Disulfide

Carbon disulfide has a vapor pressure of 2.6×10^2 mm Hg and a water solubility of 2.3×10^3 mg/l. Upon release, carbon disulfide will primarily be lost by volatilization. Since it has a low absorbitivity to soil organic matter ($K_{oc} = 54$ ml/g), it may also readily leach into groundwater. This compound is considered difficult to degrade, and therefore may persist for some length of time. Carbon disulfide is not expected to bioconcentrate significantly in aquatic organisms (Howard, 1990b).

Carbon Tetrachloride

The vapor pressure of carbon tetrachloride is 90 mm Hg, and its water solubility is 8.0×10^2 mg/l. The organic partition coefficient and octanol-water coefficients for carbon tetrachloride are 1.1×10^2 ml/g and 4.36×10^2 , respectively. Based on its high vapor pressure, carbon tetrachloride is expected to readily evaporate from soil. Its low soil adsorption coefficient suggests that it will be moderately mobile in soil, and only slightly adsorbed to sediment. Under anaerobic conditions, carbon tetrachloride is expected to biodegrade to chloroform. Studies suggest that the chemical has a low potential to bioaccumulate in aquatic organisms (Howard, 1990b).

Chloroform

Chloroform has a vapor pressure of 1.6×10^2 mm Hg, and a water solubility of 8.0×10^3 mg/l. Chloroform is expected to evaporate rapidly to the atmosphere from water and from near-surface soils due to its high vapor pressure. The octanol-water coefficient for chloroform is 93.3, and its organic carbon partition coefficient is 31 ml/g. In decreasing order of affinity, chloroform can be adsorbed to peat moss, clay, and sand. It poorly adsorbs to soils with low organic carbon content, and can therefore leach into groundwater. Under anaerobic conditions, chloroform may also be subject to slow biodegradation in soil and groundwater. The substance will not significantly hydrolyze under normal environmental conditions, and there is little or no tendency to bioconcentrate in aquatic species (Howard, 1990b).

Chloropicrin

Chloropicrin has a vapor pressure of 18.3 mm Hg, and a water solubility of 400 mg/l. If spilled on soil, chloropicrin will rapidly volatilize and may leach into groundwater. It will also photodegrade in the surface layers of water. Under simulated atmospheric conditions, the vapor photodegrades to phosgene and nitrosyl chloride, with nitrosyl chloride further photolyzing to chlorine and nitric

oxide. Chloropicrin has been found to adsorb more to mineral soil than to muck, which has a higher organic content, suggesting that adsorption to organic matter is not the principle mechanism for adsorption. Chloropicrin is not expected to adsorb to sediment, and little or no bioconcentration is expected to occur in fish (Howard, 1990c).

Cresols

Cresols have a relatively high water solubility and a low vapor pressure (water solubility = 2.15×10^4 mg/l; vapor pressure = 1.0 mm Hg). The organic carbon partition coefficient and octanol-water coefficients for the cresols are 5.0×10^2 ml/g and 93.3, respectively.

m-Cresol

This compound is mobile in soil, and therefore has the potential to leach into groundwater. It has a low potential to volatilize from water, and instead, partitioning will predominantly occur into the aqueous phase. The substance generally biodegrades readily in soil and surface water alike, while bioconcentration does not significantly occur in fish (Howard, 1990a).

p-Cresol

Based on its chemical properties, p-cresol is relatively mobile in some soils, and may leach into the groundwater. It biodegrades rapidly in soil and water. P-cresol is not expected to bioaccumulate in fish (Howard, 1990a).

Table 6-2 summarizes the physical characteristics of the compounds detected at the Orrington site.

6.4 References

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Section 7

Contaminant Characterization & Fate and Transport

7.1 Introduction

This section presents descriptions of contaminant distribution in the various environmental media at the site and nearby environs. Temporal changes in concentrations are also discussed. Finally, the fate and transport of the environmental contamination are discussed. Information presented in previous sections, combined with the results of SI sampling and analysis, provide the basis of the discussion. Each of the site media is addressed separately in the subsections below.

7.2 Contaminant Characterization

7.2.1 Surface and Near Surface Soil

Soil samples were collected to investigate certain identified spill areas within the plant, and to provide additional samples from less developed portions of the site and background locations. Additional samples were collected in conjunction with soil invertebrate (earthworm) sampling to evaluate the environment to which these biota were exposed. Prior to the SI, Acheron collected soil samples in less developed portions of the site and at off site locations. The location of Acheron samples and the results of their analyses are provided in Appendix B.

Plant Area Soils

Soil sampling in 1995 focused mainly on investigating the plant area, specifically areas of known or suspected spills or releases. Five additional samples were collected in 1995 from sitewide locations, including the landfill ridge and southern cove area, and one from a background location.

In general, the results of the 1995 soil sampling in the plant area identified the primary brine handling areas as potential source areas due to brine spills. **Figure 7-1** shows the location of in-plant soil sampling and presents the results of SI mercury, pH and cyanide analyses. The results of the second sampling round later in 1995 confirmed the results of the initial soil sampling, and provided more information on the extent of contamination. The limits of each separate spill were not established; rather, a general area of the plant was defined where soil is expected to contain elevated concentrations of mercury. This area begins south of the cell building in the vicinity of the chlorine

storage tanks, and extends west of the cell building to include the area around the sludge pits, (brine filter backwash receiver tanks) former clarifier, former coal filters, brine storage area, heat exchanger, brine system pump houses and the depleted brine receiver tanks.

With few exceptions, the near surface soil samples collected from a depth of 1.0 to 1.5 feet below ground surface contained less mercury than the corresponding surficial soil samples. In all but seven of the 49 locations sampled, the concentration of the second depth interval sample was below 20 mg/kg. The locations where concentrations remained above 20 mg/kg in the 1.0 to 1.5 foot interval included the areas south of the cell building; the sludge pits; the clarifier; the heat exchanger; and the depleted brine storage area. Exceptions to the decreasing concentration with depth trend were noted adjacent to the sludge pits (SSS-05F), northeast of the heat exchanger (SSS-07C), and at the western corner of the HCl tank farm (SSS-08C).

A ratio of the concentration at the surface to that at the 1.0-1.5 depth interval was calculated to evaluate the amount of attenuation observed with depth. The mean ratio was 8.0 while the median value was 2.4. These ratios illustrate that rapid attenuation with depth occurs for mercury in soil. However, two exceptions were noted in areas where large quantities of brine are reported to have been spilled.

Other reported spill areas within the plant area, which were identified in the 1995 Work Plan and sampled during the SI, did not contain similarly elevated concentrations of mercury. These areas include:

- east of the former chlorate plant (SSS-01A);
- the transformer yard east of the cell building (SSS-02B);
- the HCl tank farm area (SSS-08A & B);
- the chloropicrin plant area (SSS-09A & B);
- the salt storage pad area (SSS-10A & B); and
- the terminus of the railroad loading area (SSS-11A & B).

The concentrations of mercury detected in these areas were typically less than 10 mg/kg. Higher concentrations were observed in the surficial soil samples in the transformer yard (SSS-02C) and in the eroded section of the stormwater ditch immediately above the paved sump (SSS-12A).

Concentrations of mercury in near surface soil samples collected in 1995 outside of the plant area (SSS-15A through SSS-20A) were lower, ranging from 0.18 mg/kg at the background location (SSS-18A) to 10 mg/kg (SSS-16A). The concentration of 10 mg/kg was detected in the area where fill excavated from the plant during construction of the former sodium chlorate plant was deposited on

the ridge. Other samples collected on the ridge in 1995 contained 1.2 and 2.8 mg/kg mercury, respectively. The concentrations detected outside of the fill area are consistent with the values obtained by Acheron, which ranged from 0.08 mg/kg to 6.55 mg/kg. The source of this mercury in surface soils is considered to be air deposition from plant operations. A sample collected by Acheron one inch below the 6.55 mg/kg sample contained only 0.40 mg/kg, a distribution pattern which is consistent with contaminant distributions expected from air deposition.

In addition to mercury, samples were analyzed for VOCs and chloropicrin in 1995, and additional samples were collected for VOCs in 1997. In 1995, samples were collected from the 1.0 to 1.5 foot interval from six selected locations where chloropicrin was known to have been handled or spilled (SSS-09A through SSS-11B). Chloropicrin was detected at an estimated concentration of 320 $\mu\text{g}/\text{kg}$ in one of two samples collected at the end of the railroad car loading area (SSS-11B). It was also detected at an estimated concentration of 9 $\mu\text{g}/\text{kg}$ in one of two samples collected in the vicinity of the chloropicrin plant (SSS-09B). Other VOCs were not detected in any of the soil samples. In 1997, three samples (SSS-071-01, SSS-072-01, and SSS-073-01) were collected in the vicinity of the fiberglass repair building and analyzed for VOCs. VOCs were not detected in the hand auger holes with an OVM or by the laboratory in the samples collected in the vicinity of the fiberglass repair building.

Samples for total cyanide were collected in 1995 from two depth intervals at two locations (SSS-10C and SSS-10D) adjacent to the salt storage pad to investigate the possibility that sodium ferrocyanide, potentially used as an anti-caking agent, was released in this area. No cyanide was detected.

Soil samples collected in 1995 were analyzed for pH. The pH of the samples collected from within the plant area were almost all in the 8 to 11 pH standard unit range. The source of the elevated pH is most likely the sodium hydroxide added to the brine to raise its pH prior to brine treatment. The pH values for samples collected outside of the plant were between 6 and 7, which is typical of soil in coniferous forests.

Four samples were collected in 1997 in the sand blast grit area and analyzed for the eight RCRA regulated metals (arsenic, barium, cadmium, chromium, lead, mercury, selenium, and silver). These samples contained concentrations of mercury ranging from 0.64 to 310 mg/kg. These concentrations are attributed to plant maintenance activities as materials are removed from brine handling areas. Eight delineation soil samples were collected surrounding samples SSS-065-01 (29 ppm) and SSS-067-01 (310 ppm). The samples collected in the vicinity of SSS-065-01 location showed an increase in concentration of mercury in the vicinity of a scrap metal pile. The mercury concentrations detected in delineation samples surrounding SSS-065-01 and SSS-067-01 are presented on **Figure 7-2** and **Figure 7-3**, respectively.

In summary, surface and near surface soil within the brine handling areas of the plant contain concentrations of mercury greater than 10 mg/kg. The source of mercury within the manufacturing portions of the plant is most likely brine and potentially product spills. Accordingly, the extent of concentrations greater than 10 mg/kg resulting from spills is expected to be limited to the immediate manufacturing plant area. In addition, mercury concentrations up to 310 mg/kg were detected within the sand blast grit area at the northeast corner of the plant. The extent of these elevated concentrations were approximately defined with delineation samples and are expected to be limited to the sand blast grit and scrap metal storage area.

Sitewide Soils

In 1997, a total of 43 sitewide soil samples were collected to evaluate the effects of potential air deposition of mercury from plant operations. These samples included 39 collected throughout the HoltraChem site (excluding areas of plant activity) and four samples on the PERC property. To assess the relationship between soil mercury content and proximity to the HoltraChem plant, soil samples were collected in the plant vicinity and at distances of up to 4000 feet from the plant. The results of the sitewide soil sampling for mercury are presented on **Figure 7-4**. The mercury concentrations detected in the soils, with the exception of samples collected from known fill or release areas, are attributed to air deposition primarily from plant activities and secondarily from off site sources.

One sample (SSS-018) in 1995 and 3 samples (SSS-054 through SSS-056) in 1997 were collected for mercury analysis from a background location in Dorthea Dix park in Hampden Maine. Mercury in each of these samples was below the method detection limit, which ranged from 0.18 to 0.32 mg/kg.

The sitewide soil data was statistically analyzed to evaluate the distribution patterns of mercury concentrations. The data set was prepared as follows:

- The value of the detection limit was used for samples with mercury concentrations reported below the method detection limit;
- The mean value of the parent and duplicate results was used;
- Background samples were not considered in the analysis; and

- Samples SSS-024 and 025, which, based on their concentrations relative to adjacent samples, were sampled in locations of believed spills, releases, or fill areas, were not considered in the analysis. Additionally, samples collected on recently capped landfills (SSS-030 and SSS-036) which contained mercury levels below the method detection limit were not considered.

The mercury concentrations of the 39 sitewide soil samples ranged from 0.2 to 6.2 mg/kg, with a mean concentration of 1.08 mg/kg and a median concentration of 0.49 mg/kg. The concentrations of mercury in surficial soils collected at distances of over 2000 feet from the HoltraChem site (seven samples) were all below the analytical detection limit ranging from 0.20 to 0.34 mg/kg. The surficial soil data set was evaluated using geostatistical techniques, which illustrated decreasing concentrations as distance from the site increased. A variogram model was developed and used to describe the spatial correlation of the data. The variogram was then used to develop estimates of soil mercury concentrations at unsampled locations by kriging. The kriged data concentrations were contoured and overlaid onto a site base map. The estimated surficial soil mercury concentration contours are shown on Figure 7-2. The kriged estimates indicate that soil mercury concentrations in the areas outside the HoltraChem property are expected to be less than 0.5 mg/kg. The data indicate that airborne mercury originating from the HoltraChem site has not contributed measurably to off-site soil mercury concentrations.

Contours of the data indicate an area of elevated mercury concentrations due to air deposition on the landfill ridge. The prevailing wind direction is generally from the south/southwest. The highest mercury concentrations resulting from air deposition were detected north and northeast of the plant area, which correlates with the prevailing wind. Concentrations within this area ranged from 1.0 to 4.9 mg/kg. Mercury was detected at concentrations of 3.2 mg/kg (SSS-035) and 6.2 mg/kg (SSS-053) in a fill area northeast of the plant and north of Landfill 4. This area, originally mined for sand and gravel, was filled with soils from the excavation for the construction of the sodium chlorate building. Mercury concentrations potentially associated with the filled area are co-located with the area where elevated concentrations are expected due to air deposition. Higher concentrations of mercury were detected in subsurface samples collected in the fill. Therefore, it is likely that the highest mercury concentrations detected in surface soil in this area are due to filling rather than to air deposition.

The sitewide soil sampling in 1997 identified two areas, which based on their mercury concentrations relative to adjacent samples, do not indicate an air deposition source: SSS-024-01 (19 mg/kg), southeast of the north ditch, and SSS-025-01 (52 mg/kg), south of the plant and west of the outfall 001 treatment wier. Fourteen delineation samples collected surrounding SSS-024-01 contained mercury at concentrations ranging from non-detect to 39.2 mg/kg. The locations of these delineation samples are shown in Figure 7-3. With one exception, delineation samples collected

from the deeper interval (1.0 to 1.5 feet) contained lower mercury concentrations than those collected from ground surface to 0.5 feet. Based on an observation of graphite anodes in soils at this location, the mercury detected adjacent to the north ditch appears to be associated with plant fill. According to Mr. Gene Bridges of H.E. Bridges Construction, who has performed much of the excavation work around the plant, the area below the plant has been regraded, the north ditch has been excavated, and some material from the sodium chlorate plant may have been placed in this area.

Delineation samples associated with SSS-025 contained mercury at concentrations ranging from 0.4 to 42.5 mg/kg. The sampling locations in the vicinity of SSS-025 are shown on **Figure 7-6**. Mercury at SSS-025 is believed to be associated with overflows of a weir at the pH neutralization tank, or to the cleaning of sludge from the tank. Samples collected at a depth of 1.0 to 1.5 feet contained lower concentrations of mercury (1.1 mg/kg and 4.8 mg/kg) than those collected at SSS-025-01 (ground surface to 0.5 feet).

In summary, soil concentrations of mercury (1- 4 mg/kg) were documented on the landfill ridge and below the plant which are attributed to air deposition. The highest concentrations attributable to air deposition were observed on the bedrock ridge in the vicinity of the landfills. The mercury concentrations decrease rapidly with increased distance from the plant and are below analytical detection limits (0.18 to 0.32 mg/kg) within approximately 2000 feet of the plant in the prevailing downwind direction and less than 2000 feet in other directions. Surface and near surface soil with concentrations greater than 10 ppm were observed outside of the plant area in three locations: on the ridge in the area where fill was deposited during construction of the sodium chlorate plant, adjacent to the north ditch, and adjacent to the pH adjustment wier. All of these detections of mercury are associated with fill material, and the approximate limits of these fill areas have been delineated.

7.2.2 Subsurface Soil

Ridge Area

Three borings (B-407, B-411 and B-412) advanced on the ridge in 1995 characterize the area around the swale north of Landfill 4, which was reported to have been filled. Boring B-412 encountered approximately five feet of fill close to the ground surface. It was advanced to 24 feet but did not encounter as much fill material as anticipated. This boring was abandoned and a new location chosen closer to the drainage swale. B-411 was advanced in the center of the swale and penetrated seven feet of fill. Groundwater was not encountered in the overburden at this location.

Boring B-407 was advanced approximately 50 feet west of the swale north of Landfill 4. The depth of the fill in this boring is 17 feet and fill is underlain by sand, gravel and silt above bedrock. Soil samples collected during the advancement of boring B-411 were analyzed for mercury to assess contamination with depth. Concentrations as high as 64 mg/kg were detected (2'-4'). The soil concentrations dropped below 10 mg/kg at 10 feet, and below detection limits at 22 feet, with the exception of one sample from 24 to 26 feet below ground surface which contained 2.3 mg/kg. The extent of fill north of Landfill 4 was presented on Figure 5-3.

B-506 was advanced through Landfill 4 to evaluate the thickness of the landfill material, mercury concentrations in the fill, and the depth to bedrock. The highest concentration of mercury detected in soil samples was 239 mg/kg in a sample collected from 2-4 feet below the Hypalon cap. Soil mercury concentrations below that depth ranged from 1.26 to 95.51 mg/kg. The last soil sample, which was collected between 14 and 16 feet below the Hypalon cap, contained 19.64 mg/kg. Split-spoon refusal was encountered at 16 feet. The base of fill and top of bedrock coincide at 81.3 feet PD at this location.

Plant Area

Leach Field Investigations

Subsurface soil investigations were conducted on leach field Nos. 1, 2, and 8. The investigation of leach field No. 1 was performed by Acheron on July 10, 1997. Soil samples were collected using a Geoprobe direct push sampler. Mercury concentrations in the samples collected from leach field No.1 ranged from 0.140 to 13.67 mg/kg. A summary of the leach field No. 1 investigation can be found in the Acheron letter report "Summary Report - Leach Field Investigation," dated July 25, 1997.

Leach field No. 2 was sampled on August 17, 1998 using a Geoprobe sampler. A layer of black gravel with trace sand was encountered at approximately 6 feet below ground surface at 3 of the 4 sampling locations. The gravel layer was very wet and contained a slight sheen. Based on the elevations shown on plans of the former leach field piping and chambers, this observed strata is likely at the depth of the pre-1975 leach field interval.

Concentrations of mercury detected in the samples analyzed by the HoltraChem lab ranged from non-detect to 17.92 mg/kg. Mercury was detected in very low levels at boring LF2-1 (less than 0.03 ppm) and no evidence of a leach field stratum was observed; therefore it is not likely that this boring encountered the leach field distribution lines.

At borings LF2-2 and LF2-3, mercury was detected at maximum concentrations of 0.96 and 14.35 mg/kg, respectively. The highest concentrations correspond to the black, wet gravel material which is likely the leach field, at a depth of six feet below ground surface. The concentrations of mercury rapidly decreased with depth, as the strata changed to less permeable silt, then clay at a depth of 10 feet below ground surface.

A mercury concentration of 1.38 mg/kg was detected at LF2-2, in the sample collected from two to four feet below ground surface. A concentration of 10.72 mg/kg was detected from two to four feet at LF2-3. Plans show the elevation of the soil at the bottom of the concrete leach field chambers to be at an elevation of El. 96. The ground surface elevations at borings LF2-2 and LF2-3 are El. 98 and El. 99, respectively. Therefore, the mercury in the samples collected from 2 feet below ground surface (El. 96-97) is likely associated with the leaching chambers.

Geoprobe boring LF2-4 was advanced through an access manhole into a leach field chamber. Mercury concentrations of 17.92 and 1.84 mg/kg were detected in the soil within the manhole at depths of three to four feet and four to six feet below ground surface, respectively. The soil with the highest concentration of mercury was collected from a pile of soil directly beneath the access manhole, three feet below the soil surface. This mercury contamination may be associated with the distribution of mercury within the leach field chambers.

Based on the absence of elevated mercury concentrations in the geoprobe boring LF2-1, and the rapid attenuation with depth observed in each geoprobe boring, the distribution of mercury in the leach field No. 2 system appears to be limited to the close proximity of the leach lines and leaching chambers. Based on the fine grained, low permeability composition of the soils observed beneath the leach lines and chambers, the migration potential for the mercury detected in the leach field No. 2 system is low.

Leach field No. 8 was located between the HCl tank farm and the lined process lagoon in the main plant. Existing plans of the leach field indicated a series of distribution lines extending from a distribution box. However, according to Gene Bridges of H.E. Bridges Construction, who has performed much of the excavation work on site, the leach field consisted of hemlock planks which were located parallel to the bedrock scarp west of the HCl tank farm and extending toward the lined process lagoon. Mr. Bridges indicated that the planks were likely removed during subsequent work in the area.

Subsurface soils at leach field No. 8 were investigated by excavating three test pits. Test pit logs describing the subsurface materials encountered are included in Appendix D. No evidence of leach field distribution lines or hemlock planks was observed in the test pit trenches. All three test pits

were excavated through asphalt, underlain by a layer of fill consisting of brown coarse to fine sand with varying amounts of gravel. In test pit LF8-TP1, a layer of gravel was observed. This material is likely the former road surface material. In each of the test pits, a layer of sandy fill was observed at thicknesses varying from two feet at LF8-TP1 to five feet at LF8-TP3. According to Gene Bridges, the excavation contractor, this material is trench backfill used in utility trenches. Native material, consisting of orange brown and tan silt with traces of clay and fine sand, was observed at a depth of three feet in LF8-TP1 and 2.5 feet in LF8-TP2.

Concentrations of mercury in the samples collected from the leach field No. 8 test pits ranged from 0.04 to 2.50 mg/kg. A concentration of 2.22 mg/kg was detected in a thin dark brown coarse sand and fine gravel layer, approximately 2 feet below ground surface in test pit LF8-TP3. The sample collected at three feet below ground surface in the light brown fine sand with little silt layer contained mercury at a concentration of 2.50 mg/kg.

Based on the absence of observed hemlock planks or stained soils which might be associated with leach fields, and the relatively low concentrations of mercury detected in subsurface soils, it is apparent that leach field No. 8 was removed during earlier plant utility construction activities.

Brine Tank Investigation

Additional subsurface soil investigations during 1997 were conducted within the plant by Acheron, as part of the brine tank release investigation. This investigation was initiated following a leak from the north brine tank, discovered by HoltraChem on February 19, 1997. The brine tank investigation included the advancement of eight geoprobe soil borings in the vicinity of the brine tanks and subsurface utilities. Soil samples from these geoprobe borings were collected and analyzed for mercury at the HoltraChem laboratory.

The results of the brine tank investigation indicate that elevated mercury concentrations are generally concentrated in surficial soils within the plant area. Concentrations of mercury in surface soils (1-3 ft bgs) in the plant area ranged from less than 1 mg/kg to approximately 10 mg/kg. The highest mercury concentrations at depth were found in locations GP-7 and GP-8, both of which are located between the loading shed and the Outfall 001 pH adjustment weir. In each of these borings, mercury concentrations greater than 1 ppm were found to a depth of more than 7 feet. Contamination in these borings is attributed to releases from the plant, notably the brine tank, which may have traveled along the industrial sewer leading to the weir and an existing subsurface groundwater collection system.

Soil Borings

During the monitoring well installation program, mercury samples were collected to profile contamination with depth at four locations surrounding the manufacturing plant. At B-403, located along the bedrock scarp above Landfill Area 1 and downgradient of the plant, a concentration of 2.5 mg/kg was detected in the 0-2 foot sample and concentrations below 1 mg/kg were detected below that to a depth of 12 feet. Mercury was not detected above the method detection limit of 0.1 mg/kg below the 12 foot depth (mercury in the 16 to 18 foot sample was at the detection limit.)

Borings B-502, B-503 and B-510 were advanced downgradient from the main plant area, along the crest of the hill above the river. Boring B-510 encountered a maximum concentration of 4.8 mg/kg from 0-2 feet below ground surface. Below four feet, mercury concentrations were generally below 1 mg/kg. Mercury concentrations in B-502 were consistently below 1 mg/kg, with a maximum concentration of approximately 1 mg/kg at the surface. Concentrations of mercury in B-503 were also consistently below 1 mg/kg, with a maximum concentration of 0.28 in the top 2 feet. Contaminated subsurface soils in the area directly below the plant appear to be confined to surficial fill material.

In summary, contaminated soils in the areas peripheral to the plant appear to be confined to the surface or near surface. Data from in plant surface and near surface soil sampling and from the Brine Tank Investigation performed by Acheron indicate that, in general, mercury contamination in the plant is also limited to surface and near surface soils. However, areas within the plant which may have received large brine spills (e.g. the brine sludge receiver tanks, the clarifier and the heat exchanger) have localized areas where mercury contamination extends to greater depths. Investigations have focused on the periphery of the manufacturing area and on specific releases and potential releases in the plant (i.e. the Brine Tank Investigation, and leach fields investigations). The investigation of leach field No. 2 documented that low permeability silt and clay soil underlying that area of the plant limited downward migration of mercury. Where the silt and clay is not present, till deposits, which underlie the remainder of the plant area, are expected to have a similar effect.

River Area

Lined Process Lagoon Investigation

Five Geoprobe borings were advanced beneath the lined process lagoon in October of 1998. Soils recovered from beneath the lagoon liner consisted of brown to orange-brown medium to coarse sand and fine gravel, underlain by brown and grey tills. Mercury concentrations in soils recovered from beneath the lined process lagoon ranged from 0.02 mg/kg to 30.4 mg/kg. The highest concentrations

of mercury were associated with samples collected from the first two depth intervals beneath the center of the lagoon, with lenses of oxidized sand and gravel deposits, and with soils close to the sand/till interface. The lowest concentrations of mercury were found in soils collected from within the till.

Soils beneath the lagoon were observed to have low amounts of organic or fine-grained constituents, indicating a low capacity for contaminant adsorption. Mercury in the coarse grained soils appeared to be concentrated in zones of oxidation, where chemical coprecipitation of mercury with iron was favorable. Mercury was also concentrated along the sand/till interface, where groundwater penetration was restricted and finer grained materials provided a higher adsorption capacity for mercury. The maximum concentration detected at the till interface was 20 mg/kg. The till appears to significantly limit further downward migration of mercury.

Mercury contamination beneath the lined process lagoon and observations of leaks in the lagoon liners suggest that the lined process lagoon is the likely source for the subsurface soil contamination observed in the five geoprobe borings. Due to the low amounts of organic and fine-grained constituents in soils directly beneath the lagoon, adsorption of mercury to soils does not appear to be significant until the till surface is reached. The water table below the lined process lagoon is at or just above the top of the till surface. Therefore, releases from the lined process lagoon may be directly contaminating groundwater below the lagoon.

Soil Borings

Three soil borings were advanced in and below Landfill Area 1 during the 1997 investigation. B-501-O1 was advanced through Landfill Area 1, directly downgradient from the lined process lagoon. The highest soil mercury concentration detected in this boring was 17.84 mg/kg, collected from 4 to 6 feet below the Hypalon cap. Soil concentrations below this depth were less than or equal to 0.05 mg/kg until a detection of 1.1 mg/kg from 24 to 26 feet and 10.19 mg/kg from 26 to 28 feet below the liner. Based on the low concentrations of mercury detected below 6 feet, it appears that contaminated fill at this location is approximately 6 feet deep and that the base of fill is approximately Elevation 52 PD. The water level in MW-501-O1 was measured at approximately Elevation 30 PD. The higher concentrations detected below the zone of uncontaminated soil are most likely attributable to transport of mercury in groundwater.

Boring B-512 was advanced below Landfill Area 1, in sand and gravel deposits directly above the Penobscot River. Subsurface soil material in this boring consisted of sand and silt from 0-21 ft, underlain by sand and gravel to 45 feet, and till below 45 feet. The maximum mercury concentration in this boring was found at the surface, in fill material directly beneath the landfill liner. Mercury

concentrations at depth were consistently below 1 mg/kg, with a maximum concentration of 0.42 at a depth of 35-37 ft. Higher mercury concentrations in subsurface soils in this area appear to be confined to loose fill material within the landfill.

Boring B-513 was also advanced below Landfill Area 1, between MW-402-O1 and B-326-O3. Soil samples collected from this boring had mercury concentrations ranging from 0.005 to 3.8 mg/kg, dry weight. The maximum mercury concentrations (3.8 mg/kg, maximum) were found in soils directly above the till layer, at depths of 45-47 and 50-52 ft bgs. Elevated mercury concentrations in soils at this location appear to be associated with transport of contaminated groundwater.

Prior to the SI, Acheron collected samples for mercury analysis from three borings advanced in the area between the plant and the river. All of the detectable concentrations were below 10 mg/kg, and the analytical results indicate that contamination is generally limited to the first interval sampled. A mercury concentration of 8.9 mg/kg was noted at a depth of 20 to 22 feet below ground surface. This interval reportedly corresponded to the water table at this location. One of Acheron's borings (SB-A) was located in the vicinity of the former Mac's Pond. Mac's Pond was an unlined surface impoundment located approximately where the paved sump is now.

In summary, the distribution of mercury in subsurface soil beneath and downgradient from the lined process lagoon and Landfill Area 1 indicates that mercury from near surface sources such as the lagoon and fill within the landfill has passed through the coarse grained material and preferentially adsorbed to finer grained material at the base of the sand and gravel and top of the till.

7.2.3 Site Sediment and Surface Water

Sediment

In 1995, surface water and sediment samples were collected from 10 locations within stormwater drainages which carry surface water runoff from the site to the Penobscot River. Additional sediment and surface water samples were collected in the Penobscot River. These are discussed in Section 7.2.4.

Sediment samples from the ten onsite locations were analyzed for mercury and total organic carbon. Surface water samples were analyzed for mercury and the field measured parameters, pH, temperature, dissolved oxygen, salinity, specific conductance and turbidity. The distribution of mercury and VOCs in site sediment and surface water is presented on **Figure 7-7**. The distribution of mercury in site sediment and surface water indicates that runoff of mercury contaminated soils and surface water from the plant is the primary source of contamination in the ditches. No mercury

was detected in the sediments or surface water of the flooded gravel pit located in the valley east and upgradient from the plant (SSW/SSD-001).

The highest mercury concentrations in both surface water and sediment (37 $\mu\text{g/l}$ and 1000 mg/kg, respectively) were detected in the paved sump (SSW/SSD-005). This structure acts as a sedimentation basin for surface drainage Area 2 (See Figure 2-2) and discharges to the north ditch. The sediments in the paved sump were removed in May 1995 as an Interim Stabilization Measure during the SI. Their removal was documented in Bimonthly Progress Report No. 6. Sediment from the paved sump was removed again in April 1998.

A mercury concentration of 78 mg/kg was detected in sediment behind the concrete spill control dam built into the southerly stream. Although surface water is not impounded at the dam under normal conditions, this dam forms a limited sedimentation basin for runoff from surface drainage Area 1. The area of the plant which drains into the southerly stream is the portion east of the cell building where concentrations of mercury are low. The mercury concentrations in sediment in the north ditch and the southerly stream decreased downstream from the plant toward the Penobscot River. The decrease in concentration in sediment is likely due to contaminated sediment settling and being deposited closer to the plant.

One exception to this distribution trend was noted. Mercury was detected at a concentration of 19 mg/kg in SSD-002, which was obtained from the stormwater ditch downstream from Landfill 2. This sample was very fine grained and highly organic (TOC=117,000 mg/kg or 11.7%). Mercury has a strong affinity for fine grained and organic material, and the relatively high concentration at this location could be a result of the smaller grain size and higher TOC concentration. The source of mercury at this location is not known.

On June 3, 1997, Acheron performed additional sediment sampling in the southerly stream. This sampling was performed to assess the impact of a release from the brine receiver tanks, which was discovered by HoltraChem on February 19, 1997. Mercury concentrations in sediment ranged from 7.7 to 91 mg/kg. Three of the Acheron sampling locations from this investigation duplicated locations CDM sampled in September 1994. Although the concentrations detected in the two rounds are similar (78 v. 58 mg/kg, 36 v. 26 mg/kg and 6.5 v. 44 mg/kg, 1994 v. 1997, at CDM locations SSD-006, -007 and -008, respectively), the temporal distribution of concentrations indicates that higher sediment concentrations have progressed downstream from the retention basin and the section of the stream adjacent to the HoltraChem plant. In addition to the brine tank release three months prior to sampling, the concrete dam at the retention basin had been rebuilt since CDM's sampling round because it had washed out during heavy spring flows.

Surface Water

Mercury concentrations in surface water samples collected by CDM in September 1994 ranged from non-detect at SSW-001 and SSW-003 to 37 $\mu\text{g/l}$ at the paved sump (SSW-005). During their brine tank investigation in June of 1997, Acheron also conducted surface water sampling in the southerly stream. Surface water samples from this sampling event contained mercury at concentrations ranging from 0.7 to 1.4 $\mu\text{g/l}$. Acheron's surface water samples were analyzed by the HoltraChem onsite laboratory. The surface water concentrations were slightly lower than those measured by CDM at the same locations.

As part of comprehensive monitoring required by MEDEP pursuant to the Consent Agreement, HoltraChem has collected flow volume and mercury concentration measurements from weirs installed at the paved sump and in the southerly stream since April of 1998. Flow measurements and analytical samples are collected weekly, and analyses are performed at HoltraChem's onsite laboratory. Tables summarizing the results are provided in Appendix B. Spring flows in the southerly stream ranged from 22 gpm to 338 gpm in 1998; total mercury concentrations ranged from 0.4 $\mu\text{g/l}$ to 4.6 $\mu\text{g/l}$, and dissolved mercury concentrations ranged from <0.1 $\mu\text{g/l}$ to 1.3 $\mu\text{g/l}$. Spring flows at the paved sump ranged from 8 gpm to 27.5 gpm. The total mercury concentrations during this season ranged from 8.1 $\mu\text{g/l}$ to 16.8 $\mu\text{g/l}$, and the dissolved mercury concentrations ranged from 5.5 $\mu\text{g/l}$ to 9.3 $\mu\text{g/l}$.

In November 1997, CDM performed a filtration study on the surface waters of the north ditch and southerly stream to support evaluation and design of sediment stabilization measures for the site drainages. The data showed that the majority (75 - 90%) of the mercury load during wet weather/high flow conditions was associated with sediment particles greater than 20 microns (medium silt). During low flow/dry weather conditions, 73% of the mercury in the northern ditch/paved sump water was in a dissolved form (i.e. was not removed by filtering with a 0.45 micron filter). Although 70% of the mercury in the southerly stream low flow/dry weather sample was also in dissolved form, the total concentrations were much lower (0.82 $\mu\text{g/l}$). These results indicate that mercury associated with stormwater runoff is primarily adsorbed to particulate matter. Dissolved mercury loads during low flow conditions are largely attributed to baseflow recharge from groundwater. The results of the filtration analysis are provided in Appendix B.

The flow volume, concentration, and filtration study data from the surface outfalls provide data with which to estimate the flux of contaminants via surface water. The data represent five months of the year, and measurements were taken on a weekly basis. The data appear to indicate that mercury is being transported from the site to the Penobscot River via surface water in both a dissolved and a

particulate associated form. During non-storm events, the mercury flux is lower and a higher percentage of the transport is in the dissolved phase. In addition, the amount of mercury present in the dissolved phase is much greater at outfall 003 than in the southerly stream. Based on the dissolved and total mercury results of monitoring wells upgradient from the paved sump, underdrain discharge, and outfall 003 sampling, it appears that groundwater carrying dissolved mercury is discharging to the utilities which flow to outfall 003. Although contaminated groundwater is also likely to be flowing towards the southerly stream, groundwater is being collected in the area by an interceptor trench installed for that purpose.

7.2.4 Penobscot River Surface Water and Sediments

Penobscot River Sediment

In 1995, sediment samples were collected from two upriver areas (RSD-001 through RSD-002), downgradient of Landfill Area 1 (RSD-003), and from the southern cove (RSD-004 through RSD-008). The results of the 1995 Penobscot River sediment sampling are presented on **Figure 7-8**. At each location, samples were collected from the top 0.1 feet and at approximately 0.8 to 1.0 feet below the sediment surface. In areas where the organic silt was greater than one foot thick, the second sample was collected from the first granular material encountered below the organic silt. At the two background locations and downgradient of Landfill Area 1, samples were collected from the approximate halfway point of the intertidal zone and from just below the low tide mark. In the southern cove, sampling locations were distributed to characterize four distinct environments: the sand and gravel bank, an organic mat, sedge beds and the organic silt covered tidal flat.

The mercury content of the organic silt in the southern cove ranged from 0.54 mg/kg in RSD-08A-01 to 2.9 mg/kg in RSD-07D-01. The average was 1.5 mg/kg, with very little variability around this value. The even distribution of mercury concentrations in the organic silt, and the presence of similar concentrations in the organic silts at the upriver locations, suggest that 1.5 mg/kg represents an approximate background concentration for the organic silt deposits.

Greater variability in mercury concentrations was observed in the granular samples collected in 1995, especially in the second depth interval. Concentrations in this interval ranged from non-detect in numerous samples to one sample at 54 mg/kg (RSD-05C-02). Concentrations from ND (RSD-01A-02 and RSD-02A-02) to 1.2 mg/kg (RSD-01B-02) were observed in the second depth interval at the background locations. The highest concentration of mercury detected during the 1995 sampling round was in the second depth interval sample collected from the sedge bed in the middle of the cove (RSD-05C-02, 54 mg/kg).

Additional samples were collected in the southern cove in August and October of 1997, to further investigate the presence of mercury in sediments. The August 1997 sampling round (RSD-09 through RSD-16) attempted to evaluate mercury concentrations in the vicinity of potential sources, including the north ditch and southerly stream, and the PERC and HoltraChem outfalls, where sediments from plant runoff are likely to accumulate. A total of thirty-seven samples were collected from the same depth intervals as those collected in 1995. In addition, deeper samples were collected where possible from 1.3 to 1.5 feet and from 1.8 to 2.0 feet below the sediment surface.

Sediment coring in October of 1997 (RSC-001 through RSC-026) was performed to delineate the lateral and vertical distribution of mercury identified in August 1997. A total of 26 samples were collected with the sediment coring device. Sediment cores were collected from depths of 2.0 to 2.5 feet, and 2.5 to 3.0 feet below the sediment surface.

The 1995 and 1997 sediment data were statistically analyzed by each depth interval sampled. The data set was prepared as follows:

- The value of the detection limit was used for samples with mercury concentrations reported below the detection limit;
- The mean value of duplicate results was used and referred to with the parent sample name; and
- Background samples and the samples collected at the base of landfill area 1 were not considered in the analysis.

A total of 81 samples were collected from a depth of 0 to 0.1 feet over the various rounds. The concentrations at this depth ranged from below the detection limit (0.1 mg/kg) to 190 mg/kg, with a mean concentration of 6.88 mg/kg and a median concentration of 1.7 mg/kg. In 54 out of 81 locations, the samples collected at this interval contained the highest mercury concentration of the depths sampled. This pattern was particularly true across the organic muck mudflat. However the highest sediment mercury concentrations were detected at greater depths, especially in the vicinity of the HoltraChem and north ditch outfalls.

A total of 78 samples were collected from the 0.8 to 1.0 foot interval. The mercury concentrations at this depth ranged from below the detection limit (0.1 mg/kg) to one sample at 460 mg/kg. The mean mercury concentration at this depth was 18.64 mg/kg, and the median concentration was 0.71 mg/kg. In twenty-four of the sampling locations, the 0.8 to 1.0 foot sample contained the highest mercury concentration. Locations in which this sampling interval contained the highest levels of mercury

included the area near the discharge point of the HoltraChem and northern ditch outfalls (RSD-06A, -07A, -07B, -08A, -11A, -11C, -11 F, -11G, and RSC-019), within the sedge bed (RSD-05B, -05C), and in the southern portion of the cove where granular samples are overlain by organic muck (RSD-15B, -15C, -15D, -15F, -15H, -16B, RSC-010).

A total of 13 samples were collected from a depth of 1.3 to 1.5 feet below the sediment surface. The mercury concentration in these samples ranged from below the detection limit of 0.25 mg/kg to 63 mg/kg. The mean and median concentrations of these samples were 6.76 and 0.33 mg/kg, respectively. Eleven samples were collected at a depth of 1.8 to 2.0 feet. These samples contained mercury concentrations ranging from not detected (<0.2 mg/kg) to 280 mg/kg, with a mean concentration of 31.69 mg/kg and a median concentration of 0.3 mg/kg. The highest mercury concentration in the 1.3 to 1.5 foot sampling interval was found at sampling location RSD-10A (63 mg/kg). The highest concentrations of mercury at the 1.8 to 2.0 foot interval were found at sampling locations RSD-10B and RSD-10C, with concentrations of 280 and 66 mg/kg, respectively. The gravelly nature of the sediments at these locations prevented the collection of deeper samples during subsequent sampling events; therefore, at these three locations, total depth of mercury contamination was not determined.

In November of 1997, a portion of the HoltraChem outfall pipe which had become exposed on the beach face was repaired. During the repair of this outfall pipe, four sediment samples were collected at depth from within and around the excavation. Mercury concentrations in an excavation adjacent to location RSD-10A were 19.62, 0.12, and 11.35 mg/kg, at depths of one, two, and three feet, respectively. The mercury concentration in a sidewall sample collected from the outfall pipe excavation was 2.25 mg/kg, at a depth of 1.0 to 1.5 feet.

Geostatistical techniques were employed as follows to delineate the distribution of mercury in the southern cove sediment. Variogram models were developed for the 0 to 0.1 foot, 0.8 to 1.0 foot, and maximum concentration data sets to describe the spatial correlation of the data. Kriging was then used to develop estimates of sediment mercury concentrations at unsampled locations. The kriged data concentrations were contoured and overlaid onto a southern cove base map. It is important to note that the kriged contours are predictive of the most likely concentrations which would be obtained if another sample was collected within the contoured area. These contours do not strictly honor the existing concentrations at a specific point but rather the distribution of concentrations at that point and surrounding locations.

The mercury contours for the 0 to 0.1 feet, 0.8 to 1.0 feet, and maximum mercury sediment concentrations are shown on **Figure 7-9**, **Figure 7-10** and **Figure 7-11**, respectively. Individual sediment mercury concentrations are also indicated on these maps. The contour maps identify two

areas of locally high mercury concentrations in the northern portion of the southern cove, directly adjacent to the discharge of the north ditch and HoltraChem outfalls. Grab samples of sediments adjacent to the HoltraChem outfall contained mercury at concentrations as high as 460 mg/kg (RSD-11A-02) at depths of 0.8 to 1.0 feet, and 280 mg/kg (RSD-10B-04) at depths to 2 feet below the sediment surface.

The distribution of mercury in the sediment of the southern cove is consistent with what may occur from the discharge of mercury from site streams, ditches and outfalls. Additional sampling was performed in December 1997 and July 1998 to acquire bulk samples for whole sediment biotoxicity analyses. Because these analyses require homogenized two-gallon samples, large volume samples were collected from areas where some of the highest mercury concentrations had been identified in the August and October sampling rounds. Homogenized soil samples were screened at the HoltraChem onsite laboratory to evaluate if the target levels of mercury were obtained. Each time a bulk sample was collected, the concentrations in the bulk samples were much lower than the previously collected grab samples from the same locations. Based on these results, it appears that the distribution of mercury in southern cove sediments is concentrated in discrete layers, because the bulk samples when analyzed did not replicate the higher numbers. In fact, the sample concentrations from bulk samples are significantly lower than what is represented by the grab samples.

Penobscot River Surface Water

Surface water samples were collected from five locations in the Penobscot River in 1995, as shown on Figures 3-10 and 3-12. The results of 1995 surface water sampling are presented on Figure 7-8. In addition to field measured parameters, the samples were analyzed for mercury using the cold vapor atomic fluorescence technique to provide low detection limits. Four of the five samples were analyzed for VOCs. Two samples (RSW-001 and RSW-002) were collected upriver from the plant to represent background conditions, one sample (RSW-003) was collected downgradient of Landfill Area 1, one sample (RSW-004) was collected within the southern cove during a falling tide, and one sample (RSW-005) was collected offshore of the southern cove. With the exception of the sample collected in the cove during the falling tide, all 1995 river surface water samples were collected at low tide.

Mercury concentrations in the two background locations were 5.29 nanograms per liter (1 ng/l = 0.001 $\mu\text{g/l}$) at RSW-001, and 3.29 ng/l at RSW-002. The sample collected off Landfill Area 1 (RSW-003) contained 6.46 ng/l. The highest concentration detected in 1995 was in RSW-004, collected from surface water within the southern cove during the falling tide. This sample contained a mercury concentration of 70 ng/l. The sample collected offshore of the southern cove at low tide (RSW-005) contained mercury at 24 ng/l (28 ng/l in the duplicate).

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Fifty-six additional surface water samples were collected in 1997. The objectives of the 1997 river surface water sampling were to evaluate the distribution and concentration of mercury in the southern cove and the Penobscot River during low, high, ebb, and flood tides in order to identify contaminant sources to the Penobscot River and the relative impact of each of these sources.

Results of the 1997 river surface water sampling are presented on **Figures 7-12, Figure 7-13, Figure 7-14, and Figure 7-15**, representing flood, high, low, and ebb tide conditions, respectively. The 1997 surface water sampling found total mercury concentrations in all tidal conditions greater than the ambient water quality standard of 12 ng/l. Concentrations of total mercury in surface water were generally lowest during the flood and high tides. Total mercury concentrations were generally highest during the ebb tide. The highest concentration of 173 ng/l was detected against the shoreline, upriver of the HoltraChem NPDES outfall during ebb tide. Although the sampling technique attempted to prevent the inclusion of sediment in the sample, this sample was slightly more turbid than others. Resuspension of fine grained sediment is typically observed near the shoreline during the incoming and outgoing tides.

Penobscot River Studies - Dye Testing and Surface Water Sampling

During the summer and early fall of 1998, Acheron performed an investigation of the fate and transport of mercury discharges from the HoltraChem site, as outlined in an October 16, 1998 letter from MEDEP to HoltraChem. This data was required by the MEDEP for evaluation of HoltraChem's discharge permits and is useful for assessing the impact of these discharges to the Penobscot River as part of the SI. The investigation included tracking dye released from the NPDES outfall (001), the north ditch, and the southerly stream, assessing background concentrations of total and dissolved mercury, evaluating the bathymetry and hydraulics of the Penobscot River in the vicinity of the site and an evaluation of the settleable solids associated with the plant's NPDES discharge. The data generated during this investigation will be presented in a report to MEDEP to be submitted in January 1999. Copies of this data are provided in Appendix B. A summary of the results of the Penobscot River Studies which pertain to the SI are summarized below.

Visible concentrations of dye were released individually from outfall 001, the north ditch, and the southerly stream and the dispersion patterns were observed and documented. Dye released from outfall 001 at times other than batch discharges tended to remain close to the river bank at both high and low tides. Dye released during the batch discharges moved to the bottom of the river in a density flow. Dye released from the north ditch at high tide also tended to stay close to shore. During the dye study the flow from the north ditch was so small that it submerged into the gravel beach face before reaching the river at low tide. Dye released into the southerly stream at high tide remained close to the outfall of the southerly stream until the tide started to ebb and then gradually

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spread toward the mouth of the cove. As the tide continued to fall the dye tended to concentrate in the vicinity of the channel formed by the southerly stream across the mudflat. Dye released from the southerly stream during low tide flowed out of the southerly stream across the mudflat and was significantly diluted by the time it reached the main body of the Penobscot River.

Dye concentrations were calculated at the input points and measured at the point where the north ditch and southerly stream entered the southern cove and in the Penobscot River. Samples collected within 30 to 40 feet of the point where the north ditch and/or southerly stream enters the Penobscot River were all below the dye detection limit. The dye concentrations were designed so that the corresponding mercury concentrations would be below the ambient water quality criteria of 12 ng/l before the dye concentration was below the method detection limit.

Dye testing of the batch discharge revealed that the batch has sufficient salt content to create a density flow toward the deepest part of the channel. Under both high and low tide conditions, the batch discharge moved approximately perpendicular to the shore and stayed close to the bottom. The dye was observed close to the bottom for only a short period of time before the tide turned to ebb or flood. After the tide started to move, measurable dye concentrations rapidly dissipated and fell below the dye detection limit. Once again, the dye concentrations injected were sufficient to assure that the concentration of mercury associated with the batch discharge would be below the AWQC before the dye detection limit was reached. Analysis of dye versus mercury concentrations showed there was very good agreement between dye and anticipated mercury concentrations. Therefore, the results of the dye study indicate that the mercury from plant discharges including the north ditch, southerly stream and the NPDES outfall 001 rapidly disperse to less than the water quality standard of 12 ng/l.

In addition to the dye study, mercury samples were collected from upriver and downriver of the HoltraChem site and from turbid and clear water to assess the impact of resuspension of sediment to surface water quality. Clear and turbid samples were collected from two locations off the southern cove, two off of the north ditch and outfall 001 area and from three upriver locations (one below the gully north of Landfill 3, one north of the northerly streams outfall and one south of the Hampden Marina). All of the turbid water samples (upriver and near the site) contained mercury at concentrations greater than 12 ng/l. None of the dissolved samples exceeded 12 ng/l. None of the clear water samples exceeded 12 ng/l. The highest total mercury concentration (412 ng/l) was detected in a turbid sample upriver from the HoltraChem site. The correlation between total suspended solids (TSS) and total mercury was very strong and the near plant turbid water samples did not contain more mercury than the upriver turbid water samples even after normalization based on TSS content. The highest concentrations of dissolved mercury were observed in the samples collected off of the southern cove (6.03 and 6.84 ng/l) and off of the north ditch/001 outfalls (9.97

and 10.1 ng/l) but all were below the AWQC of 12 ng/l. This data indicates that the exceedances of the 12 ng/l AWQC occur only in the samples which contain significant total suspended solids as a result of resuspension of sediment. It also shows that the sediments which resuspend in the vicinity of the southern cove do not impact the surface water quality to a greater extent than the resuspension of sediment upriver. In addition, although the dissolved mercury concentrations were greater in the turbid samples in the vicinity of the HoltraChem site, none of these samples exceeded 12 ng/l and the concentration of dissolved mercury in the greater Penobscot river was not appreciably increased.

7.2.5 Air

Modeling and monitoring of mercury concentrations at and in the vicinity of the HoltraChem facility have been required by MEDEP as part of the Comprehensive Monitoring Plan outlined in the Consent Agreement. HoltraChem also performs monitoring within the cell building for worker health and safety assessment and for calculation of fugitive emissions from the plant. Prior to the establishment of Comprehensive Monitoring Plan air monitoring stations, MEDEP performed a five day air sampling event. The results of the modeling and monitoring are discussed below.

Mercury Dispersion Modeling by Earth Tech

To aid in the siting of likely locations for long-term monitoring of ambient mercury, Earth Tech performed dispersion modeling of elemental mercury emissions from the HoltraChem facility (Earth Tech, August 1997). Earth Tech identified two point sources at the facility: the air purasiv stack and the hydrogen purasiv stack. These point sources were modeled with a combined source strength of approximately 47 pounds of elemental mercury per year; emission estimates were based on stack test results (not reviewed for this report). Earth Tech also identified two fugitive sources at the facility: the ridge vents and the roof vents in the cell room. There are two ridge vents and eight roof vents located on the cell building. These fugitive sources were modeled as volume sources with a combined source strength of approximately 239 pounds of elemental mercury per year; emission estimates were based on calculations by Earth Tech.

The dispersion modeling was performed using the Industrial Source Complex-Short Term (ISCST3) model, version 97363, using the regulatory default option and accounting for aerodynamic building wake effects (downwash) on point sources. The ISCST3 model is a steady-state Gaussian model approved in the U.S. EPA modeling guideline appropriate for this application. The model was exercised with five years (1985-1989) of meteorological surface data collected by the Maine DEP at Bangor Airport, approximately 9 kilometers north of the HoltraChem facility, along with the corresponding five years of upper air data from Caribou, Maine. A Cartesian grid of 20 kilometers x

20 kilometers was used to define receptors, with receptor spacing varying from 50 meters (closest to the source) to 800 meters (farthest from the source).

The results provided were in the form of a series of isopleth (equal-concentration line) maps of the area near the HoltraChem facility for various averaging times (annual; 24 hours; one hour). Copies of these isopleth maps are provided in Appendix B. CDM performed a review of the air modeling data to evaluate its suitability for use in site characterization and risk assessment. Based on CDM's review, the results of the modeling appear to be reasonable. There are two important factors to note. First, the ISCST3 model will conservatively predict the maximum impacts of sources modeled; i.e., model validation studies have shown it can be expected to overestimate the maximum impacts by at least a factor of two. This is an acceptable degree of accuracy for mathematical dispersion models and represents the state of the art. Second, in the specific application modeled for HoltraChem, the off-site impacts from the fugitive sources (ridge and roof vents) will likely be further overestimated because the ISCST3 model does not account for downwash effects on volume sources. Therefore, emissions from these vents are assumed to disperse without the influence of building wake effects which will result in a greater mass of pollutants in the fugitive plume reaching off-site receptors than will likely happen in reality. Again, the approach used by Earth Tech in this modeling exercise represents the state of the modeling art.

Ambient Mercury Sampling by Maine DEP

The Maine DEP conducted a limited air sampling study on HoltraChem property in December 1997 to collect and analyze airborne mercury following an approved NIOSH method. A copy of the results of this sampling is included in Appendix B. Ambient air samples were collected at three sites inside the fence line on five separate days. A collocated sample was also collected along with the ambient samples at one of the outdoor locations on each of the five sampling days. Indoor air samples were collected at four locations inside the cell building on one day. One trip blank was also collected per day. No concomitant on-site meteorological data were collected. The sampling method consisted of drawing a sample of air through a glass tube filled with Hopcalite for four hours using a battery-powered pump at a nominal flow rate of 0.2 liter per minute and is appropriate for collecting particulate- and vapor-phase mercury. Sampling was generally conducted from mid morning until early afternoon on each sampling day. None of the outdoor sampling apparatus was protected from the elements and the ambient temperature was reported to be between 20° F and 30° F. Holding times for the samples varied from one to five days. Samples were analyzed at the University of Maine by cold vapor atomic absorption spectroscopy.

Analytical results for the ambient samples ranged from 0 to 4.39 $\mu\text{g}/\text{m}^3$ and results for indoor samples ranged from 10.90 to 44.90 $\mu\text{g}/\text{m}^3$. According to the NIOSH method documentation, when

sampling for eight hours, this method can provide results in the range of one to 50 $\mu\text{g}/\text{m}^3$, and when sampling for 1.67 hours, this method can provide results in the range of five to 250 $\mu\text{g}/\text{m}^3$. Because sampling for this study was conducted for four hours per sample, the minimum detectable concentration might be expected to be approximately three $\mu\text{g}/\text{m}^3$. All of the indoor samples exceeded this concentration, but only one of the ambient samples exceeded this concentration. This suggests that the results reported for all but one of the ambient samples are not reliable. This implication is supported not only by the inferred minimum detectable concentration but by two other factors. First, four of the five collocated samples provide a poor measure of reproducibility, presumably because of the low ambient concentrations. Second, the actual amount of air sampled by the battery-powered pumps is not well known. Whether due to the extreme cold temperatures encountered or other environmental factors, documented air flow rates varied from +19 to -36 percent between pre- and post-sampling period flow rate checks for all samples. No measure of flow rates during sampling is available. Therefore, the actual volume of air sampled in the ambient samples is only an estimate and could be considerably different from that reported with the results. The flow rate of the indoor samples appears to be much more stable. Based on the foregoing reasons, the sampling method used in this limited study should be considered inappropriate for ambient sampling and the majority of the ambient samples should be considered to be unreliable. Because only a single ambient sample exceeding the minimum detectable concentration was collected, it is not known whether this result is representative of typical ambient concentrations on plant property.

Ambient Air Monitoring

Ambient air monitoring is being performed by Earth Tech Inc on behalf of HoltraChem as part of the Comprehensive Monitoring Plan outlined in the Consent Agreement. Three monitoring stations and one meteorological station were established. The locations of the monitors are near Route 15 on the PERC property southeast of the HoltraChem plant and near Route 1A and Route 202 in Hampden north of the plant. Approximately one month of data is available at the time of this writing. The available data and a figure showing the monitoring locations are provided in Appendix B. Using the available data through October 18, 1997, the average concentrations of the three monitoring stations listed with increasing distance from the HoltraChem plant were 12.0, 3.45 and 3.55 ng/m^3 , respectively. Maximum 24-hour average concentrations listed with increasing distance from HoltraChem were 57.3, 9.0 and 6.43 ng/m^3 , respectively. Maximum 1-hour average concentrations listed with increasing distance from HoltraChem were 174, 29.5 and 14.8 ng/m^3 , respectively. These concentrations are all within the values predicted by the modeling effort and serve to confirm that the modeling data is appropriate for site characterization purposes until such time that additional monitoring data is available.

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7.2.6 Groundwater

The groundwater monitoring network consists of 69 monitoring wells, most of which were installed during three field seasons in 1989/90, 1994, and 1997. Monitoring well nomenclature reflects the installation date as follows: B-300 series wells were installed by Acheron during 1989 and 1990; MW-400 series wells were installed by CDM in 1994; and MW-500 series wells were installed by CDM in 1997. Monitoring wells without an MW- or B- prefix generally indicate installation by PELA prior to 1989. Screen intervals are reflected in the well suffix: wells labeled -B1, -B2 and -B3 reflect deep, intermediate, and shallow bedrock wells, respectively. Wells labeled -O1, -O2, and -O3 reflect deep, intermediate, and shallow overburden wells, respectively. The monitoring well network provides coverage along the bedrock ridge and downgradient from the plant to the south and west. The tightest network of monitoring wells is below Landfill Area 1, where the highest concentrations of mercury in groundwater have been found.

Groundwater samples have been collected in six major monitoring events. The earliest groundwater sampling rounds were conducted in 1980 and 1983, when PELA sampled the limited groundwater network available at that time. Acheron sampled the existing monitoring well network in 1989/1990, following the installation of the B-300 series wells. This sampling event provides the first baseline data for most areas of the site; however, all of the mercury samples collected by Acheron were filtered in the field, and analytical results are reported as dissolved concentrations. CDM sampled the B-300 and MW-400 series wells in 1995, as part of the initial Site Investigation. At the request of the EPA, all of the metals analyses in this sampling event were total metals. In 1997, CDM sampled the 17 newly installed (MW-500 series) monitoring wells, and 52 existing wells on site. To facilitate temporal comparisons between this event and previous sampling rounds, both dissolved and total metals samples were collected. Finally, in 1998, CDM conducted the first sampling round for the Comprehensive Monitoring Plan and sampled additional wells to assess water quality trends associated with the landfills. This monitoring event consisted of a total of 24 monitoring wells throughout the site. All groundwater samples were analyzed for mercury and conventional parameters; 65 of the wells during the 1997 round and 13 wells during the 1998 comprehensive monitoring round were also analyzed for VOCs. Sampling results from all of the CDM groundwater monitoring rounds can be found in Tables 3-8 through 3-15.

Groundwater sampling is discussed in Sections 3.4.3 and 3.4.4. Groundwater analytical results are presented in Tables 3-8 through 3-13. The distribution of mercury and chloropicrin in overburden and bedrock groundwater are provided on **Figures 7-16** through **7-19**. Each of these contaminants of concern and others are discussed below.

To facilitate the interpretation of the data, the groundwater data is discussed below by analyte. Within each analyte section, data are discussed by plant area and vertical distribution. Temporal comparisons are then drawn between data from each of the applicable sampling rounds. Finally, probable source areas and discharge points are discussed.

Mercury

Mercury contamination in site groundwater is present in four general areas: (1) beneath and downgradient of the lined process lagoon and Landfill Area 1; (2) along the northern side of the plant adjacent to the bedrock ridge and below Landfills 3 and 4; (3) at the western side of the plant and along the north ditch; and (4) along the southern side of the plant and the southerly stream. Of these areas, the highest mercury concentrations are found in wells downgradient of the lined process lagoon and Landfill Area 1. Groundwater on the north side of the bedrock ridge and in the vicinity of Landfill 2 does not contain mercury at concentrations greater than the EPA Maximum Contaminant Level (MCL) of 2 $\mu\text{g/l}$. Contour maps showing the concentrations of mercury in overburden and bedrock groundwater during 1997 are depicted in Figures 7-16 and 7-17, respectively.

Landfill Area 1

During the 1997 groundwater monitoring round, the highest concentration of mercury in overburden deposits was detected in MW-501-O1. This well was installed during the 1997 field season, and is screened across the top of till beneath Landfill Area 1 and downgradient of the lined process lagoon. The total mercury concentration in this well was 4600 $\mu\text{g/l}$ during the 1997 groundwater monitoring event. The dissolved mercury concentration was 3600 $\mu\text{g/l}$, indicating that the form of mercury present is primarily in a soluble form such as mercuric chloride. Mercury concentrations in this well decreased slightly in 1998, to 3100 $\mu\text{g/l}$ (total) and 3000 $\mu\text{g/l}$ (dissolved).

The remaining wells in the area of Landfill 1 contained similarly high mercury concentrations during the 1997 monitoring event. Monitoring well B-326-O2 had a total mercury concentration of 1600 $\mu\text{g/l}$ during this sampling event. The dissolved mercury concentration was 770 $\mu\text{g/l}$. In 1998, mercury concentrations in this well were 1500 $\mu\text{g/l}$ (total) and 1100 $\mu\text{g/l}$ (dissolved). This well is screened at the base of the sand and gravel layer, and is located just south of Landfill Area 1.

The three overburden wells and one bedrock well of the B-326 cluster provide some information on the vertical distribution of mercury in this area. In 1997, mercury concentrations in the till (B-326-O1) and shallow bedrock (B-326-B1) were 7.9 and 8.0 $\mu\text{g/l}$, respectively, and the mercury concentration in the shallow overburden well was 54 $\mu\text{g/l}$. As evidenced by the much higher

concentrations in B-326-O2, mercury in overburden below Landfill Area 1 is concentrated at the base of the sand and gravel deposits above the lower permeability till. Although concentrations of mercury in the till and bedrock are low, the above wells did not have detectable concentrations of mercury during the 1995 groundwater sampling event. Some mercury appears to have seeped into the till and bedrock strata between the 1995 and 1997 monitoring events. In 1998, only B-326-O2 and B-326-O3 were sampled in this cluster. As in 1997, mercury concentrations in 1998 were highest in the deep overburden well, directly above low permeability till deposits.

Monitoring wells MW-401-O1, MW-512-O1, MW-513-O1, and MW-402-O1 all had total mercury concentrations greater than 500 $\mu\text{g/l}$ during the 1997 sampling round. Of these, MW-401-O1 (1100 $\mu\text{g/l}$) had the highest mercury concentration, and MW-512-O1 (800 $\mu\text{g/l}$) had the lowest. Mercury concentrations in all of these overburden wells decreased in 1998. Concentrations during the 1998 monitoring event ranged from 120 $\mu\text{g/l}$ (MW-401-O1) to 700 $\mu\text{g/l}$ (MW-512-O1).

Prior to the 1995 site investigation, only one monitoring well cluster (B-326) was present below Landfill Area 1. This cluster therefore provides the longest record for temporal comparison. During Acheron's 1989 sampling, B-326-O2, screened in the sand and gravel, had dissolved mercury concentrations of 3300, 1300, and 3000 $\mu\text{g/l}$ in three sampling events. The total mercury concentrations detected during the initial SI sampling (CDM, 1995) were 2630 $\mu\text{g/l}$ in the parent and 1660 $\mu\text{g/l}$ in the duplicate sample. Concentrations of 1600 $\mu\text{g/l}$ (total) and 1500 $\mu\text{g/l}$ (dissolved) were detected in the confirmation sampling round in both the parent and duplicate samples. In 1997 and 1998, total mercury concentrations in B-326-O2 were 1600 $\mu\text{g/l}$ and 1500 $\mu\text{g/l}$, respectively, and dissolved concentrations were 770 $\mu\text{g/l}$ and 1100 $\mu\text{g/l}$. Mercury concentrations in this well cluster appear to have stabilized since the 1995 monitoring event.

Monitoring well MW-401-O1, located north of the B-326 cluster adjacent to the bedrock scarp, had total mercury concentrations of 33 $\mu\text{g/l}$, 1100 $\mu\text{g/l}$, and 120 $\mu\text{g/l}$ in 1995, 1997, and 1998. Dissolved mercury concentrations in MW-401-O1 remained relatively stable between 1997 and 1998, with results of 11 and 13 $\mu\text{g/l}$, respectively. MW-402-O1, located southeast of the B-326 cluster adjacent to the lower plant roadway, had total mercury concentrations of 580 $\mu\text{g/l}$, 960 $\mu\text{g/l}$, and 350 $\mu\text{g/l}$, in these three monitoring events. Dissolved concentrations in MW-402-O1 decreased between 1997 and 1998, from 900 $\mu\text{g/l}$ to 270 $\mu\text{g/l}$. These results indicate that the lateral distribution of mercury below Landfill Area 1 may have reached its maximum extent in 1997. Continued monitoring of dissolved and total mercury concentrations in these wells will provide additional information on source areas and attenuation over time.

Based on the mercury concentration with depth profile from boring B-501, the depth of soil fill with high concentrations of mercury is limited to approximately 6 feet at that location. This corresponds

to an elevation of 52 PD. The groundwater level in MW-501 is typically just above the till at Elevation 30 PD or approximately 22 feet below the base of the mercury contaminated fill at that location. Based on the reported approximate location of the former mercury treatment sludge cells 1a and 1b, beneath the cap and adjacent to the bedrock scarp, these cells are thought to be above the 40' elevation contour. Based on the water level elevations observed in MW-501 located upgradient and cross gradient to cells 1a and 1b, and in MW-401-O1, located downgradient of the cells, there is greater than 12 feet of separation between the cells and the water table. Therefore it is unlikely that these cells represent a significant continuing source of mercury contamination to groundwater.

Although mercury concentrations in wells below Landfill Area 1 have fluctuated over time, temporal comparisons of the data are indicative of a continuing source of mercury upgradient from these wells. A number of factors suggest that the lined process lagoon may be a continuing source. Groundwater contours for both overburden and bedrock indicate that a groundwater flow path exists from the lined process lagoon to the contaminated wells. MW-501-O1, which is the well closest to the lined process lagoon, had the highest concentrations of mercury during the 1997 sampling round. Mercury in all of the wells downgradient from the process lagoon is primarily in a soluble form, which is consistent with a source containing process water. Observations of the process lagoon liner during October of 1998 also indicate that the lagoon may be leaking: salt piles on the top lagoon liner, moisture and fluid between the liners, and a hole along the eastern wall of the liner all indicate that leaks may be present. Mercury concentrations of the brine solution circulating through the lined process lagoon are typically on the order of 15,000 $\mu\text{g/l}$; a small leak in the lagoon liner could account for the observed mercury concentrations directly downgradient from this surface impoundment.

Northern Edge of Plant/Bedrock Ridge

Monitoring well MW-506-B1 was installed in 1997, directly beneath Landfill 4. Mercury was not detected the first time that this well was sampled in 1997. Mercury concentrations of 19 $\mu\text{g/l}$ (total) and 12 $\mu\text{g/l}$ (dissolved), were detected during the 1998 sampling as part of the comprehensive monitoring plan. Monitoring well MW-410-B1 is screened in bedrock immediately below Landfills 3 and 4. The total mercury concentrations observed in this well in 1995 ranged from 16 to 50 $\mu\text{g/l}$, which were the highest concentrations observed in this area during the initial SI. The concentration of total mercury in this well was 9 $\mu\text{g/l}$ in 1997 and 62 $\mu\text{g/l}$ in 1998. Based on the sampling results from MW-410-B1 and MW-506-B1, and due to the low volume of water which is contained in fractured bedrock, the concentrations detected do not indicate a significant mass of mercury present in the groundwater system in this area. These two wells are the only locations on the bedrock ridge where the mercury concentrations are above the MCL of 2 $\mu\text{g/l}$, and groundwater from this location is expected to migrate towards the plant and discharge to overburden deposits along the bedrock

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scarp. Groundwater between the landfills on the ridge and the Penobscot River does not contain mercury at concentrations greater than the MCL.

Monitoring well P-2A was sampled in 1983 by PELA, and indicated a mercury concentration of 80 $\mu\text{g/l}$. This well is screened across the bedrock/overburden interface, and is located downgradient from MW-410-B1. The concentrations of mercury in this well during 1997 and 1998 were 14.8 $\mu\text{g/l}$ in 1995, 2.3 $\mu\text{g/l}$ in 1997 and 15 $\mu\text{g/l}$ in 1998.

The mercury concentrations in P-2A have decreased since the initial sampling round, but have fluctuated since that time. Fluctuations in concentrations may be due a limited continuing source such as percolation through the landfill cap, and/or seasonal variations in precipitation, groundwater levels or sampling methodologies.

Monitoring wells further downgradient from P-2A along the bedrock ridge have shown slightly increased concentrations of mercury since the 1995 investigation. MW-405-B1, which had a total mercury concentration below detection limits in 1995, had a total mercury concentration of 12 $\mu\text{g/l}$ in 1997 and was not sampled in 1998. Mercury concentrations in MW-405-O1 were less than 0.2 $\mu\text{g/l}$ in 1997 and 13 $\mu\text{g/l}$ in 1998, compared to a concentration of 9.6 $\mu\text{g/l}$ in this well during 1995. MW-315-O1 had mercury concentrations of 30 $\mu\text{g/l}$ and 100 $\mu\text{g/l}$ during 1997 and 1998, up from 5.2 $\mu\text{g/l}$ during 1995. The mercury concentration in the shallow bedrock well at this location remained relatively constant between 1995 and 1997, at approximately 4 $\mu\text{g/l}$.

All of these wells are located downgradient from Landfills 3 and 4. Potential sources for mercury detected in wells along the bedrock scarp include bedrock groundwater discharging to overburden deposits, and plant area sources. Groundwater along the bedrock scarp is migrating westward toward Landfill Area 1 and the Penobscot River. However, some of this water may be captured by the underdrains located beneath the former sodium chlorate plant. All of these wells will be observed through continued monitoring.

Western Edge of the Plant

During the 1997 groundwater monitoring round, three monitoring wells in the plant area near the salt storage pad had mercury concentrations higher than the MCL of 2 $\mu\text{g/l}$. The mercury concentrations in these wells were 4.2 $\mu\text{g/l}$ in MW-508-O1, 230 $\mu\text{g/l}$ in MW-510-O1, and 740 $\mu\text{g/l}$ in MW-502-O1. Monitoring wells MW-502-O1 and MW-510-O1 are screened across the top of the till, and would therefore be expected to detect contaminants traveling across the top of this layer. Monitoring well MW-508-O1 is screened in the till, and was installed to evaluate whether mercury from the plant is entering this layer. The total mercury concentrations of these three wells were lower

in 1998: 1.0 $\mu\text{g/l}$ in MW-508-O1, 120 $\mu\text{g/l}$ in MW-510-O1, and 120 $\mu\text{g/l}$ in MW-502-O1. Although the lower concentrations in the 1998 sampling round suggest that the source of the groundwater contamination in this area is not stable or increasing, two rounds of data are insufficient to establish a trend. The lower concentration of total mercury detected in the deep till well MW-508-O1 in 1998 is consistent with the hypothesis that the till is limiting downward migration of contaminated groundwater. Monitoring wells MW-502-O1 and MW-510-O1 will continue to be monitored as part of the Comprehensive Monitoring Plan.

Monitoring well B-316-O1, located adjacent to the north ditch, had mercury concentrations of 270 and 560 $\mu\text{g/l}$ in the parent and duplicate samples, respectively, during 1997. Dissolved concentrations during 1997 were 270 $\mu\text{g/l}$ and 240 $\mu\text{g/l}$ in parent and duplicate samples. In 1998, mercury was found in this well at concentrations of 2500 $\mu\text{g/l}$ (total) and 90 $\mu\text{g/l}$ (dissolved). Contamination in this well is attributed to surface spills and runoff from the plant area. The outfall of the paved sump to the north ditch is located upgradient of the B-316 cluster. Based on overburden groundwater contours, the north ditch is a losing stream in much of the area between the paved sump and the southern cove. This means that surface water is discharging to and recharging groundwater. Mercury concentrations of surface water in the paved sump have been as high as 100 $\mu\text{g/l}$ in measurements made by HoltraChem. Based on these observations, and the lack of another nearby source area for mercury contamination, it is likely that contaminated surface water from the north ditch is recharging groundwater in this area, leading to the observed groundwater contamination in B-316-O1.

Monitoring wells B-320-O1 and B-320-O3, located near the outlet of the north ditch, had mercury concentrations of 18 and 4.3 $\mu\text{g/l}$, respectively, in 1997. All three of the wells in the B-320 cluster had lower levels of mercury in 1997 than in 1995. This monitoring well cluster was not resampled in 1998. Monitoring well B-320-O1 had the highest mercury concentrations in this cluster. Mercury concentrations in this well were 28 $\mu\text{g/l}$ during 1995 and 18 $\mu\text{g/l}$ during 1997. The data from these wells indicate a discontinuous or attenuating mercury source, which has allowed the mercury concentrations in these wells to decrease over time. Concentrations of mercury in these wells, if attributable to infiltration from the north ditch, are expected to fluctuate over time, following changes in concentrations within the north ditch.

Southern Edge of the Plant

Monitoring well B-327-O1 had a mercury concentration of 840 $\mu\text{g/l}$ during the comprehensive 1997 groundwater investigation. Prior to this sampling event, this monitoring well had mercury concentrations at or near the detection limit of 0.2 $\mu\text{g/l}$. In 1998, the concentration of mercury in this well decreased to 66 $\mu\text{g/l}$. This well lies directly below the location of a documented brine leak,

which was discovered in March of 1997. Fill surrounding utility lines may represent preferential pathways for groundwater flow, leading from the brine tank area to this monitoring well (see Acheron's 1997 report, *Summary Report - Brine Tank Investigation*). Temporal comparisons of mercury concentrations in this well support a discrete time source for contamination. Mercury in this well is therefore attributed to the surficial spill which occurred between the 1995 and 1997 monitoring events.

Monitoring well B-321-B2 had total mercury concentrations of 2.7 $\mu\text{g/l}$ during the 1997 monitoring round, and 1.9, 0.6, and 5.2 $\mu\text{g/l}$ during three rounds in 1998. The dissolved mercury concentration in this well was below the detection limit of 0.2 $\mu\text{g/l}$ in all four sampling events. The groundwater samples from this well are usually turbid which is unusual for a bedrock monitoring well. When a low flow sample was collected from this well in 1995, the total mercury concentration was below the method detection limit of 0.2 $\mu\text{g/l}$. These data indicate that the mercury detected in B-321-B2 is associated with particulates, and is not in a soluble form.

The southerly stream in the vicinity of this well has a surface elevation slightly higher than the groundwater level in B-321-B2, and appears to be a losing stream. During Acheron's dye testing study, groundwater was sampled from B-321-B2 while dye was being injected into the southerly stream. Dye was detected in the groundwater sample from this well, indicating that it is receiving recharge from surface water in this area. Other monitoring wells in the area further indicate that mercury in B-321-B2 is not migrating from the plant to this cluster through groundwater: MW-504-O1 and MW-504-B1, located between the plant and the B-321 cluster, each had mercury concentrations below detection limits during the 1997 monitoring round, and the groundwater contour maps for both overburden and bedrock show the B-321 cluster to be cross-gradient from the plant.

Elevated concentrations of mercury in B-321-B2 are attributed to infiltration from surface water. Mercury in this well is always associated with particulates, consistent with a source from suspended sediments in streamwater. Temporal comparisons of groundwater concentrations in B-321-B2 are not likely to be useful without corresponding data on surface water concentrations.

Mercury was detected in the shallow bedrock well MW-511-B2 the first time this well was sampled. The concentration was 0.28 $\mu\text{g/l}$, just slightly above the method detection limit and well below the MCL of 2 $\mu\text{g/l}$. The dissolved concentration was less than the method detection limit. This well has been sampled three times since the initial round and mercury was not detected in any of the subsequent sampling rounds. Mercury has also not been detected in any of the samples collected from the deeper monitoring well MW-511-B1, the MW-505 cluster or the residential wells on Ferry Road.

Chloropicrin in Groundwater

Chloropicrin is manufactured and handled in the western portion of the plant. Chloropicrin is also known to have been spilled at the western terminus of the railroad car loading area (see Figure 2-2). The occurrences of chloropicrin in groundwater are limited to the area from the chloropicrin plant and railroad car loading area to the Penobscot River, with the exception of low level detections in wells P-13 during the 1995 and 1998 sampling rounds and MW-406-B1 during the 1995 sampling round. Contours of chloropicrin concentrations in 1997 are depicted in Figures 7-18 and 7-19.

The highest concentrations of chloropicrin were found in monitoring well B-316-O1, directly downgradient from the railroad car loading area and adjacent to the north ditch. In 1997, concentrations of chloropicrin in this well were 46,000 and 47,000 $\mu\text{g/l}$ in parent and duplicate samples, respectively. In 1998, the chloropicrin concentration in B-316-O1 was 24,000 $\mu\text{g/l}$. Chloropicrin was also detected in high concentrations in MW-402-O1, MW-513-O1, B-316-B1, and the MW-503 cluster, all within the area downgradient from the chloropicrin tank car fill area. Concentrations of chloropicrin in these wells during 1997 and 1998 ranged from 8,800 $\mu\text{g/l}$ (MW-503-O1) to 28,000 $\mu\text{g/l}$ (MW-513-O1).

Chloropicrin was also detected in wells in the western portion of the plant area, near the chloropicrin manufacturing building. The concentration of chloropicrin in MW-404-O1, adjacent to the chloropicrin manufacturing area, was 45,000 $\mu\text{g/l}$ in 1995 and 2,000 $\mu\text{g/l}$ in 1997. Groundwater from MW-510-O1, located downgradient of the salt storage pile, had a chloropicrin concentration of 120 $\mu\text{g/l}$ during the 1997 investigation. This well was not analyzed for VOCs in 1998.

Monitoring well MW-501-O1, located in the center of Landfill Area 1, had chloropicrin concentrations of 7600 $\mu\text{g/l}$ during the 1997 groundwater monitoring round and 2400 $\mu\text{g/l}$ in 1998. This well is downgradient from the chloropicrin manufacturing area and is screened across the interface between till and more permeable sand deposits. Monitoring well clusters B-326 and MW-401 lie at the northern edge of the monitoring well network below Landfill Area 1, and provide some information on the vertical distribution of chloropicrin contamination. B-326-O2 is screened in sand and gravel above the till, and had chloropicrin concentrations of 470 $\mu\text{g/l}$ in 1995, 92 $\mu\text{g/l}$ in 1997, and 70 $\mu\text{g/l}$ in 1998. The shallower overburden well and the bedrock well in this cluster, by contrast, had chloropicrin concentrations of 0.54 $\mu\text{g/l}$ and 1.5 $\mu\text{g/l}$, respectively, in 1995 and were both below the method detection limit of 10 $\mu\text{g/l}$ during the 1997 sampling event. Consequently they were not sampled in 1998. Monitoring well MW-401-B1 had an estimated chloropicrin concentration of 5600 $\mu\text{g/l}$ in 1995, 120 $\mu\text{g/l}$ in 1997, and this well was not sampled in 1998. The samples from the overburden well in this cluster were 3100 $\mu\text{g/l}$ in 1995, and below the detection limit of 10 $\mu\text{g/l}$ in 1997 and 1998. As with mercury, it appears that chloropicrin contamination is limited to higher

permeability deposits within subsurface media. In the MW-401 cluster, where the till layer is not present, chloropicrin is present in bedrock. In the B-326 cluster, where a till layer is present, chloropicrin contamination is limited to sand and gravel layers within overburden deposits.

In 1995, chloropicrin was detected in B-316-O1 at an estimated concentration of 50,000 $\mu\text{g/l}$. In 1997, chloropicrin in B-316-O1 had decreased slightly, to 46,000 $\mu\text{g/l}$ in the parent and 47,000 in the duplicate sample. In 1998, chloropicrin was detected in this well at a concentration of 24,000 $\mu\text{g/l}$. Chloropicrin at MW-402-O1 was detected at concentrations of 23,000 $\mu\text{g/l}$ and 18,000 $\mu\text{g/l}$ in the parent and duplicate samples, respectively, in 1995. Groundwater monitoring during 1997 and 1998 detected chloropicrin concentrations in this well of 21,000 $\mu\text{g/l}$ and 16,000 $\mu\text{g/l}$, respectively. Temporal comparisons of the available data indicate that chloropicrin concentrations have decreased in overburden deposits downgradient of the tank car loading area. In the bedrock well below the tank car spill area (B-316-B1), the concentration of chloropicrin increased from 500 $\mu\text{g/l}$ to 9800 $\mu\text{g/l}$ between 1995 and 1997, suggesting that contaminants have migrated from overburden into bedrock over this time period.

The most downgradient wells exhibiting chloropicrin are the shallow bedrock and deep overburden wells in the B-320 cluster, located near the edge of the southern cove below the plant. Monitoring well B-320-B1 had chloropicrin concentrations of 990 $\mu\text{g/l}$ in 1995 and 1000 $\mu\text{g/l}$ in the parent sample and 1100 $\mu\text{g/l}$ in the duplicate in 1997. Monitoring well B-320-O1 had an estimated chloropicrin concentration of 9500 $\mu\text{g/l}$ in 1995, and a concentration of 1900 $\mu\text{g/l}$ in 1997. The B-320 cluster was not sampled in 1998. Again, since chloropicrin is limited to deep overburden and shallow bedrock wells, it appears that chloropicrin has migrated along surfaces where permeability contrasts are present. There is no till present at the B-320 cluster but there are significant till thicknesses closer to the plant at the B-316 cluster.

Chloropicrin concentrations in other areas of the plant also decreased between 1995 and 1997. In the well closest to the chloropicrin manufacturing area (MW-404-O1), chloropicrin concentrations decreased from approximately 45,000 $\mu\text{g/l}$ in 1995 to 2,000 $\mu\text{g/l}$ in 1997. This significant decrease strongly suggests a discrete time source release, which has migrated or attenuated over time. Two of the wells below Landfill Area 1 also showed decreasing chloropicrin concentrations between 1995 and 1997. Chloropicrin in MW-401-B1 decreased from 5,600 $\mu\text{g/l}$ in 1995 to 120 $\mu\text{g/l}$ in 1997. Chloropicrin concentrations in the deep overburden well at this cluster (MW-401-O1) decreased from 3100 $\mu\text{g/l}$ in 1995 to less than 10 $\mu\text{g/l}$ in 1997. The concentration of chloropicrin in B-326-O2 also decreased, from 470 $\mu\text{g/l}$ in 1995 to 92 $\mu\text{g/l}$ in 1997, and 70 $\mu\text{g/l}$ in 1998. All of these results point to a discrete time source release, such as a spill, that has attenuated over time.

Chloropicrin contamination below the chloropicrin tank car loading area (B-316 cluster) is attributed to the spills which occurred in the railroad car loading area prior to the 1995 site investigation. Contamination in B-320 is attributed to flow of product along the north ditch and in preferential pathways within overburden deposits following this spill. Chloropicrin concentrations in these wells have remained relatively stable or decreased slightly over time, suggesting that contamination from this discrete time source has begun to attenuate.

Carbon Tetrachloride and Chloroform in Groundwater

Carbon tetrachloride was reportedly deposited in Landfills 2, 4 and 5. The highest concentration of carbon tetrachloride in groundwater was detected in MW-410-B1 430 $\mu\text{g/l}$ in 1995. Since that time the concentrations detected have decreased (140 $\mu\text{g/l}$ in 1997 and 110 $\mu\text{g/l}$ in 1998). The second and third highest concentrations were detected in P-2A (86 $\mu\text{g/l}$ in 1995 and 49 $\mu\text{g/l}$ in 1995) and MW-506-B1 (54 $\mu\text{g/l}$ in 1997 and 50 $\mu\text{g/l}$ in 1998), respectively. Lower concentrations were detected in monitoring well B-309-B1, located north of Landfill 4 (18 $\mu\text{g/l}$ parent, 16 $\mu\text{g/l}$ duplicate in 1995 and 21 $\mu\text{g/l}$ in 1997). Carbon tetrachloride was detected in at least one well per cluster in the rest of the wells located along the bedrock scarp adjacent to the plant (MW-409-B1; MW-405-B1; B-315-O1 and B1; MW-403-B1, -O1 and -O2; and MW-401-O1). In 1997, the concentrations detected in these wells were typically below, at or near the MCL of 5 $\mu\text{g/l}$. Carbon tetrachloride in MW-405-O1 decreased in concentration from 44 $\mu\text{g/l}$ in 1995 to less than 1 $\mu\text{g/l}$ in 1997. The concentration of carbon tetrachloride in MW-405-B1, however, increased from below detection limits to 9 $\mu\text{g/l}$ in the same period. Concentrations of carbon tetrachloride elsewhere on the site are below the MCL of 5 $\mu\text{g/l}$.

Temporal comparisons of carbon tetrachloride concentrations in the wells with the highest historical detections indicate a general decrease in concentrations between 1995 and 1997. The most significant decreases in carbon tetrachloride levels occurred in monitoring wells MW-410-B1, P-02A, and MW-405-O1. The decreasing concentrations suggest that a continuing source is not present. In general, concentrations of carbon tetrachloride in other wells along the bedrock scarp have remained relatively constant since 1995. The single exception to this rule is in MW-403-B1, located above the lined process lagoon. Carbon tetrachloride in this well decreased between 1995 and 1997, from 34 $\mu\text{g/l}$ to 5.7 $\mu\text{g/l}$. Decreasing concentrations between 1995 and 1997 suggest that the source is not a continuing one.

It is possible, based on the reported disposal of carbon tetrachloride, that Landfill 4 is the source of carbon tetrachloride along the bedrock scarp. It appears that groundwater is discharging to the plant area from the bedrock ridge in the vicinity of P-2A, and possibly elsewhere along the scarp. The hydraulic conductivity of the overburden deposits in this area is measured to be one or more orders

of magnitude greater than that of the till deposits under the plant, and till is thin or absent above bedrock in several borings adjacent to the scarp. Groundwater also appears to be discharging to the underdrains beneath the former sodium chlorate plant. Carbon tetrachloride was detected in the water in these underdrains during the 1994 and 1997 sampling programs (SSW-010, carbon tetrachloride 3.5 $\mu\text{g/l}$, and 5.1 $\mu\text{g/l}$, respectively). Chloroform, a degradation product of carbon tetrachloride, was also detected in these underdrains at a concentration of 44 $\mu\text{g/l}$ in 1994 and at a concentration of 8.8 $\mu\text{g/l}$ in 1997.

Carbon tetrachloride and chloroform were detected in both the overburden and bedrock wells in the vicinity of Landfill 4, along the bedrock scarp and in the downgradient river area. The concentrations of chloroform in groundwater are typically higher than for carbon tetrachloride. The highest concentration of chloroform (570 $\mu\text{g/l}$) was detected in MW-410-B1, downgradient of Landfill 4, the first time this well was sampled. Subsequent samples from this well have contained between 120 $\mu\text{g/l}$ (1997) and 370 $\mu\text{g/l}$ (1998). Chloroform has been detected at concentrations of 290 $\mu\text{g/l}$ in MW-501-O1, downgradient of the lined process lagoon, 220 $\mu\text{g/l}$ in B-316-O1 downgradient of the paved sump and 47 $\mu\text{g/l}$ in B-326-02 at the base of Landfill Area 1.

Because chloroform is a daughter product of anaerobic biodegradation of carbon tetrachloride and was not reportedly used at the site, the presence of greater concentrations of chloroform than carbon tetrachloride suggest that biodegradation is occurring at the site and that a fresh supply of carbon tetrachloride is not being discharged to the groundwater at a rate faster than the biological reactions are occurring. The chloroform concentrations between the former sodium chlorate plant and the Penobscot River are lower than those detected upgradient and at the former sodium chlorate plant. This distribution illustrates the possibility that much of the groundwater containing carbon tetrachloride and chloroform is being captured by the underdrains beneath the former sodium chlorate plant. Samples from the underdrain also contained carbon tetrachloride and chloroform.

Other Volatile Organic Compounds

In addition to carbon tetrachloride, chloroform, and chloropicrin, several other chlorinated and brominated VOCs were detected, generally in low concentrations. The distribution of VOCs other than the three listed above is limited to the wells on the western side of the plant area and in the area surrounding Landfill Area 1.

The most commonly detected chlorinated VOC was trichloroethylene (TCE). Detectable concentrations ranged from 1.2 $\mu\text{g/l}$ in MW-506-B1 to 10 $\mu\text{g/l}$ in MW-403-O2 and B-315-O2. The occurrence of TCE is limited to the area adjacent to the bedrock scarp, and at the base of Landfill Area 1. The highest concentrations of TCE were found in wells along the bedrock scarp,

downgradient from the former sodium chlorate plant. TCE was also detected at low levels in wells along the bedrock scarp during the 1995 investigation. The concentration of TCE in these wells remained constant, or decreased slightly, between the two groundwater sampling events.

Acetone was detected in six wells: B-316-B1, B-320-B1 (duplicate sample only), B-326-02, MW-402-01, MW-404-01, and MW-508-01. Acetone concentrations ranged from 11 $\mu\text{g/l}$ in B-326-02 (1995), to 7300 $\mu\text{g/l}$ in B-316-B1 (1997). Acetone was not detected in any of the 1998 groundwater samples. The two wells with the highest detected concentrations (B-316-B1, 7300 $\mu\text{g/l}$ and B-320-B1, 3200 $\mu\text{g/l}$) are located along the north ditch, and may have received recharge from contaminated surface water in this area. The other area where acetone has been detected is below the tank car loading area (MW-509-B1, 220 $\mu\text{g/l}$ and B-327, 200 $\mu\text{g/l}$).

The detection limits for acetone are often elevated by its very high solubility and/or the presence of other VOCs in the sample. Accordingly, the detections of acetone have not been consistent around the site, from sampling round to sampling round and even between parent and duplicate samples in the same well. Therefore, temporal comparisons are considered weak. Acetone in monitoring well B-320-B1 increased from non-detected concentrations in 1995 to 510 $\mu\text{g/l}$ in 1997. Acetone concentrations in monitoring well B-316-B1 remained relatively stable, from 6700 $\mu\text{g/l}$ in 1995 to 7300 $\mu\text{g/l}$ in 1997. Because acetone contamination is limited to areas within and below the plant, this contamination is also attributed to plant activities such as fiberglass repair.

Carbon disulfide was detected in seven wells in 1995 at concentrations ranging from 2 $\mu\text{g/l}$ in B-326-O2 to 2200 $\mu\text{g/l}$ in B-316-B1. Carbon disulfide was not detected in the 1997 sampling round and was detected at a concentration of 20 $\mu\text{g/l}$ in B-326-O2 in 1998. There is no reported use of carbon disulfide on the site.

A number of brominated hydrocarbons were detected in the monitoring wells in and downgradient from Landfill Area 1. These VOCs include bromodichloromethane, bromoform, bromomethane and dibromochloromethane. In 1997, these compounds were detected at concentrations ranging from 19 $\mu\text{g/l}$ (bromodichloromethane) to 51 $\mu\text{g/l}$ (bromoform) in MW-512-O1. Several of these compounds were detected at lower concentrations in monitoring wells B-326-O2, MW-501-01, and MW-512-01 in 1998. The highest concentrations of bromodichloromethane (54 $\mu\text{g/l}$), bromoform (120 $\mu\text{g/l}$) and dibromochloromethane (93 $\mu\text{g/l}$) were detected in the 1998 sample from MW-501-01, located downgradient from the lined process lagoon. The detection limits of the previous sample from MW-510-O1 were elevated above these levels due to the presence of chloropicrin in the sample. These compounds have not been used on site.

7.3 Fate and Transport

7.3.1 Introduction

The topography of the site dictates that the mobile fractions of the mercury and VOC contamination detected in site media during this SI will migrate towards and ultimately be discharged to the Penobscot River. Transport mechanisms which may affect site media include surface water discharges, sediment transport, leaching from contaminated soil to groundwater, and groundwater flow. Factors controlling the amount of contamination migrating offsite include natural attenuation of contaminated media, adsorption equilibria between groundwater and soils, and relative amounts of infiltration and surface water runoff from site precipitation. The following is a summary of the various transport routes and the estimated flux or potential flux through those routes.

7.3.2 Surface Water and Sediment Transport

Detectable concentrations of mercury were observed in the surface water of the north ditch and southerly stream, each of which discharge to the Penobscot River at the southern cove. The concentrations of mercury in each surface water body were measured at weirs installed in the spring of 1998. The weir for the southerly stream is located at the concrete spill control dam south of the caustic loading shed, and the weir for the northern stormwater ditch is located at the outfall of the paved sump. Surface water samples collected during 1995 indicated a decrease in mercury concentration with increased distance from the plant. Decreasing mercury concentrations in the southerly stream are likely attributable to dilution from additional surface water inputs; decreasing concentrations in the north ditch may be attributable to dilution from groundwater recharge or adsorption to fine grained particles. Despite the decreasing downstream mercury concentrations, data from the two weirs can provide an order of magnitude estimate of surface water mercury fluxes to the river.

Data from the two weirs was collected on an approximately weekly basis, beginning in April of 1998. Prior to this time, samples were collected for mercury analysis without accompanying data on volumetric flow rate. The mass flux of mercury to the Penobscot River was calculated by multiplying the total mercury concentration at each weir by the volumetric flow rate for that sampling event. A total of 18 sampling events were recorded, and an average flux of mercury, in pounds per day, was calculated. Calculations are provided in Appendix E. The calculated mass flux of mercury to the Penobscot River from the southerly stream is estimated to be 1.12E-3 lbs/day, or 0.41 lbs/year. The mass flux of mercury from the north ditch (outfall 003) is estimated to be 1.67E-3 lbs/day, or 0.61 lbs/year.

Based on the filtration study data collected during 1997, individual storm events may lead to more significant mercury loading during high flow periods. The concentration of mercury in the southerly stream was 34 $\mu\text{g/l}$ in the turbid storm flow sample vs. 0.74 $\mu\text{g/l}$ in the non-storm low flow period. Filtration of the storm flow sample showed that approximately 90% of the mercury in the sample was associated with particulates greater than 20 microns in size (equivalent to medium silt). In the sample collected from the north ditch at the paved sump, approximately 75% of the mercury contained in the sample was associated with particulates greater than 20 microns. The north ditch samples contained a higher amount of dissolved mercury under both storm and non-storm conditions.

Comparison of low flow and storm flow samples collected for the filtration study show that a greater mass of mercury is mobilized during storm events than is represented in the non storm flux calculations. Because the flow and mercury concentration data collected by HoltraChem were collected only on a weekly basis, and only two significant storm events are represented, it is probable that the above values underestimate the amount of suspended mercury discharged to the river through surface water pathways. The surface water transport pathway includes transport of dissolved and suspended mercury during low flow and perhaps more importantly, suspended sediment during storm events.

As part of the 1995 Site Investigation Report, sediment loading from site surface water bodies to the Penobscot River was modeled using a Soil Conservation Service runoff model. This model application included numerous approximations of peak flows, mean sediment concentrations, and channel and sediment characteristics, making quantitative interpretations of the model output unreliable. However, the model results did support the results of filtration analyses, indicating that significant mercury loading to the Penobscot River could occur during peak storm events.

Because of the numerous variables involved in sediment transport calculations, the flux of mercury per year could not reliably be calculated with the information available. Results from filtration analyses during 1997, however, indicate that sediment loading to the Penobscot River may be a significant migration pathway for mercury transport off site, especially during storm events.

In 1997, EPA and MEDEP recommended that an interim stabilization measure (ISM) be implemented to limit sediment transport to the Penobscot River. Results of the filtration analyses suggest that this ISM will significantly reduce mercury loading to the Penobscot during storm events by allowing suspended sediments to settle before transport to the river occurs.

7.3.3 Transport and Leachability of Soil

Mercury introduced to soils, whether from spills, leaks, or atmospheric deposition, will become adsorbed to soil particles to varying degrees. The ability of soil to adsorb mercury, and retard its migration to groundwater, is dependent on the form of mercury introduced and site specific soil properties. In a moist soil environment, heavy metals such as mercury are subject to a variety of transformation processes including dissolution and precipitation, complexation, chelation, oxidation/reduction, ion exchange, and biologically mediated methylation. Some plant uptake is also possible. The variety of reactions is very complicated, and depends on many factors such as pH, redox potential, the presence of other reacting chemicals, and microbial activity. The result is a large number of possible mercury species including, for example, elemental mercury, mercury hydroxides, chloride salts and complexes, sulfide salts and complexes, complexes with organic matter (e.g., humic and fulvic acids), methylmercury, and mercury coprecipitated with other soil minerals such as iron hydroxides. Each of these chemical forms will have different fate and transport properties. Detailed models which can predict the actual speciation are not available, nor can laboratory tests easily distinguish the various chemical forms. Furthermore, the published literature is essentially devoid of measured soil sorption constants for mercury in realistic soil environments. The available literature generally indicates that there is a relatively rapid and irreversible sorption, or incorporation, of mercury into soil. Nevertheless, in areas with more significant soil contamination some mercury may be subject to leaching to the groundwater.

Mercury sorbed to surficial soils may be transported to a limited downward extent in the soil profile as particulates migrate downward with percolating waters. Mechanical soil mixing (e.g., from construction activities, bioturbation, or by subsequent covering with clean soil or natural deposits) could also lead to elevated mercury concentrations at depths. Continual source releases of mercury contaminated waste streams or spills would also be expected to result in elevated mercury levels at depth in soils. Mercury that does penetrate below the surface layer is, essentially, unavailable for transport via surface erosion and runoff to surface waters over the time frames that are significant for this SI. Except in areas of high erosion, the mercury that has entered the topsoil will have a very long residence time in this soil compartment. This extended residence time leads to formation of stable insoluble forms of mercury in soils that result due to slower chemical reactions (e.g., oxidation/reduction).

Overland transport of mercury contaminated soil will occur primarily in areas that are not vegetated or that have been disturbed by construction activities. The amount of soil which can migrate in surface water is dependant on the severity, magnitude and duration of the storm event and on the grain size, grade and other physical characteristics of the soil. Areas within and surrounding the

manufacturing plant are most likely to have elevated mercury concentrations and to be susceptible to overland transport with surface water runoff. Based on the concentrations of mercury detected in the sediments of the southerly stream, north ditch and southern cove, overland transport of mercury contaminated suspended sediments represents a major transport pathway to the Penobscot River.

The potential for mercury in surficial soils to leach to groundwater was evaluated using batch desorption tests. The objective of the batch desorption study was to develop a desorption isotherm for mercury in the sitewide soil: a graphical representation of the distribution of mercury between soil and water when the two phases are at equilibrium. The tests were conducted on soil samples collected from two areas of the site. Sample SSS-071 was collected adjacent to the outfall 001 weir and contained mercury at a concentration of 57 mg/kg. This mercury concentration is attributed to surface spills. Sample SSS-072 was collected from the landfill ridge and contained 13 mg/kg of mercury. The mercury in this soil sample is attributed to the filling of the area with soils from the main plant or to air deposition or a combination of the two. Because the potential for mercury to become sorbed to soil is highly dependent on the total organic carbon content (TOC) of the soil, the TOC was analyzed for each sample. The TOC values were 1.26% (SSS-071) and 1.07% (SSS-072), which is within the expected range for non-agricultural surficial soils in a field/forested environment.

The desorption isotherms for mercury were developed by dispersing varying masses of soil of known mercury concentration into 100 ml of water and allowing the mixture to equilibrate. The liquid phase was then analyzed for mercury. The results indicate the amount of mercury liberated from the soil at various dilutions, and thus the potential for mercury to be released to the groundwater from the soil in aqueous solution. Conversely, the results show the capacity of the soil to adsorb and retain mercury. The results of the batch desorption test are presented in tabular form in Table 3-23. The calculations and graphical results are provided in Appendix E.

The concentration of mercury in solution was measured by the laboratory and the equilibrium concentration in the soil was calculated by the equation stated and described below.

$$C = [(M \times C_o) - (C_s \times 0.1 L)] / M$$

where: C = concentration of mercury in soil at equilibrium

M = mass of soil added to 100 ml of water

C_o = laboratory measured concentration of mercury in the soil sample collected from the HoltraChem site

C_s = laboratory measured concentration of mercury in solution at equilibrium.

The number of pore volumes of water which pass through a soil volume can be related to the infiltration of rainwater and the dilution ratios used in the batch desorption tests. For example, assuming 50% void space within the soil, an infiltration of 4 inches of rainfall (infiltration due to a single major rain event) on one cubic centimeter of soil is equal to an approximate ratio of 100 ml water to 25 grams of soil or 5.3 pore volumes. Likewise, a rainfall infiltration of 20 inches (infiltration due to one year of rainfall) is roughly equivalent to a soil-water ratio of 5 g to 100 ml of water, or approximately 27 pore volumes.

The percentage of mercury desorbed from the soil on a weight basis for each of the soil to water ratios was calculated. Mercury was not detected in the water from the 0.1 and 1 grams soil to 100 ml dilutions for either of the soil concentrations. For both of the soil concentrations (13 and 57 mg/kg), mercury was desorbed from the soil at detectable concentrations ($>0.2 \mu\text{g/l}$) from the 5, 10 and 25 grams soil to 100 ml dilutions only. Therefore, for each soil concentration, three data points were available. The percentages of mercury desorbed from the 57 mg/kg sample ranged from 0.19% to 0.35 % indicating that between 99.65 and 99.81 percent of the mercury is bound to the soil and not leachable. For the 13 mg/kg sample, the percent dissolved ranged from 0.06% to 0.23% indicating that between 99.77 and 99.94 percent of the mercury was bound to the soil and not leachable.

For the 57 mg/kg sample, SSS-071, the 5, 10 and 25 gram dilutions resulted in desorbed mercury concentrations in water of 10, 15 and $27 \mu\text{g/l}$, respectively. For the desorption tests conducted on the 13 ppm sample (SSS-072), the resulting desorbed water concentrations were 1.5, 2.2, and $1.9 \mu\text{g/l}$ for dilutions of 5, 10, and 25 grams, respectively.

The data for sample SSS-071 (57 mg/kg) shows that the mercury concentration in water increased as the mass of mercury and soil added increased. A point was not reached where an additional increase in soil mercury resulted in a reduction or a leveling off of mercury desorption from soil to water. From this relationship, it can be concluded that for 57 mg/kg soil, increasing the mass of mercury will result in increased desorption of mercury from the soil to water.

The desorption data for soil sample SSS-072 (13 mg/kg) shows an increase in the desorbed mercury concentration in water to a point, after which the concentration in water decreases for an increased mass of mercury and soil. For the 13 mg/kg soil, it appears that an equilibrium has been reached between the fraction of mercury desorbed from the soil and the fraction which remains bound to the soil. A similar equilibrium is exhibited by the relationship shown in the desorbed mercury concentration in water versus pore volume. There was little change in the concentration of mercury in water (1.5, 2.2, $1.9 \mu\text{g/l}$) with increased pore volumes. The expected percolate concentration will be approximately $2 \mu\text{g/l}$ for 5 of 25 pore volumes of water passing through soil with a mercury concentration of 13 mg/kg.

The equilibrated water concentrations measured during this study can be viewed as representative of percolate water from the surficial soil layer. In actual field conditions this percolate would then pass through subsequent layers of soil which also has capacity to sorb mercury. This attenuation of mercury in the percolate as it travels through the vadose zone serves to limit the depth to which mercury and other metals migrate downward in soil profiles. Data from soil sampling within the plant area documents that this is occurring. An average eight fold attenuation was documented between the surface (0.0' - 0.5') and the near surface (1.0' - 1.5') intervals. This attenuation over a one foot interval would indicate an attenuation factor of 64 over two feet and 512 over three feet, respectively. If mercury reaches groundwater its concentration will be further diminished due to mixing (dilution) with the groundwater.

For a soil mercury concentration of 13 mg/kg, the desorption testing showed that an equilibrium had been reached at which no more mercury was desorbed. The desorption of mercury from the 13 mg/kg soil resulted in percolate water concentrations in the 2 µg/l range for each of the soil-water ratios tested.

The results of the desorption tests are consistent with the lysimeter data from the immediate vicinity of sitewide soil sample locations SSS-035 (3.2 mg/kg) and SSS-053 (6.2 mg/kg) and with the groundwater sample results from the MW-407 cluster which is screened below the fill on the north side of the bedrock ridge. Total mercury concentrations in the water sample collected from the lysimeter located at SSS-035 were below the method detection limit of 0.2 µg/l. The water sample from the lysimeter located at SSS-053 contained 2.0 µg/l total mercury, and the dissolved concentration was less than the detection limit of 0.2 µg/l. The overburden monitoring well, MW-407-O1, contained 0.38 µg/l in the parent and 0.44 µg/l in the duplicate sample in 1995. This well was dry in 1997. The bedrock well, MW-407-B1, did not contain mercury above the method detection limits of 0.2 µg/l in either the 1995 or the 1997 sampling rounds. These data suggest that mercury associated with soils in this area is not contributing substantially to contamination of groundwater.

7.3.4 Groundwater Discharges

Mercury and VOCs were detected in groundwater on the ridge, adjacent to the bedrock scarp and at the base of Landfill Area 1. A significant percentage of the mercury in these monitoring wells is in a soluble form. Some attenuation of mercury and VOCs to soil in the saturated zone may occur which would result in retardation of transport. However, monitoring wells located near the Penobscot River contain elevated concentrations of mercury and VOCs, so these concentrations were used in contaminant flux calculations without assumptions of additional attenuation. Therefore, mercury and VOC transport is estimated to occur at the seepage velocities presented in Section 5.4.4. This is

likely to be a conservative assumption because both mercury and many VOCs do adhere to soil particles, especially organic carbon. However, because of the complexity of many of these reactions and the various media through which groundwater travels (rock, till, sand and gravel, and sediment) it is not practical to establish an attenuation/retardation factor which adequately addresses the complexity of the site.

One other factor which could result in over estimation of groundwater flux is the assumption that the concentrations measured in groundwater samples are representative of the average concentrations moving in the groundwater. At the B-326 cluster, the difference in concentrations between the water table well (B-326-O3) and the well screened at the base of the sand and gravel deposit (B-326-O2) show that the concentrations are not evenly distributed vertically. It is possible that the concentrations detected in wells screened at the base of the sand and gravel deposits are associated with dense aqueous phase saline liquids which are somewhat stagnant or at least not moving as fast as less contaminated groundwater higher up in the deposit.

The mass flux of mercury and VOCs to the river was calculated from the concentrations observed in each area and the flux of groundwater anticipated. For sitewide characterization, estimation of the area and amount of infiltration was necessary to establish the amount of groundwater discharging from each region. The controlling factors are the location of the drainage divides and the relative rate of infiltration anticipated based on the composition of the drainage area surface. Estimates of groundwater discharge to the Penobscot River were developed for water balance calculations and were discussed in Section 5.4.5. Contaminant flux calculations are provided in Appendix E. The fluxes were calculated for the scenarios described below.

On the bedrock ridge, a groundwater recharge area was established using the December 1997 groundwater contours. Flowlines anticipated to pass through the area where contamination was detected indicated discharge to the north. The amount of infiltration to this area was conservatively estimated based on the assumption that 80% of the area was covered with sand and gravel deposits. The presence of the landfill caps was ignored because the water running off these caps was assumed to infiltrate elsewhere within the area of recharge. During the 1997 groundwater sampling event, monitoring wells MW-408-O1, MW-407-O1, and B-309-O1 were dry. These wells had the highest detected mercury concentrations in this area during 1995, and also had detectable concentrations of carbon tetrachloride and chloroform in 1995. The total contaminant fluxes for this region were therefore calculated using concentrations of mercury, carbon tetrachloride and chloroform in these wells during the 1995 sampling event, and are therefore considered conservative estimates of groundwater fluxes in this area. The flux of mercury from the ridge northward to the Penobscot River was calculated to be 3.1×10^{-5} lbs/day (0.01 lbs/year). The flux of carbon tetrachloride was calculated to be 1.1×10^{-4} lbs/day (0.04 lbs/year) and the flux of chloroform was 2.6×10^{-4} lbs/day

(0.09 lbs/year).

Compared to the potential discharge occurring at the base of Landfill Area 1, discussed below, the estimated flux of contaminants from the bedrock ridge is more than two of magnitude orders lower. Therefore, these calculated fluxes do not have a significant effect on the calculated fluxes from the site. Groundwater containing mercury and potentially VOCs is also discharging toward the southerly stream as it passes the southern side of the plant. However, groundwater in this area is being collected by a groundwater collection system designed for this purpose. This groundwater is treated in the plant's wastewater treatment system and the flux to the river is therefore included in the number for the NPDES discharge. Based on the presence of dissolved mercury detected in southerly stream surface water, some contaminated groundwater may not be collected by the existing groundwater collection system. However, this flux is represented by the flux estimated for the southerly stream based on concentrations and flows.

The groundwater flux to the Penobscot River beneath Landfill Area 1 was estimated in four ways, as discussed in Section 5.4.5. Briefly, these methods were based on estimating the amount of water discharging from the plant and passing beneath Landfill Area 1 (methods 1 and 2) or estimating of the amount of water discharging from Landfill Area 1 based on modeled or measured hydraulic gradients (methods 3 and 4, respectively). Each of these methods contain assumptions with associated uncertainties. The four methods were performed to evaluate the range of groundwater fluxes, to evaluate whether there were similar results in the predicted fluxes with the various approaches, and to provide for evaluation of the ranges and degree of uncertainty in the estimates.

The groundwater flux estimates and the analytical results of groundwater monitoring were used to estimate the mercury discharge from the base of Landfill Area 1. The two methodologies estimating discharge from the plant (methods 1 and 2) provide high and low estimates for the amount of groundwater passing through the stratified drift deposits below the landfill. Using this range of groundwater fluxes, flow was assumed to be proportioned below the landfill based on the saturated thickness and measured hydraulic conductivity of each well. This proportioning is represented on cross section A-A' in Appendix E. By proportioning the groundwater flow in this manner, and multiplying the resulting flux of water in each cell by the mercury concentration in that cell, the flux of mercury off site through groundwater is estimated to be between 8.0 and 13.8 lbs/year based on 1997 concentrations, and between 7.1 and 12.2 lbs/yr based on 1998 concentrations. Calculations are provided in Appendix E.

For comparative purposes, additional calculations were made using the range of groundwater fluxes from methods 1 and 2 and multiplying by the geometric and arithmetic mean mercury concentrations in the wells beneath the landfill. Using the geometric mean of the concentrations observed in

monitoring wells MW-401-O1, MW-512-O1, B-326-O2, B-326-O3, MW-513-O1 and MW-402-O1 (569.6 ug/l), the low and high flux estimates were 5.1 lbs/year and 8.9 lbs/year based on 1997 concentrations, and 4.3 and 7.5 lbs/yr based on 1998 concentrations. Using the arithmetic mean of the same wells (824.86 ug/l) yields flux estimates between 7.5 lbs/year and 12.8 lbs/year based on 1997 concentrations, and between 7.9 and 13.7 lbs/yr based on 1998 concentrations. Although the flow cell approach is the most systematic method of calculating the flux of contaminants beneath the landfill, the arithmetic average appears to provide a useful simplification of the calculations for this hydrogeologic system.

The amount of contaminant flux was also calculated using the groundwater flux estimated with modeled and measured hydraulic gradients at the base of Landfill Area 1 (methods 3 and 4). Mercury fluxes estimated from these methods ranged from 13.4 to 26.4 lbs/year based on 1997 concentrations and 14.8 to 28.5 lbs/yr based on 1998 concentrations. It is important to note that fluxes calculated with the hydraulic gradients modeled and measured at the base of Landfill Area 1 are very sensitive to water level and survey measurements and that some discrepancies (i.e. negative gradients) were noted in the data. For example, the differences in the estimated discharge presented above result from adjustments of surveyed data of only nine one hundredths of a foot. The lower discharge numbers, which are closer to the results produced from methods 1 and 2, are the product of hydraulic gradients calculated using the adjusted survey which is considered most representative of actual site conditions.

To use all the data generated from the various methods and to produce a consensus estimate for the amount of mercury flux to the Penobscot River, the results of all the estimation methods were tabulated and evaluated. The average (mean) estimated flux is 12.7 lbs/year which is consistent with the level generated with the infiltration based calculation (method 1). The standard deviation of the various estimated flux numbers is approximately 6 lbs/year indicating significant uncertainty exists due to the sensitivity of the calculation and the accuracy of available measurements.

The total mercury flux to the Penobscot River was estimated using the mean of the calculated values discussed above (12.7 lbs/yr), the current plant NPDES discharge (2.9 lbs/yr) and the amount estimated to leave the site through surface water (1 lb/yr). Using these values, the total flux of mercury is estimated to be 16.6 lbs per year. The HoltraChem site is currently permitted to discharge 16.64 lbs per year in its NPDES permit.

There are several factors and observations which suggest that the values cited above may be an overestimate of the contaminant flux off site. 1) Attenuation and retardation of contaminants may be occurring which was not accounted for in the calculations. 2) The concentrations measured in groundwater samples from the base of the deposit may not be representative of the average

concentrations moving in the groundwater. Dense saline liquids may be somewhat stagnant or at least not moving as fast as less contaminated groundwater higher up in the deposit. 3) Preferential pathways for groundwater flow could result in greater discharge of less contaminated water than from areas where the highest concentrations have been detected.

Observations from the 1997/98 field season suggest that the calculated values may be an overestimate of the actual mercury flux. During a site visit in September 1998, seepage from the beach face was observed to be concentrated in locations downriver of Landfill Area 1. This observation indicates that discharge may not be occurring beneath Landfill Area 1 to the extent predicted in the calculations but may be taking a circuitous route and preferentially discharging where groundwater is less contaminated.

Beach face seep samples were collected at three locations by Acheron as part of their Penobscot River Studies. The samples were collected by impounding the seepage and collecting a sample using "clean" techniques (clean hands/dirty hands, acid washed tubing and bottles, etc), as the tide was coming back in and were analyzed by Frontier Geosciences for low detection levels. The sample from the seep closest to Landfill Area 1 contained mercury at 0.0863 $\mu\text{g/l}$ total and 0.0287 $\mu\text{g/l}$ dissolved. The total and dissolved concentrations detected in this seep are approximately 1200 and 2800 times lower, respectively, than the concentrations detected in the nearest well screened across the water table at the base of Landfill Area 1 (100 $\mu\text{g/l}$ total and 80 $\mu\text{g/l}$ dissolved in B-326-O3, 1998). The sample collected from a higher volume beach face seep noted around the HoltraChem outfall contained 0.0757 $\mu\text{g/l}$ total and 0.0359 $\mu\text{g/l}$ dissolved mercury. The total and dissolved concentrations detected in this seep are 56 and 5.6 times lower, respectively, than the concentrations detected in the nearest well screened across the water table (4.3 $\mu\text{g/l}$ total, and <0.02 assumed 0.02 $\mu\text{g/l}$ dissolved at B-320-03, 1997 concentrations).

The beach face samples were collected as the tide was coming in when the amount of groundwater contributing to beachface discharge is expected to be at its maximum and bank storage at a minimum. The beach face samples indicate that the amount of mercury being discharged to the river may be substantially less than that predicted by the flux calculations.

The fluxes of selected volatile organic compounds from wells downgradient of Landfill Area 1 were also calculated using the flow cell approach. The total fluxes of chloropicrin, carbon tetrachloride, and chloroform were calculated using 1997 and 1998 groundwater monitoring results. Based on the 1997 data, the flux of chloropicrin entering the river below the landfill was calculated to be between 0.20 and 0.34 lbs/day (71 to 125 lbs/year). The maximum calculated fluxes for chloroform and carbon tetrachloride were 1.7 E-2 and 3.3 E-3 lbs/day (6.2 and 1.2 lbs/yr), respectively. All of these values remained constant or decreased slightly between the 1997 and 1998 sampling events. As with

the calculated mercury fluxes, these values are likely to be overestimates of the contaminant flux below landfill area 1, since estimates of groundwater flux in this area are assumed to be conservative.

Section 8

Human Health Risk Assessment

Menzie-Cura & Associates, Inc. conducted a human health and ecological risk assessment for the chemicals detected at the HoltraChem Manufacturing site (the site) in Orrington, Maine. This chapter presents the methodology and results of the human health risk assessment. The ecological risk assessment is presented in Section 9.

The objective of the human health risk assessment (HRA) was to assess current and future potential risks to on site workers, nearby residents, and users of nearby recreational resources (*i.e.*, the Penobscot River). The assessment considered potential risks associated with exposure to chemicals in soils, groundwater on site and groundwater potentially migrating off site, on site surface water and sediment, ambient air, and Penobscot River surface water and sediment.

Total cancer risk estimates either did not apply (*i.e.* the scenario did not involve exposure to carcinogenic compounds) or were less than the State of Maine target level of 1×10^{-5} for four of five scenarios evaluated: current on site facility workers engaging in sitewide walkovers; current on site trespassers; current off site residents; and future on site facility workers engaging in excavation or construction. Total cancer risk exceeded the State of Maine target level for the future hypothetical on site residents. The noncancer hazard indexes were greater than the target index of one for the current off site receptor, but were below target index for current on site receptors. The total noncancer hazard index for the future hypothetical on site resident scenario was greater than one.

8.1 Introduction

This HRA is conducted in accordance with the "Human Health and Ecological Risk Assessment Protocol, HoltraChem Manufacturing Site, Orrington, Maine" [Camp Dresser & McKee Inc. (CDM) and Menzie-Cura, 1994]. The protocol was approved by the United States Environmental Protection Agency (USEPA) and State of Maine Department of Environmental Protection (MEDEP). Both the protocol and the HRA are based on current State of Maine and USEPA guidance, as described in the following documents:

- USEPA Region I. *Risk Update*. Number 4. November, 1996.
- USEPA Region I. *Risk Update*. Number 3. August, 1995.

- State of Maine Department of Environmental Protection and Department of Human Services. *Guidance Manual for Human Health Risk Assessments at Hazardous Substances Sites*. June, 1994.
- USEPA Region I. *Risk Update*. Number 2. August, 1994.
- USEPA. Guidelines for Exposure Assessment. FR 57(104), May 29, 1992.
- USEPA. *Guidance for Data Useability in Risk Assessment (Part A)*. Final. Publication 9285.7-09A. PB92-963356. April, 1992.
- USEPA. Human Health Evaluation Manual Supplemental Guidance: "Standard Default Exposure Factors." OSWER Directive 9285.6-03. March 25, 1991.
- USEPA. *Risk Assessment Guidance for Superfund. Volume I - Human Health Evaluation Manual (Part A)*. Interim Final. EPA 540/1-89-002. December, 1989.
- USEPA Region I. *Supplemental Risk Assessment Guidance for the Superfund Program*. Draft Final. EPA 901/5-90-001. June, 1989.
- USEPA. *Exposure Factors Handbook*. EPA 600/8-89/043. May, 1989.

Figure 8-1 outlines the five major components addressed in the HRA:

- | | |
|---|-------------|
| ■ Data Evaluation and Hazard Identification | Section 8.2 |
| ■ Exposure Assessment | Section 8.3 |
| ■ Dose-Response Assessment | Section 8.4 |
| ■ Risk Characterization | Section 8.5 |
| ■ Uncertainty Analysis | Section 8.6 |

The Data Evaluation and Hazard Identification section identifies the contaminants of concern at the site, *i.e.*, those chemicals potentially associated with significant risk. The Exposure Assessment quantifies the exposures associated with current and future land uses; identifies potential receptors, exposure pathways, and exposure media; and estimates exposure point concentrations and daily exposure doses for potential receptor groups. The Dose-Response Assessment section examines the effects of the contaminants of concern on humans and assigns quantitative estimates of toxicity. The

Risk Characterization integrates the exposure and dose-response assessments into a quantitative characterization of human health risk. Site risks are compared to target risk levels for evaluating noncarcinogenic health hazards and carcinogenic risks. Contaminant concentrations are compared to applicable standards.

8.1.1 Site Conceptual Model

A site description is presented in Section 2. The hydrologic and hydrogeologic characteristics of the site are presented in Section 5. The ecological characteristics and habitats are described in Section 9. This HRA refers to study areas within the 235-acre HoltraChem property boundaries as "on site study areas," and refers to study areas outside the HoltraChem property boundaries as "off site areas."

On site study areas are further defined as "the plant area" and "the area outside the plant area." The plant area is a mostly paved area that encompasses approximately 12 acres of active and ongoing plant operations. The on site area outside the plant area is mostly wooded and largely undeveloped. It is approximately 223 acres and includes the five landfills.

The off site study area is a cove of the Penobscot River located southwest (downgradient) of the HoltraChem property. This off site study area is referred to as "the southern cove." The Penobscot River is immediately adjacent to the site, receives water from National Pollutant Discharge Elimination System (NPDES) permitted releases, and may have received contamination originating from the site.

8.2 Data Evaluation and Hazard Identification

HoltraChem Manufacturing uses the chlor-alkali process to produce chlorine gas at the site. Sections 3, 6, and 7 of this SI describe the nature and extent of contamination at the site.

8.2.1 Data Useability

The USEPA Guidance for Data Useability in Risk Assessment (USEPA, 1992c) provides guidance for the assessment and interpretation of environmental data for use in human health risk assessments. The following discussion of data useability is presented according to the six criteria outlined in USEPA (1992c) and described below.

8.2.1.1 Reports to Risk Assessor

USEPA (1992c) provides a list of information that should be included in reports and provided to the risk assessors.

- site description
- sampling design
- analytical methods and detection limits
- results on per-sample basis, qualified for analytical limitations
- sample quantitation limits and detection limits for non-detects
- field conditions for media and environment
- preliminary reports
- meteorological data
- field reports

We were provided with reports and/or information that address all of the above points.

8.2.1.2 Documentation

USEPA (1992c) specifies the four major types of documentation that ensure sample results are properly related to geographic locations:

- sampling and analysis plan
- chain-of-custody records
- standard operating procedures (SOPs)
- field and analytical records

We obtained documentation for sampling conducted during 1989-1990 as described in Acheron, 1990. We also obtained documentation for sampling conducted by CDM in 1994, 1995, 1997, and 1998. We were not able to obtain documentation for any samples collected prior to 1989. Consequently, data collected prior to 1989 were not used in the quantitative risk assessment. The data collected by Acheron and CDM are more comprehensive and considered to be more representative of site conditions.

8.2.1.3 Data Sources

USEPA (1992c) provides the following guidance on evaluating data sources for their usefulness in characterizing a site:

CDM Camp Dresser & McKee \ Menzie • Cura & Associates, Inc.

December 22, 1998

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- analytical data results for one sample per medium per exposure pathway
- broad spectrum analysis for one sample per medium
- field measurements data for media and environment

Table 8-1 provides a summary of the compounds detected in each medium. A brief discussion of the sampling rationale for each medium is provided below.

On site Soils: The data set for on site soils includes data from the plant area and the area outside the plant area. The plant area, where most of the contamination is expected to be, is well represented by the number and location of samples.

The plant area is characterized by 214 soil samples that were analyzed for mercury. A few of these samples were also analyzed for volatile organic compounds (VOCs). The specific locations of these samples were targeted for known spill areas to ensure complete characterization of the plant area. The area outside the plant area is characterized by 196 soil samples analyzed for mercury, 8 samples analyzed for RCRA 8 metals, 26 samples analyzed for VOCs and 21 samples analyzed for chloropicrin.

On site and Southern Cove Sediment: Twenty-two on site sediment samples (from outside the plant area) and 456 southern cove sediment samples (from the southern cove in the Penobscot River) were analyzed for mercury and total organic carbon (TOC). A subset of these samples was analyzed for VOCs. On site sediment samples were taken from stormwater areas across the site. Samples from Acheron (1990) and CDM (1994, 1995, and 1997) have been combined. The samples from the southern cove of the Penobscot River ensure adequate characterization of this area.

On site and Southern Cove Surface Water: Sixty eight on site surface water samples (from outside the plant area) and 41 southern cove surface water samples (from the Penobscot River) were analyzed for mercury and water quality parameters. A subset of samples was analyzed for VOCs and RCRA 8 metals. A total of 15 samples were collected and analyzed from on site seeps, which were incorporated into the summary statistics for on site surface water in the risk assessment.

On site Groundwater: Well P-2A, near Landfill 3, was identified in 1989 as the well displaying the most contamination prior to initiating the study (Acheron, 1990). Consequently, on September 15, 1989, Well P-2A was analyzed for Appendix IX parameters to determine the list of contaminants reported in Acheron (1990; Table 3-13), which is the basis for subsequent groundwater sampling activity. There are also eleven RCRA wells around Landfill 5 which were sampled quarterly during 1989 and 1990 for the RCRA monitoring list. This list includes those contaminants for which EPA

had interim drinking water standards, general water quality parameters, and indicator parameters as shown in Table 3-13 of Acheron (1990). None of the wells near Landfill 5 had detectable concentrations of PCBs or pesticides. In 1995, CDM analyzed for Appendix IX parameters in three overburden wells (B-326-O2, B-316-O1, and P-2A). In 1997, CDM installed and sampled 35 additional overburden and bedrock wells. These wells and existing wells were sampled in 1997. Several contaminants were identified in this screening that were not included in groundwater analysis from other locations. Groundwater samples collected prior to May 1995 were excluded from the risk assessment because the current data better represents site conditions. The groundwater database consists of a total of 65 sampling points for mercury and several samples analyzed for VOCs and other metals. Chloropicrin was analyzed in 65 locations in samples collected from August 1997 through 1998.

Biota: Biota samples were analyzed for mercury only.

Air: The State of Maine Department of Environmental Protection collected 24 ambient air samples for mercury in December, 1997 at various indoor and outdoor locations in the plant area of the site. Twenty of these were from outdoor locations and four were from the cell room.

8.2.1.4 Analytical Method and Detection Limit

USEPA (1992c) documents the use of routine methods to analyze chemicals of potential concern. We reviewed the analytical methods and detection limits provided in the appendices of Acheron (1990) and Section 4.0 of this document. The analytical methods are appropriate and acceptable for risk assessment purposes. The detection limits are within acceptable ranges. Subsequent to workplan approval, an EPA method for analysis of chloropicrin in drinking water (Method 551) was issued. There is currently no EPA method for analysis of chloropicrin in soil.

8.2.1.5 Data Review

USEPA (1992c) discusses the importance of alternative levels of data review, citing timeliness and depth as the critical characteristics. A complete data review for the samples collected by CDM during 1994-1997 is provided in Section 4, and the complete data validation report is provided in Appendix H. A similar review of the samples collected prior to 1994 was not available.

8.2.1.6 Data Quality Indicators

USEPA (1992c) discusses five data quality indicators as they relate to the assessment of sampling and analysis: completeness, comparability, representativeness, precision, and accuracy. These

indicators are evaluated through a review of sampling and analytical data and accompanying documentation.

“Completeness” is a measure of the amount of useable data resulting from data collection activity. As described below, the data are generally complete.

“Comparability” refers to the confidence with which data are considered to be equivalent. The data appear comparable for all media, except ambient air.

“Representativeness” refers to the extent to which the data defines the true risk to human health and the environment. Generally, the data appear representative of site conditions. Subsurface soil concentrations are conservatively estimated by the near surface soil data, which may not be representative of conditions below the surface. The air data gathered by Maine Department of Environmental Protection in 1997 do not appear representative of site conditions, as judged by the biased sample locations and other factors discussed below.

“Precision” refers to a quantitative measure of variability, comparing results for site samples to the mean, and is usually reported as a coefficient of variation or a standard derivation or the arithmetic mean. We use standard deviation to represent as a measure of precision, and generally, we feel the data has good precision, based on their low standard deviation.

“Accuracy” refers to the closeness of a reported concentration to the true values. It is our opinion that the data have a high level of accuracy, based upon laboratory and field quality control checks.

We determined the distribution of the data in each medium by graphical methods. The distribution of data is determined to calculate the 95% upper confidence limits, as presented in Appendix F-1. The data, based on simple plots of concentration versus number of samples, was mostly normally distributed, except where indicated in the following text. Therefore, 95% UCLs were calculated based on the H-Statistic method (USEPA, 1992e). Summary tables for each of the media at the site are provided in Appendix F.

On site Soils: Appendix F1-1 provides summary statistics for mercury in on site surface soil, combining CDM and Acheron data. The Acheron data are primarily from the area outside the plant area. Summary statistics for VOCs in on site surface soil samples are also shown.

The plant area has the highest levels of mercury and a dataset that is both random and variable. The dataset is referred to as random because it is not normally distributed, as is often the case with environmental data. The dataset is referred to as variable because the statistical indicator, the

standard deviation, was large. The geographical distribution of the detected concentrations indicates the presence of somewhat localized hotspots, areas where high detected concentrations are clustered.

Outside the plant area, soils are characterized with 300 data points. This dataset is also referred to as random and variable because it is not normally distributed and has a large standard deviation.

As the limited subsurface data is not adequate for risk assessment purposes, it is assumed that exposure to mercury in subsurface soil is at concentrations found in surface soil. This is a conservative assumption because the primary sources of soil contamination at the site are surface spills and atmospheric deposition. This methodology is supported by contamination versus depth profiles conducted by both CDM and Acheron. The evaluation of subsurface soil as surface soil does not include assumptions about weathering, dilution, or other mechanisms.

The soil data are complete for surface soils, but not for subsurface soils. The soil data are comparable, accurate and representative of site conditions. The plant area soils are not highly precise, judged by the relatively large standard deviation.

On site Sediment: Appendix F1-3 provides summary statistics for the on site sediment samples. The on site sediment data are not normally distributed and show some variability, primarily because of the mercury located in the stormwater area leading from plant operations to the Penobscot River. The hotspot in the stormwater sediment was removed. On site sediment samples taken from outside the area of plant operations show lower concentrations of mercury, including two non-detects.

Southern Cove Sediment: Appendix F1-3 provides summary statistics for the off site sediment samples collected from the southern cove in the Penobscot River. The southern cove sediment data are not normally distributed.

On site Surface Water: Appendix F1-4 provides summary statistics for the on site surface water samples. The on site surface water data include both Acheron and CDM data.

Appendix F1-4 provides summary statistics for the off site surface water samples collected by CDM from the southern cove in the Penobscot River.

Off site Surface Water: The Penobscot River samples are provided for only the southern cove because it is the area most hospitable to swimmers. Both on site and southern cove surface water samples follow a lognormal distribution.

The sediment and surface water data meet the data quality indicators.

Groundwater: Appendix F1-2 provides the data summary for site-wide groundwater samples from fall 1997 through 1998 from all groundwater wells. Appendix F1-2 provides the data summary for samples collected quarterly from 1997 to 1998 from the well cluster MW-321, used to conservatively estimate potential off-site drinking water levels for current off-site residents. Residential well data has also been gathered and is shown in Appendix F1-2. The data were collected over several sampling events and show high reproducibility between sampling events, indicating a high level of accuracy. However, the standard deviation is large, indicating low precision, due to variability site-wide in groundwater conditions.

Biota: Appendix F1-6 provides summary statistics for the biota samples. Only eel data are used in the HRA. The eel data have a low level of variability and are judged consistent with data quality indicators for high data quality.

Ambient Air: Table E-10 provides summary statistics for Maine DEP's ambient air data. This data is judged inaccurate and unreliable for several reasons. First, the minimum detectable concentrations for a four-hour ambient air sample is expected to be $3 \mu\text{g}/\text{m}^3$, based on the NIOSH method (CDM, personal communication, 1998) while the reported results range from 0 to $4.39 \mu\text{g}/\text{m}^3$. Samples collected from a single event may not be representative. Four of the five collocated samples have poor reproducibility, indicating a low level of precision. The pump air flow rates were estimated due to the variability of the measured air flow rates, an indicator of low comparability. Finally, air dispersion modeling results (Earth Tech, 1998) indicate one-hour average maximum expected ambient air concentrations exceeding $3 \mu\text{g}/\text{m}^3$, but on site annual average maximum concentrations are predicted in the range of $0.1 \mu\text{g}/\text{m}^3$, well below the detectable level. The one-hour estimates are not representative of chronic exposure levels. Therefore, the Maine Department of Environmental Protection ambient air data is not judged acceptable by EPA data quality indicators, and is not used in the risk assessment.

8.2.2 Contaminants of Concern

Contaminants of concern are selected from the following datasets:

- recently collected data as summarized and evaluated in Sections 3.0 and 4.0 of this report; and
- previously collected data provided in Acheron (1990) and described in Section 2.0 of this report.

The Work Plan (CDM, 1993) provided a preliminary data evaluation and selected potential COCs based on site history and data collected by Acheron (1990) and PELA (1988). Historical sampling data and the most recent sampling data (CDM) were evaluated for selection of COCs. In certain cases, contaminants identified as potential COCs based on historical sampling results have been eliminated as COCs with the incorporation of more recent data. An expanded database mostly impacts the selection of COCs based on frequency of detection. Many of the preliminary COCs identified in the Work Plan are no longer COCs because of low frequency of detection. Other criteria used to select COCs are described below.

All detected compounds for the site are listed in **Table 8-1** and presented by medium below.

SOIL

Mercury
Chloropicrin
Ethylbenzene
Xylenes
Lead
Arsenic
Barium
Cadmium
Chromium

ON SITE SURFACE WATER

mercury
carbon tetrachloride
chloroform
iron
magnesium
manganese
potassium
sodium
calcium

GROUNDWATER

Mercury
Acetone
Bromoform

SOUTHERN COVE SURFACE WATER

mercury

ON SITE SEDIMENT

carbon disulfide	mercury
carbon tetrachloride	
Chloroform	SOUTHERN COVE SEDIMENT
Chloropicrin	mercury
Manganese	
Trichloroethene	BIOTA
	mercury
Additional VOCs	AIR
	mercury

8.2.2.1 Methodology

COCs are selected following the approach presented in the Human Health and Ecological Risk Assessment Protocol Document (CDM and Menzie-Cura, 1994). The steps involved in selecting COCs for the human health risk assessment follow.

Sort data by medium. Data are sorted by medium, combining data from different sampling events as appropriate.

Evaluate data quality. The risk assessment relies on data validated by or assumed for Gradient Corporation or CDM. Within the scope of the risk assessment, data are evaluated (*i.e.*, assessed sample quantitation techniques, assessed the assigned qualifiers, codes, and detection levels) to determine which samples can be retained for subsequent consideration in the risk assessment. The "J"-qualified data are retained for analysis, and duplicate samples are averaged and treated as a single sample.

Evaluate common laboratory contaminants. Common laboratory contaminants (*e.g.*, methylene chloride and acetone) are evaluated following USEPA guidance (USEPA, 1989). Common laboratory contaminants are eliminated as contaminants of concern if their detected concentrations are less than 10 times the concentration detected in the method blank.

Assess frequency of detection. Analytes detected in less than 5% of the samples are eliminated from the risk assessment if a statistically significant number of samples is available. Contaminants that are

detected in one or a few samples can also be eliminated from further analysis if the measured concentration is less than five times the method quantitative limit (USEPA, 1989).

Evaluate contaminant properties. The toxicity, mobility, persistence, and potential bioaccumulation of each contaminant is evaluated. This step can result in the inclusion of contaminants that were previously screened out. A contaminant that is known to have significant toxicity can be included as a COC based on its potential toxic effects. A contaminant that may volatilize or be present in multiple exposure media may also be included as a COC based on increased exposure potential.

Compare to background, standards, guidelines and criteria. Maximum detected contaminant concentrations are compared to the following, and are excluded if they do not exceed these levels:

- natural background concentrations identified for the local, state and regional areas where available (*i.e.*, United States Geological Survey data for eastern United States);
- standards that include Maximum Contaminant Levels (MCLs), Secondary Maximum Contaminant Levels (SMCLs), and Ambient Water Quality Criteria (AWQC) for the protection of human health;
- guidelines that include the State of Maine Maximum Exposure Guidelines (MEGs), Maine Bureau of Health fish advisories, and Food and Drug Administration (FDA) action levels; and
- risk-based screening criteria that include Risk Based Concentrations (RBCs) developed by USEPA Region III (RBC Table, dated 4/15/98), the July 27, 1990 Proposed Corrective Action Rule for Solid Waste Management Units (55 FR 30798).

These values are listed for comparative purposes only and do not necessarily represent applicable or relevant and appropriate requirements (ARARs) for the site. **Tables 8-2 through 8-6** summarize the selection of COCs.

8.2.2.2 Results

Soil

Table 8-2 compares the detected soil constituents to RBCs and other guidelines. The detected concentrations of mercury in on site soil range from non-detect to 350 mg/kg and exceed the RBC

for residential land use (23 mg/kg), the Resource Conservation and Recovery Act (RCRA) Corrective Action Level (20 mg/kg), and the SSL (23 mg/kg) for mercury.

There are no standards available for chloropicrin, but this compound is selected as a COC for soil because of the potential toxicity of chloropicrin at the concentrations detected in on site soil in certain areas.

Carbon disulfide, 2-butanone, and hexanone were identified in the Work Plan as potential COCs for soil based on initial sampling results. Subsequent sampling by CDM has not revealed detectable levels of these contaminants in soil. Each of these contaminants was detected in less than 5% of the samples analyzed for these parameters. They are not considered COCs for soil because of low frequency of detection. Ethylbenzene, xylenes, and several metals were detected in soils, but not at levels exceeding any of the screening criteria. They are not considered COCs for soil because they are not present at levels that exceed relevant criteria. Arsenic was measured at concentrations ranging from 8.4 to 13 mg/kg in 8 of 8 soil samples. While this level exceeds the residential RBC for arsenic (0.43 mg/kg), it is consistent with background concentrations for arsenic in soils. Arsenic is not a COC because it is present at levels consistent with background (Shacklette and Boerngen, 1984).

Groundwater

The greatest number of contaminants were reported in analyses of groundwater samples. All groundwater data collected prior to August 1997 was excluded because the more recent data is more comprehensive and more representative of site conditions. Results for each well are reported as the average of all sampling results from 1997 through 1998. Table 8-3 summarizes the groundwater COC selection process. The table lists:

- the maximum concentration detected;
- the frequency of detection;
- the tap water RBC;
- the MCL for groundwater; and
- whether the contaminant was selected as a COC.

The contaminants selected as COCs for groundwater are identified with an "x".

For most of the volatile organic compounds, the total number of samples included in the groundwater database is 65. Many of the VOCs are eliminated as COCs based on low frequency of detection. The following contaminants detected in the groundwater sampling program are eliminated

as COCs based on frequency of detection (other criteria may also apply): 1,1-dichloroethane; carbon disulfide; bromodichloromethane; chloromethane, chlorobenzene; dibromochloromethane; and methylene chloride.

The VOCs selected as COCs were selected because they were frequently detected, the maximum concentrations detected were greater than the MCL, or an MCL was not available and the reported maximum concentrations exceeded the risk based concentration for tap water. The following VOCs are retained as COCs: acetone, bromoform, carbon tetrachloride, chloroform, and trichloroethene. There are no standards available for chloropicrin, but it was selected as a COC for on site groundwater because of the potential toxicity at the concentrations detected in some areas.

The semivolatile organic compounds were analyzed less frequently than the VOCs. Twenty nine wells were sampled for Appendix IX parameters. No SVOCs were detected.

Mercury was analyzed in 65 groundwater wells out of 32 overburden wells and 36 bedrock wells available.

Other metals were analyzed in up to 65 samples. Metals measured and detected in groundwater in 1997 and 1998 include calcium, iron, magnesium, potassium, and sodium which are naturally abundant, essential nutrients, and are not toxic at the concentrations detected. These metals are not selected as COCs for groundwater because they were not detected at concentrations exceeding the MCL or the RBC for tap water.

Mercury and manganese were also detected in groundwater samples. Mercury is selected as a COC for groundwater because the detected concentrations in groundwater are greater than the MCL (2 µg/l) and the MEG (2 µg/l). Manganese is selected as a COC because concentrations exceed the tap water RBC (0.730 mg/l) and the SMCL (0.050 mg/l).

Summary statistics for well 321 samples are presented separately because these wells are used to estimate potential concentrations of COCs in off site drinking water wells.

On site and Southern Cove Surface Water

Mercury is selected as a COC in surface water both on site and in the river because the detected concentrations in these surface waters are greater than the AWQC for human health for the consumption of water and organisms (acute 2.4 µg/l; chronic 0.050 µg/l). Table 8-4 summarizes the comparison of mercury to standards and guidelines.

Several inorganics were detected not but included as COCs. Carbon tetrachloride and chloroform were detected once in on site surface water and are conservatively included as COCs, because they exceed the Ambient Water Quality Criteria.

On site and Southern Cove Sediment

Mercury was frequently detected in on site and southern cove sediment and was retained as a COC despite the lack of applicable or relevant criteria or guidelines, as shown in Table 8-5.

Biota

The maximum concentration of mercury in eel (0.88 ppm) is greater than the State of Maine Bureau of Health fish advisory level (0.43 ppm wet weight) and the RBC for fish (0.41 ppm), but less than the FDA action level of 1 ppm for organic mercury. This comparison is shown in Table 8-6. Mercury was retained as a COC in fish.

Air

Mercury was detected in 10 of 19 air samples collected by Maine DEP, and is included as a COC. However, this data was judged not to be representative of site conditions. Therefore, air dispersion modeling results were used to represent ambient air concentrations. No other compounds were measured in air. Other COCs were modeled from soil or groundwater. Volatile COCs in groundwater are modeled to ambient or indoor air. Fugitive dust concentrations are estimated for soil COCs.

Table 8-7 is the resulting list of COCs for all media.

8.3 Exposure Assessment

The Exposure Assessment section of the report creates exposure scenarios based on the location and the physical characteristics of the study area. Analysis of land use and accessibility leads to the identification of human populations (receptor groups) that could potentially be exposed to media at or near the HoltraChem property. Analysis of the physical characteristics of the study area leads to the identification of media (e.g. soil, groundwater, surface water, sediment, etc.) to which each receptor group could be exposed. The above information combines to form a complete "exposure pathway."

Identification of exposure pathways is followed by an estimation of contaminant concentrations potentially encountered by the receptor groups according to their specified exposure pathways. These "exposure point concentrations" are calculated for each medium at each study area.

The final step of the Exposure Assessment section combines the exposure point concentrations with assumptions about the human receptors, such as physical characteristics and intensity/frequency of exposure, to estimate average daily chemical intake via each pathway.

8.3.1 Exposure Setting

This section characterizes the exposure setting at and near the HoltraChem property. Analysis of the location, physical characteristics, and accessibility leads to the identification of potential human receptor groups, which are also presented here.

The broadest method of characterizing the study area refers to "on site areas" and "off site areas." On site areas represent the entire HoltraChem property. Off site areas represent a portion of the Penobscot River that is south of the HoltraChem property and hereafter referred to as "the southern cove."

The HoltraChem property (on site area) is approximately 235 acres. Of the 235-acre property, approximately 12 acres represent the on site "plant area." The plant area is a relatively flat plateau, most of which is paved and occupied by the plant buildings and working facilities. The remaining 223 acres of the HoltraChem property represent the on site area "outside the plant area." The area outside the plant area includes the five landfills (open fields) described in Section 8.1.1, but for the most part is heavily wooded. The area outside the plant area also includes a flooded gravel pit to the northeast and stormwater ditches that flow through the property to the Penobscot River.

The southern cove (off site area) is located in the Penobscot River immediately adjacent to the southwest corner of the HoltraChem property. The cove receives water from the southerly stream and northern stormwater ditch that flows from the HoltraChem property and two NPDES outfalls (the Penobscot Energy Recovery Plant outfall and the HoltraChem outfall). The southern cove is adjacent to a residential area and hence is fairly accessible.

Based on the characteristics of the study area, there are five exposure scenarios evaluated in the HRA:

Current On site Facility Workers are assumed to be exposed to contaminants during site walkovers and general maintenance activities. This exposure is assumed to occur on site across the entire property (the plant area and the area outside the plant area).

Current On site Trespassers are assumed to be exposed to contaminants during recreational activities such as hiking, biking, and swimming. The exposure is assumed to occur on site in the area outside the plant area.

Current Off site Residents are assumed to be exposed to contaminants while swimming in the Penobscot River. This exposure is assumed to occur off site in the southern cove. They are also exposed through air inhalation, the consumption of fish (eels) from the Penobscot River, and through ingestion of groundwater as drinking water.

Future On site Facility Workers are assumed to be exposed to contaminants during subsurface excavation activities. This exposure is assumed to occur on site in the plant area only. According to personal communication with plant personnel (W. Miller, 1995), this is the only area where excavations may take place.

Future Hypothetical On site Residents are assumed to be exposed to contaminants during outdoor activities such as yard work or gardening. This exposure is assumed to occur on site across the entire property (the plant area and the area outside the plant area). This exposure could occur through ingestion of homegrown produce or exposure to groundwater used as drinking water. It is assumed that a future resident could live on any portion of the property.

For each of these human receptors, there are two sets of exposure parameters. The first is assumed to represent the central tendency, or average exposure, and the second is assumed to represent a Reasonable Maximum Exposure (RME), designed to be protective of the 95th percentile.

Tables 8-8 and 8-9 provide the exposure profiles considered in this analysis. The tables present exposure pathways, exposure durations, and exposure frequencies for the average and RME cases.

Appendix F-2 provides detailed exposure assumptions, such as physical characteristics for each human receptor and exposure intensity/frequency for each pathway.

8.3.2 Exposure Pathways

The previous section identified the human receptors that are evaluated in this HRA and the portion of the study area to which each is assumed to be exposed. This section develops the exposure pathways for each scenario by addressing the specific media that are accessible at each area of exposure.

8.3.2.1 Air

Receptors may be exposed to potential contaminants via inhalation from vapors originating in groundwater and/or from fugitive dust in addition to vapors released from the cell room as fugitive emissions.

Current On site Facility Workers are assumed to experience site-wide inhalation exposures via fugitive dust during site walkovers and general maintenance activities. Volatile organic compounds (VOCs) have only been detected in one surface soil sample to date. VOCs have been detected in groundwater beneath the property. Vapors originating from VOCs in soils, groundwater, or surface water are assumed to be rapidly diluted at the ground surface by contact with ambient air. Therefore, Current On site Facility Workers are not likely to experience exposure to these vapors. However, they may be exposed to mercury in fugitive dust and vapors in ambient air.

Trespassing Teenage Children are also assumed to experience inhalation exposures due to fugitive dust outside the plant area and mercury in ambient air. Again, vapors potentially originating in soils and water on site are assumed to be rapidly diluted by contact with ambient air.

Current off site residents may be exposed to mercury in vapor form in ambient air.

Future On site Facility Workers who engage in construction or excavation activities are assumed to be exposed to both fugitive dust and vapors in the plant area. Section 8.3.3.3 provides a description of a "trench model" to estimate the exposure to vapors originating from groundwater during subsurface excavation.

Future Hypothetical On site Residents are also assumed to have site-wide exposure to both fugitive dust and vapors. However, this scenario assumes exposure to vapors will occur via indoor air. Section 8.3.4.4 provides the methodology for estimating exposure to indoor air originating from groundwater.

8.3.2.2 Soils

Receptors may be exposed to potential contaminants in surface soils through direct dermal contact, incidental ingestion, and inhalation of fugitive dust. A detailed discussion of exposures to fugitive dust is provided in Section 8.3.4.2.

Current On site Facility Workers are assumed to be exposed to site-wide surface soils as a result of normal plant operations. The property is not entirely paved and contains areas where direct access to soil is likely. Based on available data, soil in the plant area contains the highest concentrations of COCs.

Trespassing Teenage Children may contact on site surface soils outside the plant area. Access to the property is not entirely restricted, potentially attracting trespassers to large undeveloped portions of the property.

Current offsite residents are not likely to be exposed to site soils because access to the site is currently limited.

Future On site Facility Workers may contact surface and subsurface soils during utility line maintenance and/or subsurface stormwater system excavation and repair. These excavations may occur only in the plant area. The protocol document states that excavations are assumed to occur 90 days/year for the average case and 150 days/year for the RME case. This assumption has been updated based on site-specific information that the frequency of excavation is four to five excavations per year for one to two days each (W. Miller, personal communication, 1995). Hence, the exposure assumptions (Appendix F2-4) have been updated to represent actual exposures of 10 days/year.

Future Hypothetical On site Residents are assumed to be exposed to site-wide surface soils in the event that the property is redeveloped for future residential use.

8.3.2.3 Sediments

Current On site Facility Workers could contact sediments through routine maintenance activities and site walkovers. Exposure to sediment is assumed to occur outside the plant area as there are no streams or ponds in the plant area. Dermal contact with sediment is a conservative assumption for the Current On site Facility Worker because they are required to wear protective clothing.

Trespassing Teenage Children may contact on site sediments at several locations outside the plant area:

- in stormwater ditches leading into the cove; and
- in a flooded gravel pit northeast of the plant.

The HRA protocol document (Menzie-Cura & Associates, Inc. and CDM, 1994) identified potential exposure to sediments from the flooded gravel pit in the northeastern corner of the site, and described a scenario in which these exposures were considered separately. This scenario focused on teenagers who may swim in the flooded gravel pit and experience exposures to the gravel pit sediments rather than site-wide sediments. Subsequent monitoring activities indicated that mercury was not detected in either the surface water or the sediments of this flooded gravel pit. Consequently, this scenario is not quantitatively evaluated in this HRA.

Current Off site Residents using the Penobscot River for recreational activities may contact off site sediments in the southern cove. The scenario is divided into three life stages: young children, children, and adults.

Future hypothetical onsite residents are not assumed to be exposed to Penobscot River Sediments because this media is addressed for a current resident.

8.3.2.4 Surface Water

Current On site Facility Workers could contact site-wide surface water through routine maintenance activities and site walkovers. Exposure to surface water is assumed to occur outside the plant area as there are no streams or ponds in the plant area. Dermal contact with surface water is a conservative assumption for the Current On site Facility Worker because they are required to wear protective clothing.

Trespassing Teenage Children may contact on site surface water at several locations outside the plant area:

- in stormwater ditches leading into the cove; and
- in a flooded gravel pit northeast of the plant.

The water in the flooded gravel pit is deep enough that Trespassing Teenagers could potentially swim. At the stormwater ditches, hands, arms, feet and lower legs may be exposed. The surface waters are shallow stormwater channels rather than swimming locations. Current monitoring data

show that mercury was not detected in the flooded gravel pit at the northeast corner of the site; consequently, exposure at this location will not be quantitatively evaluated.

Current Off site Residents who use the Penobscot River are assumed to contact off site surface water during recreational activities, such as swimming or fishing in the southern cove.

Future hypothetical onsite residents are not assumed to be exposed to Penobscot River surface water because this media is addressed for a current resident.

8.3.2.5 Groundwater

Current workers and trespassers are not exposed to groundwater because there is no exposure pathway. Groundwater is not currently used at the site.

Future On site Facility Workers could be exposed to groundwater through the industrial nonpotable use of groundwater from the site. However, the water used at the site is Bangor municipal water, and no use of groundwater is anticipated at the site now or in the future (W. Miller, personal communication, 1995). Hence, this scenario is not quantitatively evaluated in this HRA.

Future On site Facility Workers may be exposed to groundwater during excavation in the plant area. Dermal contact with groundwater and inhalation is included in the scenario, though it is common practice to dewater during the excavation (W. Miller, personal communication, 1995).

Future Hypothetical On site Residents and Future On site Facility Workers are assumed to be exposed to vapors originating in groundwater during site-wide activity (Section 8.3.2.1). Future hypothetical on site residents are also assumed to be exposed to groundwater via drinking water ingestion.

8.3.2.6 Fish

The Penobscot River provides suitable habitat for a number of fish species. However, the transitional nature of the river segment adjacent to the site (brackish, neither fresh nor marine) limits the number of species using this reach of the river. Softshell clams and blue mussels cannot be supported in the salinity range reported for the portion of the Penobscot River adjacent to the site. The one resident species that has been observed and harvested in this portion of the Penobscot River is the American eel.

Current Off site Residents who use the Penobscot River for recreational activities, such as swimming and fishing, could become exposed (via consumption) to contaminants that have bioaccumulated in eel tissue. Eel can bioaccumulate contaminants through a direct water - gill mechanism or, in the case of mercury, through the food chain.

8.3.2.7 Home Grown Produce

Future Hypothetical On site Residents are assumed to grow vegetables in site-wide soils. Section 8.3.4.5 provides the methodology for estimating exposure to contaminants via consumption of home grown vegetables.

8.3.2.8 Game/Wildlife

Small mammals and deer who are resident in the undeveloped portions of the site may bioaccumulate contaminants (mercury) from vegetation ingestion, surface water ingestion and incidental soil ingestion. Bow hunters have been observed trespassing at the site, so it is possible that off site residents may kill and consume deer or other mammals from the site.

8.3.3 Data Used to Calculate Exposure Point Concentrations

The previous section described pathways by which each receptor group is potentially exposed to contaminants at or near the HoltraChem property. This section outlines the specific data that appropriately characterize the different media at each study area. The appropriate data are used in Section 8.3.4 to calculate exposure point concentrations (EPCs).

8.3.3.1 Air

Exposure to elemental mercury in ambient air for the Current On site Facility Workers, Current On site Trespassers, and Current Off site Resident is based on estimated mercury concentrations from air dispersion modeling conducted by Earth Tech with the Industrial Source Complex Short Term (ISCST3) Model. The annual maximum concentration was used as the EPC, which is likely to over predict the actual maximal concentration by a factor of 2 (CDM, personal communication, 1998). The ISCST3 Model provides a conservative estimate of exposure to off site receptors because it does not account for the effects of downwash. Initial off site monitoring data indicate the ISCST3 model is appropriate for offsite receptors. The predicted annual maximum concentration is 30 ng/m³, and the average measured concentration is 12 ng/m³. The 24-hour maximum concentration, the closest to a long term value since data has been collected for less than one year, was 57.3 ng/m³ (personal communication CDM, 1998).

Exposure to fugitive dust for Current On site Facility Workers, Current On site Trespassers, Future On site Facility Workers, and Hypothetical Future On site Residents is based on soil data from the appropriate areas of exposure. Soil data for the various study areas are described below in Section 8.3.3.2, and the process of estimating fugitive dust EPCs is described in Section 8.3.4.2.

Exposure to vapors for Future On site Facility Workers (during excavation) is based on 1997 and 1998 groundwater data from the plant area. Groundwater data are described below in Section 8.3.3.5, and the process of estimating vapor EPCs is described in Section 8.3.4.3.

Exposure to indoor air for Future Hypothetical On site Resident is based on groundwater data for the entire property. Groundwater data are described below in Section 8.3.3.5, and the process of estimating indoor air EPCs is described in Section 8.3.4.4.

8.3.3.2 Soils

This HRA calculates EPCs for on site surface soil using CDM and Acheron data collected from 1990 to 1997. The analytical data and summary statistics for surface soil are presented in Appendix F. For each of the following scenarios, the EPCs for exposure to fugitive dust are estimated concentrations (Section 8.3.4.2) based on the appropriate soil samples.

Exposure to surface soils for Current On site Facility Workers and Future Hypothetical On site Residents incorporates site-wide data, using data from the plant area and data from outside the plant area (Acheron 1991). The data set for mercury in site-wide soils consists of 410 samples. Mercury was detected at a minimum concentration of 0.1 mg/kg, a maximum concentration of 350 mg/kg, and an average concentration of 17 mg/kg. The data set for chloropicrin in site-wide soils consists of 21 samples. Chloropicrin was detected at a minimum concentration of 0.009 mg/kg, a maximum concentration of 0.32 mg/kg, and an average concentration of 0.2 mg/kg.

Exposure to surface soils for Current Trespassing Teenagers incorporates all soil data collected outside the plant area. The data set for mercury in soil outside the plant area consists of 196 samples. Mercury was detected at a minimum concentration of 0.1 mg/kg, a maximum concentration of 350 mg/kg, and an average concentration of 8.7 mg/kg.

Exposure to surface and subsurface soils for the Future On site Facility Worker incorporates soil data collected from only the plant area. In the absence of subsurface soil data within the plant area, the HRA uses measured soil concentrations to a depth of 18 inches. It is conservative to assume that the concentration at 18 inches is the same as at 10 feet because the primary sources of contamination are surface spills and atmospheric deposition. The data set for mercury in soils collected from the plant

area consists of 214 samples. Mercury was detected at a minimum concentration of 0.1 mg/kg, a maximum concentration of 310 mg/kg, and an average concentration of 25 mg/kg. The data set for chloropicrin in these soils consists of 14 samples. Chloropicrin was detected at a minimum concentration of 0.009 mg/kg, a maximum concentration of 0.3 mg/kg, and an average concentration of 0.05 mg/kg.

8.3.3.3 Sediment

The HRA assumes that the Current On site Facility Workers and Current Trespassing Teenagers may encounter sediment at all on site surface water bodies and stormwater areas. Mercury is the only COC in on site sediment and has a data set that consists of 10 samples. Mercury was detected at a minimum concentration of 5 mg/kg, a maximum concentration of 78 mg/kg, and an average concentration of 29 mg/kg.

Current Off site Residents of each life stage are assumed to contact off site sediment while swimming or fishing in the southern cove of the Penobscot River. The scenario does not assume contact with on site sediment and calculates the EPCs for mercury (the only detected compound) using 456 off site sediment samples (CDM). Mercury was detected at a minimum concentration of 0.1 mg/kg, a maximum concentration of 460 mg/kg, and an average concentration of 11 mg/kg.

8.3.3.4 Surface Water

The HRA assumes that the Current On site Facility Workers and Current Trespassing Teenagers may encounter surface water at the on site surface water bodies and stormwater areas. The EPCs for this pathway are calculated based on the site (Acheron and CDM) surface water data. The data set for mercury has a minimum detected concentration of 0.0005 mg/l, a maximum detected concentration of 1.2 mg/l, and an average concentration of 0.049 mg/l. Carbon tetrachloride and chloroform were the only other COCs. Carbon tetrachloride was detected in 4 of 4 on site samples, at a minimum concentration of 0.0035 mg/l, a maximum concentration of 0.005 mg/l, and an average concentration of 0.0043 mg/l. Chloroform was detected in 2 of 4 on site samples, at a concentration of 0.0088 mg/l and an average concentration of 0.0015 mg/l.

Current Off site Residents of each life stage are assumed to contact off site surface water while swimming or fishing in the southern cove of the Penobscot River. The scenario does not assume contact with on site surface water and calculates the EPCs for mercury (the only detected compound) using surface water samples from the river. The data set has a minimum detected concentration of 0.0053 µg/l, a maximum detected concentration of 0.173 µg/l, and an average concentration 0.039 µg/l.

8.3.3.5 Groundwater

Direct exposure to groundwater at the HoltraChem site is highly unlikely for two reasons:

- on site water is City of Bangor municipal water, and
- dewatering occurs during on site excavation activities.

The HRA conservatively assumes limited exposure to on site groundwater during excavation by Future On site Facility Workers. The EPCs for exposure to groundwater are calculated using data from the plant area since this is the only area where excavation activities are reported to occur (W. Miller, personal communication, 1995). The EPCs for exposure to vapors are modeled concentrations (Section 8.3.4.3) that are based on detected concentrations in groundwater from the plant area.

Future Hypothetical On site Residents are assumed to contact site-wide groundwater via direct ingestion as drinking water and through inhalation of vapors in indoor air. The EPCs for drinking water are the average concentration of all groundwater measurements since August 1997 for all COCs. These values are presented in Appendix F. The EPCs for exposure to indoor air are modeled concentrations (Section 8.3.4.4) that are based on detected concentrations in site-wide groundwater.

At Maine DEP's request, off-site residents were assumed to contact groundwater constituents by ingestion from private wells at concentrations measured in well B-321. The unfiltered data from six sampling rounds was averaged to calculate exposure point concentrations. Mercury and manganese are the only COCs in this well. Mercury was detected five times in 18 measurements at a maximum concentration of 5.2 µg/l, a minimum concentration of 0.6 µg/l and an average concentration of 0.8 µg/l. The average concentration includes one half the detection limit for the non-detects. Manganese was detected in 6 of 18 samples at a concentration of maximum concentration of 15 mg/l, a minimum concentration of 0.009 mg/l and an average concentration of 1.53 mg/l, including non-detects.

Appendix F1-3 summarizes the groundwater data sets for site-wide exposures.

8.3.3.6 Fish Tissue

Current Off site Residents who use the Penobscot River for recreational purposes may consume fish. The EPC in fish is calculated using direct field measurements of fish tissues.

The American eel is the target species for this pathway, as described more fully in the Ecological Risk Assessment protocol document, is. The fillets of individual eel samples (collected from the Penobscot River) were analyzed for mercury. Mercury was detected in eel tissue at a minimum concentration of 0.3 mg/kg, a maximum concentration of 0.9 mg/kg, and an average concentration of 0.5 mg/kg.

8.3.3.7 Home Grown Produce

Future Hypothetical On site Residents are assumed to consume home grown vegetables. EPCs for exposure to mercury in vegetables are estimated concentrations based on estimated mercury concentrations in site-wide soils. Section 8.3.4.5 describes the process of estimating EPCs associated with consuming vegetables.

8.3.3.8 Game Tissue

The State of Maine Department of Inland Fisheries and Wildlife stated that a deer hunting permit allows the permit holder to kill one (1) deer per year. They also indicated that there is a possibility people are illegally killing more than one deer per year and potentially consuming additional deer meat from roadkills (personal communication, October 1995). There is some potential risk associated with the ingestion of game that may bioaccumulate mercury. This risk cannot be quantified due to the great degree of uncertainty associated with the uptake of mercury in game tissue. The uptake of mercury in game tissue is a complex process that cannot be reliably modeled due to significant variability in the uptake of mercury in plants, distribution of mercury in plant tissue, absorption of mercury from plants, and distribution of mercury in game tissue and the foraging patterns and areas frequented by the game.

8.3.4 Exposure Point Concentrations (EPCs)

EPCs are calculated based on the approach presented in the EPA Region 1 Risk Updates (EPA, 1994; EPA, 1995). This approach uses the Upper Bound value as the EPC for all media except groundwater. The Upper Bound value is the lesser value of the 95% Upper Confidence Limit (UCL) on the arithmetic mean and the maximum concentration. The Upper Bound value is used in both the central tendency (average) and reasonable maximum exposure scenarios. Average exposure estimates are calculated by combining the Upper Bound value with average exposure parameters; likewise, RME estimates combine the Upper Bound value with high end exposure parameters.

For exposure to soil and sediment, the average EPCs are calculated by adjusting the Upper Bound value by a factor of 0.5. This accounts for the site-specific assumption (Approved Protocol

Document, 1995) that 50% of the soil/sediment is contaminated. The RME EPCs are calculated assuming that 100% of the soil/sediment is contaminated. This conservative approach is recommended by the Maine DEP for a residential scenario (Maine, 1994) and is consistent with an RME.

For exposure to groundwater, the average EPCs are calculated using the average detected concentration. The RME EPCs are calculated using the maximum detected concentration.

For exposure to surface water, both the average EPCs and the RME EPCs are calculated using the Upper Bound value.

Table 8-10 presents the average and RME EPCs for each pathway for each scenario.

8.3.4.1 Calculating the 95% UCL

The 95% UCL of the arithmetic mean is the value that equals or exceeds the true mean 95 percent of the time were a mean to be repeatedly calculated for a random subset of site data. For exposure areas with limited amounts of data or extreme variability in measured or modeled data, the UCL can be greater than the highest measured or modeled concentration. In this case, the highest measured or modeled value can be used as the concentration term.

The first step in calculating the 95% UCL is to evaluate the distribution of the data. Normality tests were conducted on each dataset for each contaminant in each medium using graphical methods. All but the eel data were lognormally distributed. As a result, the data were log-transformed.

After log-transforming the data, the arithmetic mean and standard deviation of the transformed data were calculated. Based on this information and the number of samples, we determined the H-statistic (Gilbert, 1987) and calculated the UCL based on the following equation:

$$UCL = e^{x + 0.5s^2} \frac{sH}{\sqrt{n-1}}$$

Where:

UCL = 95% Upper Confidence Limit

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e	=	constant (base of the natural log)
x	=	mean of the transformed data
s	=	standard deviation of the transformed data
H	=	H-statistic (Gilbert, 1987)
n	=	number of samples

EPCs for all media except groundwater are calculated by estimating the upper bound concentration. The upper bound concentration is the lesser of the 95% Upper Confidence Limit on the arithmetic mean and the maximum detected contaminant concentration. Groundwater EPCs for the average exposure case are the average contaminant concentration, including non-detects at one half the detection limit. For the RME case, groundwater EPCs are the maximum detected contaminant concentrations.

8.3.4.2 Estimating Fugitive Dust EPCs

Although there have been no direct air measurements taken at the HoltraChem property, exposures to fugitive dust via soil can be evaluated using the following equation:

$$EPC_A = C_S \cdot FC \cdot PM_{10} \cdot CF_M$$

where:

EPC_A	=	Exposure point concentration of contaminant in air (mg/m ³)
C_S	=	Concentration of contaminant in soil (mg/kg)
FC	=	Fraction of inhaled soil that is contaminated
PM_{10}	=	Concentration of respirable soil particulate (µg/m ³)
CF_M	=	Conversion factor for mass (10 ⁻⁹ kg/µg)

As the fugitive dust EPCs are based on soil EPCs, the modeled values represent the upper bound concentration for both average exposure and RME estimates.

8.3.4.3 Estimating Vapor EPCs

For current exposure to vapors in ambient air, the maximum average annual mercury concentration from the ISCST3 Model results was used as the EPC.

For Future On site Facility Workers conducting subsurface excavation work are assumed to contact vapors that volatilize from groundwater in the plant area. However, because groundwater migration

is dynamic, we use sitewide groundwater data to estimate vapor EPCs. The EPCs for inhalation of vapors are modeled based on concentrations detected in groundwater. Estimating EPCs for inhalation of vapors follows a three-step process:

1. estimate soil gas concentrations based on contaminant concentrations in groundwater (Lyman *et al.*, 1982);
2. estimate emissions due to excavation using soil gas contaminant concentrations (Johnson *et al.*, 1989); and,
3. estimate ambient air contaminant concentrations using the Near Field Box Model (Pasquill 1975; Horst 1979).

The ambient air contaminant concentrations that result from the third step were used as the EPCs for the inhalation of vapors by Future On site Facility Workers.

The modeling calculations for each of the above steps are provided in Appendix F-3.

Step 1. Estimate Soil Gas Concentrations

Volatilization of contaminants from groundwater is represented by the Henry's Law Constant, a value that is specific to each contaminant and was obtained through a literature search. The Henry's Law Constant, together with the concentration of the contaminant in groundwater, provides a quantitative indication of the soil gas concentration.

The concentration of a contaminant volatilizing from groundwater and diffusing into soil gas are conservatively estimated using the equation presented in Lyman *et al.*, (1982):

$$C_{i,\text{soilgas}} = C_{i,\text{groundwater}} \cdot [\text{HLC}/0.024] \cdot \text{CF}$$

where:

$C_{i,\text{soilgas}}$	=	Concentration of compound I in soil gas (mg/m ³)
$C_{i,\text{groundwater}}$	=	Concentration of compound I in groundwater (mg/l)
HLC	=	Henry's Law Constant (atm-m ³ /mol)
0.024	=	Product of Universal Gas Constant (R) and absolute temperature (T)
CF	=	Units Conversion Factor (1000)

This equation assumes that the gas-phase concentration at the groundwater surface is in equilibrium with the water and acts as a source of VOCs. The concentrations present in soil gas above the contaminated groundwater are estimated assuming the equilibrium conditions of Henry's Law.

Appendix F3-1 presents the estimates of soil gas concentrations from groundwater in the plant area.

Step 2. Estimate Emissions Due to Excavation

The following equation from Johnson *et al.*, (1989) uses the estimated soil gas concentrations from Step 1 to estimate compound emissions due to excavation, *i.e.*, vapor flux (F in g-cm²/day):

$$F = D_i^o C_{ivapor}^{eq} / H$$

where:

- D_i^o = vapor phase molecular diffusion coefficient in air (cm²/day)
 C_{ivapor}^{eq} = vapor phase concentration of chemical I (equilibrium assumed) (g/m³)
H = depth of excavation (213 cm)

The model assumes that there is a non-diminishing source of soil gas and that vapors are swept away by the wind at the ground surface (zero concentration at ground level). These assumptions lead to conservative emission estimates.

Appendix F3-1 presents the estimates of compound emissions due to excavation.

Step 3: Estimate Ambient Air Contaminant Concentrations

The last step in the modeling process involves estimating ambient air contaminant concentrations using the Near Field Box Model (Pasquill 1975; Horst 1979).

The Box Model is conducted using the simple conservation of mass and requires an estimate of an average flux out of the box, since it is assumed that the wind speed increases with height. This model is applicable to exposure scenarios such as the Future On site Facility Worker where the receptor group is on the site (GRI 1987).

The model is based on the following equation which uses the compound emissions from Step 2 to estimate the concentrations in on site ambient air (C_a in mg/m³):

where:

$$C_a = F / (H_b W_b U_m)$$

F	=	emission rate of contaminant (vapor flux) (mg/s)
H _b	=	downwind height of the box (m)
W _b	=	width of the box, crosswind dimension of area of contamination (m)
u _m	=	average wind speed through the box (m/s)

Appendix F-3 presents the estimates of ambient air concentrations for the Future On site Facility worker during excavation in a trench. These ambient air concentrations are used as the EPCs for exposure to vapor.

8.3.4.4 Estimating Indoor Air EPCs

No indoor air samples were collected at the site. Therefore, the EPCs for exposure to contaminants in indoor air for the Future On site Resident are modeled using equations for the volatilization of contaminants from groundwater to indoor air, based on the Johnson and Ettinger model (Environmental Science & Technology, 1991). This model relates groundwater concentrations to indoor air concentrations through the calculation of a volatilization factor. The equations for this model are published in the peer-reviewed American Society for Testing and Materials (ASTM) standard for Risk Based Corrective Action (ASTM, 1995).

The equations for estimating the volatilization of contaminants from groundwater to enclosed spaces (e.g., indoor air) permit the incorporation of both chemical-specific parameters (e.g., Henry's Law constant, diffusivity in air, and diffusivity in water) and site-specific variables (e.g., soil depth, enclosed-space volume/infiltration area ratio, and enclosed-space air exchange rate). We model vapor intrusion from groundwater data using ASTM default model parameters, wherever site-specific information is not available. Appendix F-3 presents the equations, references, and calculations used with the indoor air models. EPCs for indoor air are the product of the volatilization factor and the EPCs for groundwater contaminants.

8.3.4.5 Estimating EPCs for Consumption of Home Grown Produce

The EPCs for mercury in homegrown produce are estimated following USEPA Soil Screening Level guidance (USEPA 1996). This guidance document outlines a methodology for evaluating the soil-plant-human exposure pathway. USEPA has developed Soil to Plant Uptake Response Slopes for Common Contaminants (µg chemical/g dry weight produce per/g chemical/g dry weight soil). The product of the uptake slope and the concentration of mercury in soil is the EPC for mercury in homegrown produce.

$$EPC_{\text{Hg produce}} = EPC_{\text{Hg Soil}} \times UC_{\text{Hg}}$$

where:

$EPC_{\text{Hg produce}}$	=	EPC of mercury in homegrown produce
$EPC_{\text{Hg Soil}}$	=	EPC of mercury in soil
UC_{Hg}	=	Soil to Plant Uptake Response Slope for mercury

We selected an uptake slope for mercury among the following groups of produce: potato, leafy vegetables, legumes, roots, and garden fruits. Of these foods, root vegetables typically have the highest reported uptake slopes. One uptake slope for mercury in root vegetables is 0.017 g soil/g plant (USEPA, 1989). Additional studies (USEPA, 1992; USEPA, 1994) on the uptake of mercury in root vegetables provide a range of 0.001 g/g to 0.043 g/g.

By using an uptake slope for root vegetables to estimate the concentration of mercury in produce, we calculate worst case estimates of exposure to mercury in produce. Concentrations of mercury in leafy vegetables and garden fruits would be less. Potatoes and root vegetables, such as carrots and onions, are hardy in cold climates. These vegetables grow well in Maine and are a dietary source throughout the year. Leafy vegetables and garden fruits have a shorter growing season and are a dietary source for a limited number of months per year.

Calculation of $EPC_{\text{Hg produce}}$

$$EPC_{\text{Hg produce}} = (77 \mu\text{g/g}) (0.017 \text{ g/g}) = 1.31 \mu\text{g/g.}$$

8.3.4.6 Summary of EPCs

EPCs are estimated using the data specified in Section 8.3.3 and methodology outlined in this section (Section 8.3.4). Table 8-10 presents the average case and RME EPCs for each medium and scenario.

8.3.5 Exposure Doses

This section describes the formulas used in calculating Average Daily Doses (ADD) for each of the exposure pathways under consideration.

The medium of exposure for which dose estimates are provided should be similar to the medium of exposure upon which the toxicity value is based. If chemical-specific information on relative

absorption is available (EPA, 1992; EPA, 1989d), this information is incorporated into the estimated ADD for each pathway and scenario.

Exposure doses are calculated for acute effects, noncarcinogenic chronic effects, and carcinogenic effects. The ADD(day) estimates are the dose estimates for acute effects. For chronic noncarcinogenic effects, the ADD(chronic) estimates are the dose estimates. For carcinogenic effects, ADD(life) estimates are the dose estimates. The ADE is the average daily exposure.

Specific exposure frequencies for these scenarios are evaluated as part of the HRA. Below the equations for ADD (day) are shown, then the equations for ADD (chronic) and ADD (life) are presented.

8.3.5.1 ADE(day) for Exposures via Inhalation

$$\text{PotE} = \text{EPC}_A \cdot \text{EF} \cdot \text{ED} \cdot \text{CF}$$

where:

PotE	=	Potential exposure (mg/m ³)
EPC _A	=	Exposure point concentration in air (mg/m ³)
EF	=	Exposure Frequency (hours/event)
ED	=	Exposure Duration (events/day)
CF	=	Conversion Factor (days/hour)

8.3.5.2 ADE(day) for Acute Exposure via Inhalation for Chloropicrin

$$\text{ADD} = \frac{\text{EPC}_A \cdot \text{PP} \cdot \text{VR} \cdot \text{EF} \cdot \text{ED} \cdot \text{CF}}{\text{BW}}$$

where:

ADD	=	Acute daily dose
EPC _A	=	Exposure point concentration in air (mg/m ³)
PP	=	Proportion of Particulate from site (unitless)
VR	=	Ventilation rate (m ³ /hr)
ED	=	Exposure Frequency (hours/event)
EF	=	Exposure Duration (events/day)

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CF = Conversion Factor (days/hour)
 BW = Body weight (kg)

8.3.5.3 ADD(day) for Incidental Ingestion of Soil or Sediment

$$PotD = \frac{EPC_s \cdot (IngR_s \cdot FC) \cdot CF_M}{BW}$$

where:

PotD = Potential dose (mg/(kg-day))
 EPC_s = Exposure point concentration in soil or sediment (mg/kg)
 IngR_s = Ingestion rate for soil or sediment (mg/day)
 FC = Fraction of ingested soil that is contaminated (unitless)
 BW = Body weight (kg)
 CF_M = Conversion factor for mass (10⁻⁶ kg/mg)

8.3.5.4 ADD(day) for Dermal Exposure to Soil or Sediment

$$IntD = \frac{EPC_s \cdot (AR \cdot FC \cdot FA \cdot SA \cdot FE) \cdot CF_M \cdot CF_A}{BW}$$

where:

IntD = Internal dose (mg/(kg-day))
 EPC_s = Exposure point concentration in soil or sediment (mg/kg)
 AR = Adherence rate (of soil or sediment to skin) (mg/cm²/day)
 FC = Fraction of contacted soil/sediment contaminated (unitless)
 FA = Fraction of contaminant absorbed (unitless)
 SA = Skin area (m²)
 FE = Fraction of skin area exposed (unitless)
 BW = Body weight
 CF_M = Conversion factor for mass (10⁻⁶ kg/mg)
 CF_A = Conversion factor for area (10⁴ cm²/m²)

8.3.5.5 *ADD(day) for Incidental Ingestion of Surface Water*

$$PotD = \frac{EPC_{sw} \cdot IngR_{sw} \cdot ED \cdot CF_v \cdot CF_m}{BW}$$

where:

- PotD = Potential dose (mg/(kg/day))
- EPC_{sw} = Exposure point concentration in surface water (µg/l)
- IngR_{sw} = Ingestion rate for surface water (ml/hour)
- ED = Exposure duration (hours/day)
- BW = Body weight (kg)
- CF_v = Conversion factor for volume (10⁻³ l/ml)
- CF_m = Conversion factor for mass (10⁻³ mg/g)

8.3.5.6 *ADD(day) for Incidental Ingestion of Ground Water*

$$PotD = \frac{EPC_{GW} \cdot IngR_{GW}}{BW}$$

where:

- PotD = Potential dose (mg/(kg/day))
- EPC_{GW} = Exposure point concentration in ground water (mg/l)
- Ing_{GW} = Ingestion rate for ground water (l/day)
- BW = Body weight (kg)

8.3.5.7 *ADD(day) for Dermal Exposure to Surface Water*

$$IntD = \frac{EPC_w \cdot (K_p \cdot SA \cdot FE) \cdot ED \cdot CF_M \cdot CF_A}{BW}$$

where:

- IntD = Internal dose (mg/(kg-day))

EPC _w	=	Exposure point concentration in water (µg/l)
K _p	=	Permeability coefficient for transfer from water to skin (l/cm ² /hour)
SA	=	Skin area (m ²)
FE	=	Fraction of skin area exposed
ED	=	Exposure duration (hours/day)
BW	=	Body weight (kg)
CF _M	=	Conversion factor for mass (10 ⁻³ mg/µg)
CF _A	=	Conversion factor for area (10 ⁴ cm ² /m ²)

8.3.5.8 ADD(day) for Ingestion of Foods (Fish, Vegetables, etc.)

$$PotD = \frac{EPC_F \cdot IngR_F \cdot CF_M}{BW}$$

where:

PotD	=	Potential dose (mg/(kg-day))
EPC _F	=	Exposure point concentration in food (µg/g)
IngR _F	=	Ingestion rate for food (g/day)
BW	=	Body weight (kg)
CF _M	=	Conversion factor for mass (10 ⁻³ mg/µg)

8.3.5.9 ADD(chronic)

The ADD(day) is used to calculate the ADD(chronic) which incorporates the exposure frequency to express either a continuous exposure (365 days/year) or an intermittent exposure. The ADD(chronic) is estimated by adjusting the ADD(day) by the exposure frequency and days of exposure per year:

$$ADD_{chronic} = \frac{ADD_{Day} \cdot EF}{YAP}$$

where:

EF	=	Exposure frequency (days/year)
YAP	=	Yearly Averaging Period (days/year) - equivalent to Maine DEP (1994), "Days of exposure per year (DPY)"

If the exposure frequency is 365 days/year, then the ADD(day) is equal to the ADD(chronic). ADD(day) and ADD(chronic) are calculated separately for each life stage.

8.3.5.10 ADD(life)

The ADD(life) is estimated as the weighted average of the chronic ADDs for the three life stages as shown below:

$$ADD(life) = \frac{[EP_{YC} \cdot ADD(chronic)_{YC}] + [EP_{OC} \cdot ADD(chronic)_{OC}] + [EP_A \cdot ADD(chronic)_A]}{AP}$$

where:

ADD(life)	=	Average daily dose averaged over a lifetime (mg/kg/day)
EP _{YC}	=	Exposure duration as a young child (years)
EP _{OC}	=	Exposure duration as an older child (years)
EP _A	=	Exposure duration as an adult (years)
ADD(chronic) _{YC}	=	Chronic ADD as a young child (mg/kg/day)
ADD(chronic) _{OC}	=	Chronic ADD as an older child (mg/kg/day)
ADD(chronic) _A	=	Chronic ADD as an adult (mg/kg/day)
AT	=	Averaging time (years)

Dose estimates for each scenario and for each life stage (if appropriate) containing ADD(day), ADD(chronic) and ADD(life) are presented in Appendix F.

8.4 Dose-Response Assessment

This subsection discusses the toxicity of detected contaminants, and combines information on the potential toxic effects of contaminants with information on exposure (Exposure Assessment) to estimate risks.

We evaluated the potential toxic effects of a compound by evaluating available data that relate its observed toxic effects to doses at which these effects occur. The Dose-Response Assessment considers the following information:

- potential acute, subchronic or chronic non-carcinogenic health effects of compounds; and,

- potential for compounds to cause cancer.

The following toxicity factors are used to evaluate the toxic effects of compounds:

- reference doses or reference concentrations at which effects have not been observed for non-carcinogenic compounds; or
- carcinogenic slope factors for carcinogenic compounds.

Toxicity factors were identified from the following USEPA databases:

- USEPA, Integrated Risk Information System (IRIS); and
- USEPA, Health Effects Assessment Summary Tables (HEAST).

These databases provide toxicity factors for the inhalation and oral routes of exposure. Toxicity factors for oral routes of exposure are adjusted to evaluate the dermal exposure pathway, as discussed in Section 8.4.3. The Superfund Technical Support Center was contacted as appropriate. Additional toxicity factors have been developed for those compounds not listed in IRIS or HEAST (Section 8.4.1.1 and 8.4.1.2).

8.4.1 Toxicity Assessment for Non-Carcinogenic Effects

Contaminants detected at the site may have the potential for effects that are non-carcinogenic in nature. The toxicity factors which express non-carcinogenic effects are reference doses (RfDs) or reference concentrations (RfCs). The non-cancer hazard indexes for oral and dermal routes of exposure are calculated using RfDs. The non-cancer hazard indexes for inhalation routes of exposure are calculated using RfCs.

The assessment of non-carcinogenic effects is complex. There is a broad interaction of time scales with types of effects (acute, subchronic, and chronic). In various risk assessment guidance documents, most of the attention focuses on evaluating the consequences of chronic (long-term) exposure to various compounds. USEPA has focused its efforts at establishing reference doses and reference concentrations for chronic exposures.

The RfD values for contaminants of concern serve as benchmarks for assessing the potential subchronic and chronic non-carcinogenic health effects. They represent "threshold" levels below which no adverse health effects are anticipated to occur due to exposure over a lifetime. Safety

factors (e.g., uncertainty factors and modifying factors) are applied to the supporting data base to ensure that these benchmarks are sufficiently protective.

A reference dose is defined in the USEPA Integrated Risk Information System as an estimate (with uncertainty spanning perhaps an order of magnitude) of a daily exposure to the human population (including sensitive subgroups) that may be without an appreciable risk of deleterious effects during a lifetime. The critical effect refers to the health endpoint upon which the reference dose is based. Uncertainty factors are incorporated in the reference doses as divisors to the dose associated with the critical effect, which is usually a No Observed Adverse Effect Level (NOAEL) or a Lowest Observed Adverse Effect Level (LOAEL). NOAELS or LOAELS may be divided by the following standardized uncertainty factors:

- 10-fold factor for extrapolation from animals to humans;
- 10-fold factor for variability in the human population;
- 10-fold factor for use of a less-than-chronic study;
- 1 to 10-fold factor for extrapolation from a LOAEL.

The use of ten-fold uncertainty factors is traditional. However, there may be situations where data support the application of smaller uncertainty factors. There is on-going research directed at the use of physiologically-based pharmacokinetic models for interspecies extrapolation. However, at this time, no specific guidance can be provided on the use of pharmacokinetic methods for developing more accurate methods of extrapolation from animal to human data.

Modifying factors also contribute as divisors to the NOAEL or LOAEL. The IRIS review group uses a modifying factor based on collective professional judgment to further adjust the reference dose.

The result of applying various multiples of 10 is that for many compounds, the RfD is calculated to be a factor of 100 less than the NOAEL. For other compounds, the resulting RfD is as much as a factor of 1000 less than the NOAEL.

8.4.1.1 Acute Health Effects

Acute effects are those that might occur as a result of short-term exposure (usually at high concentrations) to contaminants of potential concern. "Short-term" is defined as exposure periods of less than one day up to a few days. Such effects could also occur if individuals were exposed on a continual basis to concentrations high enough to cause acute effects.

Acute exposures and associated effects might also occur as a result of the releases of gases or dusts during excavation of contaminated soils or waste deposits. In most cases, acute effects can result from skin or eye contact with "high" levels of vapors, or from direct contact with raw or highly concentrated materials.

We have developed an acute RfC for chloropicrin, the one compound detected on site at concentrations which may pose an acute health risk. A toxicity profile for this compound is presented in Appendix F. We use the RfC in a direct comparison with modeled ambient air concentrations of chloropicrin. We have assumed an acute exposure to chloropicrin to be a single, or short-term, exposure to a relatively high concentration. This assumption is consistent with chloropicrin's low persistence in the local atmosphere.

We based the proposed RfC for chloropicrin (0.5 mg/m^3) on a burning sensation of the nasal passages and inhibition of respiration observed in animal studies. The RfC is based on the RD50 concentration (the concentration of airborne chemical which causes a 50% decrease in respiratory rate). The RD50 for chloropicrin is 8 ppm [53.8 mg/m^3 (Kane, *et al.*, 1979)]. Though we obtained this value from a mouse model, it may be protective for humans as rodents are obligate nasal breathers. The proposed RfC incorporates an uncertainty factor of 100 to account for inter- and intraspecies differences. Thus, the proposed RfC is 0.08 ppm or 0.5 mg/m^3 .

8.4.1.2 Subchronic and Chronic Health Effects

Subchronic and chronic health effects are those that might occur as a result of longer-term exposure. USEPA defines subchronic exposure as up to seven years. Chronic exposure is defined as greater than seven years.

The USEPA has published both inhalation and ingestion RfD values for some but not all chemicals. These RfD values are in units of milligram of chemical per kilogram of body weight per day (mg/kg/day). RfD values were used in the equations to calculate non-carcinogenic hazard indexes. For those chemicals without published RfDs, toxicity values were cross-assigned from a structurally similar compound. Documentation is provided for the assignments of toxicity factors in **Table 8-11**.

USEPA has not developed RfDs for chloropicrin. Since this compound is one of the primary contaminants of concern at the site, we developed oral RfDs and inhalation RfCs for chloropicrin. The chronic oral RfD for chloropicrin used in the HRA is 0.002 mg/kg/day . The chronic inhalation RfCs for chloropicrin are 0.7 mg/m^3 for worker exposures and 0.0016 mg/m^3 for residential exposures.

The proposed oral RfD for chloropicrin is based on a 90-day oral gavage study in Sprague-Dawley rats (Condie *et al.*, 1994). This study identified a NOAEL of 8.0 mg/kg/day chloropicrin in corn oil. The critical effects in this study are reduced weight gain, reduced thymus weight, and inflammation of the forestomach (histopathology). A LOAEL of 32 mg/kg/day was identified.

An uncertainty factor of 5000 is applied to the NOAEL for extrapolating results of the animal study to humans. This uncertainty factor reflects factors of 10 for interspecies extrapolation, 10 for the protection of sensitive individuals, 10 for a less than lifetime exposure of the test animals to the chemical, and 5 for the lack of reproductive and developmental toxicity data.

The proposed inhalation RfCs for chloropicrin are based on threshold effects for eye irritation levels identified by the American Conference of Governmental Industrial Hygienists (ACGIH). A threshold limit of 1 ppm chloropicrin may be a reasonable no effects level when inhaled for a short period of time (ACGIH, 1986). ACGIH incorporates a margin of safety into this threshold level to account for sensitive individuals who may experience repeated exposure to chloropicrin. ACGIH sets the time-weighted average Threshold Limit Value (TLV) for workers chronically exposed to chloropicrin in air in the workplace at 0.1 ppm (0.7 mg/m³). We use 0.7 mg/m³ as the RfC for chloropicrin in the facility worker scenario.

For non-occupational exposures, we derive a RfC by converting the ACGIH TLV from a workplace exposure limit (8 hour duration for 5 days/week) to a continuous exposure limit (24 hour duration for 7 days/week). The continuous exposure limit is furthered lowered to account for sensitive individuals (safety factor of 10) and uncertainty in the database (another factor of 10). The formula to derive the non-occupational RfC is from Williams *et al.* (1995):

$$RfC(mg/m^3) = \frac{TLV}{[(168 \text{ hours/week} / 40 \text{ hours/week})(SF)]}$$

where:

TLV = Threshold Limit Value (0.7 mg/m³ for chloropicrin)
SF = Safety Factors (100)

$$RfC(mg/m^3) = \frac{0.7 \text{ mg/m}^3}{420} = 0.0016 \text{ mg/m}^3$$

For chloropicrin, although there is uncertainty in the proposed inhalation RfC, use of a highly conservative RfC results in conservative estimates of potential risk and probably does not underestimate the potential chronic inhalation risks associated with chloropicrin.

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8.4.2 Toxicity Assessment for Carcinogenic Effects

Carcinogenicity is described in two ways: (1) through the USEPA Weight-of-Evidence classification scheme (A through E), which provides information on the type and quantity of data available; and (2) the Cancer Slope Factor (CSF) in $(\text{mg}/\text{kg}\text{-day})^{-1}$ which provides a quantitative estimate of the carcinogenic potency of the contaminant to humans. The State of Maine requires that carcinogenic risk be estimated for all contaminants of concern classified as Group A, B1, or B2.

The weight of evidence regarding the potential Carcinogenicity of a compound varies as a result of variations in the available test data, adequacy of studies, types of studies, and observed responses. These factors are taken into account by USEPA in assigning weight-of-evidence categories for characterizing carcinogenic compounds. As described in the documentation supporting the IRIS database, chemicals that give rise to cancer and/or gene mutations are generally classified by USEPA as follows:

Group A: Human Carcinogen;

Group B1: Probable Human Carcinogen, limited human data;

Group B2: Probable Human Carcinogen, sufficient evidence in animals and limited evidence or no evidence in humans;

Group C: Possible Human Carcinogen, limited evidence in animals and limited or no evidence in humans;

Group D: Not Classifiable as to Human Carcinogenicity, insufficient tests for carcinogenesis or mutagenesis are available; and,

Group E: Evidence of Non-Carcinogenicity in Humans.

The CSF, as calculated by the USEPA, is usually the 95% statistical upper bound on the slope of the dose-response curve in the low-dose linear portion as estimated by the multistage linearized model. The larger the CSF, the more potent the compound. The USEPA and others estimate CSFs based on the assumption that there are no threshold levels for carcinogenic effects and that the response is linear with doses at low levels (including those dose levels encountered in the environment). Thus, there is always some level of cancer risk at every exposure concentration.

Four of the COCs are classified by USEPA as potentially carcinogenic:

bromoform;
chloroform;
carbon tetrachloride; and
trichloroethylene.

All of these compounds were detected in groundwater. None of the contaminants of concern detected in soil have been classified as carcinogenic by USEPA. Mercury is not carcinogenic. **Table 8-12** provides the weight-of-evidence classification and CSFs for the other contaminants of concern.

USEPA has not evaluated the Carcinogenicity of chloropicrin but available data show that chloropicrin is not carcinogenic. One single-term toxicity study has been conducted (National Cancer Institute, 1978). In this study, Osborne-Mendel rats and B6C3F1 mice were dosed (oral gavage) with chloropicrin. Mortality was significant only in the high dose groups, and was treatment related. In the lower dose groups (*i.e.*, 0, 16, 25, or 40 mg/kg/day), no statistically significant increases in tumor incidence were observed in the mice. In rats, tumors were observed at a higher incidence in control animals than in treated animals.

Two relatively recent studies have examined the mutagenicity of chloropicrin. The studies (at similar doses) concluded that chloropicrin is a direct-acting, weakly mutagenic substance. The systems that were used included the bacterial reversion-assay systems of *S. typhimurium* (TA100, TA98, TA1535, TA1537, and TA1538) and a strain (WP2 her) of *E. coli* (Moriya *et al.*, 1983). In these assays, chloropicrin was a direct-acting weak mutagen only in TA100 and TA98.

In the second study (Garry *et al.*, 1990), lymphocytes were treated with single doses of chloropicrin and were studied for chromosome aberrations and sister chromatic exchanges (SCEs). The authors concluded (without rigorous statistical analyses) that chloropicrin induces SCEs in the absence and presence of a microsomal activating system.

Taken together, the existing studies reveal that there is minimal carcinogenic activity of chloropicrin, although the data are insufficient for USEPA to classify chloropicrin in terms of its carcinogenicity.

8.4.3 Internal Versus Potential Doses

Toxicity values may need to be adjusted depending on the basis of the value in IRIS or HEAST. Adjustments may be necessary to match the exposure estimate with the toxicity value if one is based on an internal dose and the other is based on a potential dose. This is the case for the dermal

pathway, in which the exposure dose is expressed as an internal rather than potential dose. Adjustments may also be necessary for different vehicles of exposure. If USEPA's published toxicity value is based on a potential dose, it is necessary to convert the toxicity value to an internal dose to be in accordance with the estimated ADD. Based on chemical-specific absorption information in IRIS, the following adjustments may be necessary:

- If the published CSF is based on a potential dose, divide the CSF by the absorption efficiency to derive an adjusted internal dose CSF.
- If the published RfD is based on a potential dose, multiply the RfD by the absorption efficiency to derive an adjusted internal dose RfD.

Ingestion and inhalation exposures were calculated as potential doses. If the toxicity value for a contaminant was expressed as a potential dose, then no adjustment was made. Tables 8-11 and 8-12 present the factors used to reflect absorption efficiencies for the potential exposure media/routes of exposure.

8.5 Risk Characterization

The Risk Characterization for the HoltraChem Manufacturing site provides quantitative and qualitative analyses of potential human health risks. This subsection provides quantitative risk estimates by comparing potential exposure doses to acceptable dose levels (noncancer health hazards) or by multiplying potential exposure doses by estimates of potency (cancer risks).

We quantified risks for five exposure scenarios: facility workers (site walkovers), trespassers, and off site residents under current land use; and facility workers (excavation) and hypothetical on site residents under future land use. We make qualitative statements of potential risk for the exposure pathway that considers a current off site residents' potential exposure to contaminants in game via ingestion.

8.5.1 Method

Risks are estimated for three time frames: acute, chronic, and lifetime. Noncancer health hazards are estimated for the five exposure scenarios evaluated in the HRA. Acute inhalation risks are evaluated for future facility workers (excavation) potentially exposed to vapors in a trench. Subchronic doses were not evaluated because the risk estimates will be equal to those for the chronic dose. Chronic

doses are used to estimate noncancer health hazards. Lifetime doses are used to estimate cancer risks.

The noncancer health hazard or Hazard Index (HI) is calculated by dividing the ADD(chronic) by the chronic RfD, respectively. The Hazard Index is estimated separately for each scenario and exposed population. Tables in Appendix F4 summarize noncancer hazard indices by exposure scenario and present the following information:

- chemical-specific HIs in each exposure pathway;
- summed HIs across exposure pathways for each exposure medium; and
- Total HI for the scenario calculated as the sum of HIs for all exposure media in the scenario.

The Incremental Lifetime Cancer Risk (ILCR) is defined as the incremental probability of an individual developing cancer over a lifetime as a result of exposure to a contaminant. ILCRs are estimated separately for each scenario and each exposed population. Tables in Appendix F4 summarize cancer risk estimates by exposure scenario and present the following information:

- chemical-specific ILCRs in each exposure pathway;
- summed ILCRs across exposure pathways for each exposure medium;
- Total ILCR for the scenario calculated as the sum of ILCRs for all exposure media in the scenario.

Risks are estimated for two exposure cases: average and RME. Section 8.3 described the exposure assumptions used to construct the average and RME cases. The methodology for quantifying risk is the same for both cases.

8.5.1.1 Target Risk Levels

Target risk levels are regulatory goals set by USEPA and State of Maine. The target risk level for noncancer health hazards is a Total Hazard Index of less than one (1). The target risk level for cancer risk is a Total ILCR of less than 1×10^{-5} (one in 100,000). The State of Maine has set 1×10^{-5} as the upper bound for an acceptable Total ILCR for all media including both residential and industrial/commercial scenarios. USEPA generally considers a range of 10^{-4} to 10^{-6} (one in 10,000 to one in 1,000,000) as an acceptable risk range.

8.5.2 Results

Total cancer risk estimates were either not calculated (i.e. the scenario did not involve exposure to carcinogenic compounds) or less than the State of Maine target level of 1×10^{-5} for four of five scenarios evaluated: current on site facility workers engaging in sitewide walkovers; current on site trespassers; current off site residents; and future on site facility workers engaging in excavation or construction. Total cancer risk exceeded the State of Maine target level for the future hypothetical on site residents. The noncancer hazard indexes was greater than target index of one for the current off site receptor scenario and less than the target index for current on site receptors. The total noncancer hazard index for the future hypothetical on site resident scenario was greater than one. The following subsections present the quantitative and qualitative risks for the five exposure scenarios evaluated in the HRA. **Table 8-13** presents a summary of the quantitative risk estimates. Risk calculations are shown in Appendix F4.

8.5.2.1 Facility Workers (Site Walkovers)

Noncancer risk estimates for both the average and RME cases for the current on site facility worker scenario are below target risk levels. The average and RME Hazard Indexes are 0.18 and 0.53, respectively. The average and RME Cancer risks were below target risk levels, 9×10^{-8} and 1×10^{-6} respectively. Current facility workers traversing the entire site are assumed to contact contaminants of concern detected in: (1) on site surface soils through direct contact (ingestion and dermal absorption) and inhalation of dust; (2) on site surface water through dermal contact; (3) on site sediment through dermal contact and incidental ingestion; and (4) inhalation of ambient air. No exposure is associated with a significant risk for the RME exposure case.

8.5.2.2 Trespassers

Noncancer risk estimates for both the average and RME cases for the trespasser scenario are below target risk levels. The average and RME Hazard Indexes are 0.12 and 0.36, respectively. The average and RME Cancer risk were 2×10^{-8} and 2×10^{-7} , below the target risk level. Trespassers are assumed to contact contaminants of concern detected in: (1) on site surface soils through direct contact (ingestion and dermal absorption) and inhalation of dust; (2) on site surface water through dermal contact; (3) on site sediment through dermal contact and incidental ingestion; and (4) inhalation of ambient air. No exposure is associated with a significant risk.

8.5.2.3 Off site Residents

The total noncancer hazard indexes for both the average and RME cases for the current off site resident scenario are greater than the target risk level of one. The average and RME Hazard Indexes are 2.7 and 24, respectively. Cancer risks were not calculated for this exposure scenario because carcinogenic compounds were not detected in the exposure media for this potential receptor group. Current off site residents are assumed to inhale mercury in ambient air, ingest mercury in groundwater and eel captured from the Penobscot River, and to be exposed to contaminants in surface water and sediment in the river during recreation. Risks to off site residents who are assumed to eat game from the site have been evaluated qualitatively. Exposure to mercury and manganese in groundwater were the primary contributors to the exceedence of the target risk level.

Currently, there are off site residents near the site who have private water wells. Recent sampling and analysis of drinking water from these residences did not detect mercury in the water supplies (CDM, September 1998). If contaminants in groundwater migrated off site and there was a hydrogeological connection between site groundwater and off site water wells, there could be a potential exposure pathway for these residents. Concentrations up to 5.2 µg/l of unfiltered mercury were measured in groundwater samples collected from the monitoring well B-321 located near the southern property line. These measured concentrations are greater than the MCL for mercury (2 µg/l). Manganese was measured at concentrations up to 15 mg/l in well B-321, and up to 18 mg/l in resident wells. There is no MCL for manganese, but the SMCL is 0.05 mg/l. The elevated levels of manganese appear to be unrelated to the site, because the manganese was also measured at 18 mg/l in Norlen's well, which is hydrologically upgradient.

Mercury was not detected in filtered samples from this monitoring well, nor was mercury detected in an unfiltered sample collected by low flow purge and sample techniques. The difference between the filtered and non-filtered analytical results indicate the mercury is not in a soluble form at this location. The low flow purge and sample results are considered most representative of actual groundwater conditions. Groundwater contours and water level elevation measurements made at B-321-B2 and the nearest off site residence indicate groundwater in the vicinity of B-321-B2 is discharging to the Penobscot River. However, if an exposure pathway exists, exceedence of the drinking water standard for mercury in monitoring well B-321-B2 would indicate a potential for future risk associated with ingestion of groundwater from off site water wells. The presence of manganese at the detected concentration in B-321 also indicates the potential for risk to the off site resident. However, manganese was detected at similar concentrations in residential wells, and in the Norlen's background well.

Ingestion of mercury in game (e.g., deer) captured near the site is considered a potential exposure pathway. Deer are herbivorous and uptake of mercury by plants has been documented (Lindbergh *et al.*, 1994). Mercury measured in plant leaves comes from two primary sources: soil uptake and atmospheric deposition. The uptake of mercury by deer is less well documented. Assuming that deer and other game bioaccumulate mercury, nearby residents who eat game could be exposed to mercury in the meat. The State of Maine Department of Inland Fisheries and Wildlife states that a deer hunting permit allows the permit holder to kill one (1) deer per year but indicated that there is a possibility that people are illegally killing more than one deer per year and that others may consume additional deer meat from roadkills (personal communication, October 1995). There is some potential risk associated with ingestion of game that may bioaccumulate mercury. This risk cannot be quantified due to the great degree of uncertainty associated with the uptake of mercury in game tissue. The site is posted with no trespassing and no hunting signs.

8.5.2.4 Facility Workers (Excavation)

Risk estimates for both the average and RME cases for the future on site facility worker scenario are less than target risk levels for chronic, and lifetime (cancer) exposures. The average and RME chronic Hazard Indexes are 0.0071 and 0.061, respectively. The average and RME Cancer risk estimates are 3×10^{-13} and 3×10^{-13} , respectively. The site does not pose a significant risk to future workers engaged in excavation on site, as defined in the exposure scenario for this potential receptor group. Facility workers involved in excavations in the plant area are assumed to contact contaminants of concern detected in: (1) soil through direct contact (ingestion and dermal absorption) and inhalation of dust; (2) groundwater through dermal contact; and (3) vapors in ambient air from contaminants volatilizing from soil gas.

Acute risk associated with exposure to chloropicrin vapors in a trench are evaluated by comparing the modeled concentration of chloropicrin to an acute reference concentration. We consider chloropicrin to be the only contaminant detected on site that could be acutely toxic via inhalation based on its volatility, toxicity and the concentrations at which it was detected. The modeled concentrations of chloropicrin in the ambient air of a trench is 0.009 mg/m^3 based on the 95% Upper Confidence Limit of the mean value. These modeled concentrations are less than the acute reference concentration of 0.5 mg/m^3 . The presence of elevated concentrations of chloropicrin on site do not pose an acute risk to facility workers engaged in excavations as described in this exposure scenario.

8.5.2.5 Hypothetical On site Residents

Although the current and anticipated future use of the site is industrial, a hypothetical on site residential scenario was evaluated. This receptor group is assumed to live on a redeveloped portion

of the site for up to 30 years. Three age groups, including sensitive receptors, are included in the scenario: young children up to six years of age; older children age 6-18; and adults greater than 18 years of age. Future on site residents are assumed to contact contaminants of concern in surface soils through direct contact (ingestion and dermal absorption), inhalation of dust, and ingestion of homegrown produce. Future residents also contact groundwater through ingestion and inhalation of vapors in indoor air.

The average Hazard Indices range from 48 to 120, depending on the age of the receptor. For the RME, the Hazard Index ranges from 960 to 2,400. The average and RME Cancer risk estimates are 3×10^{-5} and 4×10^{-4} , respectively. The Hazard Indexes are greater than the target risk level of 1 for noncancer health hazards. The Cancer risk estimates exceed the target risk level of 1×10^{-5} .

The greatest potential risk for this scenario is associated with the ingestion of mercury in drinking water (HI = 110, average; and HI = 2,380, RME). Other than ingestion of groundwater, the pathways associated with the greatest risk for the RME case is ingestion of plants (HI = 12).

This scenario does not quantify potential exposures to surface water and sediment in the Penobscot River. This potential exposure is addressed in the current off site resident scenario. The estimated risks for current off site residents exposed to surface water and sediment in the Penobscot River are less than target levels and would also be less than target levels for future on site residents because of the similarity of these receptor groups.

This scenario does not quantify potential exposures to surface water and sediment on site. This potential exposure is addressed in the current trespasser scenario. The estimated risks for current trespassers exposed to surface water and sediment on site are less than target levels and would probably be the same for future on site residents because of the similarity of these receptor groups. The most likely trespassers are older children, age 6-18.

This scenario also does not quantify the risk associated with dermal contact with groundwater. Groundwater on site is not used for drinking water. If there was a complete exposure pathway, *i.e.*, groundwater on site was used for drinking water, the detected concentrations of contaminants above the MCL (carbon tetrachloride, chloroform, and trichloroethene) would indicate a potential health risk. The risks would be greater if the dermal pathway were included. The following contaminants in groundwater are associated with significant risk for a future residential scenario: acetone, carbon tetrachloride, chloroform (also via inhalation), chloropicrin, hexachloroethane, manganese, and mercury.

8.6 Uncertainty Analysis

All risk assessments are subject to uncertainty because of limitations in the sciences upon which risk assessment draws, including the fields of analytical chemistry, biology, medicine, and toxicology. To balance this uncertainty, this risk assessment relies on conservative estimates of potential exposure and risk. Both average exposure and RME cases were evaluated using upper bound contaminant concentrations and health protective exposure assumptions. It is our opinion that the risk estimates presented in this HRA do not underestimate potential risk for the exposure pathways and scenarios evaluated.

We have not quantified the uncertainty associated with the risk estimates generated in the HRA. Uncertainty is qualitatively characterized as low, medium, or high, based on professional judgment.

8.6.1 Uncertainty in the Data Evaluation and Hazard Identification

There are uncertainties associated with the analytical data. The analytical methods used to determine the contaminant concentrations in environmental media are in compliance with strict USEPA protocols. However, uncertainty is introduced into the analysis in using these data, particularly with regard to the treatment of non-detects in the risk assessment. A conservative approach was taken in assigning non-detects a value equal to one-half the sample detection limit. This is done to capture those compounds that may have been reported as non-detect due to 1) matrix interferences that may have increased the detection limit in the sample, and 2) the presence of compounds at concentrations below the practical quantitation limit.

The focus of analytical activities under the current investigation had been, to some extent, determined by previous investigations and discussions with regulators. Consequently, chemical analyses focused on certain contaminants. This introduces uncertainty when considering the full range of contaminants that may be present at the site. However, the installation of additional monitoring wells and subsequent and repeat testing of those wells decreases the likelihood that significant sources of contamination are present and have not been identified.

Uncertainty associated with the selection of COCs is considered low, given that current activities have been guided by previous site investigations and regulatory oversight.

8.6.2 Uncertainty in the Exposure Assessment

Uncertainty in the Exposure Assessment is related to quantifying potential exposures. Actual measurements of exposure to contaminants at hazardous waste sites have not been made. We instead rely on estimates of exposure, which are based on assumptions about the exposed population such as the characteristics of the receptor group, the frequency and intensity of exposure, and the concentrations to which they may be exposed. Exposure assumptions contain uncertainties, as they are estimates of human activities that may vary. The exposure assumptions used in this analysis are health protective in that they are far more likely to overestimate than to underestimate potential exposures.

Modeling of exposure point concentrations introduces uncertainty into the analysis as models are based on assumptions about a chemical's behavior in the environment, site conditions that influence the fate and transport of contaminants, and the representativeness of the analytical data used in the models. This uncertainty is addressed in part by using upper bound contaminant concentrations in the models.

We do not quantify the uncertainty associated with the exposure point concentrations for individual media, particularly if a significant number of samples is not available. A significant number is defined as 20 samples (USEPA, 1992). Qualitative descriptions of uncertainty in the exposure point concentrations are provided for the exposure media evaluated in the HRA:

On site Soils: The surface soil dataset contains a significant number of samples. Surface soils within the plant area are well characterized. The 95% UCL on the mean for surface soil samples within the plant area is almost three times the average concentration, indicating high variability in the data. This is attributable to the fact that sampling was targeted to known spill areas.

The EPCs for surface soils were used to evaluate subsurface exposures in the On site Facility Worker scenario because the sources of contamination are surface spills and atmospheric deposition. The uncertainty associated with estimating potential exposures for the On site Facility Worker scenario is medium, but is likely balanced by the presumed overestimate in the potential risk derived from conservatively evaluated exposures.

On site Sediments: The on site sediment dataset provides individual samples for each distinct sampling location, and provides a limited number of samples with which to characterize the extent of contamination. There is uncertainty associated with on site sediment exposure point concentrations because of the limited number of on site sediment samples collected and analyzed. Uncertainty in the on site sediment dataset is balanced by three factors: (1) there is good agreement between Acheron

and CDM results for mercury; (2) data are from the most heavily contaminated areas; and (3) the uncertainty associated with the concentrations measured at locations of potential exposure is low.

On site Surface Water: The on site surface water dataset contains 68 samples for mercury but only 4 samples for VOCs. There is hence uncertainty associated with exposure point concentrations for on site surface water due to the limited number of samples used to characterize exposures. The volatile nature of the VOCs would influence the fate of these compounds in surface water.

Southern Cove Sediment: The river sediment dataset contains a significant number of samples for mercury, but not for VOCs (8 samples). Although only a limited number of sediment samples were analyzed for VOCs, we do not expect to detect these compounds in sediment because they tend to volatilize.

Southern Cove Surface Water: The river surface water dataset contains seventy five samples for mercury and nine for volatile organic compounds. There is hence uncertainty associated with exposure point concentrations for river surface water due to the limited number of samples used to characterize exposures. Again, the volatile nature of the VOCs would influence the fate of these compounds in surface water.

On site Groundwater: The groundwater dataset contains a significant number of samples with which to assess the extent of contamination. Spatial heterogeneity of sampling locations is also sufficient.

Biota - Eel: There are 15 eel samples. There is little uncertainty associated with exposure point concentrations for eel because the detected concentrations are consistent with measured mercury concentrations in eel and other fish collected from other locations in Maine.

Biota - Game Tissue: Game tissue samples were not collected. Thus, risks associated with contaminant concentrations in game tissue are not quantitatively evaluated.

Ambient Air: The ambient air data is based on data gathered from three locations over a period of five days in a single week. There is uncertainty in the representativeness of this data for air exposures, particularly those occurring off-site.

Homegrown Produce: Risks were estimated for this exposure pathway for future hypothetical residents who may grow vegetables on site. Samples of homegrown produce were not collected and analyzed for contaminants. Exposure to mercury in homegrown produce was conservatively estimated using a worst case ingestion rate (*i.e.*, 200 mg/day vegetable consumption x 40% homegrown fraction = 80 mg/day) and high uptake factor (0.017 µg/g for root vegetables).

According to USEPA (1991), a proportion of homegrown produce set at 40% is a worst case assumption. Additional information on consumption patterns in Maine could be used to refine the risk estimates. The ingestion rate and uptake slope used may overestimate potential risks associated with this pathway.

8.6.3 Uncertainty in the Dose-Response Assessment

The most difficult uncertainty to control is that inherent in the Dose-Response Assessment. All risk assessments rely on toxicity factors developed by USEPA using numerous assumptions to estimate reference doses and cancer potency factors for contaminants. A significant uncertainty in this area is the extrapolation of effects data in animal studies to humans. Specific to this HRA, toxicity factors were developed for chloropicrin and are subject to the same uncertainties as the toxicity factors developed by USEPA.

8.6.4 Uncertainty in the Risk Characterization

The Risk Characterization is also subject to uncertainty. The risk estimates provided in this report are point estimates. The actual risks are a range of values based on varying levels of exposure in the population. Given the conservative nature of the analysis performed, it is our opinion that the risk estimates presented in this HRA do not underestimate potential risk for the exposure pathways and scenarios evaluated.

Recently, a panel of risk assessment experts participated in a review of a site specific Reference Dose for mercury in fish derived by ICF Kaiser, Inc. for Lacava Bay in Texas, convened by the Toxicology Excellence for Risk Assessment (TERA) organization. The panel generally agreed with the derived reference dose with noted exceptions. The reference was considered site specific because it is based on data from the Seychelles Islands where fish ingestion was the major source of mercury exposure for the studies population. The derived RfD was based on the estimated benchmark doses in maternal hair associated with adverse neurological effects in 29 month old children. A range of RfDs estimated by incorporating the maternal pharmacokinetics of the estimated dose using Monte Carlo simulation into ranged from 0.3 to 1 $\mu\text{g}/\text{kg}/\text{day}$. The recommended value is 0.4 $\mu\text{g}/\text{kg}/\text{day}$, slightly greater than EPA's published value of 0.3 $\mu\text{g}/\text{kg}/\text{day}$. We estimated the risk to an off site resident child from ingesting eels using this reference dose, and the resulting Hazard Index was 0.096 versus 0.13 using EPA's reference dose.

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Section 9

Ecological Risk Assessment

9.1 Introduction

This section is an ecological risk assessment for the HoltraChem Manufacturing site in Orrington, Maine (the site). The purpose of ecological risk assessment is to provide a systematic, and reasonably conservative, weight-of-evidence approach to estimate potential risk to site-specific ecological receptors representing several habitats on and near the site. The sources of the potential risk are chemicals and activities associated with current and historical plant operations.

In ecological risk assessment, the underlying assumption is that the estimate of risk to several carefully chosen receptors represents risk to the ecosystem in question. There can be a significant degree of uncertainty associated with each assumption made in the estimate of risk, and this risk assessment presents and discusses the sources and levels of uncertainty.

In accordance with USEPA guidance, this ecological risk assessment has been conducted using a "weight-of-evidence" approach which includes direct field observations, selected field and laboratory studies from the scientific literature, food chain exposure modeling and effects assessment, site chemical and toxicity data, and evaluation of chemical analytical data relative to environmental benchmarks.

9.1.1 Objective

The objective of this assessment is to provide a qualitative and quantitative description of the potential sources of risk to ecological components associated with the contaminants of concern (COCs) at these areas of concern:

- The Penobscot River offshore of the site;
- The tidal cove adjacent to the plant (the southern cove);
- The onsite stormwater ditches and streams;
- The onsite terrestrial environments.

9.1.2 Regulatory Guidance

This ecological risk assessment follows recent USEPA guidance for assessment of ecological risk at contaminated sites. The specific guidance documents include:

- USEPA, 1992. Framework for Ecological Risk Assessment. USEPA Risk Assessment Forum. EPA/630/R-92/001. February 1992;
- USEPA, 1993. A Review of Ecological Assessment Case Studies from a Risk Assessment Perspective. USEPA Risk Assessment Forum. EPA/630/R-92/005. May 1993;
- USEPA, 1989. Risk Assessment Guidance for Superfund, Volume II, Environmental Evaluation Manual. USEPA Office of Emergency and Remedial Response. EPA/540/1-89/001. March 1989;
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- USEPA, 1992. Developing A Scope of Work for Ecological Assessments. USEPA Office of Solid Waste and Emergency Response. ECO Update Publication 9345.0-05I. May 1992;
- USEPA, 1992. Briefing the BTAG: Initial Description of Setting, History, and Ecology of a Site. USEPA Office of Solid Waste and Emergency Response. ECO Update Publication 9345.0-05I. May 1992;
- USEPA Region I, 1991. Supplemental Risk Assessment Guidance for the Superfund Program.

9.1.3 Organization of the Risk Assessment

This section follows the general framework for ecological risk assessment, as suggested in USEPA guidance. The assessment subsections include: Background and Site Description, Problem Formulation, Exposure Assessment, Ecological Effects Assessment, and Risk Characterization.

Figure 9-1 (USEPA, 1991) shows the four major elements of ecological risk assessment (problem formulation, exposure assessment, ecological effects assessment, and risk characterization) and the general contents of each component. USEPA (1991) recognizes that each element in the process can affect others and that the process is iterative as suggested by the bidirectional arrows in the figure.

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Problem Formulation includes identification of contaminants of concern, identification of assessment and measurement endpoints, selection and characterization of receptors, and identification of exposure pathways. This assessment elaborates upon the site description provided in Section 2 and the hydrologic and hydrogeologic characteristics of the site provided in Section 5. Within the problem formulation, we have extended the site conceptual model to include contaminated media of concern for ecological exposure, exposure pathways to receptor classes, and biota as potential exposure media.

Exposure Assessment estimates the magnitude of actual and/or potential ecological exposure to a contaminant of concern. This subsection describes the method used to quantify the potential exposure to contaminants from each medium of concern. For the ecological receptors, this includes specifying species which occur at a particular area or habitat, and the likely exposure routes which include contact with sediments or water, or exposure through ingestion of contaminated prey. Within this subsection, we provide the methods and results of biological field collections used to measure or estimate exposure point concentrations. Section 3 describes the measurement of potential exposure point concentrations in physical media.

Ecological Effects Assessment summarizes and weighs available evidence regarding the potential for contaminants to cause adverse effects in exposed individuals and provides, where possible, an estimate of the relationship between the extent of exposure to a contaminant and the increased likelihood and/or severity of adverse effects. This subsection describes the potential effects associated with:

Concentrations of the contaminants of concern in environmental media (water, sediment, or food) to which biota may be exposed; and

Ingested doses of a chemical contaminant through exposure to water, sediment, or food.

This section also describes the methods and results of site-specific field and laboratory studies conducted as part of this assessment to estimate ecological effects. These include toxicity tests and benthic community surveys.

Risk Characterization summarizes and integrates the exposure assessment and ecological effects assessment into a quantitative and qualitative expression of risk. The quantitative estimates rely on the toxicity quotient method. Qualitative estimates of risk include our interpretation of the results of habitat surveys, rapid bioassessment protocol, and benthic surveys. This subsection also distinguishes between risk associated with exposure to the site and risk reflected in local background conditions. It provides an evaluation of risk to receptors in the Penobscot River, the onsite streams

and stormwater ditches, and the onsite terrestrial environment. The risk characterization uses weight-of-evidence based on:

- Established water quality or other criteria;
- Environmental contaminant concentrations
- Food chain modeling;
- Toxicity test results;
- Literature values of toxicity;
- Field surveys of receptor populations.

The risk characterization also addresses uncertainty in the analysis of ecological risk. This subsection includes identification and evaluation of sources of uncertainty in various aspects of the analysis, including model applications and the use and applicability of toxicity factors.

9.1.4 Summary of Results

This assessment addressed the potential for ecological risk due to site-related contaminants to selected ecological receptors in:

- The Penobscot River offshore of the site;
- The southern cove adjacent to the plant;
- The onsite streams and stormwater ditches;
- The onsite terrestrial environments.

The results of this ecological risk assessment indicate that the HoltraChem site presents little risk to biota in the Penobscot River from exposure to the contaminants of concern, primarily mercury. Food chain modeling using dietary doses indicated little risk to piscivorous birds and mink. Mercury concentrations in the water column of the main body of the Penobscot River upstream of the site and in the southern cove are less than current ambient water quality criteria indicating protection of local aquatic populations. The abundance and diversity of the local benthic community near the site is similar to offsite areas and the historically observed benthos in this reach of the Penobscot River, indicating no impairment from site-related influences relative to regional conditions.

Sediment toxicity testing in 1995, 1997, and 1998 suggests little potential for risk in the southern cove. Sediment toxicity testing indicated that the site sediments are not toxic to a freshwater

amphipod using 10-day tests. The benthic populations in the cove were more abundant than populations observed elsewhere in this reach of the river. Eels caught in the vicinity of the southern cove had body burdens of mercury similar to fish sampled from remote water bodies in the State of Maine, as well as in rivers that the state considers representative of background (Mower, personal communication, 1995). This also indicates no exposure above regional conditions. Metals availability based on SEM/AVS ratios indicate that metals are available in the inorganic mat/reed bed, sand and gravel sediments, and subtidal areas within the cove which occur less than one third of the total area of the cove. The SEM/AVS values for the mid-intertidal/sedge bed and the lower intertidal/fine silt, which cover greater than two thirds of the southern cove, were less than one indicating the mercury in these sediments is not available. Speciation analysis demonstrated that more than 99% of the sediment mercury is tightly bound to sediments, thus minimizing bioavailability.

There is potential for risk to aquatic biota in the onsite southerly stream and northern stormwater ditch that flow to the southern cove. We base this opinion on the observations that there appears to be a reduced macroinvertebrate community relative to the reference stream, indicating potential for impairment relative to regional conditions due to site-related influences. There are elevated conductivities (i.e. high concentrations of non-specific ions or salts) atypical of freshwater streams in the stream and ditch which could impair the macroinvertebrate community. Mercury concentrations in the stream and ditch surface water columns exceed the water quality criterion.

A potential risk exists for individual organisms that feed directly on soil invertebrates in the terrestrial habitat portion of the site. No population level risks were identified to songbirds and small mammals based on U.S. Fish and Wildlife Service benchmark doses of mercury protective of reproduction. However, there may be effects to small mammals and songbirds using lower effect levels. Food chain modeling did not indicate potential risks to the red-tailed hawk. The potential for risk to songbirds and small mammals onsite does not appear to result in decreased abundance or diversity of wildlife relative to a reference area based on qualitative observations made in these areas,

9.2 Background and Description

This subsection describes the natural resources of the region, and provides an ecological description of the site, adjacent habitat, and several background areas. We conducted a literature review and performed several site-specific surveys to provide these ecological descriptions (Survey methods are provided in Appendix F).

As suggested in USEPA (1992), the ecological description includes: a list of the habitats associated with the site; a description of the physical and biological characteristics of these habitats; observations of the occurrence of plants and animals in the area including migratory species, Federal or State listed endangered or threatened species, and species or habitats of special status.

Section 5 provides a description of the geology, topography, and general hydrology of the site and surrounding area. **Figure 9-2** shows the general habitat types on site and in the surrounding areas addressed in this subsection. These habitats include:

- the deep, steep-sided freshwater tidal run of the main stem of the Penobscot River which runs along the northern and western boundary of the site;
- a freshwater, tidal marsh in an intertidal cove of the Penobscot River adjacent to the southern end of the site, and a freshwater intertidal cove at the northern end of the site;
- one onsite stream, the southerly stream and northern stormwater ditch, an onsite flooded gravel pit, and associated wetlands;
- mixed deciduous pine woods, hemlock woods, shrublands, and fields in the terrestrial portions of the site.

9.2.1 Natural Resources of the Region

This subsection summarizes the biological resources of the Penobscot River region based on reviews of wildlife (Pegraff and Rudis, 1987) and fishery resources (Baum, 1983). The region in and around the HoltraChem plant supports a diverse variety of wildlife, aquatic life, and fish species. In the Orrington/Penobscot River area there occur approximately thirty-nine species of mammals. At least one hundred and twenty-three species of water birds, breeding birds, and summer resident birds occur in the region. Over one hundred and seventy bird species have been identified in the region throughout the year. There are approximately twelve amphibian and seven reptile species which occur along this section of the Penobscot. The Penobscot River and its tributaries in the Orrington area support approximately thirty-nine species of fish. However, the run of the Penobscot River is near the HoltraChem file a transitional area in terms of salinity regime, and supports few resident species.

The Penobscot River supports an anadromous Atlantic salmon (*Salmo salar*) run, and intensive federal/state efforts are being made to restore the salmon fishery on the river (Baum, 1983). The Penobscot salmon population is under federal review to assess its status relative to other potentially

endangered or threatened salmon runs in Maine (Nickerson, personal communication, 1995). The river also supports a number of other anadromous fish species, including herring (*Alosa spp.*), sea-run trout (*Salvelinus fontinalis*), striped bass (*Morone saxatilis*), white perch (*Morone americana*), American shad (*Alosa sapidissima*), sturgeon (*Acipenser spp.*), and a catadromous population of American eel (*Anguilla rostrata*¹)

9.2.2 The Penobscot River

This subsection describes the general physical, chemical, and biological conditions in the reach of the Penobscot River adjacent to the site. It depends upon historical data as well as data recently collected as part of this ecological risk assessment. The methods used during the current site studies to obtain data on the hydrography, sediment conditions, benthic biota, fish species, and wildlife in the Penobscot River are provided in Appendix G.

The Penobscot River runs approximately north to south along the northern and western border of the site as a deep run (Figure 9-3). At the northern and southern ends of the site, the river forms two intertidal flats. The southern flat includes a freshwater tidal marsh. Otherwise, the banks along the property boundary are steep-sided ledges falling nearly vertically into the river. Channel depths near the site approach 30 feet at mean low water. The river is about 800 feet wide in this reach. Tidal amplitudes are approximately 12.8 feet above mean low water.

9.2.2.1 Hydrography of the Penobscot River

Historical data from this reach of the Penobscot River indicate that the salinity in the Orrington area generally ranges from near freshwater (less than 1 part per thousand, or ppt) to brackish (5 ppt), especially near bottom. According to Haefner (1967), the freshwater portion of the river extends 9 to 11 nautical miles south of Bangor during peak spring discharge, and less than 2 nautical miles south of Bangor in summer. For example, just upstream of the site, average (1963 to 1965) mid-tide salinities were near zero throughout the water column in April, and between 1 ppt in surface waters and 5 ppt in bottom waters in August. Hatch (1971) reported a similar variation in salinity in this area for observations between 1966 and 1970.

¹ Anadromous fish are those which live in the ocean and migrate into rivers to spawn. Catadromous fish have the opposite life cycle pattern. Adult fish spend their lives in rivers and migrate into the ocean to breed.

Recently collected hydrographic data agree with the historical observations. During September 1994, salinity ranged from 0.1 ppt near surface to 1.2 ppt near bottom over one tidal cycle. The water column was essentially isothermal with temperatures ranging from 16.8°C to 17°C during low tide and 17.4°C to 17.5°C during high tide. Dissolved oxygen concentrations were similar for both surface and bottom waters, ranging from 9.3 mg/l to 9.5 mg/l.

During June 1995, salinities were uniformly below the detection limit of the measuring instrument, less than 0.1 ppt from surface to bottom at all stations. The waters again were isothermal and well oxygenated from surface to bottom, ranging from 18.8°C to 18.9°C, and 8.9 to 9.1 mg/l dissolved oxygen.

These data indicate a well-mixed, well-oxygenated water column with some seasonal changes in salinity in the waters adjacent to the site. The water in this reach of the river ranged from brackish in September 1994 to fresh in June 1995.

These fresh to brackish salinities provide a varying physical and chemical environment for associated biota. The salinities are not high enough to support marine softshell clams or mussels. On the other hand, the higher salinities will exclude many resident freshwater fish from the area of the site. This transition zone is thus likely to support an impoverished fauna.

9.2.2.2 *Sediment Conditions in the Penobscot River*

Haefner (1967) classified the bottom of this reach of the Penobscot River at Hampden (the area of the site) as ledge and the bottom at Winterport (south of the site) as a mixture of pebbles, silt, wood chips, and pulp fiber. Recent observations made during this assessment did not reveal the presence of pulp or paper fiber. Wood chips were observed at the northern end of the site. The sand and gravel bottom is consistent with the hydrographic observations of a well-mixed water column and the strong tidal amplitudes in this area.

The recently collected data (Table 9-1) indicate that the offshore, subtidal portions of the Penobscot River are comprised of a variety of sediment types, dominated by gravel and ledge with generally small sand deposits scattered among the ledge and gravel (Figure 9-3).

9.2.2.3 *Benthic Biota in the Penobscot River*

The historical observations in this reach of the Penobscot River indicate a depauperate freshwater fauna. Haefner (1967) concluded that "Benthic fauna were remarkably scarce from Brewer (author's note - upstream of the site) to Winterport (author's note - downstream of the site)", based on Ekman

dredge samples taken between 1963 and 1966. In particular, Haefner's data indicate only three species in the Hampden area, an unidentified freshwater Pelecypod (clam or mussel), dipteran larvae (insect), and a freshwater gastropod (snail). These data were collected prior to the construction of the HoltraChem plant.

The recent observations made as part of this assessment are consistent with these historical data. Table 9-2 summarizes benthic observations from September 1994 and June 1995. Low abundances of oligochaetes and chironomids were observed in most sediment samples taken from the run of the Penobscot River (Figure 9-3). Benthic infauna is of low abundance and low diversity (expressed as number of taxa observed). The salinity regime indicates that the area will not support marine shellfish.

9.2.2.4 Fish Populations in the Penobscot River

The reach of the Penobscot adjacent to the site has very little historical fish data. There is a smelt fishery at Winterport (Maine Department of Marine Resources (DMR), personal communication) and a commercial eel fishery in the reach of the river adjacent to the site (J. McCleave, personal communication). According to DMR personnel and personnel at the University of Maine, American eel (*Anguilla rostrata*) and killifish (*Fundulus*) are likely resident species in the vicinity of the site. According to Maine Fish and Wildlife Service (F. Kircheis, personal communication), the site is unlikely to support many resident species due to the salinity regime.

The run of the river supports migrant fish populations such as striped bass, Atlantic salmon (*Salmo*), trout, and herring (*Alosa*) (US Fish and Wildlife Service Atlantic Coast Ecological Inventory Map, Bangor, Maine, 1980). Sampling for fish species during this assessment occurred in or immediately offshore of the tidal coves at the northern and southern ends of the site. Subsections 9.2.3.1 and 9.2.3.2 provide a summary of these data for the southern and northern coves, respectively.

9.2.2.5 Wildlife Observations along the Penobscot River

During early June 1995, the river's mudflats and shorelines attracted only limited shorebird use, and few fish-eating wildlife species were observed (Table 9-3). However, this section of the river is heavily traveled by migratory anadromous and catadromous fish species, including salmon, eels, herring, and trout. During fish runs, the area may be more attractive to fish-eating animals such as bald eagle (*Haliaeetus leucocephalus*), osprey (*Pandion haliaetus*), waterfowl, great blue heron (*Ardea herodias*), mink (*Mustela vison*), and river otter (*Lutra canadensis*). Plant personnel have observed eagles over the site. Other aquatic species such as frogs, snapping turtles (*Chelydro*

serpentina), Canada geese (*Branta canadensis*), and muskrat (*Ondatra zibethicus*) likely occur in the river at the HoltraChem site. Beaver (*Castor canadensis*) sign was observed along the river shore.

9.2.3 Freshwater Tidal Coves

There are two intertidal coves associated with the site, a southern cove and a northern cove. The southern cove is immediately downstream of the site and receives runoff and groundwater from the active areas of the plant. The northern cove does not receive runoff or groundwater from the active, developed portions of the plant site, and this assessment considers it a reference area. Section 3 provides the hydrological justification for use of the northern cove as a local reference area.

9.2.3.1 The Southern Cove

At the southern end of the site, the Penobscot River forms an intertidal freshwater cove and associated freshwater marsh. Figure 3-15 is a detailed map of this southern cove and its associated marsh showing the range of the intertidal zone and associated sediment types. The cove is subject to local diurnal tides and includes an extensive intertidal range as indicated by the low and high water lines on Figure 3-15. The cove is hydrologically downgradient of plant activities, and receives water from the southerly stream and northern stormwater ditch that flow from the site and two National Pollutant Discharge Elimination System (NPDES) outfalls (the Penobscot Energy Recovery Corporation (PERC) outfall and the HoltraChem outfall).

Southern Cove Sediments

- The southern cove has four general sediment types in the intertidal zone:
- High intertidal organic mat supporting a reed and sedge bed;
- Sand and gravel mid-intertidal area covered by a recent, thin (less than 0.5 inches) layer of fine silt;
- Sand and gravel low intertidal area covered by an approximately 1 foot layer of fine silt;
- Mid-intertidal sedge bed.

The vegetated beds of the southern cove are a unique habitat in this reach of the river. We did not observe any similarly vegetated areas upstream or downstream of the site.

Southern Cove Benthos

The subtidal portions of the river offshore of the tidal cove at the southern end of the site supported a more abundant benthic infauna in both September 1994 and June 1995 observations than did the run of the Penobscot River (Table 9-2). Taxonomic diversity was low, as in the run of the river, but the abundance of oligochaetes was 100 times that observed in the main stem of the Penobscot River. This is also the area where the sediments are comprised of higher amounts of silt and organic matter (Table 9-1), which may explain the increased abundance. We observed no macroinvertebrates (e.g., bivalve shellfish) in the intertidal zone of the cove.

Southern Cove Fish

Our September 1994 fish sampling efforts offshore of the southern cove yielded 24 killifish (*Fundulus heteroclitus*), five juvenile herring, three white perch, one smelt, and one white sucker (*Catostomus commersoni*). Among these, killifish occurred nearshore in the vegetated beds. The other species were caught in a gillnet at the mouth of the cove. Small forage fish were caught among the grass beds of the southern cove in June 1995. These included: golden shiner (*Notropis cornutus*), killifish, and Atlantic silverside (*Menidia menidia*). Seining also returned numerous young of the year herring.

Live trapping indicated that eels were abundant in the water immediately offshore of the southern cove (Table 9-4). During recent field measurement programs, we observed two commercial eel trapping operations offshore of the cove.

Southern Cove Wildlife and Vegetation

The southern cove vegetation is characterized by three zones (Figure 3-14): mudflats with sparse low-growing plants; marsh vegetated with narrow-leaved cattail (*Typha angustifolia*), three-square rush (*Scirpus pungens*), seaside arrow grass (*Triglochin maritimum*), beaked spike-rush (*Eleocharis rostellata*), and other emergent species; and muddy rocky shores, vegetated with grasses, sedges, and herbaceous species, including blue flag iris (*Iris versicolor*), goldenrod (*Solidago spp.*), and water parsnip (*Sium suave*).

Very little wildlife activity was observed on the marsh/flats (Table 9-3). Raccoon (*Procyon lotor*) tracks were observed along the shore and a single spotted sandpiper (*Actitis macularia*) was observed on the flats. During peak shorebird migration later in the summer, the flats may be used by more migrant shorebird species. Marsh vegetation is extremely limited within the cove. There is insufficient emergent vegetation cover to support marshland wildlife species such as rails or

waterfowl. At high tide, the marsh vegetation likely provides habitat for small fish and other aquatic life in the river. In August 1995, a fish survey in the marsh/flats area yielded alewife (*Alosa pseudoharengus*), golden shiner, killifish, and Atlantic silverside.

9.2.3.2 *The Northern Cove*

At the northern end of the site, the Penobscot River forms an intertidal cove. It receives surface water from a single stream entering at the head of the cove. The stream does not drain active or developed areas of the plant site.

Northern Cove Sediments

The sediments in the northern cove are generally silty sand (see Table 9-1 and Section 3). These silty sands are homogeneously distributed over the intertidal area. In contrast to the southern cove, this cove does not support a freshwater tidal marsh and its associated vegetation.

Northern Cove Benthos

The northern cove supports a similar oligochaete-dominated fauna, as found elsewhere in this reach of the Penobscot River. The infaunal abundance in the northern cove is ten times greater than in the main stem of the Penobscot River but ten times less than that in the southern cove (Table 9-2).

Northern Cove Fish

Live trapping indicated that eels were abundant in the water immediately offshore of the northern cove (Table 9-4). However, seining in the northern cove did not return resident forage fish. It did return numerous young-of-the-year herring. The lack of suitable vegetative habitat in the northern cove can explain the absence of killifish and other small forage fish. As indicated above, the vegetated beds of the southern cove are a unique habitat for this reach of the river.

9.2.4 *Onsite Streams, Stormwater Ditch, and Flooded Gravel Pit*

This subsection provides recent limnological and biological observations made at a reference stream, an onsite stream, the southerly stream and northern stormwater ditch, and their associated wetland habitats. The onsite stream runs east and north, draining the woodlands west of the plant and flowing into the river at the northern cove. The southerly stream (Figure 9-2) flows east to west around the HoltraChem plant and into the Penobscot at the southern cove; and the northern stormwater ditch, a small grassy swale, flows from a paved sump on the west side of the plant,

through the lower field, and into the river at the northern end of the southern cove. The reference stream is a stream similar in morphology to the southerly stream and northern ditch and located approximately one mile downstream of the site on the opposite shore. The field observation methods used to assess aquatic conditions in the onsite streams, onsite stormwater ditch, and the offsite reference stream are provided in Appendix F.

The southerly stream was channelized and northern stormwater ditch was constructed when the plant was built and they both receive stormwater runoff from the industrial areas of the plant. Thus, even though the northern ditch was assessed for ecological purposes, it is not a stream. As set forth in 38 M.R.S.A., 480-B, (1995), "'River, stream or brook' does not mean a ditch or other drainage way constructed and maintained solely for the purpose of draining stormwater or a grassy swale".

9.2.4.1 *Southerly Stream*

The southerly stream begins at a small flooded gravel pit situated to the east and upgradient of the active plant site (Figure 9-2). An intermittent channel begins at the flooded gravel pit and runs west toward the plant. After crossing under a railroad bed, flow in the channel becomes perennial. It is shallow (less than 2 inches deep) and narrow (approximately 3 feet wide) where it enters the southern end of the southern cove on site. It receives water from the adjacent PERC plant in its lower reaches.

Southerly Stream - Limnological Characteristics

The land use around the southerly stream is generally forested with moderate local watershed erosion and obvious potential sources of contamination (the site and the PERC plant). The stream is partially shaded by the overstory and has well-vegetated banks. The sediment is generally comprised of gravel and sand, and the stream runs among large- to medium-sized boulders. We observed no sediment odors or oily sediments. The bottom was generally free of detritus and coarse particulate organic matter. The surface water exhibited a pH of 8.34, a dissolved oxygen concentration of 10.7 mg/l, warm temperature, and an unusually high conductivity (1.54 millisiemens/cm). This conductivity is approximately 25 times that measured in the Penobscot River. The source of the high conductivity may be brine or sodium hydroxide spills and/or the salt piles which are located above the paved sump. The stream exhibited little periphyton, no filamentous algae, no macrophytes, no slimes, few macroinvertebrates or other benthic invertebrates (Table 9-5), and no fish.

Southerly Stream - Associated Wetlands and Wildlife

The upstream, riparian wetland associated with the southerly stream contains small red maple trees (*Acer rubrum*), cattail, purple loosestrife (*Lythrum salicaria*), and wetland ferns. After crossing under a road downstream of the flooded gravel pit, the channel widens briefly into a small cattail-lined pool. The stream then flows around the east and south sides of the plant via a series of culverts. Stands of wetland shrubs occur along the stream. After passing under the plant's access roadway, the stream runs parallel to the rail loading area. Fill and debris constitute part of the banks of the stream along this section. It then enters into the mixed woods where it flows through a small, shallow valley. In the valley's bottom occurs some wetland vegetation including sedges (*Carex spp.*), poison ivy (*Toxicodendron radicans*), and jewelweed (*Impatiens capensis*). Before reaching the southern cove and ultimately the Penobscot River, the elevation of the stream drops over 6 feet through a series of ledges and boulders.

The stream and its associated riparian zone appear to provide limited aquatic and wetland habitat along the southern side of the HoltraChem property. The drop at the river would appear to create a barrier to fish moving up from the Penobscot. No fish were observed in the stream; however, small fish were seen in the flooded gravel pit at its head (Table 9-3). It is possible, therefore, that the stream does provide some limited fish habitat. No amphibians were observed in the stream, although bullfrogs (*Rana catesbeiana*) were noted in the flooded gravel pit. Due to the apparent lack of fish or amphibian resources in the stream, it is likely that fish-eating wildlife, such as mink and herons, do not heavily use the waterway. Raccoon tracks were noted along the course of the stream. It is likely that the course of the stream serves as a travel corridor for wildlife, such as muskrat, to move west to east from the river, around the HoltraChem plant, and into interior habitats. Other wildlife noted along the stream include yellow warbler (*Pendroica petrechia*), common yellowthroat (*Gleothlypis trichas*), and deer mouse (*Peromyscus maniculatus*).

Although the southerly stream does provide some habitat resources for area wildlife, it is likely that disturbance from the adjacent activities at the HoltraChem plant greatly reduces the quality of its habitat along much of its length. Up- and downstream of the HoltraChem plant, less disturbed wetland and aquatic habitat still occur along this stream.

Southerly Stream - Associated Flooded Gravel Pit

The small flooded gravel pit from which the southerly stream flows is surrounded primarily by shrubs. It did not appear to be vegetated with rooted, submerged or other floating-leaved vegetation. The flooded gravel pit is the largest fresh, open water habitat within the survey area. The flooded gravel pit appears to have a relatively undisturbed watershed.

Unidentified small fish were observed in the flooded gravel pit, as were bullfrogs (Table 9-3). It is likely that fish-eating species, such as mink, herons, and belted kingfisher (*Ceryle alcyon*), may forage in the water body. In addition, aerial insect-feeding species including swallows, flycatchers (*Empidonax spp.*), cedar waxwings (*Bombycilla cedrorum*), and bats likely hunt over the flooded gravel pit's surface. Depending on the number and size of predatory fish present, the flooded gravel pit may support breeding amphibians such as red-spotted newt (*Notophthalmus viridescens*), American toad (*Bufo americanus*), and pickerel frog (*Rana palustris*). Snapping turtle (*Chelydra serpentina*) and painted turtle (*Chrysemys picta*) also likely live in the flooded gravel pit.

9.2.4.2 Northern Stormwater Ditch

The northern stormwater ditch is shallow (less than 2 inches deep) and narrow (approximately 3 feet wide) and empties into the northern end of the southern cove on site. It flows east to west from a small detention basin on the west side of the HoltraChem plant, across the lower field, and into the Penobscot River.

Northern Stormwater Ditch - Limnological Characteristics

The land use around the northern stormwater ditch is generally open field with moderate local watershed erosion and obvious potential sources of contaminated runoff. The stormwater ditch is unshaded and has well-vegetated, grassy banks. The sediment is cobble and gravel. We observed no sediment odors or oily sediments. The bottom was generally free of detritus and coarse particulate organic matter. The surface water exhibited a pH of 7.51, a dissolved oxygen concentration of 10.3 mg/l, warm temperature, and an unusually high conductivity (1.55 millisiemens/cm). This conductivity is approximately 25 times greater than that measured in the Penobscot River. The source of the high conductivity may be salt piles on site. The ditch exhibited little periphyton, no filamentous algae, no macrophytes, no slimes, few macroinvertebrates or other benthic invertebrates (Table 9-5), and no fish.

Northern Stormwater Ditch - Associated Wetlands and Wildlife

The northern stormwater ditch is vegetated with wetland grasses, sedges, and herbaceous species. At its lower end, it is tidal, with the river backing up the channel for a short distance at high tide. In the field, the swale has very little discernable channel. Water is standing or slowly moving through stands of wetland grasses.

No unique wildlife species were observed along the stormwater ditch. The presence of water in the ditch does maintain wetland meadow habitat within the surrounding field area. It is likely that

wildlife using the field take advantage of this additional habitat diversity. The vegetation within the swale displayed no evidence of ecological stress.

9.2.4.3 Northerly Stream

The northerly stream is deeper (5 to 7 inches) and wider (approximately 6 feet wide) than the stormwater ditches that flow into the southern cove and the reference stream. It drains much of the woods on the northern and eastern ends of the property.

Northerly Stream - Limnological Characteristics

The northerly stream can be classified as Class B based on State of Maine default classifications for freshwater streams in this reach of the Penobscot River. The land use around this stream is generally forested with little local watershed erosion and no evidence of local sources of contamination. The stream is partially shaded by the overstory and has well-vegetated banks. The unconsolidated bottom is gravel, sand, cobble, and bedrock, and the stream runs among large- to medium-sized boulders. We observed no sediment odors or oily sediments. The bottom was generally free of detritus and coarse particulate organic matter. The surface water exhibited a pH of 7.8, a dissolved oxygen concentration of 9.4 mg/l, cold temperature, and a typical freshwater conductivity (0.138 millisiemens/cm). The stream exhibited abundant periphyton, no filamentous algae, sparse macrophytes, no slimes, and abundant macroinvertebrates (helgraminthes, snails, egg cases on the underside of rocks, and water striders). In addition, there were abundant and diverse infaunal benthic invertebrates (Table 9-5). The stream is probably of sufficient depth and width to support fish, although none were observed.

Onsite Stream - Associated Wetland and Wildlife

The area of the northerly stream is wooded and rocky. At its mouth at the northern cove, the stream is approximately 6 feet wide and shaded with Eastern hemlock (*Tsuga canadensis*), red spruce (*Picea rubens*), red oak (*Quercus borealis*), white pine (*Pinus strobus*), and white birch (*Betula papyrifera*). There is a small riparian area along the stream edge which is vegetated with sapling red maple, poplar (*Populus spp.*), and northern white cedar (*Thuja occidentalis*); shrub cover of speckled alder (*Alnus incana*) and maple-leaved viburnum (*Viburnum acerifolium*); and 70% ground cover of sedges, grasses, sensitive fern (*Onoclea sensibilis*) and goldenrod. The stream's watershed appears to consist primarily of sections of the property beyond the past or present operational area of the plant. Therefore, disturbance levels within the stream course are presumed to be low.

Winter wren (*Troglodytes troglodytes*), raccoon, and white-tailed deer (*Codocoileus virginianus*) were observed along the stream. Although no fish were observed at the stream's mouth, there is no apparent drop or other obstruction to upstream passage as there is in the southern stormwater ditch. It is expected that this relatively undisturbed stream supports a diverse aquatic and terrestrial biota.

9.2.5 Reference Stream

We chose a reference stream which had the approximate sediment and morphological characteristics of the onsite stream and ditch. This reference stream is approximately one mile downstream from the site and on the west bank of the Penobscot River.

Reference Stream - Limnological Characteristics

The reference stream is shallow (approximately 3 inches deep), and narrow (approximately 3 feet wide). It is a Class B stream, based on State of Maine default classifications for freshwater streams in this reach of the Penobscot River. The very mouth of the stream may be classified SC, based on tidal influence. The land use around this stream is generally forested with no to moderate local watershed erosion and no obvious potential sources of contamination. The stream is well shaded and has well-vegetated banks. The sediment is generally sand, cobble, and boulders. We observed no sediment odors or oily sediments. The bottom was generally free of detritus and coarse particulate organic matter. The surface water exhibited a pH of 8.00, a dissolved oxygen concentration of 11.3 mg/l, cold temperature, and a typical freshwater conductivity (0.153 millisiemens/cm). The stream exhibited abundant periphyton, no filamentous algae, no macrophytes, no slimes, and several macroinvertebrates (small snails, slugs, helgramithes, water striders, and egg cases on the underside of rocks). There was a paucity of infaunal invertebrates (Table 9-5), and no fish were observed.

The reference stream is similar to the southerly stream in that it drains through dense mixed woods, drops down into the river (thereby impeding fish movement), and has a similar rock/sand substrate. The woods overhanging the reference stream is vegetated with hemlock, white pine, red oak and northern white cedar and a limited understory of tree saplings and seedlings, moss, ferns, and jewelweed. Black-and-white warbler (*Mniotilta varia*) and black-capped chickadee (*Parus atricapillus*) were observed along both the southerly stream and the reference stream (Table 9-3).

9.2.6 Terrestrial Habitats

During a recent habitat survey (see Appendix G for methods), we identified five major terrestrial habitat types within and adjacent to the HoltraChem property: mixed deciduous/pine woods, hemlock woods, shrubland (upland and wetland), field, and the factory area (see Figure 9-2). The

size, location, vegetational complexity, and juxtaposition with neighboring habitat types greatly influences each habitat's potential to support a diverse, unimpaired community of organisms. In addition, this subsection describes several offsite reference areas to compare the condition of similar habitats adjacent to and distant from the HoltraChem plant.

9.2.6.1 Mixed Deciduous/Pine Woods

Mixed Woods - Vegetation

Mixed woods is the dominant upland habitat surrounding the HoltraChem plant. One stand of mixed woods (approximately 11 acres) occurs on the south side of the plant between the railroad loading area, the plant's access road, and the Penobscot River. The woods are located along both sides of the southerly stream. The dominant trees in the area are white pine and red oak, with scattered Eastern hemlock, gray birch (*Betula populifolia*), American elm (*Ulmus americana*), northern white cedar, and balsam fir (*Abies balsamea*). The percentage of oak to pine varies widely, with pine dominant southwest of the dirt road that runs through the area, and oak dominant on the road's northeastern side. Within the woods, there is approximately 20% shrub cover consisting of tree saplings, willow (*Salix sp.*), and honeysuckle (*Lonicera sp.*). Ground cover includes grasses, goldenrod, sensitive fern, hawkweed (*Hieracium sp.*), bracken fern (*Pteridium aquilinum*), and Canada mayflower (*Miainthemum canadense*).

Mixed woods also cover the northern and eastern sides of the HoltraChem property. Woods immediately north of the plant (approximately 5 acres) contain a variety of wooded stands which vary in species composition and maturity. At the east end of the woods, on a knoll overlooking the plant, is a stand dominated by red oak. Other stands farther west are dominated by white pine. In between these mature stands are a number of second-growth areas, which were cut down but are currently being revegetated by hemlock, birch, oak, and pine. Directly north of the plant is an area of dead and blown down pine revegetating with balsam fir and grey birch. Throughout this area of mixed woods, shrub cover, consisting primarily of tree saplings and patches of lowbush blueberry (*Vaccinium vacillans*), varies from approximately 40% to almost 80%. Ground cover species, which occur in densest concentration in woodland openings, include bracken fern, Canada mayflower, tree seedlings, bunchberry (*Cornus canadensis*), grasses, and upland sedges (*Carex pensylvanica*).

An extensive area of mixed deciduous/pine woods (approximately 186 acres) occurs to the east of the HoltraChem plant. This area is relatively isolated from the plant's activities but has been actively logged. The logging activity has created a patchwork of wooded stands of mixed species composition and maturity. As in the area north of the plant, the proportion of white pine to oak and other deciduous trees varies greatly. This creates a great variety of sub-habitats within this area of

mixed woods cover type. Two drainage channels flow through the western area of mixed woods, adding to its habitat diversity. The onsite stream runs east to west along much of the southern edge of the mixed woods, entering the river at the north cove. The southerly stream begins within the mixed woods at a small flooded gravel pit. It then flows west-ward from the woods, around the HoltraChem plant, and into the river at the southern cove.

Mixed Woods - Wildlife

The mixed woods cover type appears to support the greatest variety of wildlife species within the study area (Table 9-3). The woods provide diverse feeding, nesting, and cover habitats for species such as white-tailed deer, porcupine (*Erethizon dorsatum*), squirrels, deer mice, warblers, woodpeckers, sparrows, thrushes, and ruffed grouse (*Bonasa umbellus*). The June 1995 survey found that the mixed woods, even areas immediately adjacent to the HoltraChem plant, appear to support a variety of wildlife species. However, qualitative observations appeared to indicate that somewhat more wildlife activity was present in the eastern section of the mixed woods versus the wooded areas immediately adjacent to the plant. This observed phenomenon may be due in part to the eastern area's larger size, its diversity of subhabitats, and its isolation from plant activities.

9.2.6.2 Hemlock Woods

Hemlock Woods - Vegetation

The hemlock woods (approximately 19 acres) is a stand of trees consisting almost completely of Eastern hemlock. The stand occurs to the north of the HoltraChem plant, along a steep hillside that descends into the Penobscot River. A few scattered red oak and northern white cedar occur among the hemlocks. The hemlock canopy is so dense that sunlight barely penetrates to the woodland floor. Little shrub or ground cover occurs, resulting in low plant species diversity within this cover type.

Hemlock Woods - Wildlife

The area may be important as a wintering area for local white-tailed deer (Table 9-3). Active deer trails criss-cross the area. The only other species observed within the hemlock stand were red squirrel (*Tamiasciurus hudsonicus*), brown-headed cowbird (*Molothrus ater*), and black-capped chickadee. The shade provided by the hemlocks along the shore of the Penobscot River may create some protected aquatic habitats along the river's edge.

9.2.6.3 Upland Shrublands

There are no extensive areas of shrubs within the HoltraChem study area. Small upland shrubby areas are located on the edges of the fields and woods. Small wetland shrublands occur along the southerly stream.

Upland Shrublands - Vegetation

Three areas of upland shrubs, containing both shrubs and small trees, occur adjacent to the southwestern, northern, and western edges of the HoltraChem plant. Each of these areas appears to have been disturbed and in the process of revegetating back to woodland. The southwestern area is bordered by mixed woods, field, and the railway loading area for the HoltraChem plant. Species within this area include speckled alder, young quaking aspen (*Populus tremula*), staghorn sumac (*Rhus typhina*), and small white pine and black cherry trees (*Prunus serotina*). The canopy is open with interspersed areas of grasses, goldenrod, ferns, hawkweed and dandelion (*Taraxacum officinale*). The northern shrub area occurs along a steep slope adjacent to the plant. The area is grown up with small gray birch, brambles (*Rubus sp.*), sweet fern (*Comptonia peregrina*), and other low shrubs. The third shrubby area is to the east of the HoltraChem site, in the "V" formed by the railroad tracks running into the plant. This area is dominated by revegetating small birches, cherry, and scattered shrub clumps. Open areas are vegetated with grasses, goldenrods, and other herbaceous species.

Upland Shrublands - Wildlife

Although upland shrubland habitat is limited within the study area, it does support a number of edge species (Table 9-3). Because the shrublands serve as an interface between the site's wooded and field habitats, a variety of species were observed using the shrublands. Cedar waxwings (*Bombycilla cedrorum*), American robins (*Turdus migratorius*), song sparrow (*Melospiza melodia*), indigo bunting (*Passerina cyanea*), deer mouse, and white-tailed deer were all observed within the cover type. Woodland species including black-capped chickadee, black-and-white warbler, blue jay (*Cyanocitta cristata*), yellow-rumped warbler (*Dendroica coronata*) also likely use the shrubland habitats.

9.2.6.4 Wetland Shrublands

Wetland Shrubland - Vegetation

Wetland shrublands are limited to small areas adjacent to the southerly stream and the small flooded

gravel pit. These wet areas are vegetated with wetland shrubs and small trees including willow, speckled alder, grey birch, larch (*Larix laricina*), red maple, and arrowwood (*Viburnum recognitum*). The shrubs and small trees are interspersed with wetland ferns, grasses, sedges, rushes (*Juncus spp.*), and herbaceous species.

Wetland Shrubland - Wildlife

Most of the wildlife species observed in the upland shrublands also occur in the wetter areas (Table 9-3). In addition, several wetland wildlife species, including yellow warbler, common yellowthroat, and red-winged blackbird (*Agelaius phoeniceus*) appeared to occur primarily in the wetland shrublands.

9.2.6.5 Fields

Fields - Vegetation

Field habitats within the survey area are located on a series of closed landfills along the northern edge of the HoltraChem plant (approximately 7 acres) and in the area between the Penobscot River and the western side of the plant ("the lower field") (approximately 4 acres). These fields are vegetated primarily with grasses, goldenrod, clovers (*Trifolium spp.*), vetches (*Vicia spp.*), milkweed (*Asclepias syriaca*), dandelion and other herbaceous species. Less than 5% of the fields contain scattered small shrubs and trees, such as speckled alder, willow, quaking aspen, and grey birch. The northern stormwater ditch runs in a small, grassy, swale runs through the center of the lower field into the river.

Fields - Wildlife

The field habitats in the survey area are relatively small and appear to be too small to support breeding populations of open field species, such as bobolink (*Dolichonyx oryzivorus*) or Eastern meadowlark (*Sturnella magna*). There also was no evidence of meadow vole (*Microtus pennsylvanicus*), either from the small mammal traps (see Subsection 9.4) set in the field or from observations of tunnels and other sign. Voles are typically found in field habitats, but their populations can be cyclical. It is possible that the survey occurred during a natural low period in vole numbers.

The species that were observed in the fields also use adjacent shrubland and wooded habitats (Table 9-3). Song sparrow, American goldfinch (*Carduelis tristis*), mourning dove (*Zenaidura macroura*),

and woodchuck (*Marmota monax*) were all observed along the field edges. The field also provides open foraging areas for aerial insect feeders such as swallows, swifts, flycatchers, and bats.

9.2.6.6 Factory Area

The HoltraChem plant site is heavily developed with multiple buildings, parking areas, driveways, and storage sites. Vegetation and natural habitat availability is extremely limited. The primary habitat presently consists of the plant's buildings, which may be used by bats, swallows, American robins, European starling (*Sturnus vulgaris*), rock dove (*Columbia livia*) and house sparrows (*Passer domesticus*). Other species which may occur in such a heavily industrialized areas include garter snake (*Thamnophis sirtalis*), American toad, mice, and Norway rat (*Rattus norvegicus*).

9.2.7 Background Area - Dorothea Dix Park

Dorothea Dix Park in Hampden, Maine was surveyed as a terrestrial reference area to compare on- and offsite conditions within major habitat types in the region. It is approximately one mile downstream and across the river from the HoltraChem site. It is primarily a wooded parcel containing mixed woods, small areas of shrubland, and a small flooded gravel pit with surrounding wetland habitat. Wildlife use and habitat quality within the park were evaluated to compare with observed ecological site conditions in the vicinity of the HoltraChem plant. Small mammal trapping was also conducted in the park to gather a reference sample for the HoltraChem mammal mercury analysis (see Subsection 9.4).

Mixed Woods

Most of the park is wooded with mixed white pine, red oak, red maple, black cherry, sugar maple (*Acer saccharum*), red spruce, and northern white cedar. The understory consists of approximately 80% shrub cover of sapling trees, arrowwood, and Japanese barberry (*Berberis thunbergii*). There is approximately 30% ground cover by Canada mayflower, tree seedlings, ferns, wild sarsaparilla (*Aralia nudicaulis*) and false solomon's seal (*Smilacina racemosa*).

While the plant species composition is similar between the reference woods and the mixed woods at the HoltraChem site, the woods at the Dorothea Dix Park are slightly wetter. This may be a function of the reference area being on the east side of the Penobscot River, or it might be due to local site conditions. The reference woods also have not been logged, so there is less of a patchwork of different age classes of woods. Although these differences exist between the two wooded areas, the observed wildlife species composition is similar to that found at the HoltraChem site (Table 9-3). Red-eyed vireo (*Vireo gilvus*), black-capped chickadee, ovenbird (*Seiurus aurocapillus*), blue jay,

pileated woodpecker (*Dryocopus pileatus*), American redstart (*Setophaga ruticilla*), veery (*Catharus fuscescens*), Eastern chipmunk (*Tamias striatus*), red squirrel, and white-tailed deer are among the species observed in both areas.

Shrubland

Shrubland is limited within the reference area. Some upland shrub habitat occurs on the west side of the property adjacent to the park's access driveway. The area contains patches of Japanese barberry, honeysuckle, and sapling trees surrounded by lawn. Some wetland shrubland occurs around the small flooded gravel pit on the north central side of the park. The wet shrub area is vegetated primarily with willows, meadowsweet (*Spirea latifolia*), and sedges. The wildlife species composition between the reference and HoltraChem site shrublands is quite similar. American robin, mourning dove, red-eyed vireo, common yellowthroat, black-capped chickadee, cedar waxwing, and white-throated sparrow (*Zonotrichia albicollis*) are among the species observed in both areas (Table 9-3).

9.2.8 Sensitive Species and Habitats

9.2.8.1 Rare, Threatened, and Endangered Species

No federal or state threatened or endangered species were observed during the June 1995 survey. Bald eagle, a federally threatened species, does regularly occur along the Penobscot and plant personnel have sighted bald eagles in the area of the facility. State biologists within the Maine Department of Fisheries and Wildlife have no records of threatened or endangered species in the vicinity of the HoltraChem property (Hall, 1995; Kemper, 1995). However, there are a number of species which Maine lists as either "Species of Indeterminate Status"² or "Watch List" species. The following species listed under these two categories may potentially use the HoltraChem area: Species of Indeterminate Status - black-crowned night-heron (*Nycticorax nycticorax*), big brown bat (*Eptesicus fuscus*), little brown myotis (*Myotis lucifugus*), and Keen's myotis (*Myotis keenii*); Maine Watch List species - Cooper's hawk (*Accipiter cooperi*), red-shouldered hawk (*Buteo lineatus*), greater yellowlegs (*Tringa melanoleuco*), lesser yellowlegs (*Tringa flavipes*), least sandpiper (*Calidris minutilla*), dunlin (*Calidris alpina*), semipalmated sandpiper (*Calidris pusilla*), and Eastern

² "Species of Indeterminate Status" are "indigenous wildlife believed to be of endangered, threatened, or special concern status, but about which insufficient data are available."

"Watch List" species "do not meet the rigorous requirements of inclusion as [endangered, threatened, special concern, or indeterminate status], but do warrant special attention."

bluebird (*Sialia sialis*). The sensitivities of these species to ecological disturbances potentially present at the HoltraChem site are difficult to assess. Herons and sandpipers, in particular, are likely susceptible to disturbance of the tidal flats and river. Bats are particularly susceptible to disturbance of summer and winter roosting areas (either woodland trees or buildings in the factory area).

The primary fisheries issue along the Penobscot River is the protection and enhancement of the Atlantic salmon run. Recently, there was a citizen petition to list the Penobscot salmon for protection under the federal Endangered Species Act. The petition has not been accepted, but federal and state authorities are closely monitoring the health of the river's Atlantic salmon run (Nickerson, personal communication, 1995). The river at the HoltraChem site in Orrington is not considered by state biologists to be juvenile rearing habitat. However the river is an integral part of the salmon's migratory pathway. Adult salmon enter the river to spawn from May through October. Repeat spawners are out-migrating back down the river from December to March. Smolts (young salmon) migrate to the sea during April or May. According to state biologists, these are the potentially sensitive periods for activities on the river such as discharges (Evers, 1995). The potential for juvenile salmon to be in any tributary streams in the area is also a possible issue. It would appear that the southerly stream, which empties into the southern cove, is too small and impassible for fish. However, the onsite stream could potentially have some salmon habitat.

The Maine Natural Areas Program has records of seven rare plant species occurring in the Orrington area. Six of the seven species are aquatic or semi-aquatic and occur either in tidal marshes/flats or along river shorelines. The seventh species is an upland growing vine. One state-threatened plant has occurred in the area: the estuary bur-marigold (*Bidens hyperborea*). Maine-listed special concern species are: Parker's pipewort (*Eriocaulon parkeri*), pygmyweed (*Crassula aquatica*), and purple clematis (*Clematis occidentalis*), the sole upland species. Maine Watch List species are: water pimpernel (*Samolus valerandi*), spongy arrow-head (*Sagittaria spatulata*), and mudwort (*Limosella australis*). The June 1995 survey did not locate any of these species. However, it is likely that some of these plants may occur on the property since the southern cove marsh/mudflat is the most extensive exposed intertidal habitat present for at least several miles up and down the river. The potential sensitivity of these rare plants to potential ecological disturbance caused by past or present operations at the HoltraChem facilities is not known.

9.2.8.2 Sensitive Habitat

The Maine Natural Areas Program rates freshwater tidal marshes as an "imperiled habitat" in Maine (Maine Natural Heritage Program, 1991). This rating is due to a habitat's rarity, generally defined as only six to twenty occurrences or few remaining individuals or acres. According to the Maine Natural Heritage Program (1991), freshwater tidal marsh communities are defined as:

"Intertidal marshes at the heads of estuaries, upstream of the limit of brackish water. There may be up to three somewhat distinct vegetation zones: expansive mudflats...at the lowest level, marshes dominated by tall graminoids (*Typha angustifolia* and *Scirpus pungens*, among others) at mid-tidal elevations, and muddy rocky shores, sparsely vegetated with low herbs at the highest intertidal elevations."

The southern cove marsh and flats appear to meet the definition of a freshwater tidal wetland in that they contain at least one obligate freshwater species, the blue flag iris. Brackish tidal marshes, which occur downstream of the HoltraChem site, are not as rare as freshwater areas; however, they are still rated as a "rare" community in Maine (having on the order of twenty to one hundred occurrences in the state). Brackish marshes generally have a salinity of 0.5 to 18 parts per thousand. The June 1995 salinity measurements from the river at the HoltraChem site (0.1 to 1.2 ppt) indicate that the southern cove is borderline fresh/brackish. However, the presence of some obligate freshwater plants appears to indicate that the southern cove marsh and flats are a freshwater community.

9.3 Problem Formulation

USEPA guidance recognizes problem formulation as the initial task of an ecological risk assessment. The problem formulation establishes those elements of the assessment which focus and bound the analysis. These boundaries are defined by the contaminants of concern (COCs), the receptor organisms, the pathways by which these receptors are exposed to the contaminants of concern, and the choice of endpoints for assessment of risk to the receptors.

- Subsequent sections of the problem formulation:
- Identify the chemical data which characterize the site and select contaminants of concern from those data;
- Select and evaluate appropriate assessment endpoints and their associated measurement endpoints;
- Select and characterize receptors that may be present on the site;
- Develop a site conceptual model and discuss pathways by which receptors may contact COCs.

9.3.1 Identification and Selection of Contaminants of Concern

9.3.1.1 Description of Available Data

This section identifies the data used in the ecological risk assessment and selects contaminants of concern from those data. Section 4 evaluated the existing site data with regard to usability. The data used in the ecological risk assessment are from samples of environmental media and biota collected by CDM and Menzie-Cura & Associates, Inc. in 1994 and 1995. Table 9-6 summarizes the available data and their use in the ecological risk assessment. These data include:

- Two hundred and fourteen composite surface soil samples (SSS series) from the site and one background location in Dorothea Dix Park approximately 1 mile to the southwest of the site (SSS-18). The samples were analyzed for mercury. Section 3.8 describes sample collection and presents results (Table 3-14). These sample locations are on Figure 9-2, except for the background location.
- Thirty-three onsite surface water samples (SSW series) analyzed for mercury. One of these samples, SSW-10, was also analyzed for volatile organic compounds (VOCs). Section 3.6 describes sample collection and presents results (Table 3-16). Sample locations are on Figure 9-2.
- Ten Penobscot River surface water samples (RSW series) analyzed for mercury; four of these samples (RSW-001 and RSW-003 through RSW-005) were also analyzed for VOCs. Section 3.7 describes sample collection. Locations are on Figure 9-3 for the Penobscot River upstream of the site and Figure 3-6 for the southern cove. Tables 3-29 and 3-30 in Section 3 present the results for mercury and VOCs, respectively.
- River sediment samples were analyzed for mercury and whole sediment, chronic toxicity in 10 day growth and survival bioassays using the freshwater amphipod (*Hyalella azteca*). Two samples of river sediment were collected in 1995 (RSD-07A, RSD-07B) and both analyzed for sediment and chronic toxicity to freshwater amphipods. Six samples of river sediment were collected in 1997 (RSD-022, 023, 025, 029, 030 and 031) and analyzed for mercury in bulk sediment and pore water, and chronic toxicity to freshwater amphipods. Two samples of river sediment were collected in 1998 (RSD-032 and 045) and analyzed for mercury, total organic carbon and chronic toxicity to freshwater amphipods.

- Eleven onsite stream and stormwater ditch sediment samples (SSD-01 and SSD-03 through SSD-11) collected during the SI and analyzed for mercury. Sampling was discussed in Section 3.6. Locations are on Figure 9-2. Results are on Table 3-28.
- Two hundred and twenty two Penobscot River sediment and sediment core samples analyzed for mercury and total organic carbon (TOC) (RSD and RSC series). Sample locations are on Figure 9-3 and 3-6. Section 3.7 describes the rationale for these sampling locations and sample collection methods. Mercury and TOC results are on Table 3-32 in Section 3. Four of these samples were analyzed for VOCs (RSD-01A-02, RSD-03A-02, RSD-07A-02, and RSD-07B-02) and five were analyzed for acid volatile sulfides (AVS) and simultaneously extracted metals (SEM) (RSD-4B-01, RSD-5A-01, RSD-6B-01, RSD-7C-01, and RSD-8A-01). These results are presented in Tables 3-33 and 3-34, respectively.
- Fifteen eel samples from the Penobscot River (ES-01 through ES-15) consisting of fillets of individual eels. These samples were analyzed for mercury. Sampling locations are on Figure 9-3. Section 3.8 discusses sample methods. Results are presented on Table 9-2. We used the fillet data from the eels as an analogue of whole body concentrations. The ecological risk assessment protocol stipulated the collection of forage fish samples for analysis for mercury to use in the ecological assessment. However, despite sampling for forage fish for five days in spring and one in summer, only one forage fish sample was collected which consisted of several whole killifish combined in a composite. The mercury concentration in this sample was lower than the concentrations in the eel fillets. Therefore, to be conservative, we used the eel data to assess risks to wildlife consuming fish from the Penobscot River. These fillet data probably represent whole body concentrations, based on the technical report on the 1995 Surface Water Ambient Toxic Monitoring Program (January 1997) which states that “there was no difference in mercury levels between predator fillets and whole predators in a previous study of fish from lakes in Maine” (Sowles, et al., 1996)
- One composite whole body forage fish sample from the southern cove in the Penobscot River (RMS-01). This sample was analyzed for mercury. Section 3.8 discusses sampling methods. The sample location is on Figure 3-6, and results are reported on Table 3-38 in Section 3.
- Ten composite whole body earthworm samples (SIS-01 through SIS-011) from the same locations outside the plant area as the surface soil samples (Figure 9-2). These samples were analyzed for mercury. Sampling was discussed in Section 3.8, and results are reported on Table 3-37.

- Ten whole body small mammal samples from outside the plant area. Three of these samples (two deer mice and one short-tailed shrew (*Blarina brevicauda*)) came from the Dorothea Dix Park background location (SMS-R5, SMS-R6, and SMS-R7). The onsite samples were deer mice (SMS-B28, SMS-C7, SMS-E3, SMS-E7, SMS-E16). These samples were analyzed for mercury. Sampling was discussed in Section 3.8. Onsite sampling locations are on Figure 9-2. Results are presented on Table 3-38 in Section 3.

The ecological risk assessment does not use subsurface soil or groundwater data because the direct ecological exposures to these media are likely to be less than the surface soil exposures to foraging mammals and birds. We believe the top two feet are an appropriate exposure zone because the levels of contamination decrease rapidly with depth therefore the upper two feet zone represents the highest level of contamination in soil. Also the exposure for most of the burrowing mammals in Table 9-3 occur in first two feet. For example, EPA (1993) notes that the shrews generally nest and build runways in the top 10 cm of soil; red fox dens are generally the abandoned dens of woodchuck and badgers, 1 to 3 meters below the surface (below the most contaminated soil onsite). Therefore, the small mammal specified in the approved RAPD is likely the most exposed individual given that its dens and runways are in the most contaminated zone.

Table 9-7 presents the range, mean, and 95% upper confidence limit on the arithmetic mean for the small mammal, earthworm, eel, and onsite soil samples. There was only one forage fish sample from the south cove. Because of the small numbers of biota samples, no additional statistical analyses were performed on these data. Section 8 presents these statistics for Penobscot River and site surface water and sediment data.

9.3.1.2 Selection of Contaminants of Concern

The ecological risk assessment selected contaminants of concern (COC) from the recently collected data described in Section 9.3.1.1. Note that the samples used in the ecological risk assessment were analyzed for mercury or mercury and VOCs only. These are the likely site contaminants, based on site history and the use of chemicals at the plant. Section 2 describes the historical use of chemicals at the site, and provides the field study's rationale for measuring only those chemicals associated with past plant activities.

The criteria used to select COCs for the ecological risk assessment were simple:

- If a measured contaminant (mercury or a VOC) was detected in sediment, surface soil, surface water, or biota, it was retained as a COC;

- If mercury concentrations were within the range of measured values in reference areas or naturally occurring values from remote areas, it was dropped as a COC for that particular medium.

The remainder of this section applies these criteria and selects COCs.

Onsite Stream and Ditch Sediment

Mercury concentrations in the onsite stream and ditch sediments ranged from less than 0.1 mg/kg to 78 mg/kg in samples collected for SI. (A concentration of 1,000 mg/kg was measured in sediments in the retention basin. These sediments have since been excavated.) A maximum concentration of 91 mg/kg was detected by Acheron in 1997 (Appendix B). Although the assessment did not develop a site-specific background concentration for stream and ditch sediment, these concentrations are elevated above Penobscot River reference sediments (0.1 to 1.7 mg/kg). Therefore, mercury is a COC for onsite stream and ditch sediment.

Penobscot River Sediment

One VOC, acetone, was detected at a concentration of 88 ug/kg in one (LCP-RSD-7B-02) of four Penobscot River sediment samples analyzed for VOCs.

Concentrations of mercury ranged from less than 0.1 to 1.7 mg/kg in sediments from the upstream reference areas (Figure 9-3). Mercury concentrations in the southern cove sediments generally ranged up to 10 mg/kg. (One deeper sediment sample had a mercury concentration of 54 mg/kg.) These values exceeded the mercury concentrations in the northern reference areas. Therefore, acetone and mercury are COCs for Penobscot River sediment.

Onsite Stream and Ditch Surface Water

Mercury concentrations in unfiltered onsite stream and ditch surface water ranged from less than 0.2 to 37 ug/l. Although the assessment did not develop a site-specific background concentration for stream and ditch surface water, these concentrations are elevated above reported mercury concentrations in surface water from remote areas (less than 0.001 to 0.004 ug/l) reported by Porcella (1994). Therefore, mercury is a COC for onsite stream and ditch surface water.

The VOCs carbon tetrachloride and chloroform were detected in one location in onsite surface water (SSW-10). This surface water sample was collected from a catch basin and was the only sample analyzed for VOCs. This sample was analyzed for VOCs because the catch basin receives

groundwater from underdrains in this area. Therefore, carbon tetrachloride and chloroform are COCs in onsite surface water. Note that Acheron collected four onsite surface water samples in 1989 and 1990 did not detect VOCs.

Penobscot River Surface Water

VOCs were not detected in Penobscot River surface water. Mercury concentrations in unfiltered samples of surface water upstream of the HoltraChem plant ranged from 0.00359 to 0.00529 ug/l. In the southern cove, mercury concentrations exceeded these background concentrations and ranged from 0.0267 to 0.173 ug/l. Therefore, mercury is a COC in Penobscot River surface water.

Onsite Soil

Mercury concentrations in onsite soil samples ranged from 0.10 to 350 mg/kg, while the concentrations at the Dorothea Dix Park reference location was 0.18 mg/kg. Since onsite soil concentrations exceeded local background, mercury is a COC in onsite soil.

Penobscot River Biota

One forage fish sample was collected from the southern cove and analyzed for mercury. Mercury was detected in this sample. Since a background mercury concentration was not developed for forage fish, mercury was selected as a COC in forage fish.

Eels

Concentrations in eels collected from near the southern cove had mercury concentrations ranging from 0.259 to 0.678 mg/kg wet weight. Eels collected near the northern cove had mercury concentrations ranging from 0.271 to 0.876 mg/kg wet weight. These concentration ranges are similar and based on these data, it is not possible to distinguish eels caught near the southern cove from eels caught at the reference area. Data from State of Maine (MEDEP, personal communication, July 20, 1998) in the Winterport area of the Penobscot River identified a similar range of mercury concentrations (0.309 to 0.736 ppm). This same data set revealed a similar mercury concentration range in eels from the West Branch of the Piscataquis River at Falmouth, Maine (0.235 to 0.742 ppm). However, to assess risks due to mercury that may enter the aquatic food chain, we retained mercury as a COC in Penobscot River eels.

Onsite Biota

Mercury concentrations in onsite earthworm samples ranged from 0.087 to 2.82 mg/kg wet weight. The mercury concentration in the offsite reference location (Dorothea Dix Park) earthworm sample was 0.044 mg/kg wet weight. Mercury was selected as a COC in earthworms since the onsite concentrations exceeded the background concentration.

Mercury concentrations in deer mice caught on site ranged from 0.060 to 0.198 mg/kg wet weight, while mercury concentrations in deer mice and a short-tailed shrew caught in the Dorothea Dix Park background location ranged from 0.016 to 0.087 mg/kg. Since mercury concentrations in small mammals at the site exceeded those at the background location, mercury was selected as a COC for small mammals.

9.3.2 Selection and Evaluation of Assessment and Measurement Endpoints

This subsection defines, specifies, and evaluates assessment and measurement endpoints. Assessment endpoints are expressions of the environmental value to be protected at a site. Since assessment endpoints are usually not directly measurable, the assessment employs measurement endpoints. These are biological or measurable ecological characteristics which reflect the assessment endpoint (USEPA, 1992). Where the assessment endpoint is not directly measurable, the use of measurement endpoints may result in some uncertainty in the risk characterization.

Ultimately, the risk characterization depends upon the evaluation of the assessment endpoint as reflected in the measurement endpoints. This evaluation depends on quantitative expressions such as toxicity quotients and qualitative professional judgement in a weight-of-evidence approach. The weight-of-evidence approach is the process by which measurement endpoint(s) are related to an assessment endpoint to evaluate whether a significant risk of harm is posed to the environment. The approach is planned and initiated at the Problem Formulation stage, and results are integrated at the Risk Characterization stage. This subsection provides the rationale for each assessment endpoint and the choice of associated measurement endpoints.

9.3.2.1 Selection of Assessment and Measurement Endpoints

The specific assessment endpoints and their associated measurement endpoints for the Penobscot River addressed in this risk characterization are:

Penobscot River Assessment Endpoint	Penobscot River Measurement Endpoint
Maintenance of piscivorous bird and wildlife populations in the vicinity of the site.	Modeled dose of mercury to piscivorous birds and mammals based on measured body burdens of mercury in local fish species.
Health and maintenance of local fish populations.	Measurement of eel body burdens of mercury to compare to a reference area.
	Measurement of water column concentrations of COCs to compare to standards and criteria.
Ability of the benthic invertebrate community to support fishery resources.	Abundance and diversity of local benthic community relative to general conditions in this river reach Measurement of sediment concentrations of Solid phase bioassays with sediment.

The assessment and measurement endpoints for aquatic and terrestrial portions of the site addressed in this risk characterization are:

Onsite Assessment Endpoint	Onsite Measurement Endpoint
Viability of the benthic invertebrate community in the onsite stream and stormwater ditch.	Observations of stream and ditch benthic communities and comparison to a reference stream.
	Measurement of sediment mercury
	Measurement of water column
Maintenance of bird and wildlife populations on site.	Observations of onsite bird and wildlife populations and comparisons to a reference area.
	Modeled dose of mercury to songbirds and small mammals based on measured body burdens of mercury in earthworms; modeled dose of mercury to hawks based on measured body burdens of mercury in small mammals.

9.3.2.2 Evaluation of Assessment and Measurement Endpoints

This subsection evaluates, in narrative form, whether the selected assessment endpoints meet the criteria which USEPA suggests for assessment endpoints. USEPA guidance (USEPA, 1992, and references cited therein) suggests six criteria for such evaluations:

- ecological relevance;
- economic importance;
- measurable;
- susceptible and sensitive to chemical-induced stress or other stresses;
- unambiguously defined;
- logically and practically related to the remedial decision of interest.

In addition, this subsection defines and provides seven attributes which reflect USEPA recommended considerations for the selection of measurement endpoints (USEPA, 1992).

These attributes are:

- *Closeness of Correspondence to the Assessment Endpoint:* This attribute refers to the extent to which the measurement endpoint is representative of, correlated with, or applicable to the assessment endpoint. If there is no association between a measurement endpoint (e.g., a study that may have been performed for some other purpose) and the assessment endpoint of interest, then that study should not be used to evaluate the stated assessment endpoint.
- *Site-specificity:* This attribute relates to the extent to which data, media, species, environmental conditions, and habitat types used in the study design reflect the site of interest.
- *Stressor-specificity:* This attribute relates to the degree to which the measurement endpoint is associated with the specific stressor(s) of concern. Stressors might include a particular chemical, waste, or physical alterations. Some measurement endpoints may respond to a broad range of stressors so that it is difficult to interpret results with regard to the stressor of concern, while other measurement endpoints are more specific to a particular stressor.
- *Availability of an objective measure for judging environmental harm:* This attribute relates to the ability to judge results of the study against well-accepted standards, criteria, or objective measures. Examples of objective standards or measures for judgment might include ambient water quality criteria, sediment quality criteria, biological indices, and toxicity or

exposure thresholds recognized by the scientific or regulatory community as measures of environmental harm.

- *Sensitivity of the measurement endpoint for detecting changes:* This attribute relates to the ability to detect a response in the measurement endpoint. The sensitivity of the measurement endpoint may be affected by natural or analytical variability.
- *Quantitative:* This attribute relates to the degree to which numbers can be used to describe the magnitude of response of the measurement endpoint to the stressor. Some measurement endpoints may yield qualitative or hierarchical results while others may be more quantitative.
- *Correlation of stressor to response:* This attribute relates to the degree to which a correlation is observed between levels of exposure to a stressor and levels of response, and the strength of that correlation.
- *Use of a standard method:* This attribute relates to the extent to which the study follows specific protocols recommended by a recognized scientific authority for particular methods. Examples of standard methods are study designs or chemical measures published in the Federal Register or the Code of Federal Regulations, developed by the American Society for Testing and Materials (ASTM), or repeatedly published in the peer-reviewed scientific literature.

This ecological risk assessment uses a qualitative scheme to evaluate each measurement endpoint relative to these attributes. We characterized an individual measurement endpoint as either adequate or inadequate in terms of meeting the defined evaluation attributes.

Evaluation of Penobscot River Assessment and Measurement Endpoints

Evaluation of the Assessment Endpoint - "Ability of the Benthic Invertebrate Community to Support Fishery Resources" - and Its Related Measurement Endpoints

The suggested assessment endpoint "ability of the benthic invertebrate community to support fishery resources in the Penobscot River" is a reasonable assessment endpoint, and it meets the evaluation criteria.

- *Ecological Relevance* - The coves and nearshore waters of this reach of the Penobscot river support a benthic infaunal community (Section 9.2). These nearshore invertebrates are a prey base for fish, aquatic mammals, and wading birds which likely occur and/or were observed in

this reach of the river. The benthic invertebrates are involved in significant ecological processes, such as sediment aeration and nutrient recycling.

- *Economic Importance* - Invertebrates are not of direct economic importance in this reach of the river, but they are of indirect economic importance as prey for economically valuable game and sport fish inhabiting this river reach, such as eels.
- *Measurable* - The abundance and diversity of invertebrate communities and the chemicals which affect them are measurable quantities;
- *Susceptible and Sensitive to Chemical-Induced Stresses* - Invertebrates live in physical contact with sediment and hence sediment contaminants, and the literature shows mercury can affect invertebrates.
- *Unambiguously Defined* - The specific importance of invertebrates as forage to fish populations is clear and based on food chain relationships;
- *Logically and Practically Related to the Remedial Decision* - Invertebrates live in or on the sediments more intimately than other components of river systems, and reducing their risk should be a conservative measure of the effectiveness of remedial decisions.

The selected measurement endpoints associated with this assessment endpoint are:

- abundance and diversity of local benthic community;
- solid phase bioassays;
- measurement of sediment concentrations of mercury.

Table 9-8 provides our qualitative evaluation of measurement endpoints relative to the evaluation attributes. In general, we believe that nearly all of the measurement endpoints fulfill the evaluation attributes and reflect the assessment endpoint. The exceptions are:

"Sediment mercury concentrations," which have a low correspondence to the assessment endpoint due to the fact that there are no data which relate changes in benthic community structure directly to the concentrations of contaminants in sediments; and

"observations of abundance and diversity of invertebrate community," which we ranked low in terms of the availability of an objective measure for judging environmental harm. In the absence of such an objective measure, we suggest comparing site benthic habitats to similar benthic habitats outside the direct influence of the plant activities and using this qualitative comparison as a measure of the potential environmental harm for this measurement endpoint.

In addition, although these measurement endpoints reflect the health of local benthic communities, they do not provided information on the extent to which this benthos supports local fisheries.

Evaluation of the Assessment Endpoint - "Health and Maintenance of Local Fish Populations" - and Its Related Measurement Endpoints

The assessment endpoint "health and maintenance of local fish populations" is a reasonable assessment endpoint and it meets the evaluation criteria.

Ecological Relevance - Eels are major predators in this reach of the Penobscot River, and the southern cove supports several species of forage fish (Section 9.2). Fish populations generally play a major role in such aquatic ecosystem level properties as the maintenance of zooplankton diversity and nutrient cycling.

Economic Importance - Eels are important economically in this reach of the Penobscot River. They constitute a commercial fishery for adults and elvers.

Measurable - The health and maintenance of local fish populations are measurable quantities.

Susceptible and Sensitive to Chemical-Induced Stresses - There are toxicological and field studies supporting the sensitivity of fish to chemical induced stress from mercury.

Unambiguously Defined - The health and maintenance of local fish populations is unambiguous and clearly distinct from the assessment of migrating fish or wide-ranging fish. The term "local" means populations whose feeding and migrating range is generally on the same scale as the area of the coves proximate to the site.

Logically and Practically Related to the Remedial Decision - Fish live and feed near or on the sediments, and are continuously exposed to surface water. Their protection as a local resource will be affected by remedial decisions regarding Penobscot River sediment and surface water.

The candidate measurement endpoints associated with this assessment endpoint are:

- measurement of eel body burdens of mercury;
- measurement of water column concentrations of mercury in the Penobscot River.

Table 9-9 provides an evaluation of measurement endpoints relative to the evaluation attributes. In general, we believe that nearly all of the measurement endpoints fulfill the evaluation attributes and reflect the assessment endpoint. The exception is "Eel body burdens of mercury," which we ranked inadequate in terms of the availability of an objective measure for judging environmental harm. In the absence of such an objective measure, we suggest comparing body burdens of mercury for eels from site-influenced waters to body burdens of mercury for eels from a nearby reference area outside the direct influence of the plant activities and using this qualitative comparison as a measure of the potential environmental harm for this measurement endpoint. Although the measurement endpoints reflect the assessment endpoint, none of the measurement endpoints includes a direct measurement of fish health.

Evaluation of the Assessment Endpoint - "Maintenance of Piscivorous Bird and Wildlife Populations in the Vicinity of the Site" - and Its Related Measurement Endpoint

The assessment endpoint "maintenance of piscivorous bird and wildlife populations in the vicinity of the site" is a reasonable assessment endpoint and it meets the evaluation criteria.

Ecological Relevance - Piscivorous birds and semi-aquatic mammals occur on or near the site (Section 9.2) and prey on sediment-dwelling invertebrates and pelagic fish. They are often the top predators in this reach of the Penobscot River (e.g., osprey and bald eagle).

Economic Importance - Birds and mammals may be important economically as a recreational resource or as game.

Measurable - Bird and mammal population parameters and the exposure of these populations to chemicals are logistically difficult to measure; however, doses of contaminants can be estimated based on USEPA exposure factors (1993).

Susceptible and Sensitive to Chemical-Induced Stresses - There are toxicological studies supporting the sensitivity of aquatic birds and mammals to chemical-induced stress from mercury.

Unambiguously Defined - This endpoint explicitly refers to species of birds and wildlife whose major route of exposure is through aquatic food chains.

Logically and Practically Related to the Remedial Decision - Aquatic mammals and birds forage or dive for prey near or on the sediments and surface waters, and their protection as a local resource will be affected by remedial decisions regarding contaminated sediment and surface water.

The candidate measurement endpoint associated with this assessment endpoint is: modeled dose of mercury in food. The dose will be calculated from measured concentrations in prey. **Table 9-10** provides a matrix which makes a qualitative evaluation of the measurement endpoint relative to the evaluation attributes. We believe that the measurement endpoint fulfills all but one of the evaluation attributes. The exception is "sensitivity of the measurement endpoint for detecting changes," for which we judge the measurement endpoint to be inadequate. This reflects uncertainty in this analysis due to the limited availability of wildlife toxicity data for bird and wildlife populations.

Evaluation of Onsite Assessment and Measurement Endpoints

Evaluation of the Assessment Endpoint - "Viability of the Benthic Invertebrate Community in the Onsite Stream and Drainage Ditch" - and Its Related Measurement Endpoints

The assessment endpoint "viability of the benthic invertebrate community in the onsite stream and stormwater ditch" is a reasonable assessment endpoint and it meets the evaluation criteria.

Ecological Relevance - The onsite stream and stormwater ditch are generally shallow and unlikely to support fish. Recent observations indicate that they do support invertebrates (Section 9.2). These invertebrates are a possible prey base for birds and small mammals foraging along the stream and stormwater ditch. The benthic invertebrates are involved in significant ecological processes such as sediment aeration and nutrient recycling. These invertebrates include the larval stages of insect species.

Economic Importance - Invertebrates are not of direct economic importance in the onsite stream and stormwater ditch.

Measurable - The abundance and diversity of invertebrate communities and the chemicals which effect them are measurable quantities.

Susceptible and Sensitive to Chemical-Induced Stresses - Invertebrates live in physical contact with sediments, surface water, and their associated contaminants. The literature shows that mercury can affect invertebrates.

Unambiguously Defined - The specific importance of invertebrates to the ecology of a surface water body is clear. They include organisms which are completely aquatic as well as the larval forms of insects.

Logically and Practically Related to the Remedial Decision - Invertebrates live in or on the sediments more intimately than other components of aquatic ecosystems. They are generally sessile and in contact with the overlying water column as well. Reducing their risk should be a conservative measure of the effectiveness of remedial decisions affecting sediment and surface water.

The selected measurement endpoints associated with this assessment endpoint are:

- observations of the benthic communities in the stream and stormwater ditch;
- measurement of sediment concentrations of mercury;
- measurement of water column mercury in the onsite stormwater ditch.

Table 9-11 provides our qualitative evaluation of measurement endpoints relative to the evaluation criteria. In general, we believe that each of the measurement endpoints are strong to moderate in their ability to meet the evaluation criteria. The exceptions are:

"Sediment mercury concentration" which has a low correspondence to the assessment endpoint,

"Observations of the benthic community" which we ranked low in terms of the availability of an objective measure for judging environmental harm. In the absence of such an objective measure, we suggest comparing site benthic habitats to similar benthic habitats outside the direct influence of the plant activities and using this qualitative comparison as a measure of the potential environmental harm for this measurement endpoint.

*Evaluation of the Assessment Endpoint - "Maintenance of Bird and Wildlife Populations Onsite"
- and Its Related Measurement Endpoints*

The suggested assessment endpoint for judging ecological risk at the site, "maintenance of bird and wildlife populations" is a reasonable assessment endpoint and it meets the evaluation criteria.

Ecological Relevance - Literature review and recent observations indicate the site and surrounding areas support a diversity of bird, mammal, reptile, and amphibian populations (Section 9.2). The site exhibits a diversity of woods, fields, and shrublands which will support wildlife.

Economic Importance - Birds and mammals may be important economically as a recreational resource or as game.

Measurable - Bird and mammal populations and their exposures to chemicals are logistically difficult to measure; however, doses of contaminants can be estimated based on USEPA (1993) exposure factors.

Susceptible and Sensitive to Chemical-Induced Stresses - There are toxicological studies supporting the sensitivity of birds and mammals to chemical induced stress from mercury.

Unambiguously Defined - The survey and background review clearly indicate the use of the site by bird and wildlife populations.

Logically and Practically Related to the Remedial Decision - Onsite small mammals and birds live and feed near or on the site soil. Their protection as a local resource will be affected by remedial decisions regarding surface soils.

The candidate measurement endpoints associated with this assessment endpoint are: observations of onsite bird and wildlife species, and modeled dose of mercury in food. The dose will be calculated from measured concentrations in prey. **Table 9-12** provides our qualitative evaluation of the measurement endpoints relative to the evaluation attributes. In general, we believe that the measurement endpoints largely satisfy the evaluation attributes. However, there are exceptions where the two measurement endpoints are inadequate in terms of an evaluation attribute. The measurement endpoint "Modeled dose of mercury in food" is inadequate in terms of the "sensitivity of the measurement endpoint for detecting changes"; this reflects uncertainty in this analysis due to the limited availability of wildlife toxicity data for bird and wildlife populations. We also ranked the measurement endpoint "observations of onsite birds and wildlife populations" as inadequate for two attributes:

"availability of an objective measure for judging environmental harm" (we suggest comparing observations to a reference area which is out of the influence of the plant activities);

"quantitative" (the observations were all qualitative observations of the presence of birds, mammals, and their habitat types).

9.3.3 Selection and Characterization of Receptors

This subsection of the ecological risk assessment identifies the receptor species and provides the rationale for their selection as representative of the species that occur or are likely to occur near the site. This subsection also provides an ecological characterization of each receptor for eventual use in developing the exposure assessment.

The selected receptors represent those types of organisms most likely to encounter the contaminants of concern at the site. They include a reasonable (although not comprehensive) cross-section of the major functional and structural components of the ecosystem under study based on:

- Relative abundance and ecological importance within the selected habitats;
- Availability and quality of applicable toxicological literature;
- Relative sensitivity to the contaminants of concern;
- Trophic status;
- Relative mobility and local feeding ranges;
- Ability to bioaccumulate contaminants of concern;
- Economic importance or federal/state endangerment status; and
- Observed visible evidence of stress.

The selected species represent the ecological community and its sensitivity to the contaminants of concern. They are:

- aquatic receptors - benthic invertebrates, killifish, American eel, great blue heron, mink, osprey, and bald eagle;

- terrestrial receptors - soil invertebrates, short-tailed shrew, American robin, and red-tailed hawk.

9.3.3.1 Aquatic Receptors

Benthic invertebrates

Infaunal benthic invertebrates are potential receptor species in the Penobscot River, onsite stream, and onsite stormwater ditch because they:

- Have the greatest exposure to sediments;
- Provide food for bottom-feeding fish species (in the river);
- Are relatively immobile (sessile) in habit and therefore their general health and condition reflects local conditions;
- They are the only benthic invertebrates in this area (there are no observable macroinvertebrates such as crabs or mussels in this reach of the river or onsite water bodies).

Note that historical and recent observations indicate a sparse and species-poor benthic fauna in this reach of the Penobscot River.

Representative Fish Species

Representative resident fish species were selected to reflect local sediment and water quality conditions. The killifish is a likely and abundant local resident with a limited foraging range that has been observed in the southern cove. Discussions with personnel from Maine Department of Marine Resources (T. Creaser, T. Squires), Department of Fish and Wildlife (F. Kircheis), Bureau of Land and Water Quality (D. Quartermarch; B. Mower), and the University of Maine, Orono (J. McCleave) indicated that the hydrographic variability in this reach of the river discourages most resident freshwater species. However, recent studies in this river reach indicate that American eel has a small enough home range to likely reflect local conditions. Therefore, we have selected killifish and American eel as receptors species.

American Eel (*Anguilla rostrata*)

Eels are catadromous, breeding in the Sargasso Sea and moving in a true larval stage to the coast of

North America about a year after hatching at sea. Metamorphosis to the adult form takes place during the winter on approaching inshore waters. Larvae or leptocephalus, become small, transparent eels (glass eels), then pigmentation starts to develop. By the time the eel reaches the streams and rivers of the coast, it has become completely pigmented, is 2.5 to 3.5 inches long and is called an "elver".

Resident eels in the Penobscot River develop a yellow pigment. They move freely into the muddy, silty bottom of estuaries, rivers and lakes. Eels tend to avoid cool spring-fed waters, and lie buried in the daylight hours in summer. They spend the winter buried in the mud. Once within the Penobscot River, the eels' range is generally restricted to a 1,000 to 1,200 foot length of river, although occasionally they may migrate a mile or more and back on the tide (McCleave, personal communication, 1995).

Eels are voracious carnivores, feeding mainly at night and consuming a wide variety of fish and invertebrates. They have relatively few enemies in fresh water, but the young eels do contribute to the diet of larger fishes, and the large eels are sometimes caught by piscivorous birds and large marine species.

These organisms are potential receptor species representing piscivorous fish because they are:

- Resident in this reach of the Penobscot River;
- Exposed to sediments as well as surface water;
- A higher order predator feeding on fish and invertebrates;
- Abundant in this reach of the river (based on the presence of a commercial fishery);
- Prey to other fish and marine birds; and
- Known to bioaccumulate contaminants.

Eels therefore represent higher order, piscivorous fish in this reach of the river.

Killifish (*Fundulus heteroclitus*)

Killifish occur primarily in Atlantic coastal and brackish waters from northeastern Florida to the

Gulf of Saint Lawrence region. They can occasionally be found to have entered freshwater streams and rivers.

Killifish occur most commonly in saltmarsh flats, estuaries, and tidal areas, especially where there is submerged or emergent vegetation. They can be found upriver to the head of the tide.

Spawning takes place in the spring and summer, but the commencement and duration of spawning is significantly affected by temperature. In the New England regions, spawning occurs primarily in June and the first half of July, but can extend to early August. The eggs become adhesive on extrusion, and may stick together or attach to bottom materials. They hatch in 9 to 24 days at 13 to 17 C, with hatching time dependent upon temperature. Adults rarely exceed 3.5 to 4 inches in the Gulf of Maine, though adults have exceeded 5 inches in length in Canada.

The killifish has a flattened head and a mouth that opens dorsally, which is adapted for surface feeding. They are omnivorous feeders whose stomach contents have included diatoms, amphipods and other crustaceans, mollusks, fish eggs, small fishes, and vegetation such as eel grass. The females will devour their own eggs.

Killifish have been found to be food sources for otter, mink, brook trout, bullfrogs, and piscivorous birds of prey, dependent upon the habitat overlap of the predator and prey.

These organisms are potential receptor species representing forage fish because they are:

- Resident in this reach of the Penobscot River;
- Exposed to sediments as well as surface water;
- Omnivorous nearshore feeders;
- Prey to other fish, wading birds such as heron, and mammals feeding nearshore.

Killifish therefore represent forage fish in this reach of the river.

Aquatic Birds and Mammals

We have selected great blue heron, osprey, bald eagle, and mink to represent aquatic birds and mammals feeding in the nearshore. The characteristics of these receptors summarized here are used in the exposure assessment in Section 9.4.

Great Blue Heron (*Ardea herodias*)

The great blue heron inhabits salt and freshwater environments, typically shallow waters and shores of lakes, flooded gravel pits, marshes and oceans. In marsh environments, the great blue heron is an opportunistic feeder; they prefer fish, but they will also eat amphibians, reptiles, crustaceans, insects, birds, and mammals. The diet varies but may include up to 100% fish. A Nova Scotia study found 6% forage fish (Atlantic silverside and mummichog), 52.6% eels, and 41.4% other fish in the diet of great blue heron (USEPA, 1993). A food ingestion rate for adult breeding birds of 0.18 g food/g body weight/day has been reported.

Great blue heron tend to forage near nesting sites (USEPA, 1993). A study in Minnesota measured the distance between nesting and foraging grounds to range from 0 to 2.7 miles. A Carolina study found the same distance to be 4 to 5 miles. The maximum distance great blue heron will fly between foraging areas is 9 to 13 miles (USEPA, 1993). The size of the feeding territory in a freshwater area in Oregon was 1.5 acres, while the feeding territory in an estuarine area was 21 acres.

Great blue heron are migratory and are present in Maine from approximately April to October.

Great blue heron were observed feeding in the shallow waters of the intertidal cove adjacent to the site. They are likely to be exposed to contaminants of concern at the site through their diet of fish.

These organisms are potential receptor species because they:

- Have been observed feeding in this reach of the Penobscot River;
- Consume nearshore forage fish;
- Are a higher trophic level predator in the river.

Great blue heron, therefore, represent piscivorous birds in this reach of the river.

Osprey (*Pandion haliaetus*)

Osprey can be found along inland rivers as well as coastal areas. They generally nest near open water with a plentiful supply of fish. They are almost completely piscivorous but will eat birds, frogs, and crustaceans, if available. They generally eat slow-moving fish from near the water's surface or from shallow water. A report of percent fish in the diet by size indicated that 3.3% of fish were 4 inches or less and the remaining fish in the diet were greater than 4 inches in length. A food

ingestion rate of 0.21 g/g/day was reported for adult females during courtship in southeast Massachusetts (USEPA, 1993). Foraging radii ranging from 0.4 to 6 miles have been reported.

Osprey are migratory.

These organisms are potential receptor species because they:

- Consume mostly fish;
- Are a higher trophic level predator in the river.

Osprey, therefore, represent piscivorous birds in this reach of the river.

Bald Eagle (*Haliaeetus leucocephalus*)

The bald eagle is designated as an endangered species by the State of Maine and a threatened species by the federal government.

Bald eagles are generally found in coastal areas, near lakes, or near rivers. Their preferred breeding sites are in large trees near open water. They are usually found in areas with minimal human activity.

According to the US Fish and Wildlife Service Atlantic Coast Ecological Inventory for Bangor, Maine (1980), the Penobscot River from Bangor south to Verona Island is an important wintering area for bald eagles. However, there are no known bald eagle nests in the vicinity of the site (Ladner, 1995), and the Maine Department of Fisheries and Wildlife does not have records of threatened or endangered species in the vicinity of the site (Hall, 1995; Kemper, 1995).

Bald eagles, although primarily carrion feeders, are opportunistic and will eat whatever is plentiful including fish, birds, and mammals. Reported food ingestion rates range from 0.064 to 0.14 g/g/day. A study of adult breeding bald eagles in Connecticut estimated a food ingestion rate of 0.12 g/g/day (USEPA, 1993). A study of bald eagle diets in Maine indicated that their diets consisted of 76.7% fish, 16.5% birds, and 6.8% mammals (USEPA, 1993).

Foraging areas vary according to season and location. In Connecticut, a foraging length of 2 to 4.5 miles along a river has been reported (USEPA, 1993).

These organisms are potential receptor species because they:

- Have a federally endangered status;
- Consume fish;
- Are a higher trophic level predator in the river;
- Are sensitive to contaminants that biomagnify in the food chain.

The bald eagle, therefore, represents predatory birds in this reach of the river.

Mink (*Mustela vison*)

Mink are opportunistic feeders and eat a broad range of vertebrate and invertebrate prey from both terrestrial and aquatic habitats (Chapman and Feldhamer, 1982). Common food items include mammals, fish, birds, amphibians, crustaceans, insects, and reptiles. No single individual food item appears to be consistently more important in the mink diet than another. Studies report diets of 61% to 85% fish, 4% to 11% crustaceans, 2% to 3% amphibians, 5% to 6% birds and mammals, and 1% to 17% vegetation (USEPA, 1993). Measured food ingestion rates range from 0.12 to 0.16 g/g/day (USEPA, 1993).

Mink are reported to have home ranges of 0.6 to 3 miles, with males ranging farther than females (Chapman and Feldhamer, 1982; USEPA, 1993). Foraging areas for females have been reported to be in the range of 20 to 50 acres. Mink are present within their home ranges year round.

Only a few individuals of the mink population would be expected to be present within a foraging area. Mark and recapture studies have found densities of 1 to 3 animals per 100 acres for a river system in Montana. Other studies indicate that individual mink within a population are dispersed over wide areas (Chapman and Feldhamer, 1982).

Mink are a potential receptor species because they:

- Are an omnivorous higher order predator;
- Represent mammals foraging in the nearshore environment;
- Have toxicological data available for their exposure to mercury.

9.3.3.2 Terrestrial Receptors

Soil Invertebrates

Soil invertebrates are useful as receptors because they have the greatest exposure to soil contaminants and are generally restricted in their geographical range. These invertebrates provide food for small mammals and birds in the terrestrial environment. The health and condition of soil invertebrate communities, in terms of species richness and abundance, are a qualitative indicator of local conditions.

Terrestrial Birds and Mammals

American Robin (*Turdus migratorius*)

The American robin breeds in open woodlands, woodland edges, and clearings. Their major foods are wild and cultivated fruits, earthworms, and insects. Their food ingestion rate ranges from 0.89 to 1.52 g/g/day. Their diet varies from 32% soil invertebrates in the summer to 93% in the spring with the remaining percentage consisting of fruit (USEPA, 1993). Their territories range from 0.4 to 2 acres and can reach densities of over 100 birds per 100 acres. Their preferred habitats are grassy fields and orchards. They are migratory and may remain on site from March to November.

These organisms are potential receptor species representing passerines (songbirds) because they:

- Are exposed to soil contaminants through their food;
- Have a small home range;
- Occur on site.

Red-tailed Hawk (*Buteo jamaicensis*)

Red-tailed hawks are common in many types of habitats, including woodlands and old fields. They feed primarily on small mammals but will also eat birds, lizards, snakes, and insects. Food ingestion rates have been reported from 0.086 to 0.11 g/g/day (USEPA, 1993).

Territory sizes vary according to season and location and range from 150 to 4,400 acres. An average home range of 1,700 acres has been reported for wood lots in Michigan (USEPA, 1993).

Red-tailed hawks are migratory.

These organisms are potential receptor species because they:

- Consume small mammals;
- Are a higher trophic level predator;
- Are sensitive to contaminants that biomagnify in the food chain.

The red-tailed hawk, therefore, represents predatory birds in terrestrial habitat in the vicinity of the site.

Short-Tailed Shrew (*Blarina brevicauda*)

The short-tailed shrew is a voracious predator of small invertebrates and other vertebrates. They occur in timbered and open habitat, and are especially common along stream banks and meadows with tall grass. Earthworms are the major diet item but millipedes, insects and occasional mice and voles are also eaten. Reported food ingestion rates range from 0.49 to 0.62 g/g/day. Reported diets consist of 43.4% soil invertebrates and 31.4% earthworms. The foraging range is 0.07 to 0.5 acres (USEPA, 1993), and they are expected to be within that range year round.

Short-tailed shrew are potential receptor species representing small mammals because they:

- Represent small predatory mammals on site;
- have a high exposure to soil contaminants through their ingestion of earthworms and soil invertebrates;
- Are likely to occur in the study area .

9.3.4 Site Conceptual Model and Identification of Exposure Pathways

9.3.4.1 Development of Site Conceptual Model

Figure 9-4 presents the site conceptual model for the ecological risk assessment. Section 6.0 discussed sources of contaminants to environmental media, and Section 7.0 discussed the distribution of contaminants in these media. As demonstrated in Figure 9-4, past disposal practices,

on-going surface water and groundwater discharges, and aerial deposition have resulted in elevated concentrations of COCs, primarily mercury, in site media. The exposure media addressed in the ecological risk assessment are surface water and sediment on site and in the Penobscot River, and surface soils on site. Figure 9-4 represents the migration of COCs from onsite sources to these media and the pathways by which selected receptors are exposed to COCs in these media.

9.3.4.2 Identification of Exposure Pathways

An exposure pathway describes the links between the sources of contaminants and the ecological receptors subject to exposure. The ecological risk assessment identifies potential exposure pathways by considering the sources, the media involved in transport, the potential for bioaccumulation, and the characteristics of the receptors.

The potential links between contaminants and potential ecological receptors in the Penobscot River are:

1. Sediment to benthic organisms in the Penobscot River through direct contact and ingestion;
2. Benthic organisms to pelagic or demersal fish in the Penobscot River through ingestion;
3. Water column to fish in the Penobscot River through direct contact, respiration, and ingestion;
4. Penobscot river fish to piscivorous birds and mammals through ingestion.

On the HoltraChem property, the potential links between contaminants and potential ecological receptors are:

1. Sediment to benthic organisms in the onsite stream and stormwater ditch;
2. Water column to water column invertebrates in onsite stream and stormwater ditch;
3. Soil to invertebrates on the terrestrial portion of the site;
4. Soil invertebrates to mammals and songbirds on the terrestrial portions of the site;
5. Small mammals to birds of prey on the terrestrial portions of the site;

Exposure pathways transport contaminants from environmental media to selected receptors. The likely pathways of exposure at the site are ingestion of sediment, soil, and prey; and direct exposure to sediment and water. For aquatic organisms, ingestion may include direct ingestion of sediment by bottom-feeding or bottom-dwelling organisms, or ingestion of bottom-dwelling organisms and subsequent bioaccumulation through benthic or pelagic food chains. Dermal exposure may result in bioconcentration through the water column, or chemical partitioning between the sediments and infaunal organisms. For terrestrial organisms, ingestion may include direct ingestion of soil during foraging or ingestion of invertebrates and subsequent bioaccumulation through the terrestrial food chain.

9.4 Exposure Assessment

The exposure assessment describes exposure concentrations or doses for the selected receptors. Exposure concentrations are estimates of the concentration of mercury to which the receptors are exposed under site-specific assumptions. An exposure concentration may be used to develop an exposure dose to which a toxic effect is related.

9.4.1 Exposure to Contaminants in Sediments

The sediments in the Penobscot River are a potential exposure medium for three receptors: benthic invertebrates, eels, and killifish. The sediments of the onsite stream and stormwater ditch are a potential exposure medium for benthic invertebrates. These water bodies are unlikely to support fish. The exposure assessment estimates the exposure of receptors in the Penobscot River and onsite stream and ditch to mercury in sediment. It also estimates the exposure of receptors in the Penobscot River to acetone in sediment.

Exposure of benthic organisms is on a station by station basis. Therefore, the measured sediment concentration at each station is used as the exposure concentration. Section 3 provides the methods for measuring sediment mercury. **Table 9-13** summarizes the mercury sediment concentrations. Acetone was detected at 88 ug/kg in one of four Penobscot River sediment samples analyzed for VOCs. Figures 9-2 and 3-6 show the sampling locations in the onsite southerly stream, northern ditch, and southern cove, respectively.

The exposure assessment uses the body burdens of mercury in eels and a composite forage fish sample to estimate exposure of these receptors from all sources (sediment, water, and food chain). This assessment does not provide a discrete estimate of fish exposure via each medium.

The exposure assessment also uses the results of sediment analyses for total organic carbon (TOC), and acid volatile sulfides (AVS) and simultaneously extracted metals (SEM) to estimate the bioavailability, and hence potential exposure, to mercury in the Penobscot River and the southern cove. The ratio of SEM to AVS (SEM/AVS) is indicative of the availability of metals for uptake by biota. The concentration of total organic carbon (TOC) in sediment can also indicate of the availability of mercury for uptake by biota. Some portion of mercury in the natural aquatic environment may be methylated to form the organometallic methylmercury (Me-Hg). Because it has a lower free-energy state (is more stable) when not associated with water (is hydrophobic), methylmercury tends to sorb to the fraction of organic carbon in surface water that is dissolved (DOC) or particulate (POC), or TOC in sediment which includes DOC or POC in the sediment interstitial (pore) water.

Methylmercury is the highly toxic, bioavailable and bioaccumulating form of mercury. The amount of methyl mercury present in the southern cove sediments ranged from 0.004% to 0.38% and averaged 0.16 % of the total mercury present in the sediments. Mercury which is not tightly bound to the sediment could also be considered potentially bioavailable because it could be liberated in the digestive acids in fish or other animals. The amount of these forms of mercury were evaluated using a 1 N HCl extraction fluid which represents a stronger acid than would be found in the digestive tracks of animals. The amount of weakly bound forms of mercury in southern cove sediments ranged from 0.21% to 0.46% and averaged 0.33% of the total mercury present in the sediments. The remaining amount of mercury in the sediment is strongly bound to the sediment particles or is in the insoluble mercuric sulfide (HgS) form. HgS was analyzed directly by extracting the sample with 12 N HNO₃ and analyzing the residual for total mercury by digestion in aqua regia. The only form of mercury not extracted by the 12 N HNO₃ is expected to be HgS. The amount of HgS present in the southern cove sediments ranged from 0.53% to 1.43% and averaged 0.83% of the total mercury present in the sediments. The difference between the amount of total mercury and the amounts calculated from the sequential extraction is the fraction which is so strongly bound to the sediment that it can be extracted only with 12 N HNO₃ and is therefore not likely to be bioavailable. Based on these analyses it appears that only 0.5% of the mercury present in the southern cove is bioavailable and approximately 99.5% of the total mercury is tightly bound.

The hydrophobicity of methylmercury or its tendency to sorb to sediment TOC can be quantified by its octanol/water partition coefficient (K_{ow}). The sorption capacity of site sediment can be determined by the mass fraction of TOC in sediment (foc). For sediments with foc ≥ 0.2% by weight, TOC is often the predominant phase for hydrophobic chemical sorption (USEPA, 1993). The partition coefficient (K_p), or the ratio of mercury concentration in sediment (C_s) to that in pore water (C_w), is given by:

$$K_p = C_s/C_w = f_{oc} \cdot K_{oc}$$

where, K_{oc} is the organic carbon normalized partition coefficient for sediment. K_{oc} and K_{ow} are interpreted as measures of the contribution of hydrophobic forces to sorption under a state of equilibrium (Hassett and Banwart, 1989). Therefore, equilibrium partitioning (EP) between the solid and aqueous phases in whole sediment becomes a necessary component of estimating benthic exposure.

Adams et al. (1985) and others (USEPA, 1994) have observed that bioavailability of hydrophobic chemicals, as expressed in the stressor-response curve or as bioaccumulation, is correlated not to the whole sediment chemical concentration (dry weight), but to the concentration in the pore water fraction of sediment. This implies that pore water is the primary route of exposure for some benthic organisms. Measurement of pore water concentrations, or estimation using total sediment concentrations and relevant sediment properties such as TOC, is then used to quantify the availability of mercury to infaunal organisms. Toxicological data has indicated equality of exposure by water only and whole sediment on a pore water basis, thus, making it technically feasible to evaluate pore water exposure as though it were water only exposure (USEPA, 1993).

DiToro *et al.* (1990) have shown that insoluble metal sulfides (typically iron and manganese sulfide) control metal availability in anoxic sediments by reacting with available divalent metals. These reactive sulfides can be released with acid treatment and are analyzed as acid volatile sulfides. Since the toxicity of metals in sediment correlates with divalent metal activities in interstitial water, the AVS method assumes that if all of the divalent metals are bound to AVS, no toxicity to sediment-dwelling biota will result. A comparison of AVS and SEM concentrations on a molar basis provides a measure of whether all of the SEM are bound to AVS. If the results indicate that SEM are less than AVS, metals are potentially completely bound in insoluble sulfides and are thus not available to biota. In this case, these metals have been largely rendered non-toxic to sediment-dwelling biota. However, if SEM exceeds AVS, there are more metals available than sulfides and metals are bioavailable.

TOC and SEM/AVS analyses can only predict if the metal may be bioavailable in sediment; not whether they are toxic to biota. Other adsorption processes may bind the metal to prevent bioavailability. For example, AVS concentrations have been observed to decrease dramatically near the upper inch of sediment where bioturbation and scour are present. AVS also has been shown to vary seasonally, having lower concentrations in the winter, and higher in the summer.

Table 9-14 presents mercury, TOC, and partition coefficients data from six sediments collected in the southern cove in 1997 (RSD-022, 023, 025, 029, 030 and 031). Based on the physical properties

of the sediment they were identified “granular” or “organic”. The granular sediments (RSD-022, 023 and 025) were comprised of sand and fine gravel, with 0.27 to 0.73% TOC. In contrast, the organic sediments (RSD-029, 030 and 031) consisted of silt/clay and some sand, with 8 to 14% TOC. Whole sediment concentrations ranged from 1.9 mg/kg (RSD-030) and 2.9 mg/kg (RSD-031) to 9.9 mg/kg (RSD-022) total mercury. Whereas, the unfiltered pore water fraction (includes colloidal and dissolved) ranged from 0.000174 mg/L (RSD-031) and 0.000188 mg/L (RSD-030) to 0.004863 mg/L (RSD-022) total mercury; and filtered pore water (dissolved only) ranged from 0.00000248 mg/L (RSD-031) and 0.0000122 mg/L (RSD-030) to 0.000188 mg/L (RSD-022) total mercury.

With the exception of RSD-025 and 030, K_p was observed to increase with increasing TOC (**Figure 9-5**). Samples RSD-022, 023, 029 and 031 demonstrated a strong positive correlation ($r = 0.921$) between TOC in the sediment and the proportion of total mercury in the solid phase versus that in the aqueous phase (filtered or unfiltered pore water), compared to the correlation coefficient for all six samples ($r = 0.637$). Unfiltered pore water had 3-4 orders-of-magnitude less total mercury, and filtered pore water had 4-6 orders-of magnitude less total mercury, relative to whole sediment (see Table 9-14). It is not certain why RSD-025 and 030 had lower partition coefficients compared to the other samples. One explanation might be partitioning between solid and aqueous phases was not at equilibrium (not steady state) in these samples. Assuming a positive correlation between TOC and K_p in site sediments, total mercury should be less bioavailable for exposure of benthic organisms in “organic” sediments (RSD-022, 023 and 025) compared “granular” sediments (RSD-029, 030 and 031).

Table 9-15 provides AVS and SEM data for five sediment sampling locations within the southern cove (RSD-4B-01 in the organic mat, RSD-5A-01 in the sedge bed, RSD-6B-01 in sand and gravel, RSD-7C-01 in fine silt, and RSD8A-01, a subtidal sample; sample RSD-8A-03 is a duplicate of RSD-8A-01). Two sediment samples had SEM/AVS ratios less than “1”: one sample came from the mid-intertidal/sedge bed (RSD-5A-01) and the other came from the lower intertidal/fine silt bed (RSD-7C-01). Sufficient concentrations of acid volatile sulfides are present in these samples to bind metals and reduce their activities in interstitial waters. Since metals are effectively bound by AVS in these areas, the metals are unavailable to organisms in sediment. The bulk mercury concentrations in these samples were 1.4 and 1.7 mg/kg, respectively. Therefore, it is likely that mercury in sedge bed and fine silty sediments in the cove is bound to AVS and unavailable to benthic organisms. These two sediment types cover approximately 2/3 of the area of the southern cove.

Three locations within the cove had SEM/AVS ratios greater than “1”. These locations were in the organic mat/reed bed (5 for RSD-4B-01), sand and gravel (6 for RSD-6B-01), and the subtidal

location within the cove (average of 11.5 for RSD-08-01). The elevated SEM/AVS ratios in these locations suggest that metals in these regions may be available to benthic organisms.

9.4.2 Exposure to Contaminants in Surface Water

The receptors eels, killifish, and benthic invertebrates are exposed to mercury in surface water of the Penobscot River. Benthic invertebrates may be exposed to mercury in the surface waters of the onsite stream and ditch.

The assessment relies on measured surface water concentrations of mercury in the Penobscot River and onsite stream and ditch. VOCs were not detected in Penobscot River surface water.

The VOCs, chloroform and carbon tetrachloride, were detected at low concentrations (chloroform at 44 ug/l and carbon tetrachloride at 3.5 ug/l) in one onsite sample from a catch basin (SSW-10). The catch basin is not an ecological exposure zone and was sampled for VOCs because it receives direct groundwater discharge. However, we have conservatively assumed that these concentrations represent onsite exposure via surface water.

We assessed measured surface water concentrations on a station by station basis. The analytical data were presented in Section 3 of this report. **Table 9-16** presents the measured concentrations of mercury in surface water, and Section 9.6 discusses the results relative to water quality criteria. There are elevated concentrations of mercury in the onsite stream and ditch, (assuming the measured total mercury is dissolved).

9.4.3 Exposure to Contaminants in Soil

This assessment addresses exposure to mercury in soil in an indirect manner. No criteria or guidelines are available with which to evaluate contaminant concentrations and potential effects on soil invertebrates. We used soil mercury concentrations and mercury concentrations (measured and modeled) in earthworms to calculate dietary doses of mercury to invertebrate-eating wildlife for use in a population effects model. The soil concentrations were presented in Section 3 of this report.

Mercury concentrations in onsite surface soil were elevated above surface soil concentrations in the local reference area, Dorothea Dix Park.

9.4.4 Exposure to Contaminants in Food

The food chain is the most likely exposure pathway for the higher trophic level aquatic receptors (great blue heron, bald eagle, osprey) and the terrestrial receptors (songbirds, small mammals, and red-tailed hawk). We have estimated exposure of these wildlife via the food chain using food chain models, a population effects model, and measured concentrations of mercury in their potential prey: eels and forage fish in the Penobscot River for aquatic receptors; and earthworms and small mammals (deer mouse) for terrestrial receptors on the site.

9.4.4.1 Measured Exposure Point Concentrations in Prey - Earthworms

The methods for collection of earthworms and measurement of body burdens of mercury are provided in Appendix G. Figure 9-2 shows the sampling locations. Table 9-17 presents the results of the earthworm and soil analysis. The concentration of mercury in earthworms reflects the mercury concentration in the soil where they were found. Onsite mercury concentrations in soil and earthworms exceeded the concentration measured in a soil and earthworm sample at the reference location in Dorothea Dix Park.

9.4.4.2 Measured Exposure Point Concentration in Prey - Eels

Table 9-4 provides the length, weight, moisture content, and wet weight fillet concentration of mercury in the eels caught offshore of the southern cove and offshore of the northern reference cove. The laboratory data were reported in dry weight and have been converted to wet weight using individual moisture contents reported by the laboratory.

The ranges and mean mercury concentrations in the fillet of eels from the northern reference area (0.271 mg/kg to 0.876 mg/kg, mean = 0.53 mg/kg) and the southern cove (0.259 mg/kg to 0.678 mg/kg, mean = 0.44 mg/kg) are essentially the same, indicating no differential exposure to mercury between the two groups. Note that these two groups occupy different home ranges, based on an eel home range of 1,000 to 1,200 feet (J. McCleave, personal communication, 1995).

These exposure concentrations are similar to recently measured eel concentrations (State of Maine data from B. Mower, 1998) at Winterport on the Penobscot River (0.309 to 0.736 ppm) and at a reference site, the west branch of the Piscataquis River (0.235 to 0.742 ppm). The concentrations are also similar to lake trout body burdens from three non-industrialized lakes in Maine, Sebago Lake, Moosehead Lake, and Big Eagle Lake (Haines and Perry, 1994). These measurements included mean lake trout concentrations ranging from 0.557 mg/kg in Sebago Lake to 1.031 mg/kg in

Moosehead Lake. Lake trout are higher order predators than eels and can be expected to have higher body burdens.

9.4.4.3 Measured Exposure Point Concentration in Prey - Small Mammals

The methods for capturing small mammals on site are provided in Appendix G. Figure 9-2 provides the sampling locations. Table 9-18 presents the analytical results. Only deer mice were caught on site. Their mercury concentrations ranged from 0.060 mg/kg to 0.198 mg/kg (wet weight) from deer mice on site. We captured two deer mice from the reference site with whole body mercury concentrations of 0.087 mg/kg and 0.016 mg/kg (wet weight), and a short-tailed shrew with a whole body mercury concentration of 0.064 mg/kg (wet weight).

9.4.4.4 Food Chain Model

The food chain model in this assessment uses the average, upper confidence limit on the mean, and maximum measured whole body burden of mercury in the food item.

The food chain model uses mercury concentrations in prey tissue samples to estimate a dose (in mg of mercury/kg receptor's body weight/day) to the receptor based on the consumption of food. For each receptor, the model assumes that the intake of mercury from exposure routes other than food is minimal.

The generalized exposure model is:

$$\text{Dose} = (\text{EPCf}) * (\text{If}) * (\text{a})/(\text{fa}) * (\text{MigFac})$$

Dose = receptor's dose of mercury (mg mercury/kg body weight/day, wet weight)

EPCf = exposure point concentration of mercury in food (mg mercury/kg, wet weight).

If = food ingestion rate of the receptor (kg food wet weight/kg body weight/day)

a/fa = onsite foraging area/total foraging area of the receptor. Where this ratio is greater than 1, this is assumed to be 1.

MigFac = migration factor (fraction of the year that the receptor is in the area).

Table 9-19 provides the exposure factors for each of the receptors used in the food chain model. The values were estimated from data provided in USEPA's Wildlife Exposure Factors Handbook (USEPA, 1993) and were described previously in Section 9.3.3.

Tables 9-20 and 9-21 provide the modeled mercury dose for great blue heron, osprey, bald eagle, mink, and red-tailed hawk.

9.4.4.5 Population Effects Foraging Model

We used a population effects foraging model to assess potential exposure and risk to small mammals and songbirds using the shrew and robin as representative species. Freshman and Menzie (1996) provide the details of the model. Appendix G provides a copy of this publication.

The Population Effects Foraging (PEF) model predicts the percent of a wildlife population that will be adversely affected as a result of exposure to contaminated soil at a site. It is a food chain model in that it assumes that the wildlife population will ingest earthworms that bioaccumulate the contaminant of concern. The model takes into account the size of the foraging area, the toxic effects level, which is the resulting reference dose from a soil contaminant concentration above which the individual will see an adverse effect, and the food ingestion rate, of the population of interest. The model proceeds by:

- Choosing a random location on the site for a receptor to begin foraging;
- Calculating the average soil mercury concentration within the receptor's foraging area around the starting location;
- Calculating the mercury tissue concentration in the earthworms in the foraging area;
- Calculating a dose of mercury to a robin or shrew eating the earthworms within the foraging area;
- Comparing the dose to an effect level to calculate a hazard index;
- Storing the hazard index;
- Repeating for every individual in the population;
- Calculating the percentage of the population in which the hazard index exceeds one.

The model calculates dose as:

$$\text{Dose} = (\text{Cfa}) * \text{BAF} * (\text{If})$$

where:

Dose = receptor's dose of mercury (mg mercury/kg body weight/day, wet weight)

Cfa = concentration of mercury in soil within a random foraging area (mg Hg/kg soil)

If = food ingestion rate of the receptor (kg food wet weight/kg body weight/day)

BAF = Soil to food bioaccumulation factor kg soil/kg food.

The BAF is an empirically derived value from the soil and earthworm concentrations measured in 1995 and 1997. We used the average value of 0.27. The standard deviation on this average was 0.13.

We divided the 235-acre HoltraChem site into a 50x31 grid (1550 squares) to model dose to two populations, the shrew and the robin. Table 9-22 shows the model input parameters. The population effects foraging model used soil concentrations based on the output of an air deposition model (see section 3) for most of the site. The concentration grid predicted by the air deposition model was modified with measured data to represent hot spots and the landfill surface. The current conditions assumed that there was no exposure in the plant area because it is a paved or compacted surface (the model grid was filled as background concentrations to achieve this assumption). We believe this to be a reasonable assumption because the paved and compacted nature of the plant area surface precludes it as a food source. Although some birds may inhabit the buildings, they would have to forage for soil invertebrates in areas away from the immediate plant site. Note also, that we conducted model runs assuming very high mercury concentrations. We ran the model under the assumption that the mercury concentrations ranged from background (0.19 ppm) to as high as 1000 ppm in soil.

9.5 Ecological Effects Assessment

The effects assessment identifies the range of toxic endpoints and discusses potential biological effects of contaminants. These toxicological endpoints may include lethality, reproductive

impairment, behavioral modifications, or various sublethal toxic effects. They may also include secondary effects such as loss of habitat.

9.5.1 Approach for Assessment of Ecological Effects

The approach for assessing effects combines field observations of habitat and species, comparison of chemical data for COCs with published criteria and guidelines protective of the receptor groups or associated with ecological effects, laboratory toxicity testing, and comparison of modeled mercury doses with literature values associated with effects.

For aquatic receptors in the Penobscot River, the effects assessment includes:

- the development of mercury doses protective of or associated with toxic effects to piscivorous birds and wildlife;
- identification of criteria and standards protective of fish and other aquatic organisms for comparison to water column and pore water concentrations;
- identification of criteria, standards, or guidelines protective of benthic organisms and fish for comparison to sediment concentrations; and
- sediment toxicity testing.

In assessing potential effects on aquatic receptors in the Penobscot River, the Risk Characterization also addressed general observations of habitat, fish health, and benthic community.

For aquatic receptors in the onsite stream and stormwater ditch, the effects assessment includes:

- identification of criteria and standards protective of fish and other aquatic organisms for comparison to water column concentrations. (Note that these water bodies are unlikely to support fish);
- identification of criteria, standards, or guidelines protective of benthic organisms for comparison to sediment concentrations; and
- the results of a benthic community survey.

For onsite terrestrial receptors, the effects assessment includes:

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the development of mercury doses protective of or associated with toxic effects to birds and wildlife;

In assessing potential effects to onsite terrestrial receptors, the Risk Characterization will also address general observations of habitat and bird and wildlife use of the site.

9.5.2 Development of Toxicity Values for Birds and Wildlife

The effects evaluated in this assessment are lethal effects, reproductive or developmental effects, and other systemic effects. Mercury is the only COC for which we have developed toxicity values. (Tissue analyses were performed only for mercury.)

<u>Effect</u>	<u>Toxicity Value</u>
Lethal effects	Lowest reported or estimated non-lethal dose
Reproductive or developmental effects	Lowest reported or estimated No Observed Adverse Effect Level (NOAEL) for reproductive or developmental effects. Examples of effects to be considered include reduction in eggshell thickness, malformations of young, decrease in litter size, embryotoxicity and reduction in number of eggs per clutch.
Systemic effects	Lowest reported or estimated NOAEL. Examples of effects to be considered include reduction in growth, hepatic enlargement, etc.

These values were developed based on the following sequence of selection criteria:

- If an appropriate state or federal agency has proposed a toxicity value as a criterion for the protection of wildlife, this value is selected;
- In the absence of such proposed criteria and if data are available on NOAELs for the receptor species or for species that are phylogenetically and ecologically similar to the selected

receptor species (e.g., from the same family of birds or mammals), the lowest NOAEL is selected;

- In the absence of appropriate NOAELs, if Lowest Observable Adverse Effects Levels (LOAEL) are available for phylogenetically similar species, these may be selected.
- In cases where NOAELs (or LOAELs) are available as a dietary concentration (e.g., mg contaminant per kg food), a consumption rate for birds or mammals is estimated based on food intake summaries (e.g., Wing, 1956; DeGraaf and Rudis, 1983; USEPA, 1993), and a corresponding daily NOAEL (or LOAEL) dose may be calculated. This consumption rate is expressed as a percentage of the animal's body weight on a wet weight basis or in units of mg/kg/day (mg of food per kg of body weight per day).

9.5.2.1 *Development of a Toxicity Value for Birds*

Eisler (1987) reports that mercury toxicity to birds varies with the form of the element, dose, route of administration, species, sex and age. Acute oral toxicities of various forms of mercury have been reported to range from 2.2 to 31.0 mg/kg body weight for most avian species tested. The lowest of these concentrations represents an LD50 for mallards for methylmercury. These concentrations are roughly equivalent to 0.22 to 3.1 mg/kg/day, assuming an ingestion rate of 10% body weight per day. The proposed U.S. Fish and Wildlife Service criterion for the protection of birds is 0.64 mg/kg/day organomercury based on reduced reproductive ability in grey and ring-necked pheasants. A diet of 1 mg/kg methylmercury resulted in brain and plasma enzyme alterations (Eisler, 1987). This value results in a dietary dose of 0.1 mg/kg/day associated with chronic effects. This assessment incorporates the U.S. Fish and Wildlife Service reproductive effects endpoint into a population effects model to assess potential effect on small mammal populations because reduced reproductive effects are directly applicable to population level effects. We also used the 0.1 mg/kg/day as an endpoint in the model to evaluate potential chronic effects.

This assessment uses 0.64 mg/kg/day as a reproductive benchmark dose and 0.1 mg/kg/day as a benchmark dose for other chronic toxic effects.

9.5.2.2 *Development of a Toxicity Value for Mink*

Several studies summarized in Eisler (1987) have been conducted on Mustelid exposure to methylmercury. Eisler (1987) notes that this group of animals is more prone to mercury exposure due to their diet of fish and other aquatic organisms. Various investigators have reported subchronic and chronic lethal concentrations for mink and river otter of 1 to 5 mg/kg in the diet. Wren et al.

(1987) fed mink a dietary concentration of 0.5 mg/kg methylmercury with no effects on survival. This represents a dose of 0.07 mg/kg/day assuming a food ingestion rate of 0.13 mg/kg/day. We have used this value as a chronic lethal benchmark for mink.

9.5.2.3 *Development of Toxicity Values for Small Mammals*

Small mammals, such as mice, voles, and shrews, may be exposed to mercury in contaminated soil, food, and water. ATSDR (1989) reported a NOAEL for rats exposed to organic mercury of 0.05 mg/kg/day for developmental effects. Eisler (1987) reported subchronic lethal doses of organomercury compounds ranging from 0.25 to 0.5 mg/kg/day for a variety of mammals, and subchronic doses ranging from 0.1 to 0.5 mg/kg/day for reproductive effects in monkey, pigs, and dogs. The proposed US Fish and Wildlife criterion for the protection of mammalian wildlife is 0.25 mg/kg/day (Eisler, 1987) based on reproductive and other chronic effects.

Of these values, we used 0.25 and 0.05 mg/kg/day to assess potential risk to small mammals. The value 0.25 mg/kg/day is a value above which Eisler reported birth defects, histopathology, and elevated mercury tissue level in mammals. The value 0.05 mg/kg/day is a dose at which no ocular effects were observed in rats receiving an administered dose of methylmercuric chloride.

9.5.3 *Observed Effects - Habitat Evaluation*

9.5.3.1 *Evaluation of Mixed Woods Habitat*

It is difficult to assess what effect past and present plant activities might have in limiting wildlife use of the mixed woods cover type. Even in the area north of the plant, where pines were killed off by past plant discharges, there appeared to be healthy regeneration of woodland vegetation and active use of the area by the region's wildlife. Both the areas with pine die off and those which had been logged supported forest edge species, such as chestnut-sided warbler (*Dendroica pennsylvanica*), white-throated sparrow, brown-headed cowbird, and woodchuck. Forest interior species, which require larger tracts of undisturbed habitat were primarily found in the eastern section of the woods. These species include ovenbird, pileated woodpecker, wood thrush (*Hylocichla mustelina*), and red-backed vole (*Clethrionomys gapperi*).

9.5.3.2 *Evaluation of Field Habitat*

The field habitats in the survey area are relatively small and appear not to be large enough to support breeding populations of open field species, such as bobolink (*Dolichonyx oryzivorus*) or Eastern meadowlark (*Sturnella magna*). There also was no evidence of meadow vole (*Microtus*

pennsylvanicus), either from the small mammal traps set in the field or from observations of tunnels and other sign. Voles are normally common in field habitats, but their populations can be cyclical. It is possible that the survey occurred during a natural low period in vole numbers.

9.5.3.3 Evaluation of Wildlife Abundance

There are some qualitative observations from the survey to indicate that some wildlife species may be at lower than expected population levels in the vicinity of the HoltraChem plant. The difficulty in catching small mammals, particularly in the lower field, may indicate that small mammal species have been adversely affected from the past or current operations of the plant. Songbird numbers in the woods, fields and shrublands near the plant also appeared to be somewhat lower than observed in either the reference woods or the relatively undisturbed eastern end of the HoltraChem property. Amphibian populations also appeared to be low in the area, with only the small flooded gravel pit supporting frogs.

9.5.4 Toxicological Endpoints for Fish and Aquatic Invertebrates

Fish and invertebrates may be exposed to mercury in contaminated sediment, water and food. The nature of exposure will depend on the behavior and habitat of the species in the Penobscot River or onsite stream and stormwater ditch. Bottom-dwelling fish observed near the site, such as eels and suckers, may be primarily exposed to mercury from the sediment. However, there appear to be little data to establish relationships between mercury concentrations in sediment and adverse effects in fish.

We used USEPA ambient water quality criteria (AWQC) to evaluate mercury exposure of fish and invertebrates through surface water. The AWQC are not intended to protect 100 percent of the organisms within an aquatic ecosystem; rather, they are intended to "protect most species in a balanced, healthy aquatic community" (USEPA, 1987). Acute and chronic AWQC are available for both marine and freshwater habitats.

This assessment uses both freshwater and marine AWQC to assess aquatic risks because the Penobscot River salinity ranges from fresh to slightly brackish. We have used freshwater criteria for the onsite stream and ditch. The acute and chronic freshwater AWQC for mercury are 1.4 and 0.77 ug/l, respectively. The acute and chronic marine AWQC for mercury are 1.8 and 0.94 ug/l, respectively.

We used USEPA's Lowest Observed Effect Level (LOEL) divided by a safety factor of ten to assess risk to fish and invertebrates from exposure to chloroform and carbon tetrachloride in onsite surface

water. AWQC are not available for these compounds, but EPA has provided LOEL for both chloroform and carbon tetrachloride. The resultant freshwater toxicity factors for chloroform are 2,890 ug/l (acute) and 124 ug/l (chronic). The resultant freshwater toxicity factor for carbon tetrachloride is 3,520 ug/l (acute). USEPA provides no chronic LOEL for carbon tetrachloride.

9.5.5 Toxicological Endpoints for Benthic Invertebrates

The National Oceanic and Atmospheric Administration (NOAA) has reported ranges of contaminant concentrations in sediment associated with effects to benthic invertebrates (Long and Morgan, 1990; Long et al., 1995). NOAA assembled data on concentrations of inorganic and organic contaminants in sediments from many sources. From this information, effects-based sediment criteria were calculated using a variety of methods, including the Equilibrium Partitioning (EP) approach; the spiked-sediment bioassay approach, which involves exposing organisms to pristine sediments spiked in the laboratory with known amounts of chemicals and observing mortality and/or sublethal effects; and several methods that evaluate concurrently collected sediment and biological data. Using the concentrations observed or predicted by these methods, they then calculated the lower 10th percentile (Effects Range-Low, ER-L) and 50th percentile concentrations (Effects Range-Median, ER-M) at which effects have been observed. The data used by Long and Morgan (1990) included marine, estuarine, and freshwater sediments. Long et al. (1995) updated these ranges for marine and estuarine sediments which resulted in slightly different ER-L and ER-M values. This assessment uses both sets of values for the Penobscot River and the values developed by Long and Morgan (1990) for the onsite stream and ditch.

They are:

	ER-L	ER-M
Long and Morgan (1990)	0.15 mg/kg	1.3 mg/kg
Long et al. (1995)	0.15 mg/kg	0.71 mg/kg

These NOAA documents do not propose either the ER-L or ER-M as sediment criteria. However, the values presented in the document provide some perspective on levels of sediment contamination and may be used to assess contaminant levels in a qualitative manner.

There are no criteria or guidelines for acetone, the only VOC detected southern cove sediments.

Because effects concentrations found for pore water are equal to those found in water only exposures (USEPA, 1993; 1994), and because exposure to whole sediment is principally through the pore water fraction (USEPA, 1994; Adams et al., 1985), the effects assessment evaluated effects on benthic invertebrates by comparing pore water concentrations to AWQC.

Effects concentrations used in the ERA for total mercury or methylmercury were obtained from USEPA (1985) as follows:

Form of mercury	Final Chronic Value (FCV)	Final Residue Value (FRV)
Mercury II	1.302 ug/L	0.2 ug/L
Methylmercury	not available	0.012 ug/L

The final chronic value (FCV) for freshwater of 1.302 ug/L mercury II was used to estimate direct effects on benthic invertebrates. Whole sediment with concentrations above this in pore water may cause adverse effects in benthic invertebrates exposed to the sediment. Final residue values (FRVs) for freshwater of 0.20 ug/L mercury II, and 0.012 ug/L methylmercury, were used to estimate levels in pore water below which benthic invertebrates exposed to whole sediment would not present a consumption hazard to humans. The FRVs derived by USEPA (1985) and used in the ERA were based on the U.S. Food and Drug Administration (FDA) action level 1.0 mg/kg mercury. Overall, the freshwater values used in the ERA were lower (more protective) than saltwater values in USEPA (1985). FCV and FRVs were specifically compared to total mercury (mercury II) or methylmercury chemical data for pore water.

9.5.6 Solid Phase Toxicity Testing with Penobscot River (Southern Cove) Sediments

We collected sediment samples from the southern cove on the Penobscot River for sediment toxicity testing during the field phase of the original assessment on June 6, 7, and 8, 1995, and during two subsequent sampling rounds, December 17, 1997, and July 30, 1998. The testing was solid phase ten day toxicity testing with a freshwater amphipod, *Hyallela azteca*. The ten day test is chronic for *Hyallela azteca* (Ingersoll and Nelson, 1990).

Sensitivity Of Hyallella and Amphipods Generally As a Test Organism in The Penobscot River

Hyallella is the standard EPA sediment toxicity test organism, and we believe that it is sufficiently sensitive to mercury. This test organism was not present in our benthic samples from September 1994 and June 1995 which showed low abundances of oligochaetes and chironomids in most sediment samples taken from the Penobscot River. However, *Hyallella* is at least as sensitive and probably more sensitive a test organism than the native fauna which we observed.

We reviewed the literature regarding the sensitivity of *Hyalella*. This review incorporates marine and freshwater studies which is appropriate given the hydrographically transitional nature of the river in the area sampled.

The *Guidance Document on Sediment Toxicity Tests and Bioassays for Freshwater and Marine Environments* (Hill et al., 1993) presents a range of freshwater and marine test species previously used in sediment testing. It provides assessments of their suitability for use in sediment bioassay and toxicity testing, using sensitivity as one of the selection criteria. Specific criteria used to evaluate sensitivity include: sensitivity to toxicants relative to other species; sensitivity of life-stage tested in relation to exposure; and discriminatory ability for a wide range of test chemicals. The guidance document scores test organisms for sensitivity as:

- 1 - low compliance with the criterion;
- 2 - reasonable compliance; or
- 3 - high compliance.

Freshwater test organisms assessed with a 3, showing high compliance with sensitivity criteria, include but are not limited to chironomids (*Chironomus riparius* and *C. tentans*), the amphipod (*Hyalella azteca*) and the cladoceran (*Daphnia magna*). On the other hand, oligochaetes (*Tubifex tubifex* and *Lumbriculus variegatus*) have a sensitivity assessment of 1, showing low compliance with sensitivity criteria.

Marine test organisms with a sensitivity assessment of 3 include but are not limited to the amphipod (*Leptocheirus sp.*). The eight other amphipods listed as marine test organisms have sensitivity assessments of 2, showing reasonable compliance with sensitivity criteria.

In addition, Hill et al. (1993) recommend specific test species based on the overall assessment of a number of selection criteria including sensitivity to toxicants. For marine sediments, an amphipod (preferably *Corophium volutator*) is recommended as the standard test organism. For freshwater sediments, recommended species include chironomids (*C. riparius* and *C. tentans*) and the amphipod (*H. azteca*). According to Hill et al. (1993), for estuarine sediments, the amphipods (*H. azteca* and *C. volutator*) are likely to be suitable test species.

Hill's analysis is significant in that the most abundant organism observed among the infauna in the reach of the Penobscot under study are oligochaetes and chironomids which are ranked as equally sensitive or less sensitive than *H. azteca*.

Long and Morgan (1990) present the sediment effects data for mercury which were used to establish the ER-L and ER-M values. The second lowest mercury concentration (0.08 ppm), detected in sediment at Waukegan Harbor, Illinois, resulted in a "highly toxic" effect to the amphipod (*H. azteca*). The fourth lowest mercury concentration (0.17 ppm), detected in sediment at Lake Union, Washington, resulted in 95 percent mortality in amphipods. As shown by the sediment effects data, some of the lowest concentrations of mercury detected in sediment were associated with adverse biological effects in the amphipod, demonstrating the sensitivity of this test organism to mercury toxicity in particular.

Although not based on sediment toxicity, Ambient Water Quality Criteria for Mercury-1984 (USEPA, 1985) also demonstrate the general sensitivity of amphipods to mercury. The USEPA ranked twenty-eight freshwater species from most resistant to most sensitive based on genus mean acute toxicity values for mercury. The amphipod (*Gammarus* sp.) was ranked second to cladocerans (*D. magna* and *D. pulex*) as the most sensitive species.

Data from the Briefing Report to the EPA Science Advisory Board on the Equilibrium Partitioning Approach to Predicting Metal Bioavailability in Sediments and the Derivation of Sediment Quality Criteria for Metals (USEPA, 1994a) also demonstrates the general sensitivity of amphipods to toxic contamination. In laboratory saltwater sediment spiking experiments with metals using the amphipod (*Ampelisca abdita*), this estuarine, tube-building, infaunal amphipod is commonly used in sediment toxicity testing because it is considered a relatively sensitive species unable to avoid the sediment.

In a comparison of the relative sensitivity of three freshwater benthic macroinvertebrates to ten contaminants (Phipps et al., 1995), the amphipod (*H. azteca*) showed the lowest water-only, 10-day LC50 values for eight of the ten contaminants, including five metals. In general, the chironomid (*C. tentans*), was the next most sensitive species relative to the oligochaete (*Lumbriculus variegatus*) with respect to the ten contaminants tested. Similarly, West et al. (1993) found the amphipod (*Hyaella azteca*) to be the most sensitive to the lethal effects of test sediments contaminated with copper, followed by the chironomid (*C. tentans*) and then the oligochaete (*L. variegatus*).

Becker et al. (1995) compared the performance of the amphipod (*H. azteca*) and the chironomid (*C. tentans*) in evaluating sediment toxicity. Ten-day freshwater sediment toxicity tests were used to evaluate the following biological end points: survival, biomass and body length. Significant concordance was found among end points of the two tests with respect to relative toxicity of sediments.

The USEPA document *Methods for Measuring the Toxicity and Bioaccumulation of Sediment-associated Contaminants with Freshwater Invertebrates* (USEPA, 1994b) used several criteria for selecting the test organisms presented in the current edition of the manual. The existence of a database demonstrating relative sensitivity to contaminants was one of the primary criteria used in selecting *H. azteca*, *C. tentans* and *L. variegatus* for the manual. The document presents a brief review of the literature demonstrating the relative sensitivities of these three test organisms. In most cases, *H. azteca* was found to be the more sensitive test species in the evaluation of sediment toxicity; however, in one study, *C. tentans* was usually comparable to or more sensitive than the amphipod to the pesticides tested. *L. variegatus* has proven to be far less sensitive to sediment toxicity than either *H. azteca* or *C. tentans*.

Toxicity Testing, June, 1995

The 1995 samples were from surficial sediments (approximately the top 3 centimeters). Samples subjected to toxicity testing came from the intertidal silt bed in the cove. Sample locations are on Figure 3-6. The bulk mercury concentrations from these two stations were (RSD-07A, 2 ppm and RSD-07B, 1.9 ppm).

We collected samples for toxicity testings simultaneously with samples for mercury analysis. Therefore, the sample locations for toxicity testing were selected before the results of the mercury analysis were available and we did not collect a sample for toxicity testing from every sediment sample location. As a result, samples collected for toxicity testing were not necessarily from locations with the highest surface mercury concentrations. We attempted to compensate for this by collecting more samples for toxicity testings than necessary and by collecting from locations near the outfall of the northern stormwater ditch where mercury concentrations were expected to be the highest.

The samples which we submitted for toxicity testing were those samples with the highest measured mercury concentrations from among the bioassay samples collected and held. These samples corresponded to the locations where two of the highest concentrations of mercury were detected in deeper sediments. Based on the sampling data, the extent of the higher mercury concentrations appears to be limited to coarse grained sediment near the mouth of the stream and ditch. Although mercury levels in some deeper samples were higher than surface samples tested, we used surface samples only for toxicity testing since this is where the majority of ecological exposures would occur.

Our intention in sampling the surface sediments for toxicity was to recreate a realistic exposure zone, undiluted by sediments upon which the organisms do not feed. We believe that sampling the top 3

centimeters is a reasonable representation of the exposure zone for freshwater benthic infauna. The tested organism was *Hyalella*, and the most common infaunal invertebrates in the southern cove and offshore areas were chironomids and oligochaete. These organisms may burrow from 8 to 20 centimeters, depending on type, but generally feed within the top several centimeters. The following descriptions of their distribution within the sediment demonstrate that their likely exposures due to feeding are on or near the surface of the sediments.

The life cycle of chironomids is dominated by the larval stage. The larvae are usually found in the upper 8-10 cm of sediment. Larvae may filter feed, but the majority of their diet comes from surface deposit feeding. They usually scrape the top 1-2 mm of sediment around the openings of their burrows and feed on detritus and bacteria attached to sediment particles. They may feed on pieces of higher vegetation as well (McCall and Tevesz 1982).

The amphipods such as *H. azteca* (used in toxicity testing) and *Pontoporeia* (found in the Penobscot River) live in the top 2 cm of sediment, but most are found in the top 1-cm. They are surface deposit feeders with a diet of bacteria and algae (McCall and Tevesz 1982).

Oligochaetes are the deepest feeding organisms. They mostly found in the top 20 cm of sediment, but they feed mainly in the upper 2-8 cm. They ingest silt and clay particles and feed mainly on attached bacteria (McCall and Tevesz 1982).

The laboratory control station for the 1995 sampling was Mashpaug Pond in Union Connecticut, not the North Cove on the Penobscot River. The aquatic toxicology laboratory, New England Bioassay, routinely uses this station as a method control because the sediments from this pond are low in organic and metal contamination. Table F-1 in Appendix F provides recent analytical data from the pond.

Details of the sample collection methods, sampling rationale, and laboratory methods for these tests are provided in Appendix F. New England Bioassay tested eight replicates at each concentration for growth and survival, following EPA test methods. The results (Appendix F) indicated that there was no measurable toxicity associated with these sediments from the southern cove. Survival and growth were not significantly different from the control reference sediment at a p value > 0.05 (see Appendix F for statistical output).

Results were:

Sample June, 1995	% Survival	Growth in mg	Statistically Significant from Control
Reference Control	98	0.34	No
RSD-07A	91	0.37	No
RSD-07B	89	.031	No

These data indicate that the sediment does not exhibit toxicity based on the solid phase testing using *Hyalella*. However, we conducted further toxicity testing during December, 1997 in an attempt to obtain and test sediment samples over a wider range of mercury concentrations sediment depths.

December, 1997 Toxicity Testing

Aquatec Biological Sciences conducted the 1997 toxicity testing. Appendix F provides the details of methods and results. These samples were from stations shown on Figure 3-6. The bulk mercury concentrations from these six stations were (RSD-022, 9.9 ppm; RSD-023, 8.5 ppm, RSD-029, 5.5 ppm; RSD-031, 2.8 ppm; RSD-025, 2 ppm; and RAD-030, 1.9 ppm). The samples were from sediment depths of approximately 0 to 30 cm.

The laboratory control sample was a formulated sediment made according to EPA protocols (see Appendix F).

Aquatec Biological Sciences tested eight replicates at each concentration for growth and survival, following EPA test methods. The results (Appendix F) indicated that there was no measurable toxicity associated with these sediments from the southern cove. All samples tested had greater survival and growth than the laboratory control sample. Therefore, Aquatec did not perform any statistical analyses.

Results were:

Sample – December, 1997	% Survival	Growth in mg
Control Sediment	94	0.08
RSD-022	95	0.12
RSD-023	100	0.11
RSD-025	98	0.12
RSD-029	99	0.13
RSD-030	96	0.12
RSD-031	98	0.12

These data indicate that the sediment does not exhibit toxicity based on the solid phase testing using *Hyalella* over a range of mercury concentrations in sediment from 0.19 to 9.9 ppm.

July, 1998 Toxicity Testing

Aquatec Biological Sciences conducted the 1998 toxicity testing. Appendix F provides the details of methods and results. These samples were from stations RSD-045 and RSD-032 in the southern cove. The bulk mercury concentrations from these two stations were (RSD-045, 48 ppm; RSD-032, 0.14 ppm).

The laboratory control sample was a formulated sediment made according to EPA protocols (see Appendix F).

Aquatec Biological Sciences tested eight replicates at each concentration for growth and survival, following EPA test methods. The results (Appendix F) indicated that there was no measurable toxicity associated with these sediments from the southern cove. All samples tested had greater survival and growth than the laboratory control sample.

Results were:

Sample – July, 1998	% Survival	Growth in mg
Control Sediment	98	0.0131
RSD-045	100	0.136
RSD-032	100	0.156

These data indicate that the sediment does not exhibit toxicity based on the solid phase testing using *Hyalella* over a range of mercury concentrations in sediment from 0.14 to 48 ppm.

9.6 Risk Characterization

This assessment uses a "weight-of-evidence" approach to characterize risks associated with the presence of the contaminants of concern, principally mercury, to the study areas. This approach involves incorporating field observations with information from the literature and selected predictive techniques.

This assessment makes a qualitative and quantitative assessment of risks to ecological receptors with regard to toxic effects. It uses information generated from the Exposure Assessment and Ecological

Effects Assessment and relies upon field observations as well as the toxicity quotient approach to provide an overall weight-of-evidence concerning the nature of risks.

Field data gathered from surface water, sediment, soil, and biota, and modeled projections of dietary doses to higher trophic level receptors are used to assess the health effects and condition of receptors that may be related to the presence of contaminants.

The risk characterization employs a toxicity quotient approach using estimated toxicity benchmark mercury doses for wildlife. We recognize that the toxicity quotient alone does not determine risk, and that other lines of evidence such as onsite field observations should also be incorporated into the general weight-of-evidence.

The direct numerical comparison of a measured or modeled exposure concentration or dose to such a benchmark is a toxicity quotient.

$$\text{Toxicity Quotient} = \frac{\text{Modeled Dose or Concentration}}{\text{Benchmark Dose or Concentration}}$$

We interpret the resulting quotient using the following protocol:

- Toxicity Less Than "1" - no risk;
- Toxicity Quotient Exceeds "1" but less than "10" - small potential for environmental effects;
- Toxicity Quotient Exceeds "10" but is less than "100"- significant potential for environmental effects;
- Toxicity Quotient Exceeds "100" - effects may be expected since this represents an exposure level at which effects have been observed in other species.

Note that the toxicity quotient method provides some insight into general effects upon individual animals in the local population. If effects are judged to be at the average individual level, they are probably insignificant at the population level. However, if risks are present at the individual level they may or may not be important at the population level.

9.6.1 Risk Characterization in the Penobscot River

Using a weight-of-evidence approach, we base risk characterization in the Penobscot River on three assessment endpoints (Section 9.2.2):

- maintenance of piscivorous bird and wildlife populations;
- health and maintenance of local fish populations; and
- ability of the benthic invertebrate community to support fishery resources.

9.6.1.1 Evaluation of Assessment Endpoint: Maintenance of Piscivorous Bird and Wildlife Populations

Measurement Endpoint: Modeled Doses of Mercury to Piscivorous Birds and Mink

Table 9-19 compares the estimated mercury dose to great blue heron, bald eagle, and osprey to the toxicity benchmarks for birds of 0.64 mg/kg/day wet weight for reproductive effects and 0.1 mg/kg/day wet weight for other chronic systemic effects. The site-related doses are based on measured concentrations in eels and forage fish from the Penobscot River and deer mice from the site. Section 9.4 described how these doses were calculated, and Section 9.5 described the development of the toxicity benchmark.

Reproductive effects toxicity quotients for great blue heron, osprey, and bald eagle using average, upper confidence limit, and maximum calculated mercury doses are less than "1". This indicates that the site does not present a risk of reproductive effects to these species feeding in the vicinity of the site.

Toxicity quotients for other chronic systemic effects to great blue heron and bald eagle using average, upper confidence limit, and maximum calculated mercury doses are less than "1". This indicates that the site does not present a risk to these species feeding in the vicinity of the site.

The toxicity quotients for the osprey are less than "1" indicating that the site does not pose a risk to this species feeding near the site.

Table 9-19 also compares the estimated mercury dose to a mink toxicity benchmark of 0.07 mg/kg/day wet weight. The site-related doses are based on measured concentrations in eels and forage fish from the Penobscot River and deer mice from the site. Section 9.4 described how these

doses were calculated, and Section 9.5 described the development of the toxicity benchmark. Toxicity quotients are less than "1" indicating that the site does not pose a risk to mink feeding on and near the site.

9.6.1.2 Evaluation of Assessment Endpoint: Health and Maintenance of Local Fish Populations

Measurement Endpoint: Comparison of Water Column Concentrations of Mercury to Criteria

Table 9-16 compares mercury data for surface water samples collected from the Penobscot River and in the southern cove to freshwater and marine ambient water quality criteria (AWQC). These surface water samples had mercury concentrations lower than the acute AWQC for fresh and marine waters.

In addition, recent sampling of surface water which distinguished between total and dissolved mercury indicates that dissolved mercury in the water column does not exceed the AWQC of 0.77 ug/l. In this study, Acheron (Acheron draft data report, 1998) obtained mercury samples from upriver and down river of the HoltraChem site and from turbid and clear water to assess the impact of resuspension of sediment to surface water quality. The Acheron sampling locations included clear and turbid samples from:

two locations off the southern cover;

two locations off the north ditch and outfall 001 area; and,

three upriver locations (one below the gully north of Landfill 3, one north of the northerly streams outfall, and one south of the Hampden Marina).

The highest total mercury concentration (0.412 ug/l) was detected in a sample from upriver of the HoltraChem site. None of the samples exceeded the AWQC of 0.77 ug Hg/l. There was an apparent correlation between total suspended solids and total mercury, and the near plant turbid water samples did not contain more mercury than the upriver turbid water samples.

The highest concentrations of dissolved mercury were observed in the samples collected off the southern cove (0.00603 and 0.00684 ug/l) and off the north ditch/001 outfalls (0.00997 and 0.0101 ug/l) but both were below the AWQC of 0.77 ug/l. These data show that the sediments which resuspend in the vicinity of the southern cove do not impact the surface water quality to a greater extent than the resuspension of sediment upriver. In addition, although the dissolved mercury

concentrations were greater in the turbid samples in the vicinity of the HoltraChem site, none of these samples exceeded 0.77 ug/l

9.6.1.3 Evaluation of Assessment Endpoint: Ability of the Benthic Invertebrate Population to Support Fishery Resources

Measurement Endpoint: Sediment Bioassays

Solid phase toxicity tests conducted during June and July, 1998 and December, 1997, indicated that the sediments in the south cove do not exhibit sediment toxicity at mercury concentrations up to 37 ppm and over a range of sediment depths to 30 cm. These data indicate little potential for risk to sediment dwelling organisms.

Measurement Endpoint: Abundance and Diversity of Local Benthic Community

Benthic samples collected in 1994 and 1995 generally exhibited low population densities and low diversities off-shore of the site and off-shore of the north cove reference area. These observations agree with the historical observations of scarce benthic fauna in the reach of the Penobscot River from Winterport to Brewer. We note that the most abundant and diverse samples were from the southern cove adjacent to the site. This observation is consistent with the variety of habitat and bottom type in the southern cove which leads to an expectation of higher population abundance relative to other areas of the river. However, there are no other similar areas away from the influence of the HoltraChem facility. In fact, the southern cove with its freshwater tidal vegetation beds is a relatively unique habitat for this reach of the river. One expects such a diverse area to support a more abundant fauna than the other areas and the observations meet this expectation. The relatively abundant benthic fauna in the southern cove indicate little potential risk to sediment dwelling organisms.

Measurement Endpoint: Comparison of Sediment Mercury Concentrations to NOAA Guidelines

Table 9-13 is a summary of mercury data for sediment samples collected from the Penobscot River and in the southern cove. Most of the Penobscot River surficial sediment samples had mercury concentrations exceeding the ER-L value of 0.15 mg/kg.

Over half of the surficial sediments samples exceeded the ER-M of 1.3 mg/kg.

Long et al. (1995) published an updated ER-M for mercury of 0.71 mg/kg for estuarine and marine sediments (the ER-L of 0.15 mg/kg remained unchanged). Three additional surficial sediment

samples would exceed the ER-M based on this lower value. These data indicate some potential for risk to sediment swelling organisms.

One VOC, acetone, was detected at a concentration of 0.88 ug/kg in one of four Penobscot River sediment samples analyzed for these compounds. There are no criteria or guidelines for this compound in sediment. However, the concentration is low and the compound was detected only once in deeper sediment. Therefore, the potential for ecological exposure to this compound is low.

Measurement Endpoint: Potential for Exposure to Metals Based on SEM/AVS Ratios

Table 9-14 provides AVS and SEM data for five sediment sampling locations within the southern cove (RSD-04B-01 in the organic mat, RSD-05A-01 in the sedge bed, RSD-06B-01 in sand and gravel, RSD-07C-01 in fine silt, and RSD-08A-01, a subtidal sample; sample RSD-08A-03 is a duplicate of RSD-08A-01). Two sediment samples had SEM/AVS ratios less than "1": one sample came from the mid-intertidal/sedge bed (RSD-05A-01) and the other came from the lower intertidal/fine silt bed (RSD-07C-01). Sufficient concentrations of acid volatile sulfides are present in these samples to bind metals and reduce their activities in interstitial waters. Since metals are effectively bound by AVS in these areas, they are unavailable to organisms that may be exposed to metals in the sediment. Therefore, it is likely that mercury in sedge bed and fine silty sediments in the cove is bound to AVS and unavailable to benthic organisms. These sediment types represent approximately 2/3 of the area of the cove.

Three locations within the cove had SEM/AVS ratios greater than "1". These locations were in the organic mat/reed bed (5 for RSD-04B-01), sand and gravel (6 for RSD-06B-01), and the subtidal location within the cove (average of 11.5 for RSD-08-01). The elevated SEM/AVS ratios in these locations suggest that metals in these regions may be available to benthic organisms.

9.6.2 Risk Characterization in the Onsite Stream and Stormwater Ditch

Using a weight-of-evidence approach, we base risk characterization in the onsite stream and stormwater ditch on the assessment endpoint (Section 9.2.2):

viability of the benthic invertebrate community.

9.6.2.1 Evaluation of Assessment Endpoint: Viability of the Benthic Invertebrate Community

Measurement Endpoint: Observations of Benthic Communities Relative to a Reference Stream

The northern ditch and southerly stream that drain areas of the HoltraChem property where plant activities occur exhibit few benthic invertebrate organisms and low species diversity. There was a similar paucity of infaunal invertebrates in the reference stream. The northern ditch and southerly stream also had few macroinvertebrate organisms in contrast to the reference stream which had several macroinvertebrate species.

The onsite stream at the northern end of the site exhibited a more abundant benthic infaunal community and macroinvertebrate community than the southerly stream, northern ditch or the reference stream.

Measurement Endpoint: Comparison of Water Column Concentrations to Criteria

For ecological risk assessment purposes, Table 9-16 compares mercury data for surface water samples from the northern ditch and southerly stream to freshwater ambient water quality criteria (AWQC). Several of these samples had total mercury concentrations exceeding the chronic freshwater AWQC. However, the most recent recommended ambient water quality criteria (EPA 12/10/98) are expressed as dissolved mercury. Ambient water quality criteria are not applicable to stormwater conveyances. Surface water samples collected from the farthest upstream areas in the northwestern region of the site, such as from the flooded gravel pit and from the portion of the southerly stream flowing out of the flooded gravel pit, were found to have mercury concentrations lower than the method detection limit of 0.2 ug/l. Several locations along the northern ditch and southerly stream had total mercury concentrations greater than the acute freshwater AWQC indicating a potential for acute biological effects in exposed aquatic organisms.

The northern ditch and southerly stream exhibited elevated conductivities higher than the reference stream and unusually high for freshwater. The onsite stream's conductivity was typical of freshwater and similar to the reference stream. The conductivity of a water body is a fundamental parameter in determining the ability of organisms to inhabit it because the specific conductance of freshwaters is closely proportional to the concentrations of the major ions. For example, Wetzel (1983, p182) presents data demonstrating that the concentrations of calcium, magnesium, sodium, potassium, carbonate, sulfate, and chloride increase proportionally with increasing conductivity. That is, as conductivity increases, so does salinity. Freshwater conductivities in general do not exceed

approximately 0.01 to 0.330 millisiemens/cm (Welch, 1952). Freshwater organisms are isotonic with their environment and tend to be stenohaline (i.e. they are adapted for survival only over the narrow range of salinities represented in freshwater Purves and Orians, 1987, p. 666). Freshwater organisms do not have the ability to survive at increased conductivities. Therefore, the increased conductivities in the southern and northern stormwater streams may be precluded the existence of an abundant flora and fauna.

For chloroform and carbon tetrachloride, we used derived toxicity factors because USEPA does not provide AWQC for these compounds (Section 9.5.4). The toxicity factors derived in Subsection 9.5.4 were:

chloroform - 2,890 ug/l (acute) and 124 ug/l (chronic);

carbon tetrachloride - 3,520 ug/l (acute).

The onsite surface water concentrations were 44 ug/l for chloroform and 3.5 ug/l for carbon tetrachloride at the stormwater catch basin where VOCs were measured. This indicates no potential for risk to onsite benthic invertebrates from exposure to these compounds.

Measurement Endpoint: Comparison of Pore Water Concentrations of Mercury to USEPA Final Chronic Value or Final Residue Values

An overall comparison of total mercury or methylmercury detected in pore water (Table 9-14), to levels protective of benthic invertebrates (Section 9.5.5 - FCV = 0.001302 mg/l mercury II, FRV = 0.00020 mg/l mercury II, FRV = 0.000012 mg/l methylmercury), indicated only unfiltered pore water in RSD-022, 023, 025, and 029 exceeded protective levels (mercury II only), and could pose a potential risk to benthic invertebrates. Unfiltered pore water in RSD-022 and 025 had total mercury concentrations which exceeded both the FCV and FRV for mercury II, but RSD-023 and 029 exceeded only the FCV for mercury II.

More importantly, filtered pore water concentrations or the bioavailable (dissolved) fraction of the samples, were always below protective levels. Therefore, this more accurate, less conservative, result indicated no or low potential risk to benthic invertebrates for chronic (biological) or residue (food chain) effects. This result was supported by findings of no biological, and no statistically significant, effects on freshwater amphipods following 10 day exposures to sediment sampled December 1997 (see Solid Phase Toxicity Testing in Section 9.5.6). In addition, no biological and no statistically significant effects were observed in toxicity tests using sediment collected June 1998 in which higher concentrations of mercury were detected.

Measurement Endpoint: Comparison of Sediment Mercury Concentrations to NOAA Guidelines

For ecological risk assessment purposes Table 9-13 compares sediment mercury concentrations in the onsite stream and ditch to NOAA guidelines. These guidelines are not applicable to stormwater conveyances. Most of these samples had mercury concentrations exceeding the both the ER-L and ER-M for mercury. The only location where the mercury concentration in the surficial sediment sample was less than the ER-L was from the flooded gravel pit at the head the southerly stream. However, this comparison to the NOAA ER-L and ER-M should be interpreted in light of the low bioavailable fraction of mercury in sediments as shown by the extraction and speciation procedures described in section 9-4.

9.6.3 Risk Characterization in Onsite Terrestrial Environment

Using a weight-of-evidence approach, we base risk characterization onsite on the assessment endpoint (Section 9.2.2):

maintenance of bird and wildlife populations onsite.

9.6.3.1 Evaluation of Assessment Endpoint: Maintenance of Bird and Wildlife Populations Onsite

Measurement Endpoint: Comparison of Onsite Observations of Bird and Wildlife Populations to a Reference Area

From the information obtained from the habitat and wildlife survey, it is apparent that the HoltraChem site contains a variety of terrestrial and aquatic habitats which support a diverse flora and fauna. Vegetation growth appears healthy throughout most of the property, with formerly disturbed areas, such as the stand of dead pines on the north side of the plant, revegetating with healthy-appearing new growth. Area wildlife appear to be using most available habitats present on the site.

Comparisons between areas immediately adjacent to the HoltraChem plant and the undisturbed reference areas indicate that no major differences in wildlife usage were observable. However, qualitative observations indicate that some smaller scale differences in population numbers and activity may be present. The level of resolution of the survey was not detailed enough to monitor population or productivity trends.

While it was not possible to collect detailed information on the health of wildlife populations during the single week-long survey and sampling period, the distribution and abundance of the species observed does not indicate any signs of any major habitat degradation. Although a number of areas have been affected by the activities of the HoltraChem plant, comparisons within undeveloped areas of the property and the reference areas indicate that wildlife and fish populations continue to use the ecological resources of the HoltraChem site.

Measurement Endpoint: Modeled Doses of Mercury to Robin, Short-Tailed Shrew, and Red-Tailed Hawk

Table 9-21 compares the estimated dose of mercury to red-tailed hawks to toxicity benchmarks. The site-related doses are based on measured concentrations in deer mice for the hawk. Section 9.4 described how these doses were calculated, and Section 9.5 described the development of the toxicity benchmarks.

Toxicity quotients for reproductive and other chronic effects for the red-tailed hawk using average, upper confidence limit, and maximum calculated mercury doses are less than "1". This indicates that the site is not presenting a risk to this species.

Table 9-23 shows the results of the population effects model runs for birds and small mammals. The results show that there is little effect on populations of birds and mammals due to exposure to mercury in site soils, based on the Fish and Wildlife Service recommended doses. This indicates little potential for population level effects. However, the model is very sensitive to the toxicity factor used. For example, the table also shows that nearly all the birds and mammals may be affected using lower benchmarks for non-reproductive effects. However, for birds, this benchmark is for a chronic effect (i.e. alteration of enzymes) which indicates exposure, but is not linked to reproduction. For mammals the chronic benchmark is for ocular effects from exposure to methyl mercury chloride, but is not linked to reproduction. The reproductive benchmark is appropriate for predicting population level effects because these benchmarks are directly tied to reproductive impairments.

9.6.4 Summary of Risks

It is Menzie-Cura & Associates' opinion that the HoltraChem site presents little risk to biota in the Penobscot River from exposure to the contaminants of concern, primarily mercury. We base this opinion on the observations that:

the exposure modeling indicates that toxicity quotients for piscivorous birds and mink are less than "1" using dietary doses based on average and upper confidence limit concentrations in eels. The toxicity quotient slightly exceeds "1" using the maximum eel body burden which represents background in the northern cove;

dissolved and total mercury concentrations in the water column of the main body of the Penobscot River and in the southern cove are less than ambient water quality criteria indicating protection of local fish populations from exposure to the most likely bioavailable fraction of the dissolved mercury;

abundance and diversity of the local benthic community near the site is similar to offsite areas and the historically observed benthos in this reach of the Penobscot River, indicating no risk above regional conditions due to site-related influences.

It is also our opinion that there is little potential for risk in the biologically-active areas of the southern cove sediments. We base this opinion on the observations that:

sediment toxicity testing indicated that the sediments are not toxic to a freshwater amphipod;

pore water concentrations of total mercury and methyl mercury were less than levels protective of benthic invertebrates; and the benthic populations in the cove were more abundant than populations observed elsewhere in this stretch of the river;

TOC and SEM/AVS data indicate reduced availability of metals such as mercury or methyl mercury in sediments in the southern cove;

most of the mercury (99.5%) in sediment is tightly bound to sediment;

eels caught in the vicinity of the southern cove had body burdens of mercury similar to those found in fish from remote water bodies in the State of Maine as well as in rivers that the state considers representative of background. This indicates no risk above regional conditions due to site-related influences.

It is Menzie-Cura & Associates' opinion that there is potential for risk to aquatic biota in the onsite southerly stream and northern ditch within the plant area and between the plant and the Penobscot River. We base this opinion on the observations that:

there appears to be a reduced macroinvertebrate community relative to the reference stream, indicating potential for risk above regional conditions due to site-related influences .

there are elevated conductivities atypical of freshwater streams in the southerly stream and northern ditch (which could be causing the reduced macroinvertebrate community);

mercury concentrations in the water column of these water bodies exceed acute water quality criteria.

It is Menzie-Cura & Associates opinion that there appears to be little potential risk at the population level to small mammals and birds using the US Fish and Wildlife criteria for protection of birds and of mammals. However, there is uncertainty in this opinion based on using the same model with lower toxicity factors which indicate potential population level risk. Toxicity quotients for the red-tailed hawk were less than "1" indicating that this risk does not extend to higher order predatory birds. The potential for risk to song birds and small mammals on site does not appear to result in decreased abundance or diversity of wildlife relative to a reference area based on qualitative observations made in these areas.

9.6.5 Uncertainty

This section identifies site-specific factors and key assumptions that contribute the greatest degree of uncertainty in the ecological risk assessment. Whenever possible, we have presented a qualitative discussion regarding whether our estimate over- or under-represents ecological risk. However, in several cases, this is not possible as the degree of uncertainty is unknown, or degrees of uncertainty vary among the lines of evidence that comprise the weight-of-evidence assessment of risk.

9.6.5.1 Uncertainty in Problem Formulation

The problem formulation establishes the elements of the assessment which focus and bound the analysis. It identifies contaminants of concern (COCs), selects assessment and measurement endpoints, and selects receptors. Of these, the greatest uncertainty is likely to result from the identification of COCs based on site data limited to analyses for VOCs and mercury. These analytes for the SI were selected based on CDM's review of previous reports that included site use, site history, and the results of sampling and analysis. The same processes have been conducted at the site since it was developed for industrial use and potential contaminants used and disposed of at the site have been well documented. Therefore, although there is some uncertainty in restricting the analyses to these chemicals, it is likely that this assessment has addressed ecological risks related to compounds known to be related to past and on-going site activities.

9.6.5.2 Uncertainty in Exposure Assessment

The exposure assessment estimates the magnitude of actual and/or potential ecological exposure to COCs. A number of assumptions made in the exposure assessment introduce varying degrees of uncertainty into the assessment. They are:

the restriction of exposure in cove sediments to the top inch;

the small number of soil and earthworm samples used to characterize mercury concentrations on site;

assumptions included in the food chain models;

assumption that the VOC concentrations in the onsite catch basin represent the VOCs in onsite streams;

the use of AVS measurements to evaluate the availability of metals.

In the assessment of ecological exposure to contaminants in sediment in the southern cove, we used surficial sediments as the exposure medium. We assumed that ecological receptors would only be exposed to the surficial sediments. However, mercury concentrations were higher in some deeper sediments (depths of 10 to 12 inches), particularly near the outlets of the streams that flow into the southern cove. Since the sediment regime in the cove probably varies from season to season and year to year, the potential exists for these deeper sediments to be uncovered and to become an exposure medium for ecological receptors. The restriction of the assessment to shallow sediments may result in an underestimate of risk.

There are inherent uncertainties in the use of food chain models because they incorporate simplifying assumptions to represent a complex ecological system. They assume that the primary exposure route is via ingestion of food and, in general, do not account for ingestion of contaminants in soil or sediment during foraging, grooming, etc. We have accounted for the soil ingestion exposure pathway in the robin and shrew exposure models. Food chain models also assume that the amount of contaminant available in food is completely available to the receptor. This may overestimate risk depending on the fraction of contaminant available.

The assumption that the VOC concentrations in the catch basin represent these compounds in onsite streams is uncertain. The catch basin sample was analyzed for VOCs because it receives groundwater discharge from underdrains nearby. Therefore, this location is likely to have the

highest concentrations of VOCs in surface water. Using these values to represent surface water onsite may result in an overestimate of risk.

We have used the ratio of total mercury in whole sediment to that in pore water (K_p) assuming equilibrium partitioning, and the ratio of simultaneously extracted metals (SEM) to acid volatile sulfides (AVS) to predict availability of mercury in sediments. This method only predicts potential availability and does not provide any information regarding whether available metals are, therefore, toxic.

9.6.5.3 *Uncertainty in Ecological Effects Assessment*

There may be considerable uncertainty in the development of toxicity benchmarks for wildlife. Scant data are available on effects of contaminants on wildlife. Therefore, it is usually necessary to extrapolate from laboratory studies on species other than those selected as receptors. Toxicity benchmarks based on no observed adverse effects levels (NOAELs) may overestimate risk because the level that produces an effect may be much higher than the reported no effect level. Note, however, that when toxicity benchmarks are based on a measured level that is associated with an effect, use of this value may underestimate risk. The concentration or dose of that compound that will not produce an effect remains unknown, but may be higher than the reported effect level.

The comparison of wildlife observations onsite to a reference area is a qualitative measure of potential effects. However, it is likely to identify only the most severe effects such as stressed vegetation, fish kills, etc. Other effects, such as effects on the reproductive success of local populations, may be manifested on the site and be difficult to identify without additional in-depth studies. Even if population effects are identified, it may not be possible to distinguish the effects of site contaminants from naturally occurring variations in wildlife populations.

Use of water only effects data to evaluate potential effects of pore water exposure on benthic invertebrates contributes some level of uncertainty (may over or under-estimate risks) to the ERA. It was assumed that the FDA action level of 1.0 mg/kg mercury would be protective of humans that consume benthic invertebrates exposed to sediment mercury. Bioconcentration factors (BCFs) used by USEPA (1985) for mercury II ($BCF = 4,994$) or methylmercury ($BCF = 81,700$) in developing the FCV or FRVs were assumed to accurately estimate site-specific BCFs.

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Section 10

Corrective Action Objectives and Preliminary Media Protection Standards Proposal

10.1 Introduction

This section presents the Corrective Action Objectives and Media Protection Standards developed based on applicable guidance and the results of the Site Investigation, and Human Health and Ecological Risk Assessments. Corrective Action Objectives are the general objectives of the Corrective Measures to be developed for the site. They are intended to provide adequate protection of human health and the environment from the release or potential release of hazardous waste or hazardous constituents to the environment from Areas of Concern. Media specific Corrective Action Objectives will be developed in conjunction with Media Protection Standards for groundwater, surface water, sediment, soil and air.

Media Protection Standards are the specific numerical criteria to be achieved by the Corrective Measures that will be implemented at the site. Media protection standards are developed based on the results of (1) the site characterizations provided in Sections 2 through 7 and (2) the Human Health and Ecological Risk Assessments presented in Sections 8 and 9, respectively. Upon submittal of this Site Investigation (SI) report, the EPA and MEDEP will consider and comment on the proposed preliminary media protection standards (PMPS) presented in this section. The proposed PMPS may be revised based on EPA or MEDEP comments. Once established by the EPA, the PMPS become the basis for evaluating corrective measures. Until the Corrective Measures Study (CMS) demonstrates the PMPS for a medium can be achieved, they are still considered preliminary.

Remedies for the site will be selected based on the results of corrective measures evaluations performed in the CMS. Remedial alternatives will be evaluated based on their ability to meet the general corrective action objectives and the specific PMPS, and other criteria such as technical feasibility, environmental impacts related to remedial construction, potential risks, and cost. Some PMPS may not be technically or economically achievable.

10.2 Regulatory Criteria

In the following section, provisions of the Consent Decree, regulations and guidance, and the human health and ecological risk assessments relevant to the establishment of PMPS are summarized.

10.2.1 Consent Decree

The Consent Decree requires the establishment of Corrective Action Objectives and PMPS based on the following:

Paragraph 46.a - Groundwater - For “groundwater that is a current or potential source of drinking water,” the Consent Decree Paragraph 46.a provides that Maximum Contaminant Levels (MCLs) established by EPA in 40 C.F.R. Part 141, Subpart B be used as preliminary MPS. If an MCL is not available, Paragraph 46.a provides that a preliminary MPS be established in accordance with paragraphs 46.f and 53.h(1)-(5) of the Consent Decree.

The paragraph is silent on establishment of MPS for groundwater that is not a current or potential source of drinking water.

Paragraph 46.b - Surface Water - The consent decree states:

- “b. For constituents in surface water, the preliminary MPS shall be specified as the more stringent of the following concentrations, unless the HEA (Health and Environmental Assessment) demonstrates that less stringent concentrations are within acceptable risk limits, in accordance with EPA policy:(emphasis added)
- “(1) Water Quality Standards established pursuant to section 303(c) of the Clean Water Act, 33 U.S.C. § 1313 (c), 40 C.F.R. Part 131, by the State of Maine, if such standards are expressed as numeric values; or
- “(2) Numeric interpretations of the State of Maine’s narrative water quality standards, if appropriate, where water quality standards expressed as numeric values have not been established by the State; or
- “(3) MCLs established under 40 C.F.R. Part 141, Subpart B for constituents in surface waters designated by the State of Maine for Drinking Water supply; or
- “(4) For constituents in surface waters designated by the State of Maine for drinking water supply for which values in subparagraphs (1)-(3) immediately above are not available, the preliminary MPS shall be established in accordance with paragraphs 46.f and 53.h(1)-(5) of this Consent Decree.

Paragraph 46.c - Air The Consent Decree stipulates establishment of preliminary MPS on a health and environmental risk basis (per paragraphs 46.f and 53.h(1)-(5)) assuming exposure “through inhalation of the air contaminated with the constituent, as measured or estimated at the site boundary,

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or another location closer to the Area of Concern if necessary to protect human health and the environment.”

Paragraph 46.d - Soil The Consent Decree stipulates establishment of preliminary MPS on a health and environmental risk basis (per paragraphs 46.f and 53.h(1)-(5)) assuming exposure “through ingestion of and dermal contact with soil contaminated with the constituent. Leaching of hazardous constituents from the soil shall be considered in setting the preliminary MPS.”

Paragraph 46.e - Sediments The Consent Decree stipulates establishment of preliminary MPS on a health and environmental risk basis (per paragraphs 46.f and 53.h(1)-(5) or by using "EPA sediment criteria guidance"). There is currently no published EPA sediment criteria guidance.

Paragraph 46.f - This paragraph provides guidance for the establishment of risk-based Preliminary MPS as follows:

“f. Unless otherwise established pursuant to this paragraph 46, preliminary MPS shall be established as follows:

“(1) For known or suspected carcinogens, preliminary MPS shall be established at concentration levels which represent an excess upper bound lifetime risk to an individual of between 10^{-4} and 10^{-6} . The 10^{-6} risk level shall be used as the point of departure in proposing MPS.

“(2) For systemic toxicants, preliminary MPS shall represent concentration levels to which the human population (including sensitive subgroups) could be exposed to on a daily basis without appreciable risk of deleterious effect during a lifetime.”

These statements are immediately followed by Paragraph 46.g which allows consideration of background concentrations in the establishment of preliminary MPS:

Paragraph 46.g - “If Hanlin [HoltraChem] can demonstrate to the satisfaction of EPA that a specific concentration of a constituent in a medium at the facility is naturally occurring or consistent with area-wide concentrations, the MPS for the constituent in the medium shall not be below that specific concentration.”

Paragraphs 53.h (1)-(5) - These provisions stipulate the content of the MPS recommended in the Corrective Measures Study (CMS) Report:

“53. The CMS report must include the following...

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- “h. A recommendation regarding Performance Standards, Media Protection Standards and points of compliance, all of which shall be based upon the Corrective Measure evaluation process undertaken pursuant to paragraph 54 and the following factors:
- “(1) multiple contaminants in the medium (the procedures for considering the risks posed by multiple contaminants are described in ‘Guidelines for the Health Risk Assessment of Chemical Mixtures’ (51 FR 34014));
 - “(2) exposure threats to sensitive environmental receptors;
 - “(3) other site-specific exposure or potential exposure to contaminated media from areas of contamination identified in the SI [Site Investigation];
 - “(4) the reliability, effectiveness, practicality or other relevant factors of the Corrective Measure (or the anticipated Corrective Measure). This factor includes for soil contamination, the concentration levels which, through leaching, could result in exceedances of groundwater protection standards, and;
 - “(5) other relevant and appropriate criteria such as proposed Maximum Contaminant Levels (‘MCLs’), Maximum Contaminant Level Goals (‘MCLGs’), proposed state standards, health advisories, or federal government guidance.”

10.2.4 RCRA Corrective Action, Subpart S, Proposed Rules

The preamble of the RCRA Corrective Action Program, proposed Subpart S, published in the Federal Register, Vol. 55, No. 145, Friday, July 27, 1990, provides guidance for the RCRA Corrective Action Program. Although not promulgated as final regulations, EPA considers Subpart S to be its current guidance on RCRA and its relationship to other environmental laws.

In Subpart S, Section VI, B. Definitions, Paragraph 2, Release, states that "many facilities have releases from solid waste management units that are issued permits under other environmental laws. For example, stack emissions from a solid waste refuse incinerator at a RCRA facility are likely to be authorized under a State-issued air permit. Another example would be NPDES (National Pollutant Discharge Elimination System, under the Clean Water Act), or State-equivalent, permits for discharges to surface water from an industrial wastewater treatment system. EPA does not intend to utilize the section 3004(u) corrective action authority to supersede or routinely reevaluate such permitted releases. However, in the course of investigating RCRA facilities for corrective action purposes, EPA may find situations where permitted releases from SWMUs have created threats to human health and the environment. In such a case EPA would refer the information to the relevant permitting authority or program for action.”

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10.2.3 RCRA Corrective Action Guidance (OSWER Directive 9902.3-2A)

OSWER Directive 9902.3-2A (EPA, 1994) provides guidance for the establishment of cleanup targets for RCRA corrective action programs. Though not always strictly applicable to the HoltraChem site program, the guidance in this document may be relevant or appropriate to consider in establishing cleanup targets where the Consent Decree and regulations are not explicit.

Section V.B. covers the establishment of medium-specific cleanup standards. It provides considerable discretion in the establishment of these standards. The overriding criterion is protection of human health and the environment. As the paragraph states, "The Permittee/Respondent shall provide information as required by the implementing agency to support the agency's selection/development for media cleanup standards or any releases that may have adverse effects on human health and the environment due to migration of waste constituents."

For groundwater, MCLs, background constituent concentrations and alternate standards (e.g alternate concentration limits) are considered.

For all media (including groundwater), the following are considered:

- Waste characteristics and volumes
- Effectiveness of current control methods
- Geologic, hydrologic and climatological characteristics
- Background concentrations
- Actual and potential contaminant migration pathways
- Potential impacts to human health and the environment

10.2.4 Clean Water Act

Section 304(a) of the Clean Water Act requires EPA to develop and publish criteria for water quality accurately reflecting the latest scientific knowledge.

Section 402 of the Clean Water Act pertains to the National Pollutant Discharge Elimination System (NPDES) and describes the types of discharges requiring an NPDES permit. Paragraphs (2)(B), (3)(A) and (4)(A) require that stormwater from industrial facilities be permitted.

10.2.5 MCLs

The EPA has established maximum contaminant levels (MCLs) for water used as drinking water. The MCLs which have been promulgated for the contaminants of concern detected at the site are summarized in **Table 10-1**.

10.2.6 Ambient Water Quality Criteria

The Clean Water Act also requires EPA to publish and periodically update ambient water quality criteria. These criteria are intended to reflect the latest scientific knowledge on:

- The kind and extent of identifiable effects on health and welfare (including plankton, fish, shellfish, wildlife, plant life, shorelines, beaches, aesthetics and recreation) which may be expected from the presence of contaminants in any body of water, including groundwater;
- The concentration and dispersal of contaminants or their byproducts through biological, physical and chemical processes;
- The effects of contaminants on biological community diversity, productivity and stability.

These ambient water quality criteria are not rules and do not have regulatory impact. The criteria present scientific data and guidance on the environmental effects of contaminants intended to be used in the derivation of regulatory requirements. The latest recommended ambient water quality criteria were published in the December 10, 1998 Federal Register (Vol. 63, No 237, Pgs. 68354 through 68364). The water quality criteria for contaminants of concern at the HoltraChem site are summarized in **Table 10-1**.

10.2.7 Maine Water Pollution Law

Maine Water Pollution Law, administered by the Department of Environmental Protection, sets forth environmental policy relating to waters of the State including groundwater and surface water. It specifically addresses groundwater protection and pollution control and sets forth classifications for most waters of the State.

Section 420 states that “no person, firm, corporation or other legal entity shall place, deposit, discharge or spill, directly or indirectly, into the groundwater, inland surface waters or tidal waters of this State, or on the ice thereof, or on the banks thereof so that the same may flow or be washed into such waters, or in such manner that the drainage therefrom may flow into such waters, any of the following substances:

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“1. Mercury. Mercury, and any compound containing mercury, whether organic or inorganic, in any concentration which increases the natural concentration of mercury in the receiving waters....

“A. Any person, firm, corporation, or other legal entity who, on January 1, 1971, was discharging any of the substances mentioned in the subsection in connection with an industrial process shall not be deemed in violation of this subsection if on or before December 31, 1971, it filed with the board a statement indicating the amount of the substance so discharged on that date.”

“B. Notwithstanding paragraph A, whenever the commissioner finds that a concentration of 10 parts per billion of mercury or greater is present in any waters of this State, or that danger to public health exists due to mercury concentrations of less than 10 parts per billion in any waters of this State, the commissioner may issue an emergency order to all persons discharging to such waters prohibiting or curtailing the further discharge of mercury, and compounds containing mercury, thereto...

The remaining portions of section 420 outline the conditions under which the board may adopt site-specific criteria or alternative statewide criteria. Generally these criteria are established in the framework of the discharge licensing system and are based on "sound scientific rationale and are protective of the most sensitive designated use of the water body, including, but not limited to, human consumption of fish and drinking water supply after treatment". Human health risk is also considered.

Chapter 530, Section 5 establishes the Surface Waters Toxics Control Program. Section (5)(A)(2)(I) establishes the statewide numerical criteria for toxic constituents with national water criteria by stating "except as naturally occurs, levels of toxic pollutants in surface waters must not exceed federal water quality criteria as established by USEPA pursuant to Section 304(a) of the Clean Water Act, or alternative criteria established below." Subparagraph (ii) states that "Alternative Statewide Criteria must be adopted through rulemaking and must protect the designated uses of the assigned classification (38 M.R.S.A. Section 420(2)(B)) equally as well as the USEPA criteria." Subsequent paragraphs in the Section allow the establishment of site-specific criteria which must protect designated uses equally as well as applicable statewide criteria. Requirements for information and toxicity testing in support of establishing site-specific criteria are also presented.

10.2.8 NOAA Screening Guidelines

The National Oceanic and Atmospheric Administration (NOAA) has established screening guidance values for various media. The screening values were developed for internal use by the Coastal Resource Coordination (CRC) branch of the NOAA. Measured concentrations are compared to the screening values to identify substances that might pose a threat to resources of concern to NOAA. The screening values are conservative to give a high degree of confidence that sources eliminated from further consideration pose no potential threat to resources of concern to NOAA.

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For water samples, NOAA used promulgated ambient water quality criteria (EPA, 1985). For surface water, NOAA compares the measured concentrations directly to the ambient water quality criteria. For groundwater, NOAA compares the measured concentrations to 10 times the ambient water quality criteria to account for dilution expected during groundwater migration and discharge.

For soils, NOAA compares measured concentrations to average concentrations found in natural soils of the United States. These screening values are not enforceable and are not intended to indicate that a threat to human health or the environment exists. Rather, they are based on background measurements and, as such, are strong indicators of no-impact conditions. Again, exceedance of these values does not imply human health or environmental impact. Exceedance implies the need for site-specific evaluation.

For sediments, NOAA compares dry weight concentrations to Effective Range-Median (ER-M) values originally reported by Long and Morgan (1990; Tech Memo NOS OMA 52). The ER-M value is the concentration equivalent to that calculated at the lower 50th percentile of available sediment toxicity data which has been screened for those samples which indicate toxicity effects were observed or predicted. As such, it represents the mid-range of concentrations at which effects were observed in the studies compiled by Long and Morgan. Where ER-M values were not available, Apparent Effects Thresholds (AET) relating chemical concentrations in sediments to at least one biological indicator of injury, are used. AETs were developed by Washington State from data collected in Puget Sound, Washington. These sediment screening values are intended as general guidelines and indicators of no impact. Exceedance of these guidelines does not indicate that human health and the environment are impacted; exceedance merely indicates the need for further, site-specific evaluation.

10.3 Human Health and Ecological Risk Assessment Findings

A human health risk assessment and ecological assessment, prepared by Menzie-Cura & Associates, are presented in Sections 8 and 9, respectively. Contaminants of concern were short-listed by media based on their frequency and consistency of detection and potential significance. Total human health risk and toxicity effects were estimated. A qualitative assessment of ecological impacts was also presented.

10.3.1 Contaminants of Concern and Exposure Scenarios

Groundwater: The human health risk assessment determined that the contaminants of concern in the groundwater are:

Volatile Organics

Acetone
Bromoform
Carbon Tetrachloride
Chloroform
Chloropicrin
Trichloroethene

Metals

Mercury
Manganese

Water, for human consumption and all process needs, has been obtained from the City of Bangor municipal supply since 1988 and municipal water supply is the likely source of all future site water needs. Therefore, a drinking water exposure scenario was not considered realistic for the human health risk assessment. Dermal exposure is assumed for future onsite facility workers during excavation activities. Assumptions concerning exposure to vapors originating in groundwater are explained in the discussion of the air medium.

Soil: The risk assessments determined that the contaminants of concern in soil are:

Chloropicrin
Mercury

Soil exposure scenarios were considered in the human health risk assessment for current onsite workers, future construction workers, and trespassing teenagers.

Although future residential use is unlikely, hypothetical onsite residents were also considered and were assumed to consume home-grown vegetables which accumulate the soil contaminants of concern.

Onsite Surface Water: The risk assessments determined that the contaminants of concern in onsite surface water are:

Chloroform
Carbon tetrachloride
Mercury

Onsite surface water exposures for current onsite facility workers and current trespassing teenagers were considered in the human health risk assessment.

Onsite Sediment: The risk assessments determined that the only contaminant of concern in onsite sediment is mercury. Current onsite workers and trespassing teenagers are assumed to contact these sediments.

River Surface Water: The risk assessments determined that the only contaminant of concern in river surface water is mercury.

The human health risk assessment considered contact with offsite surface water by people at each life stage while swimming or fishing in the Penobscot River.

River Sediment: The risk assessment determined that the only contaminant of concern is mercury. Off-site residents are assumed to contact river sediments while swimming in the Penobscot River.

Air: The human health risk assessment assumed that current onsite workers and trespassing teenage children are exposed to mercury and VOCs by inhaling fugitive dust from soil contaminants. Future onsite workers and future hypothetical onsite residents are assumed to be exposed to fugitive dust contamination (soil contaminants) and vapors from mercury and VOCs in groundwater (workers during excavation activities and hypothetical residents via groundwater contaminants accumulated in indoor air.) Current offsite residents are assumed to be exposed to mercury vapors in ambient air.

Biota: The only contaminant of concern in the biota assessment is mercury. Fish tissue measurements are used in the Human Health Risk Assessment. Fish and wildlife observations and tissue measurements are used in the ecological assessment.

10.3.2 Risk Characterization and Risk Assessment Conclusions

The results of the human health risk assessment did not indicate the presence of unacceptable risk due to the chemicals of concern based on current site conditions, but did indicate potential risk if on site groundwater migrates off site. Risk was identified for some future hypothetical land use scenarios. However, these involved onsite residential use under current site conditions, a scenario which is unlikely to occur. In addition, the site presents little potential risk to biota in the Penobscot River from exposure to mercury.

The human health risk assessment made the following specific conclusions:

- Cancer and non-cancer risks for current facility workers and trespassing teenagers were less than the Consent Decree target of 1×10^{-6} and a hazard index of 1. This is true of both the average and the reasonable maximum exposure calculations.

- The non-cancer risk for current offsite residents exceeds a hazard index of 1 because it assumes a resident is ingesting drinking water from an onsite well. This is true of both the average and the reasonable maximum exposure calculations. Cancer risks were not computed because there were no carcinogenic contaminants of concern for this scenario.
- Cancer risks for future onsite workers were below the Consent Decree target of 1×10^{-6} for average and reasonable maximum exposure calculations. Cancer risks for future hypothetical on site residents exceed the target risk level assuming consumption of onsite groundwater under current conditions.
- Non-cancer hazard indices for future onsite workers were below the target of 1 for both average and reasonable maximum exposure cases.
- The non-cancer hazard indices for future hypothetical onsite residents were 120 and 2,400 for the average and reasonable maximum exposures, respectively. A hazard index of 1 indicates no toxic effects. The results were driven by the scenario of ingestion of mercury in drinking water. Ingestion of homegrown produce was also a significant risk contributor. These indices are for residential use under current site conditions, a scenario which is unlikely to occur.

The ecological assessment made the following conclusions:

- The site presents little potential risk to biota in the Penobscot River from exposure to mercury. This opinion was based on greater abundance and diversity of aquatic life in the southern cove than at similar environments near the site, non-toxic results of sediment bioassay samples and no observed increase in mercury concentrations in eels caught off the southern cove when compared to background locations and locations from remote areas of the state.
- There is little potential risk to biota exposed to the water column in the southern cove. This opinion is based on water quality sampling in the vicinity of the southern cove which indicate that the mercury concentrations, especially dissolved concentrations (the most bioavailable fraction) are below the chronic ambient water quality criteria. The higher total mercury concentrations in surface water are probably due to resuspension of sediment (see Section 7). The mercury in this sediment has been shown through speciation analyses to be strongly bound and therefore is not likely to be bioavailable. The southern cove sediments do not pose an ecological risk based on direct measurements of sediment toxicity and eel tissue mercury concentrations and the observation of abundant benthic populations.

- There is potential for risk to aquatic biota in the two plant stormwater ditches between the plant and the Penobscot River. This opinion is based on observations of reduced macroinvertebrate community relative to the reference stream, elevated and atypical specific conductivities and salinities, and mercury concentrations in excess of water quality criteria.

- Although the risk assessment model indicates there may be potential risk to songbirds and small mammals feeding directly on soil invertebrates (worms) on the terrestrial portion of the site, observation of diversity and abundance of wildlife onsite does not indicate that the ecosystems have been significantly affected. Concentrations of mercury measured in earthworm tissue were used as input to an exposure model for creatures feeding on earthworms. The model showed potential for risk to songbirds (robin) and small mammals (shrew). Soil concentrations of 2.2 mg/kg (dry weight) and 2.6 mg/kg (dry weight) of mercury would result in Toxicity Quotients of 10 and 40 for the short-tailed shrew and robin, respectively. However, this potential risk to individual organisms does not extend to population level effects, based on the application of a population model using the reproductive endpoints for songbirds and mammals.

10.4 Corrective Action Objectives

In this section, Corrective Action Objectives are established for environmental media.

10.4.1 Groundwater

Corrective Action Objectives for groundwater are as follows:

- Protect human health and the environment
- Protect drinking water supplies
- Protect surface water.

10.4.2 Surface Water

Corrective Action Objectives for surface water are as follows:

- Protect human health and the environment
- Protect groundwater

10.4.3 Soil

Corrective Action Objectives for soil are as follows:

- Protect human health and the environment
- Protect surface water
- Protect groundwater
- Protect air
- Protect sediment

10.4.4 Sediment

Corrective Action Objectives for sediment are as follows:

- Protect human health and the environment
- Protect surface water
- Protect groundwater

10.4.5 Air

Corrective Action Objectives for air is as follows:

- Protect human health and the environment

10.5 Proposed Preliminary Media Protection Standards and Points of Compliance

10.5.1 Introduction

PMPS were developed to meet the corrective action objectives listed in subsection 10.4. Preliminary MPS were either based on (1) existing ARARs, (2) risk-based concentrations protective of exposure, or (3) protection of secondary media, e.g. discharge of groundwater to surface water. PMPS were developed for chemicals of concern identified to be primary contributors to risk or hazard indices in the Risk Assessment. Numerical values were developed using site specific approaches and data wherever possible.

The HoltraChem site is currently zoned "Industrial" and "Shoreland Industrial" which at the present time allows for limited residential development. Although future hypothetical residential scenarios

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were considered in the human health risk assessment, for the purpose of developing preliminary MPS, both current and future site use is considered to be industrial. EPA recognizes that RCRA facilities are typically industrial properties that are actively managed rather than abandoned sites. Therefore, consideration of non-residential uses is especially likely to be appropriate for RCRA facility cleanups. (EPA Directive No. 9355.7-04, May 25, 1995). It is recognized that additional institutional controls consistent with the intended future use, i.e. continued industrial use, may need to be secured.

The HoltraChem site is unique in several respects including its topography, hydrogeology, rural location and proximity to a large, tidally influenced, transitionally saline river which is a point of regional groundwater discharge. Therefore, the PMPS were developed based on site specific and published analytical and scientific data, and the results of the site specific human health and ecological risk assessments. As a result, they reflect levels which would be protective of human health and the environment for this particular site based on characterization of this particular site. In some instances (for some media), no measurable or unacceptable risks were observed and establishment of a numerical PMPS was based on data from published guidance, literature or based on the no observed adverse effects levels documented at the site.

Proposed PMPS are summarized in **Table 10-2**.

10.5.2 Groundwater

Groundwater at the site exceeds MCLs for several of the contaminants of concern. However, the site is served by the Bangor Municipal Water System. Groundwater at the site is not currently and will not in the foreseeable future be used as a drinking water source. In addition, based on the direction of groundwater flow (toward the Penobscot River) and the distribution of groundwater contamination (north and west of the plant), contaminated groundwater from the site is not likely to migrate offsite where it could be a drinking water source. Therefore, MCLs and drinking water standards are not proposed as PEPS for groundwater discharging directly to the Penobscot River. However, there are several residences along Ferry Road, downriver from the site that use drinking water wells located on their properties. These wells are not directly downgradient of the plant. However, MCLs are proposed as PMPS for groundwater along the southern property boundary to be protective of these wells.

The following factors were considered when establishing a groundwater PMPS:

- Effects on human health and the environment (Consent Decree)
- The potential for groundwater to cause a violation of surface water criteria
- Maine Water Pollution Law, governing waters of the State

The risk assessment evaluated incidental ingestion of groundwater by workers during construction and volatilization of chemicals from groundwater to indoor air. Neither of these exposure pathways exceeded either a hazard index of 1 or the acceptable cancer risk range. Site groundwater discharges directly and indirectly to the Penobscot River.

Southern Plant Boundary

Sampling and analysis results and hydraulic gradient measurements indicate that site groundwater flows to the Penobscot River, not toward the residential wells downriver of the plant. However, because residences downriver use groundwater as their drinking-water source, periodic confirmation of the absence of offsite migration is warranted. Because bedrock groundwater is used as a drinking water source by the residents of Ferry Road, MCLs, where available, are proposed preliminary MPS for groundwater contaminants of concern:

<u>Constituent</u>	<u>PMPS ($\mu\text{g/l}$) for Southern Plant Boundary</u>
Acetone	100**
Bromoform	5**
Carbon Tetrachloride	5.0*
Chloroform	100 (as total trihalomethanes)*
Chloropicrin	10**
Trichloroethene	5.0*
Mercury	2.0*
Manganese	50*

* = EPA promulgated MCL or secondary MCL

** = Proposed site specific risk based PMPS

In the absence of promulgated MCLs, the PMPS for VOCs at the southern plant boundary were developed using the consumption of drinking water (offsite resident) scenario.

Manganese was identified as a COC for groundwater based on its presence in site groundwater above the secondary MCL. However, concentrations of manganese above the secondary MCL (0.050 mg/l) are not uncommon in bedrock groundwater in Maine and have been noted in upgradient wells (18 mg/l). Therefore, the MPS for manganese is the secondary MCL or background levels whichever is higher.

The proposed point of compliance is the line of monitoring wells installed on PERC property comprised of the B-321, MW-505 and MW-511 clusters. The monitoring frequency has been quarterly during 1998 and monitoring of PERC property wells and selected Ferry Road residential

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wells is expected to continue as part of the Consent Agreement Comprehensive Monitoring Plan. The frequency and scope of future monitoring may be adjusted based on evaluation of initial monitoring results.

Groundwater Discharging to the Penobscot River

Groundwater discharging from the site may impact the Penobscot River quality if there is sufficient contaminant flux. Media protection standards were developed considering AWQC as the maximum allowable concentration in the Penobscot River. VOCs, including chloropicrin, were not detected in any of the river surface water samples. However, they were detected in the wells adjacent to the Penobscot River. VOCs would be expected to volatilize and photodegrade upon release from groundwater to surface water. VOCs do not typically bioaccumulate and accordingly their ambient water quality criteria, where promulgated, are well above the range concentrations detected in site groundwater. Therefore, mercury was considered the only COC in groundwater that would impact surface water.

The flux of mercury to the Penobscot River was calculated in Section 7 and was used to develop the PMPS. Based on the calculations, the majority of the mercury flux occurs from the sand and gravel deposits with high contaminant concentrations and high hydraulic conductivity located at the base of Landfill Area 1. The contribution of mercury flux from the ridge to the river is estimated to be three orders of magnitude less and therefore does not significantly contribute to total site mercury flux in groundwater. A back calculation of the contaminant flux required to exceed AWQC for mercury was performed using the following assumptions. The calculations are presented in Appendix E.

- River Flow = 7Q10 low flow, 4000 cubic feet per second (cfs). The 7Q10 flow is the lowest flow that can be expected to occur over an extended period of time. It was therefore selected as an appropriate flow rate for comparisons with chronic ambient water quality criteria.
- Volume of potentially contaminated water discharging from the site = 685 cubic feet per day, which is the volume of water calculated to pass through the primary discharge routes where contaminated groundwater has been observed.
- The arithmetic mean of the mercury concentrations in the wells which monitor groundwater discharging through the most contaminated and the most hydraulically conductive deposits is approximately 824 $\mu\text{g/l}$, based on the December 1997 groundwater sampling round. The proportional flow cell approach indicates the arithmetic mean more closely reflects the most likely discharges.

- The flux of mercury off site through groundwater is estimated to be approximately 3.5×10^{-2} lbs/day (12.7 lbs/year) almost all of which is coming from between the MW-401 cluster and the B-316 cluster.

The current flux of mercury calculated from NPDES discharge monitoring data is approximately 8.0×10^{-3} lbs/day (2.9 lbs/year). As a result of reduced flows and construction of a new waste water treatment system, this flux is down significantly from the 1.8×10^{-2} lbs/day (6.6 lbs/year) estimated in 1995. The existing NPDES permit allows 4.56×10^{-2} lbs/day (16.6 lbs/year) of mercury as an average monthly rate (averaged by month), and a maximum discharge of 10.49×10^{-2} lbs/day. The additional flux estimated from the two stormwater ditches during non-storm conditions is 2.8×10^{-3} lbs/day. The combined contributions from the stormwater ditches, the maximum NPDES permitted discharge and existing river mercury concentrations were considered when calculating the groundwater flux which would result in an exceedance of ambient water quality criteria. Assuming maximum NPDES permitted discharges and current estimates of groundwater flux, an average groundwater concentration approximately 25 times the current average would be required before the ambient water quality criteria of 0.050 ug/l would be exceeded.

The background mercury surface water concentration is 4.3 ng/l (total) which was established by averaging two upstream river samples collected during the 1995 SI sampling and the three clear water upriver samples collected by Acheron during the Penobscot River Studies. The current flux of mercury through groundwater theoretically results in an increased surface water concentration of approximately 2.0 ng/l. A slightly smaller increase is expected from the current NPDES discharge and stormwater ditches, combined. Taking into account all three contributions (groundwater, current NPDES discharge and stormwater ditches), the Penobscot River surface water concentration is expected to rise from 4.3 ng/l to 6.4 ng/l. The average of the three surface water samples collected by Acheron in clear water off the southern cove was 6.47 ng/l.

An average value of 850 $\mu\text{g/l}$, roughly equivalent to the concentrations measured in 1997, is proposed as an initial PEPS that is protective of surface water in the Penobscot River. This initial value is proposed until removal of the lined process lagoon, a suspected source of groundwater contamination, has occurred. This proposed PMPS is protective of human health and the environment based on the data collected at the site and the human health and ecological risk assessments. Source control and/or elimination combined with natural attenuation will result in a decrease in the groundwater concentrations.

The proposed point of compliance for monitoring groundwater discharging to the Penobscot River is the arithmetic mean concentration of the six monitoring wells screened in stratified drift at the base of Landfill Area 1: MW-401-O1, MW-512-O1, B-326-03, B-326-02, MW-513-O1 MW-401-O1. The concentration in some wells may exceed the arithmetic mean concentration while others are

below it and yet the net discharge to the river will not result in an exceedance of the chronic AWQC in the Penobscot River.

Maine Water Pollution Law prohibits the discharge of mercury in “any concentration which increases the natural concentration of mercury in the receiving waters,” with some exceptions. Section 420, Section 1, Paragraph A contains waiver provisions which allow some discharges which were existing and declared at the time the law went into effect in 1972, to continue. The HoltraChem plant’s NPDES discharge is grandfathered under this provision. The flux of mercury in groundwater discharging to the Penobscot River does not cause an exceedance of the ambient water quality criteria in the southern cove or Penobscot River. However, it may increase the natural concentration in the surface water in the immediate vicinity of the site.

A separate groundwater MPS is not proposed for onsite groundwater upgradient of the point of compliance for discharge to the Penobscot River. The three landfills on the ridge are immediately upgradient of the manufacturing plant which is immediately upgradient of Landfill Area 1. Therefore, the proposed point of compliance monitoring will be representative of groundwater from the landfill ridge area, the plant area and Landfill Area 1. The downgradient boundary of the contiguous waste unit or source area is the base of Landfill Area 1. Monitoring performed within the plant area and around the landfills as part of the Comprehensive Monitoring Plan will provide data on the presence or absence of continuing sources but these monitoring points are not proposed as points of compliance.

10.5.4 Penobscot River Surface Water

The elevated total mercury concentrations in surface water are probably due to resuspension of sediment (see Section 7). During the Penobscot River Studies conducted in 1998 mercury samples were collected from upriver and downriver of the HoltraChem site and from turbid and clear water to assess the impact of resuspension of sediment to surface water quality. The correlation between total suspended solids (TSS) and total mercury was very strong. These data indicate that total mercury concentrations are very dependant on river conditions which impact the total suspended solids present in the water column. Measurement of total mercury is likely to be highly variable both over time and depending on where in the river the sample is taken. Speciation analyses of surface water have shown that less than three percent of the total mercury in surface water is methylmercury. Speciation analyses of sediment have shown that 99.5% of the mercury in sediment is strongly bound and therefore is not likely to be bioavailable.

On October 1, 1993, EPA’s Office of Water issued a memorandum entitled Policy and Technical Guidance on Interpretation and Implementation of Aquatic Life Metals Criteria. This memorandum specifically addressed the use of dissolved vs. total recoverable metals for water quality criteria. The

memorandum states “It is now the policy of the Office of Water that the use of dissolved metal to set and measure compliance with water quality standards is the recommended approach, because dissolved metal more closely approximates the bioavailable fraction of metal in the water column than does total recoverable metal. This conclusion regarding metals bioavailability is supported by a majority of the scientific community within and outside the Agency.”

On December 10, 1998, EPA published its compilation of recommended water quality criteria developed pursuant to section 304(a) of the Clean Water Act. According to EPA, these criteria were established based on relationships between pollutant concentrations and environmental and human health effects, and reflect the latest scientific knowledge. The recommended criteria for mercury in fresh water are 1.4 ug/l maximum and 0.77 ug/l for chronic exposures. A lower value of 0.050 ug/l is recommended for protection of human health through the consumption of water (incidental ingestion) and organisms (e.g. fish), and 0.051 ug/l is recommended in water considering the consumption of organisms only. The footnotes to the compilation state that all the metals values are expressed in terms of dissolved metals in the water column.

Regarding numerical water quality criteria and statewide criteria for toxic pollutants with national water quality criteria, Maine Bureau of Water Quality Control regulations Chapter 530, Section 5, A.2.a.i. states that “except as naturally occurs, level of toxic pollutants in surface water must not exceed federal water quality criteria as established by USEPA, pursuant to Section 304(a) of the Clean Water Act”. The notice published in the December 10 federal register reads “These water quality criteria are the Agency’s current recommended 304(a) criteria reflecting the latest scientific knowledge. They are generally applicable to the waters of the United States. EPA recommends that States and Tribes use these water quality criteria as guidance in adopting water quality standards pursuant to section 303(c) of the Act and the implementing of federal regulations at 40 CFR part 131.” Prior to the publication of the new recommended criteria, MEDEP used EPA’s previously recommended criteria of 0.012 ug/l measured as total recoverable mercury. This earlier number was developed using a bioconcentration factor for methylmercury and applying it to all species of mercury. Reflecting current scientific knowledge, EPA has revised the criteria and specified dissolved total (unspeciated) mercury as the measurement method.

The lower values protective of human health through consumption will likely be used to set surface water standards. Based on the discussion provided above, the proposed preliminary media protection standard for mercury in Penobscot River surface water is the ambient water quality criterion of 0.050 ug/l measured as dissolved total mercury. The point of compliance is in the Penobscot River outside of the mixing zone of the plant’s NPDES discharge. VOCs were not detected in four surface water samples collected from four locations in the river and are not considered COCs for surface water.

10.5.5 Penobscot River Sediment

The risk assessment found no human health or ecological risks associated with current sediment contamination in the cove area of the Penobscot River.

As noted in Section 7, elevated concentrations were observed in the vicinity of the outfalls of the northern stormwater ditch, southerly stream and near the plant's wastewater outfall. Elsewhere, in the cove, in the organic muck, and in the reed beds, concentrations typically ranged between 1 and 2 mg/kg which is similar to the concentrations detected at upriver locations. Because these organic silt deposits are likely to be mobile in the river environment, sediments with similar concentrations would be expected to be deposited in this cove in the future even in the absence of potential contributions from the plant.

To develop a preliminary MPS for sediments, two methods were considered. One method used site-specific partitioning coefficients and the other used whole sediment toxicity bioassay results. The result of the most conservative of the two methods was selected as the preliminary MPS.

The following calculation procedure from USEPA (1993) was performed to support development of a site-specific PMPS for sediment assuming equilibrium partitioning.

$$\text{MPS} = \text{Kd} * \text{FCV (or FRV)}$$

Site-specific partition coefficients (Kd) were calculated in Section 9.4.1 and presented in Table 9-14 for "granular" sediment containing 0.2 - 1.0% total organic carbon (TOC) and "organic" sediment containing approximately 10% TOC. The sediment PMPS were specific for 0.2 - 1.0% TOC sediment (using a mean Kd = 53,619), and ~10% TOC sediment (using a mean Kd = 548,871). A Final Chronic Value (FCV) of 0.001302 mg/l for Hg(II), and Final Residue Values (FRVs) of 0.00020 mg/l for Hg(II) and 0.000012 mg/l for methylmercury, were presented in Section 9.5.5, as developed by USEPA (1985). Based on the site specific Kds developed from the porewater analysis and the FCV and FRVs listed above, concentrations of 70 to 715 mg/kg Hg(II) would be protective for chronic (biological) effects, and 10.7 to 110 mg/kg Hg(II) protective of food chain (residue) effects.

Methylmercury is the highly toxic, bioavailable and bioaccumulating form of mercury. Based on the calculation described above, a concentration of 0.64 to 6.6 mg/kg methylmercury in sediments would be protective for residue (food chain) effects. However, the amount of methylmercury present in the southern cove sediments ranged from 0.004% to 0.38% and averaged 0.16 % of the total mercury present in the sediments tested. The concentration of methylmercury detected in cove sediments ranged from 0.00038 mg/kg to 0.033 mg/kg in the speciation analysis. In addition, the dissolved

concentrations in porewater ranged from 0.016 ng/l to 1.24 ng/l. Based on these data it does not appear that much methylation of mercury is occurring in the southern cove.

The human health and ecological risk assessment were focused on the HoltraChem site and immediate vicinity. Based on the assumptions used in the human health risk assessment, ingestion of fish with the concentrations measured in the eels collected in the vicinity of the site and exposure to average sediment concentrations in the southern cove each contribute approximately 0.1 to the total non-cancer site risk which is below the hazard index threshold of 1.0.

The southern cove is subaerially exposed twice daily and there are few resident fish predator species in this fresh to saline transitional tidal reach of the river. Eel samples were collected and analyzed for total mercury. Eels were selected as the top resident predator species and most likely to show potential food chain effects. The mercury concentrations of eel fillets collected near the site were not statistically different from eel fillets collected upriver and from other rivers in the state.

Whole sediment bioassays (toxicity tests) were also performed on samples collected from the southern cove in 1995, 1997, and 1998 to develop a preliminary MPS as described in Section 7. The concentration of mercury in the sediments bioassayed ranged from 1.9 mg/kg (1.6 mg/kg, HMC lab) to 37 mg/kg (48.9 HMC lab). No chronic adverse effects on survival or growth of freshwater amphipods *hyalella azteca* were observed in exposure to any of the samples. Based on the bioassay results, the 37 mg/kg mercury concentration is considered to be a no observed adverse effects level (NOAEL) for sediments in the southern cove with less than or equal to 1% organic carbon. The NOAEL could extend to 49 mg/kg based on the HoltraChem lab results. These results are consistent with the site-specific partitioning analysis described above which predicted a biological effects level greater than 70 mg/kg total mercury. Higher NOAELs are predicted for sediments with greater amounts of organic carbon.

Based on the results of sediment chemistry analyses, a total mercury concentration of 70 mg/kg would not be expected to adversely affect biota such as benthic invertebrates. Therefore, a PMPS of 70 mg/kg, the predicted no observed adverse effect level, is proposed for sediments in the southern cove area of the Penobscot River. The point of compliance is the average sediment concentration in the vicinity of the north ditch and outfall 001.

It is important to note that the PMPS proposed for southern cove sediments was developed using site specific data and is considered protective of the southern cove and the Penobscot River in the vicinity of the HoltraChem site based on the particular characteristics of those environments. The HoltraChem site and southern cove characteristics are unique in several respects including: large tidal range and flow volumes and the transitional or brackish water quality. Because it is subaerially exposed twice a day, the sediments in the cove are generally aerobic which, based on available literature and site specific chemical analyses, seems to limit production of methylmercury. In lower

energy environments (slow moving freshwater rivers and lakes) similar and lower concentrations have been shown to result in methylation and bioaccumulation. Therefore, the sediment PMPS presented and discussed above should be considered specific to the southern cove.

10.5.6 Onsite Surface Water and Sediment

The north ditch discharges to the Penobscot River approximately 600 feet downstream of the manufacturing plant. Discharge from this stormwater ditch is currently permitted under the plant's general stormwater discharge permit and the plant's NPDES permit (identified as Outfall 003 in the permit). However, outfall 003 is not permitted for mercury discharges.

The southerly stream also receives stormwater runoff and groundwater from the manufacturing plant. Groundwater along the southern portion of the manufacturing plant is collected in a groundwater interceptor trench designed for this purpose. However, based on dissolved mercury concentrations measured in the southerly stream, it is likely that a portion of the groundwater discharges to the stream. The southerly stream was in existence before the plant was constructed and is therefore considered "waters of the State of Maine". However, due to its low non-storm flow rates and steep drop to the Penobscot River, the southern stream would not be expected to support fish.

VOCs are not anticipated to be in site surface water because they were not detected in 12 surface water samples collected by Acheron. The concentrations of mercury in sediment present at the site ranged from non-detectable to 96 mg/kg. The source of the highest detected concentration (1000 mg/kg in the paved sump) was excavated and disposed offsite. Some sediments in the ditches could migrate to the Penobscot River during large storm events.

The same PMPS for mercury is proposed for onsite sediment as was proposed for the Penobscot River (70 mg/kg). This concentration is considered protective of human health and the environment and would not result in additional material transport to the southern cove above the PMPS set for sediment there. The national recommended ambient water quality criteria of 0.77 ug/l measured as dissolved total mercury is proposed as the PMPS for surface water in the southern stream. Based on EPA's most recent water quality criteria compilation, this value is considered protective of fresh water aquatic life under chronic conditions. In addition, for the protection of freshwater aquatic life in the southerly stream, a PMPS for specific conductivity is proposed as the background concentrations in the southerly stream measured upstream of the HoltraChem Plant. To be protective of the southern cove sediments, the PMPS for onsite surface water must also address suspended solids. Therefore, a suspended solids mercury concentration of 70 mg/kg is also proposed a PMPS for onsite surface water. The point of compliance for the sediments in the north ditch and southerly stream is the average sediment concentration along the ditch and stream. For measurement purposes, the point of compliance for surface water in the southerly stream is proposed at the point where the stream leaves the manufacturing plant near the pH adjustment weir.

Ambient water quality criteria are not proposed as a PMPS for the waters of the north ditch because it is part of the stormwater collection system, does not support fish populations and is not considered "waters of the State". Stormwater outfall 003 is monitored at the paved sump for mercury, free chlorine and pH. Outfall 003 is not currently permitted for mercury and State law prohibits permitting of new sources. However, the impact of stormwater outfalls on the environment are typically assessed after accounting for a mixing zone and dilution into the receiving body. Permissible total maximum daily loads (TMDLs) are then determined. Establishment of performance standards for stormwater in the north ditch will be deferred until permitting requirements and total maximum daily loads (TMDLs) can be established.

10.5.7 Soil

The Consent Decree stipulates establishment of preliminary MPS for soil based on health and environmental risk. The Consent Decree also requires that leaching of hazardous constituents from the soil be considered in setting the preliminary MPS. Each of these was considered in the development of a PMPS for soil and are discussed below. In addition, the MEDEP Remedial Action Guidelines for Hazardous Substances in Soils (MEDEP, 1997) were considered as an ARAR.

Human Health

A mercury concentration in soil considered protective of human health for the current and future trespasser scenarios is 140 mg/kg. These scenarios are more sensitive to mercury exposure than the site worker scenario and therefore result in a lower number than would be indicated for protection of the site worker. A mercury concentration in soil considered protective of human health considering the hypothetical future onsite resident growing and eating vegetables is 6 mg/kg. The human health based MEDEP remedial action guidelines for soils contaminated with mercury are 60 mg/kg for the residential scenario, 320 mg/kg for the trespasser and 610 mg/kg for the adult worker.

Ecological Risk

The ecological risk assessment indicates there may be potential risk to songbirds and small mammals feeding directly on soil invertebrates (worms) on portions of the site. Based on its diet largely consisting of earthworms, the shrew was considered to be the most sensitive receptor followed by the robin. The concentrations of mercury in the worms correlate to the concentrations of mercury in soil. The food chain model indicates soil concentrations above the range 1.2 to 10 mg/kg, are those in which mercury could begin to have an adverse effect on the life processes on small burrowing mammals. However, there is inherent uncertainty associated with estimating risk based on food chain exposure models, including a dependence on conservative assumptions.

Because of the limited size of the area of concern and the specific niche which the sensitive receptors occupy, there does not appear to be a significant impact on species populations or any adverse effect on the ecosystem as a whole. For example, shrews which feed primarily on earthworms may be impacted by the mercury concentrations in the soil within the most contaminated areas of the site. However, the deer mice, which feed primarily on plants, do not appear to be impacted. Therefore, the relative paucity of shrews in a limited area will not represent a missing link in the food web to higher trophic species like the fox and the owl which feed over a large area, hunt most where they are successful and can also live on deer mice and other small game. Observations of diversity and abundance of wildlife on site did not indicate that the ecosystems have been significantly affected. These observations are supported by the application of a population effects model using the reproductive endpoints for songbirds and mammals which showed that the potential risk to individual organisms does not extend to population level effects.

The inputs to the model were adjusted to evaluate what concentration over an extensive area would result in predicted population level effects. Based on the foraging patterns of the animals modeled, soil mercury concentrations would have to exceed 1000 mg/kg over large areas of the plant and surrounding vicinity before 10% of the populations would be affected. Because this number was greater than numbers protective of human health, the ecological risk assessment was not used to establish a media protection standard proposal for soil.

Leaching to Groundwater

Contaminants in soil can be a source of contamination to groundwater. As discussed in Section 7.2.3, the variety of potential reactions controlling the leachability of mercury from soil is complex and depends on many factors such as pH, redox potential, the presence of other reacting chemicals and the species of mercury itself. Detailed models which can predict the actual leaching rate are not available and the published literature is essentially devoid of measured soil sorption constants for mercury in realistic soil environments. The available literature generally indicate that there is a relatively rapid and irreversible sorption, or incorporation, of mercury into soil.

The potential for mercury in surficial soils to leach to groundwater was evaluated using batch desorption tests and water samples collected from lysimeters. The objective of the batch desorption study was to develop a desorption isotherm for mercury in the sitewide soil: a graphical representation of the distribution of mercury between soil and water when the two phases are at equilibrium. The tests were conducted on soil samples collected from two areas of the site. For a soil mercury concentration of 13 mg/kg, the desorption testing showed that equilibrium had been reached such that no more mercury was desorbed. The desorption of mercury from the 13 mg/kg soil resulted in percolate water concentrations in the 2 $\mu\text{g/l}$ range for each of the soil-water ratios tested. The equilibrated water concentrations measured during this study can be viewed as representative of percolate water from the surficial soil layer. Site soils have a capacity to retain mercury at

concentrations less than or equal to 13 mg/kg without yielding percolate at concentrations above the groundwater MCL of 2.0 $\mu\text{g/l}$. Therefore, soil with 13 mg/kg or less should not represent a continuing source for groundwater contamination.

In actual field conditions this percolate would then pass through subsequent layers of soil which also have capacity to sorb mercury. This attenuation of mercury in the percolate as it travels through the vadose zone serves to limit the depth to which mercury, and other metals migrate downward in soil profiles. The results of the desorption tests were supported by the collection of percolate samples from lysimeters placed on the bedrock ridge. One of the lysimeters was installed approximately 18 inches below ground surface in the area where the 13 mg/kg soil used in the desorption test was collected. The total mercury concentration of this lysimeter sample was 2 $\mu\text{g/l}$ and the dissolved sample concentration was less than the method detection limit.

To support development of numerical PMPS, site specific mercury partitioning coefficients (K_d) were also calculated for site soils, using the results of batch desorption analyses. Although the desorption results did not allow a representative Freundlich isotherm to be established, point estimates of K_d were calculated using individual water and soil mercury concentrations from the six batch desorption analyses. Site specific desorption results indicated a range of K_d values from 2,107 to 8,647, with an average of 5,493. The mean of the site specific results is in agreement with the upper range of values tabulated in the EPA soil screening guidance documentation (EPA, 1996).

The MEDEP Remedial Action Guidelines for Hazardous Substances in Soils acknowledges that the soil screening levels which are presented in EPA's Final Soil Screening Guidance (EPA, 1996) are "highly conservative" and "The DEP encourages development of less conservative site specific remedial guidelines using sound investigation and risk assessment techniques" (MEDEP, 1997). To identify a site specific soil concentration protective of groundwater, the site specific K_d values obtained from the desorption tests were used in place of the default value in the model which generates soil screening levels protective of groundwater. The model used by EPA in the Soil Screening Guidance and MEDEP in the Remedial Action Guidelines for Hazardous Substances in Soils assumes a K_d of 50 mg/kg and a dilution attenuation factor of 20. The model indicates soil concentrations of 85 mg/kg and 220 mg/kg would be protective of groundwater based on the minimum and mean site specific K_d values, respectively.

Based on the results of the soil desorption and lysimeter tests and the guidelines for development of site specific values protective of groundwater (EPA and MEDEP), a PMPS of 85 mg/kg is supportable. This value generated using the minimum site specific K_d is close to, yet slightly higher than, the level established by MEDEP for the residential scenario (60 mg/kg, based on ingestion). The risk assessment and site specific data show that this level would be protective of human health, terrestrial ecological populations and site groundwater. However, because site soil has a potential to be transported by overland runoff and become sediment, a lower value may be required to protect

sediment. Therefore, a PMPS of 60 mg/kg, the MEDEP recommended Remedial Action Guideline for Hazardous Substances in Soils under the residential scenario, is proposed for site surface soil. The proposed surface soil PMPS is protective of human and ecological health and Penobscot River sediment under current conditions. Containment of soils by paving within the manufacturing site would significantly reduce the potential for infiltration and/or runoff with overland flow.

The proposed PMPS for soil is protective of human health, the environment and groundwater for all current and future scenarios except onsite residential use. If the plant is redeveloped for residential use in the future, there would be a risk to human health based upon onsite drinking water consumption and ingestion of homegrown produce. Therefore, a future land use restriction should be placed on the property if the proposed PMPS are adopted.

10.5.8 Air

There were no human health or ecological risks associated with contaminant vapors or dust in air. The human health risk assessment calculated the highest estimated mercury concentration in air from onsite releases related to spills and fugitive dusts to be $5.7 \times 10^{-3} \mu\text{g}/\text{m}^3$. The maximum 24-hour average concentrations measured to date as part of the Comprehensive Monitoring Program have been $5.7 \times 10^{-2} \mu\text{g}/\text{m}^3$ at the monitoring stations on the PERC property near Route 15. The average concentration at this location to date is $1.2 \times 10^{-2} \mu\text{g}/\text{m}^3$.

EPA Region III has established a risk-based concentration which results in a hazard index of 1. This value, $0.31 \mu\text{g}/\text{m}^3$ ($3.1 \times 10^{-1} \mu\text{g}/\text{m}^3$) is proposed as the preliminary MPS for air. Based on modeling and monitoring performed to date, plant related concentrations are approximately one order of magnitude below the proposed media protection standard. The point of compliance is proposed as the property line for offsite residences.