FINAL
REMEDIAL INVESTIGATION REPORT
STRATFORD ARMY ENGINE PLANT
Stratford, Connecticut

September 2004

U.S. Army Assistant Chief Of Staff
For Installation Management
National Capital Region Field Office
Alexandria, Virginia

REMEDIAL INVESTIGATION REPORT
STRATFORD ARMY ENGINE PLANT

TABLE OF CONTENTS

Section No. **Title** **Page No.**

TABLE OF CONTENTS

Section No. Title Page No.

TABLE OF CONTENTS

Section No. Title Page No.

TABLE OF CONTENTS

Section No. Title Page No.

TABLE OF CONTENTS

Section No. Title Page No.

TABLE OF CONTENTS

ACRONYMNS

REFERENCES

FIGURES

TABLES

APPENDICES

APPENDIX A CTDEP REMEDIATION STANDARD REGULATIONS
A-1 CTDEP REMEDIATION STANDARD REGULATIONS

TABLE OF CONTENTS

Section No. **Title** **Page No.**

TABLE OF CONTENTS

Section No. **Title** **Page No.**

TABLE OF CONTENTS

TABLE OF CONTENTS

Section No. Title Page No.

REMEDIAL INVESTIGATION REPORT
STRATFORD ARMY ENGINE PLANT

LIST OF FIGURES

Figure No. Title

LIST OF FIGURES

REMEDIAL INVESTIGATION REPORT
STRATFORD ARMY ENGINE PLANT

LIST OF FIGURES

Figure No. **Title**

LIST OF FIGURES

LIST OF FIGURES

Figure No. Title

LIST OF FIGURES

LIST OF FIGURES

Figure No. Title

LIST OF FIGURES

Figure No. Title

LIST OF TABLES

REMEDIAL INVESTIGATION REPORT
STRATFORD ARMY ENGINE PLANT

LIST OF TABLES

Table No. Title

REMEDIAL INVESTIGATION REPORT
STRATFORD ARMY ENGINE PLANT

LIST OF TABLES

Table No. Title

LIST OF TABLES

LIST OF TABLES

Table No. Title

LIST OF TABLES

Table No. Title

EXECUTIVE SUMMARY

The United States Army (U.S. Army) has prepared a remedial investigation (RI) for Stratford Army Engine Plant (SAEP) to characterize the nature and extent of contamination and evaluate potential risk to human health and the environment attributable to the Site. In October 1995, SAEP was placed on the Base Closure and Realignment (BRAC) list, known as BRAC 95. U.S. Army BRAC properties must be investigated to determine the nature and extent of environmental contamination. The environmental investigation, and evaluation of potential remedial alternatives, at the SAEP facility are being conducted pursuant to the Comprehensive Environmental Response, Compensation and Liability Act (CERCLA), the National Contingency Plan, and Executive Order 12580.

CERCLA §121(d)(1) requires remedial actions to attain a degree of cleanup that assures protection of human health and the environment [42 U.S.C. §9621(d)(1)]. This CERCLA requirement is implemented through means of a risk assessment which, based on site-specific exposures, identifies contaminant exposure pathways that present either a current or potential future unacceptable risk. When such a risk is identified, a response action is required to address the unacceptable risk. The unacceptable risks presented by contamination at SAEP, will be addressed in accordance with the Connecticut Remediation Standards outlined in Title 22a-133k of the Connecticut Administrative Regulations.

SAEP is located in Stratford, Connecticut, on the Stratford Point peninsula adjacent to the Housatonic River in the southeast corner of Fairfield County. SAEP was a government-owned, contractor-operated facility. The U.S. Army owns the land and the buildings and has responsibility for the jurisdiction, control, and accountability of SAEP. There are approximately 78 acres of improved land with 52 buildings, paved roadway, parking lots and grounds. Included in the improved land are an estimated 15.7 acres along the Housatonic River where fill was placed over tidal flats. An estimated 2 acres of this property comprise a Causeway constructed in the 1930's to provide access to the river channel. There are also approximately 49 acres of riparian rights consisting of tidal flats and river channel.

SAEP has a long industrial history and was used to develop, test, and manufacture aircraft and engines, and other aerospace products, for 65 years. From 1929 until 1948 the plant was used to develop and manufacture aircraft. The earliest buildings were constructed in 1929 for the Sikorsky Aircraft Corporation. The plant was expanded during World War II to accommodate mass production. During this time the shoreline was extended to provide land area for new buildings. The plant was idle from 1948 until 1951. From 1952 until it closed in 1997 the plant was used to produce reciprocating aircraft engines and turbine engines for both commercial and military applications. The manufacturing of turbine engines required the use of oils, solvents, plating solutions, fuels and other chemicals. Industrial operations included machining; electrochemical machining; electroplating; corrosion prevention; degreasing; painting; and engine testing. Support activities included maintenance, storage of raw materials and wastes, industrial and chemical wastewater treatment and waste recovery. Most of the wastes generated resulted from production in Buildings B-2. Wastes were also generated as a result of engine and engine component testing and research and development in Buildings B-3, B-6, B-16 and B-19.

The ground surface is relatively flat with elevations less than 10 feet mean sea level. Buildings and pavement cover most of the site. Surficial deposits consist of glacial outwash, alluvium, silt and fill. Fill is found at the surface throughout most of the site and is thickest along the Dike. The underlying silt deposit consists of dark organic fine silts with very fine sands, which are occasionally clayey. The thickness of the silt layer varies from as much as 30 feet adjacent to the Dike at the north end of the facility, to nonexistent toward the interior of the facility. Alluvium and outwash deposits underlie the fill and silt. The alluvium consists of sand with clay, silt, and cobbles. The glacial outwash beneath the alluvial and above the bedrock surface is a fine to medium sand with some silt. Bedrock ranges in depth from about 49 feet to 184 feet below ground surface (bgs).

Groundwater occurs at depths ranging from approximately 5 to 8 feet bgs. Groundwater flow in the shallow aquifer is generally from west to east and southeast toward the Intertidal Mudflats. The horizontal gradient beneath Building B-2 is relatively flat but increases near the Dike to the northeast. The estuarine silts act as a low permeability boundary to groundwater flow in this area resulting in the increased gradient. Groundwater flow in the deeper portions of the aquifer is from also west to east. Vertical gradients indicate slight upward to neutral gradients from deep monitoring wells screened on top of bedrock, to shallower wells. Horizontal gradients increase in the deep aquifer in the area of Building B-19. These steep gradients are attributable to high bedrock elevations in this area with the attendant lowered transmissivity of the overburden aquifer. The estimated horizontal groundwater velocity ranges from near zero feet/year beneath Building B-2, to 255 feet/year near the tidal flats. The higher velocities occur in the more permeable sections of the overburden deposits.

The RI has been a multiphase effort. The Phase I and Phase II RI reports were completed in 1993 and 1996, respectively. When SAEP was placed on the 1995 BRAC list the focus of the site characterization shifted to a comprehensive investigation that would allow for property transfer. The Phase III RI was conducted between 1998 and 2004, and included sampling and analysis of soil, soil vapor and indoor air, groundwater, surface water, sediment and biota. A human health baseline risk assessment was performed to provide an assessment of the potential health risks associated with reasonably foreseeable future land use and exposures to chemicals at the site in the absence of any remedial action. A baseline ecological risk assessment was conducted to characterize potentially exposed ecosystems that may be at risk, and to identify biological populations that represent potential receptors therein.

Over 500 soil samples were collected from locations throughout the site to characterize releases associated with areas of concern (AOCs). In soil, contaminants detected include chlorinated solvents, fuel-related volatile organic compounds (VOCs), polychlorinated biphenyls (PCBs), polycyclic aromatic hydrocarbons (PAHs), total petroleum hydrocarbons (TPH), and hexavalent chromium. VOCs and PAHs were detected throughout the site at levels below CTDEP Remediation Standard Regulations and are not associated with specific release areas. Releases of contamination to soils have occurred where solvents and fuels were used and stored including the Hazardous Waste and Waste Oil Area and the Jet Fuel Storage Area. At these locations contaminants including TPH, VOCs and PAHs were detected above Connecticut Remedial Standard Regulations (RSRs). A release also occurred at the B-2 Chromium Plating Facility where chromium was detected above RSRs in the soils beneath the concrete floor.

The groundwater investigations used a combination of direct-push sampling and monitoring well installation to identify the nature and extent of the contamination. Groundwater monitoring indicates that chlorinated solvent contamination underlies most of the property and extends from the water table to over 150 feet bgs. Compounds detected include 1,1,1-trichloroethane (1,1,1-TCA), 1,1-dichloroethane (1,1-DCA), 1,1-dichloroethene (1,1-DCE), tetrachloroethene (PCE), trichloroethene (TCE), cis-1,2-dichloroethene (cis-1,2-DCE), and vinyl chloride. Releases of TCE have occurred beneath the former B-2 Chromium Plating Facility and near Building B-48. Releases of 1,1,1-TCA have occurred beneath the manufacturing areas of Building B-2. VOC and BTEX contamination also exists beneath the Hazardous Waste and Waste Oil Area and along the shore at the Tidal Flats and the former Jet Fuel Storage Area.

TCE concentrations in shallow groundwater are above the industrial/commercial (I/C) volatilization criteria (VC) across much of the site. Concentrations of 1,1-DCE and vinyl chloride in groundwater near the eastern property boundary in the vicinity of Building B-19 exceed their respective I/C VC. Soil vapor analyses indicate TCE and 1,1-DCE beneath much of the main site with the highest concentrations beneath Building B-2. TCE in soil vapor above the I/C VC was detected beneath Buildings B-2, B-3, B-10 and B-12. PCE concentrations are also found above the I/C VC, but less frequently.

Impacts to the shallow groundwater in the near shore Tidal Flats include the chlorinated solvents TCE, cis-1,2-DCE, and vinyl chloride, which were detected in groundwater adjacent to the Dike. The source of these chlorinated VOCs appears to be the unsaturated zone soils and shallow groundwater associated with the former Hazardous Waste Storage Areas and Former Jet Fuel Storage Area. Light non-aqueous phase liquid is present between the former Hazardous Waste Storage Area and the Oil Abatement Treatment Plant and between Buildings B-34 and B-16.

Groundwater downgradient of the site is below Surface Water Protection Criteria (SWPC), indicating VOCs are being naturally attenuated and/or transport times have not been sufficient for the higher concentrations of VOCs beneath the facility to migrate beneath the Tidal Flats. Reducing conditions measured in groundwater near and north of the Dike may be stimulating reductive de-chlorination of the TCE to cis-1,2-DCE and vinyl chloride.

Chemical transformations also limit the migration of the chlorinated solvents 1,1,1-TCA, PCE, and TCE in groundwater. These compounds degrade through the processes of natural attenuation. 1,1,1-TCA will abiotically degrade to 1,1-DCE, and biotically to 1,1-DCA and chloroethane; and PCE degrades to TCE, which then may degrade to dichloroethene. It is widely observed that cis-1,2-DCE is the preferential form of dichloroethene resulting from PCE and TCE degradation, and correlations between TCE and 1,2-DCE concentrations at SAEP support this assumption. The 1,1-DCE has resulted from abiotic degradation of 1,1,1-TCA, which correlates spatially with 1,1-DCE concentrations.

A hexavalent chromium plume with maximum concentrations of 950 milligrams per liter (mg/L) was detected beneath the Building B-2 Chromium Plating Facility and the alley between Building B-2 and B-10. The mobility of the highly soluble hexavalent chromium, present as the chromate ion, is limited by reducing conditions present beneath, and around the perimeter of, the B-2 Chromium Plating Facility. Reduction of hexavalent chromium to trivalent chromium, which has far lower solubility, has retarded the mobility of hexavalent chromium groundwater contamination.

Surface water and sediment samples were collected from the Tidal Flats, Marine Basin, and the Outfall 008 Drainage ditch in 1992, 1994, and 1999. VOCs, semivolatile organic compounds, PCBs and metals were detected in surface water and sediment samples. There are no RSR criteria for surface water and sediment, so the evaluation of chemical contamination was based on the results of the risk assessment.

The Human-Health Baseline Risk Assessment (HHBRA) was conducted to characterize health risks at the site in accordance with U.S. Environmental Protection Agency (USEPA) performance standards for risk characterization. The HHBRA included hazard identification, exposure assessment, toxicity assessment, and risk characterization. Hazard indices (HIs) less than 1 are generally interpreted by USEPA to be indicative of acceptable risk for non-cancer effects associated with chemical exposure. The USEPA guidelines indicate that the total excess lifetime cancer risk due to exposure to the chemicals at the site, by each complete exposure pathway, should not exceed a range of 1E-06 to 1E-04.

Based on the anticipated future land use, the HHBRA considered exposure to the following human populations: commercial/industrial employees, including commercial workers (e.g., office work), and industrial workers (e.g., manufacturing work); recreational anglers and shellfishermen; commercial anglers and shellfishermen; construction workers; and utility workers.

For each receptor group, the following exposure areas were evaluated: Main Site; Former Chromium Plating Area and the South Lot. For the recreational receptors, the tidal flats, Marine Basin and Outfall 8 were also considered as exposure areas. For all receptors, future use of the facility for residential development was considered implausible; the HHBRA incorporates the assumption that a land use restriction will be in place to prevent residential development without prior evaluation of such use.

The results of the HHBRA indicate that health risks associated with potential exposures to contaminants of potential concern (COPCs) in indoor air and surface soil, and indoor air and subsurface soil, under possible future commercial/industrial land use conditions do not exceed an HI of 1 (highest HI = 1). However, cancer risks for combined exposures to soil and indoor air at the Main Site exceed the USEPA cancer risk range. The risks in excess of the USEPA cancer risk range are due to potential exposures to indoor air at Building B-2. The cancer risks associated with surface soil and subsurface soil (highest risk = 7E-05) are at the upper end of the USEPA acceptable cancer risk range and exceed the CTDEP cancer risk limit. Risks in soil are primarily contributed by arsenic and PAHs.

Since RSRs are not available for surface water and sediment, the results of the HHBRA is important for evaluating exposure to surface water and sediment. Risks associated with potential exposures to COPCs in surface water, sediment, and groundwater (groundwater applicable to Tidal Flats only) under future recreational use conditions at the Tidal Flats, Marine Basin, and Outfall 008 Drainage Area are within the USEPA cancer risk range (highest cancer risk = 4E-05), but risks associated with incidental contact with shallow groundwater at the Intertidal Flats (cancer risk = 3E-05) exceed the CTDEP cancer risk limit. The estimated HI value for future recreational use (wading) at the Outfall 008 Drainage ditch exceeds a value of 1 (highest HI is 3) under the assumption that chromium detected in ditch sediments is present as hexavalent chromium. It is likely that the total chromium in the sediments is in the trivalent form because of the anaerobic conditions in this medium; under that assumption, the HI for potential recreational exposures to sediments at the Outfall 008 Drainage Ditch would be below a value of 1.

The results of the HHBRA indicate that risks associated with ingestion of finfish, ribbed mussels, and oysters associated with recreational fishing under hypothetical future conditions exceed an HI of 1 and/or the USEPA cancer risk range. Health risks associated with consumption of biota taken from the Intertidal Flats are higher than risks associated with consumption of biota taken from the Marine Basin. However, risks associated with recreational consumption of finfish and ribbed mussels taken from the Site are less than or equal to risks associated with recreational consumption of finfish and ribbed mussels at background locations. Based on available information, it does not appear that frequent recreational finfishing or shellfishing has occurred at the Tidal Flats or Marine Basin. Moreover, in order for sport fishing to occur at these areas to an extent that could permit a person to be exposed to biota in a manner consistent with the exposure assumptions used to estimate risks, the shellfish and finfish population densities in these areas would need to improve considerably.

The results of the HHBRA indicate that risks associated with hypothetical future commercial angling for finfish and commercial shellfishing are generally within the USEPA cancer risk range and below an HI of 1. The results of the HHBRA indicate that ingestion of finfish and oysters at the Tidal Flats exceed an HI of 1 and/or the USEPA cancer risk range, but risks associated with surface water and sediment contact, and consumption of other biota (finfish at the Marine Basin and ribbed mussels at the Marine Basin and Tidal Flats) are within USEPA risk management criteria. Risks associated with consumption of finfish and ribbed mussels taken from the Site are less than or equal to risks associated with consumption of finfish and ribbed mussels at background locations. Based on current trends in aquaculture, commercial fishing is unlikely to occur at the Tidal Flats or Marine Basin in the future. Moreover, in order for commercial fishing to occur at these areas to an extent that could permit a person to be exposed to biota in a manner consistent with the exposure assumptions used to estimate risks, the shellfish and finfish population densities in these areas would need to improve considerably.

The Baseline Ecological Risk Assessment (BERA) was conducted to characterize ecological risks at the site in accordance with USEPA performance standards for risk characterization. The BERA included a problem formulation, analysis (exposure assessment and effects assessment), and risk characterization. The BERA indicates that risks to macroinvertebrates are present in the Outfall 008 Drainage Area, but acceptable at other locations. Risks to the majority of other ecological receptors (i.e., fish and wildlife) are generally acceptable in all areas evaluated. Potential risks to listed bird species (piping plover) were identified at the Outfall 008 Drainage Area, and potential risks to the piping plover were also identified at the Tidal Flats. A summary of ecological risk characterization is presented below.

The results of the BERA indicate that there is no significant risk to macroinvertebrates in the Tidal Flats. Although there is a possible shift in the benthic community assemblage, the overall richness and abundance are not affected. The results of the sediment toxicity tests and benchmark comparisons did not indicate any particular analyte responsible for the observed effects, and observed effects may be due to habitat differences or other non-chemical stressors.

The results of the BERA indicate that there is a potential risk to macroinvertebrates in the Outfall 008 Drainage, due to inorganics (barium, chromium, and copper) and Aroclor-1260 in sediment. The Outfall 008 Drainage represents a physically disturbed habitat of lower ecological value than the surrounding Marine Basin and Tidal Flats. The results of the BERA, particularly the sediment toxicity test results, benthic community results, and the comparison of tissue concentrations to CBRs, indicate that there is no significant risk to macroinvertebrates in the Marine Basin.

The results of the BERA indicate that there is no significant risk to forage fish inhabiting the Tidal Flats or Marine Basin. Tissue concentrations are comparable to tissue concentrations from reference locations. Although there are some elevated concentrations in sediments, tissue concentrations are similar to those from reference areas. In addition, species observed at these locations are typical of those that would be expected; the presence of juveniles and fish in breeding condition are indicative of a healthy fish community.

The results of the BERA indicate that there is no significant risk to semiaquatic birds (represented by the black duck, great blue heron, and sandpiper) that may forage at the Marine Basin. At the Tidal Flats, there is no significant risk to the black duck and great blue heron, but a potential risk to sandpipers due to chromium in sediment and mercury (assumed to be methyl mercury) in biota. The food chain model assumes that the sandpiper feeds exclusively at the Tidal Flats; however, it is unlikely that the entire local population of sandpipers feeds exclusively at the Tidal Flats, and therefore risks to populations of this species are unlikely. At Outfall 008, chromium concentrations in sediment may pose a risk to these species if they frequently forage at this location. Given the limited habitat quality at Outfall 008, this is considered unlikely.

The results of the BERA indicate potential risk to listed bird species that may forage at the Tidal Flats and Outfall 008. Given the limited habitat, these species are unlikely to frequently forage at Outfall 008, and risks are likely overestimated at this location. There is uncertainty as to whether and how frequently plovers utilize the Tidal Flats. The food chain model assumed that the sandpiper (surrogate for the plover) feeds exclusively at the Tidal Flats but the Tidal Flats near SAEP were monitored regularly between May and August 1997 and no piping plovers were observed feeding there (LeBlanc, 1997). Based on those observations, it appears unlikely that piping plovers utilize the Tidal Flats, or if so, with only limited frequency, and exposures and risks to this species may have been overestimated.

1.0 INTRODUCTION

1.1 PURPOSE

The Stratford Army Engine Plant (SAEP) was formerly a government-owned, contractor-operated facility located in Stratford, Connecticut. In October 1995, SAEP was placed on the Base Closure and Realignment (BRAC) list. Pursuant to the Defense Base Closure and Realignment Act of 1990, the BRAC Environmental Restoration Program mandates that environmental contamination on U.S. Department of the Army (U.S. Army) BRAC properties be investigated prior to disposal and re-use.

The purpose of the Remedial Investigation (RI) is to determine the nature and extent of contamination, evaluate fate and transport of contamination, and evaluate the risk to human and ecological receptors. The RI is the source of data used in the feasibility study (FS) to develop and evaluate appropriate alternatives for remediation of Site contamination.

The RI has been a multi-phase effort. The Draft RI Report was submitted to the U.S. Environmental Protection Agency (USEPA) and Connecticut Department of Environmental Protection (CTDEP) in September 2000. In response to CTDEP and USEPA comments, supplemental investigations were conducted and a Revised Draft RI report was submitted in January 2003. CTDEP provided comments on the Revised Draft RI report in October 2003.

1.2 REPORT ORGANIZATION

The RI Report is organized as follows:

Volume I	Table of Contents, Text, and Tables
Volume II	Figures
Volume III	Appendices A through K on CD-ROM (viewable in Adobe® Acrobat Reader): CD-ROM located at the end of Volume II
Volume IV	Appendices L through X on CD-ROM (viewable in Adobe® Acrobat Reader): CD-ROM located at the end of Volume II

1.3 REGULATORY FRAMEWORK

The SAEP was closed pursuant to the Defense Base Closure and Realignment Act of 1990. Environmental investigation is being conducted pursuant to the Comprehensive Environmental Response, Compensation and Liability Act (CERCLA), the National Contingency Plan (NCP), and Executive Order 12580. The BRAC Environmental Restoration Program mandates that environmental contamination on BRAC properties be investigated and remediated in accordance with the intended future use of the site, as necessary, prior to disposal and reuse. The site is not listed as a National Priorities List (NPL) site by the USEPA; therefore, the CTDEP is acting as the lead regulatory agency for the investigation and remediation activities conducted at SAEP.

CERCLA §121(d)(1) requires remedial actions to attain a degree of cleanup that assures protection of human health and the environment [42 U.S.C. §9621(d)(1)]. This CERCLA requirement is implemented through means of a risk assessment, which based on site-specific exposures, identifies contaminant exposure pathways that present either a current or potential future unacceptable risk. When such a risk is identified, a response action is required to address the unacceptable risk.

CERCLA §121(d)(2)(A) requires that on-site remedial actions must meet the standards and criteria that are otherwise legally applicable to the substance, pollutant, or contaminant or that are relevant and appropriate under the circumstances [42 U.S.C. § 9621(d)(2)(A)]. An applicable or relevant and appropriate requirement (ARAR) may be based on a legally promulgated federal standard or legally promulgated and enforceable state standard as long as the state standard is more stringent. The compliance with ARARs mandate arises under CERCLA 121(d)(2)(A) when an on-site response action is required.

At SAEP, the unacceptable risks presented by contamination in the groundwater and soil will be addressed in accordance with the Connecticut Remediation Standards outlined in Title 22a-133k of the Connecticut Administrative Regulations.

1.3.1 Connecticut Remediation Standards

As discussed in Subsection 1.6.1, four lagoons/surface impoundments operated at the Chemical Waste Treatment Plant (CWTP) managed hazardous waste operated by a contractor at the site. Hazardous wastes were disposed of and treated in the lagoons/surface impoundments after July 26, 1982. The facility operated under Resource Conservation and Recovery Act (RCRA) interim status authority. These lagoons/surface impoundments were closed with waste in place under the RCRA Subtitle C requirements and thus continue to be subject to the interim status regulations.

Under the Connecticut RCRA regulations, the requirements for corrective action at interim status disposal facilities are applicable to owners or operators of a “disposal facility” required to operate under, currently operating under, or authorized to operate under interim status [Conn. Agencies Regs. §22a-449(c)-105(h)(2)]. “Disposal facility” is defined in part as a facility that has not been issued a hazardous waste permit at which hazardous waste was disposed of in a surface impoundment after July 26, 1982 [Conn. Agencies Regs. §22a-449(c)-105(h)(1)(A)]. Since the lagoons/surface impoundments at the CWTP managed hazardous waste after July 26, 1982, SAEP would meet the definition of a “Disposal facility”. Those facilities subject to the regulations are required to investigate and remediate all releases of hazardous waste and hazardous substances at or from the facility in accordance with the requirements for corrective action at interim status disposal facilities [Conn. Agencies Regs. §22a-449(c)-105(h)].

The corrective action regulations for interim status facilities were promulgated in August of 2002. The regulations require owner or operators of disposal facilities to submit an “environmental condition assessment form” or “ECAF” to the state by August 27, 2003, which supplies information about environmental conditions at the facility [Conn. Agencies Regs. §22a-449(c)-105(h)(3)]. SAEP, notified by the state that they were required to submit an ECAF, filed an ECAF before the administrative deadline. The ECAF is intended to provide an initiating mechanism for corrective action other than an order or permit in Connecticut’s RCRA program. After an ECAF has been submitted, the state notifies the owner or operator of the need of investigation and remediation in accordance with the regulations. SAEP received such notification dated November 12, 2003 addressing the implementation and coordination of corrective action at the facility in accordance with Connecticut Agencies Regulations §22a-

449(c)-105(h). Any remediation conducted under the corrective action regulations for interim status facilities must be in accordance with the remediation standards of Conn. Agencies Regs. §22a-133k, also known as the Remediation Standard Regulations or RSRs [Conn. Agencies Regs. §22a-449(c)-105(h)(5)(C)].

The Connecticut RSRs establish cleanup requirements for soil and groundwater. The applicable RSRs to SAEP are risked-based standards for the exposure pathways pertinent to this site including, dermal contact with surface soil, vapor inhalation at the ground surface or in overlying buildings, leaching of soil contaminants to groundwater, and contaminated groundwater migration to surface water. The RSRs are based on the excess lifetime cancer risk (ELCR) level of $1x10^{-6}$ for individual carcinogens, excess lifetime cancer risk level of $1x10^{-5}$ for multiple carcinogens and a level that does not exceed the hazard quotient (HQ) of 1 for non-carcinogens. For the human health/ecological exposure pathways addressed by the RSRs, the outcome of the human health/ecological risk assessment will not be determinative of the need for remedial action since, regardless of risk, the RSRs must be met. In addition, the need to establish remedial action goals through risk assessment for these exposure pathways will not be necessary since by meeting the RSRs, which is already required, compliance with the CERCLA §121(d)(1) mandate for protection of human health and the environment will be assured since the risk levels used in the RSRs either reflect or are more stringent than the CERCLA risk range. However, the results of the human health/ecological risk assessment for exposure pathways covered by the RSR standards may be used as a basis to establish alternative cleanup standards under the RSRs.

1.3.1.1 RSR Cleanup Requirements for Soil. In the soil media, the CTDEP RSR Residential Direct Exposure Criteria (DEC) or Industrial/Commerical (I/C) DEC, and GB Classified Aquifer Pollutant Mobility Criteria (GB PMC) will be used for comparison to analytical data. However, since an environmental land use restriction (ELUR) will be implemented for the entire site to prohibit residential use, the residential DEC will not be applicable to the site. Use of the residential DEC for analytical comparison is being presented to show the necessity of the ELUR, due an exceedance of a residential DEC. If the numeric I/C DEC presented in Appendix A of the RSRs is exceeded, an alternative direct exposure criterion, may be proposed provided cancer risks do not exceed the State's acceptable level of 10^{-6} for individual contaminants and 10^{-5} for the collective risk posed by multiple contaminants. Non- carcinogenic risks for soils may not exceed a Hazard Index (HI) of 1.0. For areas where soil concentrations exceed GB PMC criteria, the RSR allow for site-specific dilution criteria to be developed. For GB areas, the results from Synthetic Precipitate Leaching Procedure (SPLP) analyses can be compared to site-specific criteria, which consist of GA groundwater protection criteria multiplied by a site-specific dilution factor.

1.3.1.2 Groundwater and Soil Vapor. Groundwater beneath the SAEP site is classified as GB groundwater, which is defined by CTDEP Water Quality Standards (WQS) (CTDEP, 1996) as "Groundwater within a historically highly urbanized area or an area of intense industrial activity and where public water supply service is available. Such groundwater may not be suitable for human consumption without treatment due to waste discharges, spills, or leaks of chemicals or land use impacts." The groundwater remediation standards for Class GB (non-drinking water) provide cleanup criteria to address the groundwater to surface water discharge and the volatilization of groundwater to the ground surface. Consequently, the CTDEP RSR groundwater remediation standards applicable to SAEP include Surface Water Protection Criteria (SWPC) and groundwater volatilization criteria (VC). The RSR includes numeric SWPC and VC criteria; however, there are provisions for calculating alternative or site-specific criteria. The RSRs require compliance with the SWPC immediately upgradient of the point at which the groundwater discharges to surface water (e.g., the point-of-compliance). The Groundwater Model will be used to predict whether groundwater concentrations exceed calculated alternative SWPC at this point-of-compliance. Since the CTDEP Commissioner must approve alternative SWPC, the groundwater model must be completed and approved before the results are used to calculate alternative SWPC.

Regardless of the type of land use, the RSRs also require compliance with the residential groundwater VC when groundwater has been affected by a release of a hazardous substance and groundwater is located within 30 feet of the ground surface or an existing building. The RSRs include the following exceptions to meeting this requirement.

1. Buildings used solely for industrial or commercial activity, shall be remediated to the I/C VC provided that an ELUR is in effect to limit future use of the site.
2. Remediation to the groundwater VC is not required when soil vapor concentrations are below the VC for soil vapor or engineering controls are provided to prevent migration of vapors into the overlying building. Since all buildings remaining at SAEP will be used for industrial or commercial activity, remediation would not be required if soil vapor concentrations are below the I/C soil vapor VC. The efficacy of using engineering controls to prevent vapor migration into buildings will be evaluated in the FS.
3. The groundwater VC do not apply if there are no buildings over the contaminated groundwater and an ELUR is in effect to limit future construction of buildings.

1.3.2 Connecticut Water Quality Standards

In addition to the groundwater SWPC included in the RSRs, a groundwater plume discharging to a wetland, intermittent stream, or other surface water body must in compliance with the Connecticut Surface Water Quality Standards.

The Connecticut Surface Water Quality Standards are contained in Appendix D of Connecticut's WQS, and are issued pursuant to 22a-426 of the Connecticut General Statutes. By statute, the WQS apply to all interstate waters or portions thereof and all other waters within the state as the Commissioner of Environmental Protection (hereinafter Commissioner) determines necessary (Conn. Gen. Stat. §22a-426). As in most states "waters" is defined broadly to include "all tidal waters, harbors, estuaries, rivers, brooks, watercourses, waterways, wells, springs, lakes, ponds, marshes, drainage systems and all other surface or underground streams, bodies, or accumulations of water, natural or artificial, public or private, which are contained within, flow through or border upon this state or any portion thereof" (Conn. Gen. Stat. §22a-423). Therefore, the surface water quality standards will be applicable to the Marine Basin, Housatonic River, Tidal Flats and any other surrounding surface water bodies.

The surface water standards consist of both numeric and narrative criteria. The narrative criteria establishes a broad prohibition on any chemical constituents which alone or in combination results in "acute or chronic toxicity to aquatic organisms or impairs the biological integrity of aquatic or marine ecosystems outside of any allocated zone of influence" or bioaccumulates in aquatic organisms such that the health of aquatic organisms or wildlife is impaired or pose a health risk to consumers of aquatic life. The standards require the consideration at a minimum of the criteria in Appendix D of the WQS in determining compliance with this narrative standard. Numeric criteria are listed in Appendix D of the WQS that closely parallel the federal water quality criteria. The numeric criteria consist of aquatic life criteria (freshwater and saltwater) and human health criteria (for consumption of water and organisms, and organisms only). The surface water standards apply the pertinent aquatic life criteria (freshwater or saltwater) to all surface waters of the state. The human health water and organisms criteria apply only to those surface waters designated for use as an existing or potential water supply. All other surface waters must meet the human health criteria for organisms only.

1.3.3 Summary

Under the legal and regulatory framework of CERCLA, remedial action and cleanup standards at SAEP will be primarily driven by the CERCLA §120(a)(4) mandate to meet the legally applicable state laws at non-NPL facilities. Under this mandate, two legally applicable state requirements will drive the remedial actions/cleanup standards at the site: (1) the Connecticut RSRs for soil and groundwater, and (2) the Connecticut Surface Water Standards. Since these criteria are required to be met, regardless of the presence or absence of unacceptable risk, the risk assessment process in this RI Report serves a modified use other than the traditional use of a risk assessment in a RI Report. For those exposure pathways/media covered by the above applicable requirements, the risk assessment will not be decisive of the need for remedial action. Instead, the exceedence of the RSR standards/surface water standards will determine the need for remedial action. For these exposure pathways/media, the human health risk assessment in Section 11.0 and the ecological risk assessment in Section 12.0 will be primarily utilized as a basis to develop alternative criteria under the RSRs when determined to be pertinent and to clearly demonstrate compliance with the CERCLA protectiveness mandate in the administrative record. For exposure pathways/media not covered by the above applicable requirements, the risk assessment will be used in the traditional sense to identify media/exposure pathways (i.e., sediment and ecological receptors) that require remedial action to meet the CERCLA protectiveness mandate.

1.4 SITE DESCRIPTION

The SAEP is located in Stratford, Connecticut, on the Stratford Point peninsula in the southeast corner of Fairfield County. The site is on the border of the Bridgeport and Milford U.S. Geological Survey (USGS) Quadrangles. Latitudinal and longitudinal coordinates of the SAEP are approximately 41° 10’ North and 73° 07’ West. The location of the SAEP is shown on Figure 1-1.

SAEP consists of approximately 124 acres, of which about 76 acres are improved land and 48 acres are riparian rights. The 76 acres of improved land consist of 49 buildings, paved roadway and grounds, and five paved parking lots. The 48 acres of riparian rights property consist of Tidal Flats of the Housatonic River. A site map is provided as Figure 1-2.

SAEP has a long industrial history and was used to develop, test, and manufacture aircraft, engines, and other aerospace products for 65 years. The plant closed in 1997. Access to the Site is restricted with a perimeter fence and security personnel. SAEP is bounded by a paved parking lot and wetlands to the north; the Housatonic River to the east; an open field, a drainage channel, and small businesses to the south; and hangar buildings, the Sikorsky Memorial Airport, several small businesses, and Frash Pond to the west. Land in the vicinity is zoned light industrial, business, commercial, or residential. There are several businesses located west of Main Street, across from SAEP, including a small strip mall, service stations, and a restaurant.

Nearby recreational areas include Short Beach Park ½-mile to the southeast, and public wildlife areas, including Nells Island and the Great Meadow Salt Marsh across the Housatonic River from SAEP.

1.5 SITE HISTORY

The site history has been divided into two periods; the first (from 1929 until 1951) includes aircraft development and production, the second (from 1951 until 1997) includes the development, testing, and manufacture of piston and turbine engines.

1.5.1 1929-1951: Aircraft Production

In 1929, 26 acres of agricultural land was purchased and developed by Sikorsky Aviation Corporation for the development and manufacture of amphibious aircraft for the commercial airline industry. Also at this time, the Sikorsky Aviation Corporation became a subsidiary of the United Aircraft and Transport Corporation. In 1930, another 11 acres of land was acquired for the Stratford facility (Woodward Clyde Consultants [W-C], 1991).

In 1935, after governmental separation of aircraft manufacturing companies from transportation services, the portion of United Aircraft and Transport Corporation holding Sikorsky (and Chance-Vought Corporation in East Hartford), became United Aircraft Corporation. Sikorsky continued to manufacture seaplanes and in 1938, United Aircraft Corporation – Sikorsky Division began developing the helicopter at the Stratford Plant (Sikorsky, 2003).

In 1939, United Aircraft Corporation moved its' Chance-Vought Division to the Stratford Plant and merged it with its' Sikorsky Division to form the Vought-Sikorsky Division. This division manufactured seaplanes and developed helicopters and combat aircraft in the Stratford facility. In September 1939, the first helicopter prototype flight was completed at the facility. Also during this time, significant renovation and expansion of the plant occurred, in preparation for war, including installation of a conveyor line system and regular spot welding.

In 1942, separate divisions of United Aircraft Corporation were re-established for Sikorsky and Vought so that the Sikorsky Division could focus on production of helicopters, while the Chance-Vought Division could focus on production of combat aircraft. During this period, the Chance-Vought Division was mass-producing combat aircraft at the Stratford Plant, including the Kingfisher and the Corsair, for use in World War II (W-C, 1991). Chance-Vought engineers were conceiving new planes, including a torpedo bomber and Chance-Vought's first jet aircraft (Vought, 2003). The Sikorsky Division continued production of the helicopter until it left the Stratford Plant in 1943 because of overcrowding (W-C, 1991).

Following the end of WWII, the Chance-Vought Division continued to work on the development and production of jet engines. In April 1948, the Chance-Vought Division began its' move to Texas, as part of the Navy's plan to distribute military manufacturers across the country and out of vulnerable coastal areas. The move was completed in June 1949 (Vought, 2003).

The Stratford plant remained inactive until 1951. During this period of inactivity, the plant suffered severe damage from storms and floods. In February 1951, the U.S. Air Force reactivated the facility as Air Force Engine Plant #43, under the management of the AVCO Lycoming Division of AVCO Corporation (Environmental Science & Engineering [ESE], 1981).

Site Development

Site development included an initial phase between 1929 and 1930 and a World War II expansion phase from approximately 1939 to 1945. No development occurred between 1945 and 1951. Figure 1-3 shows the site layout during the period.

Initial site development occurred between 1929 to 1930 and included the construction of an Administration Building (1929) along South Main Street (currently eastern Building B-1), a Factory Building (1929) (currently eastern Building B-2), a Service Building (1929) (currently eastern Building B-10), an Engineering Building (1930) (currently southern Building B-3), and several storage buildings between the Factory Building and the Service Building (AFM FIC, 1931 and 1937). The Causeway was constructed for access to the river channel and launching seaplanes. From 1930 until 1938, the layout of the facility remained fairly consistent.

In the period from 1939 to 1943, existing buildings were expanded and new buildings were constructed as the facility manufactured combat aircraft for World War II and helicopters. When United Aircraft Corporation re-located the Chance-Vought Division to the site in 1939, site facilities were expanded considerably. The Factory Building (currently Building B-2) was quadrupled in size with an expansion to the west (1939 and 1941), the addition of employee facilities along South Main Street (1940 and 1941), and an addition to the east (circa 1942). Three buildings were constructed on the north side of the Factory Building, including a Painting and Sand Blast Building (circa 1941), the Hammer Shop (1939), and the Deflector Shop (1939) (currently the former B-2 Chromium Plating Facility). The current fire suppression tank and its' associated pump house (currently Building B-42), a new boiler room (1939), and two transformer pads were also added along the north side of the Factory Building during this time. Also, the Administration Building (currently Building B-1) was expanded (1941).

A Commercial Hangar (1940) (currently northern Building B-3) was constructed north of the existing Engineering Building (currently southern Building B-3), and two drafting buildings were added, one to the south of the Engineering Building (1942) (currently southern Building B-3) and one to the north of the Commercial Hangar (1942) (currently Building B-3A). The Storage Building (1940) (Building B-11; currently western Building B-10) was constructed west of the Service Building (currently eastern Building B-10). The Tool, Die, Jig, and Maintenance Building (1941) (currently Building B-12) was constructed west of the storage building and a Test House (circa 1941) was located west of the Tool, Die, Jig, and Maintenance Building. A group of buildings was constructed behind the Service Building and the adjacent Storage Building in 1943, including a Fire Truck and Truck Repair Building (currently Building B-9), a barrel rack, a storage building for paint, alcohol, and thinners (currently Building B-7), an auto storage building (currently Building B-8), and an incinerator.

A second fire suppression tank and associated pump house (currently Building B-43) were added near the shoreline and the eastern facility property boundary (between 1941 and 1943). Several of the existing storage buildings were still present and several new storage sheds were constructed (AFM FIC, 1941 and 1943).

In order to facilitate the continued expansion, the shoreline was extended into the Tidal Flats both north and south of the Causeway in the 1940s. Based on aerial photographs and site plans, approximately 10 acres of land was added in 1944 by hydraulic dredging of the Housatonic River (W-C, 1991).

An expansion of the Factory Building (currently Building B-2) was completed to the north and west and modifications to both the Factory Building and the Administration Building (currently Building B-1) were completed to enclose various abutting buildings under one roof. The result was the current footprint for Buildings B-1 and B-2. The Engineering Building underwent a similar transformation, to include its' various expansions and the southern Drafting Building under one roof, creating the current footprint of Building B-3. Other major structure additions completed in 1944 and 1945 included the construction of Buildings B-3T (cafeteria), B-4 (parts storage), B-5 (testing facility), Building B-6 (engine testing, parts storage, and painting), B-13 (scrap metal storage), B-14 (an incinerator), B-15 (lubricating oils and fire department), and several buildings in the area within the current footprint of Building B-19 (contractor-owned engine research).

The 1949 aerial photograph shows several buildings directly north of the Tool, Die, Jig, and Maintenance Shop (currently Building B-12) and Building B-7 but the use and building numbers are not known. These buildings are not present on subsequent aerial photographs or facility maps (USEPA, 1990a).

Facility Operations

Manufacturing of seaplanes during the 1930s involved metal working, riveting, welding, doping, painting, lead oxide dipping, heat-treating, woodworking and aircraft testing (AFM FIC, 1937). These operations were conducted in the Factory Building (Building B-2) and Service Building (Building B-10) (see Figure 1-3). Two fuel-oil USTs were located adjacent to the original boiler room and four gasoline USTs were located adjacent to the Service Building. Paints, thinners and dope were stored in sheds between the Factory Building (Building B-2) and the Service Building (Building B-10). Any wastes generated during this period would have included oils, paints, and thinners.

Mass-production of aircraft during the 1940s involved similar operations on a larger scale. The expanded Factory Building was used for assembling aircraft bodies and wings. There were two areas within this building where painting occurred. The first was in the southwest corner (prior to the final 1944 expansion) and the second was in the northeastern corner. Heat treating and anodizing (plating) were conduced adjacent to the southernmost paint area. Maintenance was conducted in the buildings that are currently Buildings B-9, B-10, and portions of B-12 (AFM FIC, 1931 to 1943).

After conversion of the fuel-oil-fired boiler system to coal in the early 1940s, several of the larger fuel oil tanks were no longer used. Underground concrete septic tanks are identified on the 1943 fire insurance map (AFM FIC, 1943) and referenced in a 1981 facility report (ESE, 1981). Table 1-1 contains information on the approximate location, size, and contents of tanks during this time period. Paints and solvents were stored in the building that is currently Building B-7 (AFM FIC, 1931 through 1943). A barrel rack was located adjacent to this building (AFM FIC, 1943 and USEPA, 1990); however, the contents of the barrels within this rack are unknown.

Aerial photographs indicate the shoreline area where hydraulic fill was placed was used for open storage of crates, drums, and other containers. The largest open storage area during this time was located near the current entrance to the Causeway in a 1943 aerial photograph (USEPA, 1990). This photograph also shows four outfalls from the plant discharging to the intertidal flats area, and several 1944 photographs show the pipelines being extended through the hydraulic fill area (W-C, 1991). Active discharge from several of these outfalls into the river can be identified on the aerial photographs (USEPA, 1990).

The Causeway was constructed between 1929 and 1931. Possible debris, mounded material, and/or rubble are identified in aerial photographs between 1960 and 1980 (USEPA, 1990). Reportedly, the Causeway was used as a training area for the plant's fire department; fires were started and extinguished on the Causeway (W-C, 1991).

From approximately 1941 to 1949, paint and solvents from the paint shop, located in the northeast corner of the Factory Building (Building B-2), reportedly were piped directly to a septic tank located between this building and the Engineering Building (Building B-3). A shop drawing dating from approximately 1950 describes the septic tank as receiving overflow from the paint spray shop through a 12-inch tile pipe. Most of the paint reportedly used during this period was zinc chromate primer (ESE, 1981). The tank was located southeast of Building B-70, as depicted on Figure 1-3.

1.5.2 1951-1997: Engine Production and Testing

In February 1951, the U.S. Air Force purchased the property and reactivated the Stratford Plant as Air Force Engine Plant #43, under the management of the AVCO Lycoming Division of AVCO Corporation (AVCO Lycoming). AVCO Lycoming was under contract to produce the Curtiss-Wright nine cylinder R-1820 radial engine and major components of the J-47 jet aircraft engine. Before production could begin, massive repair and renovation of the plant and new machinery were necessary. Water-damaged buildings were repaired, a dike along the Housatonic River was constructed for flood protection, and new machinery was installed. Manufacturing began in June 1951, and in March 1952, the first R-1820 engine was completed (ESE, 1981).

In late-1952, AVCO Lycoming was awarded a contract to develop the first American-produced gas turbine engine specifically designed for helicopters. The result was AVCO Lycoming's T53 and T55 series engines used extensively on the "Huey" helicopters in the late 1950s and 1960s, and produced at the Stratford facility. Also in 1952, AVCO Corporation relocated its' Turbine

Operation Division from Williamsport, Pennsylvania to Stratford. Throughout the 1950s, the development and production of helicopter engines continued (ESE, 1981).

During the 1960s, in addition to production of helicopter engines, AVCO Lycoming was contracted by the U.S. Air Force for the development of re-entry vehicles for Intercontinental Ballistic Missiles and the space program. This contract was a result of AVCO Lycoming's successful firing and recovery of the first re-entry vehicle in 1959. Additional government contracts during this time involved the development of engine test systems for programs such as the Surveyor Lunar Survey Vehicle, Apollo Re-entry Vehicles, and the U.S. Department of the Navy's (U.S. Navy) Deep Sea Rescue Vehicle. Work also continued on the application of gas turbine engines to fixed-wing aircraft (ESE, 1981).

During the period from 1970 to 1975, AVCO Lycoming augmented its government-based productions by engaging in international and commercial activities. The company expanded into development, production, repair, and maintenance of turbine engines for military, commercial, and industrial use. In June 1976, the Stratford facility was transferred from the U.S. Air Force to the U.S. Army and was given its current designation, Stratford Army Engine Plant. Throughout this transition, AVCO Lycoming continued as the operating contractor at SAEP. In 1977, upon selection of AVCO Lycoming's AGT-1500 engine to power the U.S. Army's new Abrams M-1 Main Battle tank, a decision was made to complete a major equipment renovation at SAEP. Following this renovation, AVCO Lycoming continued to produce helicopter and tank engines for the U.S. Army, as well as engines for commercial and international use (ESE, 1981).

AVCO Lycoming merged with Textron in December 1985, and subsequently formed the Textron Lycoming Stratford Division. Operations continued to be the production and testing of turbine engines for both military and commercial aircraft and land vehicles (W-C, 1991).

In 1994, the gas turbine aircraft engine group of the Stratford Division of Textron Lycoming was sold to AlliedSignal and the U.S. Army contract for operation also was transferred to AlliedSignal. AlliedSignal continued to develop, manufacture, and test turbine engines for the U.S. Army and the U.S. Navy, for foreign military sale, and commercial use (ABB-ES, 1996). Manufacturing was terminated in late 1997 and the plant was vacated in August 1998.

Site Development

In 1951, repair and renovation was necessary before manufacturing could begin. Water-damaged buildings were repaired, the dike along the Housatonic River was constructed for flood protection, and new machinery was installed, including the equipment necessary for electroplating. In 1952, Building B-17 was constructed south of Building B-6 for steam generation. In 1953, construction was completed on an engine testing facility (Building B-16), the associated cooling tower and cooling tower pumping station (Building B-33), and the former Jet Fuel Tank Farm and associated pumphouse (Building B-34). A second product storage facility, the former Oil House Tank Farm and its' associated pumphouse (Building B-35) were built and six stormwater pump stations (Buildings B-36 through B-41) were installed equipped with flood-control pumps (W-C, 1991 and USEPA, 1990).

Although no documentation on construction dates were located, aerial photographs and site maps indicate a loading dock was added to the southeast corner of Building B-2 between 1951 and 1953, and a testing facility was added to the northeast portion of Building B-6 between 1956 and 1960 (USEPA, 1990 and FIA, 1956).

In 1958, the CWTP (Building B-18), lagoons, and the pumphouse at Building B-63 were constructed to treat electroplating and related waste streams prior to discharge to the Housatonic River. Also in 1958, an additional building (Building B-51) was added in the area near Building B-19 (W-C, 1991). Building B-51 and several other buildings added later to the Building B-19 area ultimately would be consolidated in 1988 to form the existing footprint of Building B-19.

During the period from 1960 to 1970, construction included facilities for materials and supplies storage, production, and component testing. In 1961, six large Quonset Huts (Buildings B-44 through B-49) were constructed north of Building B-12; however, one of these buildings (B-49) was reportedly taken down in 1968 and its' exact location is unknown. The Quonset Huts were used for storage of tools and equipment. Also in 1961, a building for storage of surplus equipment (Building B-53) was erected east of Building B-6. Two Production Material Warehousing buildings (Buildings B-54 and B-55) were constructed immediately north of Building B-2 in 1963, and a Missile Storage Magazine (Building B-59) was built on the Causeway in 1968 (W-C, 1991). A chemical storage area was constructed adjacent to the northwest portion of Building B-15 sometime during this decade (USEPA, 1990).

Production facilities constructed during this decade included the Plasma Spray Facility (Building B-52) erected near the northwest corner of Building B-2 in 1962, and the Missile Assembly building (Building B-58) built west of the former Oil House Tank Farm in 1967 (W-C, 1991).

Buildings associated with component and engine testing constructed during this time included the Engine Fuel System Test Facility (Building B-7A) in 1962, three buildings (Buildings B-19F, B-56, and B-57) added to the area near Building B-19 from 1963 to 1965, and Building B-6A, the Engine Mechanical Component Test Facility, constructed immediately east of Building B-17 in 1966.

A High-pressure Natural Gas Pumping Station (Building B-60) and a Refrigeration Plant (Building B-61) were constructed north of Building B-6 at the end of the decade (W-C, 1991).

In 1971, an additional building (Building B-62) was added to what would later become the Building B-19 complex and in 1975; the Oil Abatement Treatment Plant (OATP) (Building B-64-2) and associated pumphouse (Building B-64-1) were constructed to address new National Pollutant Discharge Elimination System (NPDES) requirements (W-C, 1991). Two office facilities (Buildings B-77 and B-79) were built at the end of the decade, or early in the next decade (USEPA, 1990).

Several buildings were demolished during this time period, including a storage building (Building B-64) (location of this building is unknown) in 1970, a storm system pumphouse (Building B-39) directly north of Building B-16 in 1971, and the cafeteria south of Building B-3 (Building B-3T) in 1976 (W-C, 1991). In 1980, the former Oil House Tank Farm pumphouse (Building B-35) and three Quonset Huts (Building B-45 through B-47) were demolished and modifications to Building B-3A were completed (W-C, 1991). The incinerator (Building B-14) located north of Building B-13 was demolished sometime during this decade, although the foundation remained in place for use during construction of a storage area (Building B-81) also during this time (USEPA, 1990).

Modifications to Buildings B-15 and B-52 were completed in 1981. Building B-67 (General Stores) and Building B-69 (Resident Engineer) were constructed in 1985 and also in this year, Buildings B-10 and B-11 were connected to create one building (Building B-10). Modifications to the chemical waste treatment system were initiated in 1986 as a result of a Consent Decree that required the evaluation and updating of existing process and treatment operations (CDM Federal Programs Corporation [CDM FPC], 1992). These modifications included the construction of the Cyanide Destruction Facility (CDF) (Building B-70) adjacent to and east of the former B-2 Chromium Plating Facility, construction of equalization tanks northwest of Building B-18 (e.g., the CWTP), and construction of a Solids Handling Building (Building B-71), also northwest of the CWTP. An emergency generator (Building B-68) for the OATP system was installed near Building B-81 and B-13 in approximately 1986 (FIA, 1986 and W-C, 1991).

In 1987, concrete pads, berms, and roofs were added to four existing waste storage areas (Buildings B-73 through B-76) located near the former Oil House Tank Farm, and near the CWTP (Building B-18). In 1988, the numerous buildings previously constructed in the area of Building B-19 were joined to create the current footprint of Building B-19, and in 1989 a fuel pumping station (Building B-72) was constructed east of Building B-6. A Guard Station (Building B-78) and scale were added to the westernmost facility entrance off Main Street. In 1990, Buildings B-54 and B-55 near the northwest corner of Building B-2 were demolished for the construction of Building B-65 in 1991. Building B-65 was constructed near the northernmost corner of Building B-2 in 1991 for parts storage (W-C, 1991).

Several smaller structures were constructed early in this decade (circa 1991), including a storage shed (Building B-80) northwest of Building B-2, and a guard house (Building B-82) at the facility entrance off Sniffens Lane (AlliedSignal, 1995). The approximate layout of the facility from 1951 to 1997 is shown on Figure 1-4.

Facility Operations

Manufacturing. From 1951 until 1997 the site was used for the development, production, and testing of airplane, helicopter, and tank engines. Manufacturing and support activities included the following:

- machining;
- electrochemical machining (ECM);
- electroplating;
- corrosion resistance;
- cleaning;
- engine testing;
- miscellaneous processes; and
- maintenance activities.

Machining. Machining processes included broaching, drilling, grinding, lathing, stamping, and forging, which involve high-pressure, metal-on-metal contact between tools and work pieces. Most of the machining processes were associated with production work done in the central portion of Building B-2. Machining was also done for production of recuperators in Building B-10 (formerly Buildings B-10 and B-11) and for experimental development and testing in Buildings B-3 and B-6 (FIA, 1956 and W-C, 1991).

During the machining process, cutting oils and coolants were used to reduce surface friction, cool the tools and work pieces, and remove metal chips from the work surface. Lubricating oils, greases, and hydraulic fluids were used to lubricate internal machine parts and for transfer of hydraulic energy, respectively, and a fluorescent metal penetrant was used to aid in the inspection of finished pieces for hairline fractures and other defects.

Most of the wastes generated by machining processes were associated with coolants, such as water-soluble oils (W-C, 1991). Non-contact fluids used for lubrication and hydraulics were not major contributors to the waste stream; they generally were contained in reservoirs at each tooling machine and therefore, were not as prone to contamination by metal chips and dust.

Prior to the early- to mid-1980s, waste coolants, oils, and penetrants were disposed of into the stormwater treatment system, likely via floor drains in machining areas of Building B-2. These lines drained waste materials directly to pumphouse B-37 and the OATP for treatment and disposal (CDM FPC, 1992). After this time, waste coolants, oils, and penetrants were removed by vacuum cart and disposed of as hazardous waste oil, or as halogenated solvent waste if the material was mixed with a solvent, which was historically used for cleaning in the machining process. These wastes were dumped into concrete pits west of Building B-13, and transferred via subsurface piping (CDM FPC, 1992) to the former Oil House Tank Farm for storage (W-C, 1991).

In 1989, a penetrant wastewater system was installed, consisting of two disposable carbon filters. Cleaned water from this system was discharged to the OATP, and the spent carbon filters were recycled off-site (W-C, 1991). In May 1990, a coolant recycling unit was installed that virtually eliminated waste coolant oil from the waste stream. Waste lubricating oils, greases and hydraulic fluids were recycled as waste oil (INEL, 1991).

Electrochemical Machining. ECM is a process whereby a metal work piece is placed into a salt brine bath and then cut using an arc welder. In this process the metal is removed by dissolution into the ECM brine, a process that results in more rapid cutting of parts. The ECM process was conducted within a brine tank in the northernmost portion of Building B-4. The process generated sludges that were disposed of off-site and wastewater that was discharged to the OATP (ESE, 1981). The ECM process was decommissioned in 1987 (ABB-ES, 1996).

Electroplating. Electroplating is a process whereby metallic coatings are ionically deposited on other metal surfaces, typically by the induction of an electrical current to a solution. Electroplating at SAEP historically included chromium, nickel, copper, and cadmium plating. Copper and cadmium plating were completed using cyanide-bearing plating solutions (ESE, 1981).

Significant electroplating operations began in 1951 primarily in the northeast corner of Building B-2 in an area known as the former B-2 Chromium Plating Facility and in the southernmost portion of Building B-3 (ESE, 1981). This area of Building B-3 more recently housed the mainframe computer for SAEP. A new floor was constructed above the old, and numerous cables were noted beneath new floor grates during the 1991 site reconnaissance (W-C, 1991). A fire insurance map from 1956 indicates plating was also conducted in the northernmost portions of Building B-6 and in southern Building B-3A (FIA, 1956).

Because the success of a plating operation depends largely on the degree of cleanliness of the receiving metal prior to the application of the coating, the plating process was typically associated with stripping and cleaning processes. The stripping process was completed with acids such as chromic acid, sulfuric acid, and hydrochloric acid. Cleaning was conducted with agents such as chlorinated solvents (e.g., 1,1,1-trichloroethane [1,1,1-TCA] and Freon-based solvents), non-chlorinated solvents (e.g., Varsol, methyl ethyl ketone [MEK], and thinners), and alkaline cleaners (e.g., permanganate salts and sodium hydroxides). In the 1980s and 1990s, 1,1,1-TCA was widely used for cleaning; the solvent was contained in vapor degreasing units in the plating facility (ESE, 1981 and INEL, 1991). It is unknown what stripping and cleaning agents were used at the site prior to the 1980s.

Prior to construction of the chemical waste treatment system in 1958, wastewater from electroplating operations was neutralized within processing tanks and discharged to the storm water system, which ultimately drained, without further treatment, to the Housatonic River. After 1958, chemical tanks (e.g., acids) as well as water rinsing tanks were routed to the CWTP or the CDF for treatment, via the chemical waste collection system.

After construction of the waste transfer system in the early- to mid-1980s, chlorinated solvent wastes from the vapor degreasers were removed by vacuum cart and pumped into waste pits northwest of Building B-13 or into waste solvent tanks at the former Oil House Tank Farm (INEL, 1991). It is unknown how solvent wastes were handled prior to this time.

Corrosion Resistance. The corrosion of metals is a major cause of failure for metallic parts, due to a loss of mechanical strength or ductility. Various surface treatment technologies were used at SAEP to impart corrosion resistance to metallic surfaces. These processes were mostly completed in the northeastern portion of Building B-2, near the former B-2 Chromium Plating Facility, and included:

- anodizing;
- chemical conversion coatings;
- HAE anodic finish for magnesium;
- black coating (black oxide and phosphate);
- passivation for steel;
- painting;
- oil slushing; and
- plasma spray.

The anodizing process involved the electrical placement of a non-metallic coating on aluminum and magnesium. In addition to the anodizing conducted in Building B-2, a separate anodizing process was located near the electroplating operations in Building B-3. In 1991, the process involved degreasing using 1,1,1-TCA, part cleaning using an alkaline or acid cleaner, and immersion of the part in chromic acid solution. This produced an oxide coating on the metal that was corrosion resistant or provided a foundation for other finishes, such as paint (INEL, 1991).

Chemical conversion coating involved the placement of non-metallic coatings on metal parts using oxidation-reduction reactions. Cleaning was an important initial step to this process, and 1,1,1-TCA, alkaline cleaners, MEK, acetone, and toluene were used for this step (INEL, 1991).

Magnesium was widely used for military applications, however because it is a very active metal, its use depended upon effective protective measures. SAEP used the HAE anodic finish on its magnesium products. As part of this process, the chrome pickle treatment and the dichromate treatment were the two surface treatments used for corrosion protection of magnesium. Both processes involved degreasing of the part using 1,1,1-TCA, alkaline cleaning, and a chromic acid dip. The dichromate treatment also involved the use of hydrofluoric acid and sodium dichromate. The HAE anodic finish involved degreasing with 1,1,1-TCA, alkaline cleaning, and either a single or double treatment of HAE coating (INEL, 1991).

Black coatings were used to impart a black finish, decrease light reflection, provide a base for panting, and improve corrosion resistance. At SAEP, a black oxide coating was created on the surface of ferrous metal parts by immersing the part in a concentrated alkaline solution of sodium hydroxide, sodium nitrate, and water. Magnesium-based and zinc-based phosphates were also used to produce black coatings. All processes involved degreasing (1,1,1-TCA, toluene, MEK, or acetone), cleaning of the part (alkaline cleaner or thinner), part immersion, and rinsing (INEL, 1991).

Passivation treatments were intended to improve the corrosion resistance of parts made from steel. The passivation procedure involved degreasing using 1,1,1-TCA, alkaline cleaning, passivation using nitric acid or a nitric/dichromate solution, chromate treatment, and a chromic acid rinse (INEL, 1991).

Painting processes included the coating of epoxies, enamels, silicones, and zinc chromate primers. Painting occurred near the former B-2 Chromium Plating Facility in Building B-2 and in the northern portion of Building B-6 (FIA, 1956). In 1981, a wet-spray method was used to place non-leaded epoxy paint on parts. Wastewaters from this process were sent to the OATP (ESE, 1981). However, by 1991, painting was conducted in Building B-2 using dry-spray methods; there were three dry-paint spray booths measuring approximately 4-feet by 4-feet by 10-feet, and equipped with air filters. Used filters, paint sludge, and waste thinners were drummed, and entered the waste collection stream as hazardous (ignitable) solid waste, or as non-halogenated solvents (W-C, 1991).

For temporary corrosion protection, SAEP used blended slushing oil. The slushing oil provided a thin coating on the metal that could be easily removed for further processing at a later time. Spent slushing oil was formerly collected by vacuum cart, discharged into a concrete pit west of Building B-13, and pumped to a waste oil tank in the former Oil House Tank Farm.

Plasma spray involved the application of a sprayed metal coating onto a metallic substrate to protect against atmospheric corrosion and improve appearance. The plasma-arc spray process was located in Building B-52 at SAEP and used a gun to melt and propel the coating material at high velocity to form a protective coating. A sodium hydroxide-based product was used for stripping the plasma coating at the site (INEL, 1991).

Prior to construction of the chemical waste treatment system in 1958, the vast majority of acid and chemical wastes generated during corrosion resistance activities was neutralized within processing tanks and discharged to the storm water system, which ultimately drained, without further treatment, to the Housatonic River. After 1958, chemical tanks (e.g., acids) as well as water rinsing tanks were routed to the CWTP or the CDF for treatment, via the chemical waste collection system.

After construction of the waste transfer system in the early- to mid-1980s, chlorinated solvent wastes and waste oils were removed by vacuum cart and pumped into waste pits northwest of Building B-13 or into waste solvent tanks at the former Oil House Tank Farm (INEL, 1991). It is unknown how solvent wastes were handled prior to this time.

Cleaning. The degree of success of metal finishing processes depends largely on the cleanliness of the receiving part. Cleaning was a necessary part of electroplating, corrosion prevention, and heat treatment processes and was widely conducted in Buildings B-2 and B-3 (INEL, 1991). Additional activities included the cleaning of engine parts during engine testing in Building B-16, cleaning of parts during maintenance activities, and general facility cleaning. Cleaners used throughout SAEP have included:

- chlorinated solvents;
- non-chlorinated solvents;
- emulsion cleaners;
- alkaline cleaners;
- acids;
- abrasives; and
- miscellaneous cleaners.

Historically, chlorinated solvents were used for degreasing at numerous locations throughout SAEP; at one time, 38 vapor degreasing units were used in Building B-2 and B-3. Reportedly, 1,1,1-TCA was the most predominantly utilized solvent; however, the use of trichlorotrifluoroethane, a Freon-based solvent, was documented in one vapor-degreasing unit in 1991 (INEL, 1991). 1,1,1-TCA reportedly also was used for engine cleaning and component testing in Buildings B-16 and B-19 and parts cleaning during maintenance activities in Building B-9. Freon-based solvents were widely used for cleaning during printed circuit board assembly in Building B-6 and were contained in spray cans that were used throughout the facility for wipe and touch-up cleaning operations (INEL, 1991). It is suspected, although not documented, that other chlorinated solvents historically may have been used at the site, including tetrachloroethene (PCE) and trichloroethene (TCE). According to personnel interviews, carbon tetrachloride, TCE, and Varsol solvents were used for cleaning test engines in addition to 1,1,1-TCA (W-C, 1991). Concentrations of PCE and TCE were identified above discharge limits at Outfall (OF)-008 (INEL, 1991). The presence of high concentrations of these chlorinated solvents in groundwater beneath the facility further supports the likely use of these two solvents.

Spent chlorinated solvents were removed from degreasing tanks using a vacuum cart and then pumped into a waste solvent tank at the former Oil House Tank Farm via underground pits and transfer lines near Building B-13. Although this disposal process was established, chlorine contamination was still a major problem for the facility; machining oils and coolants were often contaminated with chlorine and discharge limits for chlorinated solvents were often exceeded at OF-007 and OF-008 (INEL, 1991). It is unknown how solvent wastes were handled prior to construction of the waste transfer system.

Non-halogenated organic solvents such as alcohols, MEK, acetone, paint thinners (e.g., 1,1-dioxane and toluene), and mineral spirits (e.g., Varsol) were used at SAEP, primarily for wipe cleaning. The use of Varsol (a mineral spirit and cold cleaner) was documented at SAEP prior to 1985; however, after 1985 cold cleaners reportedly were replaced with a Penetone product. Facility records from 1989 indicate that both Varsol and the Penetone product were purchased and used at the facility. Waste non-chlorinated solvent was dumped from tanks and treated at the CWTP (INEL, 1991).

In 1990, an emulsion cleaner replaced the use of 1,1,1-TCA for engine cleaning after testing at Building B-16. Emulsion cleaning utilizes an organic solvent, detergent, and water mixture. Spent emulsion wastewater was collected by vacuum cart and stored in a tank at the former Oil House Tank Farm. Water was pumped out of the tank and treated at the OATP. Waste oil was sent off site for recovery (INEL, 1991).

Alkaline cleaners, composed of alkaline salts such as potassium or sodium hydroxides and sequestering agents, dispersants, and wetting agents, were used to remove greasy materials or as a secondary cleaner following emulsion cleaning. Three alkaline cleaners were used in the former B-2 Chromium Plating Facility, including a permanganate salt, a de-rust cleaner, and a soak cleaner. Alkaline cleaners were also used in the anodizing room in Building B-2 as an aluminum cleaner and for paint stripping. Waste cleaner was discharged to floor drains in the former B-2 Chromium Plating Facility and the anodizing room, and treated at the CWTP (INEL, 1991; W-C, 1991).

Alkaline cleaning was usually followed by acid cleaning to neutralize residual alkaline films and remove the last traces of oxides. Acid cleaning prepared the metal surface for pickling, a process that used acid to chemically remove surface impurities from the metal. Five acids were most widely used during these processes, including hydrochloric, sulfuric, nitric, chromic, and hydrofluoric acids. Waste acids were dumped into floor drains and treated at the CWTP (INEL, 1991).

Abrasive blasting and abrasive slurry cleaning using sand, glass beads, and grit was also performed at Building B-2. These types of cleaning were used to remove rust or scale buildup and to smooth edges or other imperfections following machining. Waste from these processes was typically collected by container and entered the waste collection stream as non-hazardous solid waste. In cases where contamination by solvents or oils was possible, the waste material was tested before being disposed off-site (INEL, 1991).

Miscellaneous other cleaning processes were completed at SAEP, including vapor blast cleaning, anodic cleaning, periodic reverse cleaning, etching, and general facility maintenance. These processes utilized cleaners that were used during other cleaning processes, such as alkaline cleaners and acids, or cleaners that were not considered hazardous (INEL, 1991).

<u>Engine Testing.</u> Engines were tested in test cells in Building B-16, B-6, and B-19 beginning in the early 1950s. Engine tests ran from several minutes to thousands of hours, but a typical production test lasted between three and six hours. In addition to engine testing, some assembly and disassembly of engine parts was performed at these locations. Various engine components were tested at other locations throughout the facility, including engine fuel-system testing in Buildings B-7 and B-7A, mechanical component testing in Building B-6A, electrical systems testing in Building B-79, and flow-control testing in Building B-58.

According to personnel interviews, solvents were used for cleaning test engines, including carbon tetrachloride, TCE, Varsol and 1,1,1-TCA (W-C, 1991). Waste solvent and rinse water from these operations reportedly were dumped into floor drains of the test cells in Building B-16, which were piped to an oil/water separator west of the building and initially discharged to the Housatonic River. After 1976, these wastes were piped directly to the OATP (W-C, 1991). The drainpipes corroded over time, with nine drains reportedly closed between 1978 and 1982 (CDM FPC, 1992). The 1991 Preliminary Assessment Screening (PAS) document indicates, “The drains at B-16 are now plugged with a sludgelike mixture that potentially contains halogenated solvent, oil and grease, and fuel.” The remaining drains were reportedly plugged between 1989 and 1991 (INEL, 1991).

Emulsion cleaning replaced solvent cleaning of test engines in 1990. Emulsion cleaning wastes were collected in vacuum carts and placed into underground pits near Building B-13 for transfer to waste oil tanks located at the former Oil House Tank Farm (W-C, 1991). Following closure of the concrete pits in 1993 (ABB-ES, 1996), wastes were transported with a vacuum cart directly to the waste oil tanks. Waste oil/fuel and wastewater were allowed to separate in the waste oil tanks, and wastewater was drained to the OATP for treatment. Waste oil/fuel was disposed off-site (ABB-ES, 1996).

Mercury manometers were formerly used at the engine testing facilities in Buildings B-6 and B-16 to measure engine system pressure. Occasionally, a high pressure surge would blow the mercury out of the manometer tube, and onto the ceilings, walls, and floors, and into the floor drains. Mercury was reportedly handled carefully because of its cost; however, little mercury was recovered from the described accidental spills (ABB-ES, 1996).

Miscellaneous Processes. Miscellaneous processes conducted at SAEP included coating of metal parts with solid film lubricants, masking, peening, heat-treating, printed circuit board manufacturing, and photographic processing.

Solid film lubricants were used to lubricate bearing surfaces under operating conditions or in environments where conventional lubricants were not satisfactory, such as high temperatures, in vacuums, in dusty or dirty conditions, and when conventional lubricants could contaminate other parts. Solid film lubricants were much like paint in content and application, containing chemicals such as toluene, xylene, alcohols, and ketones. These greases, lubricants and epoxies were used in relatively small quantities throughout manufacturing and support operations at SAEP (INEL, 1991).

When plating was to be completed only on a specific area of an item, the other areas required masking. Masking usually consisted of immersing the entire part in the masking material, then selectively removing the maskant for the areas requiring plating. Typical maskants used in the past at SAEP included wax, lacquer, tape, rubber, and plastic; however, wax was the most widely used. No information is available regarding waste disposal of maskants (INEL, 1991).

Shot peening, performed adjacent to the Building B-2 boiler room, was a process whereby a metallic surface was subjected to impact, to smooth sharp points or edges. Ceramic and glass beads were typically used at SAEP for shot peening, and wastes generated by this process were containerized and enter the waste collection stream as non-hazardous solid waste (INEL, 1991).

Heat treatment of metal parts improves their metallurgical properties. Heat treatment at SAEP historically was conducted in Building B-2, initially in the western corner of the building (AFM FIC, 1943), and later next to the boiler room. Prior to heating, the metal parts were cleaned in various alkaline or acid baths. After cleaning, the metal part was subjected to heat in one of several furnaces and then quenched in oil or water (ESE, 1981).

Printed circuit boards were manufactured in Buildings B-2 (ESE, 1981) and B-6 (INEL, 1991). The manufacturing process involved etching the copper boards with ferric chloride solution. Waste ferric chloride was collected in a 55-gallon drum and sent off-site for disposal (ESE, 1981).

A photographic laboratory was located in the western portion of Building B-2 (FIA, 1956) for processing black and white and color photographs. Developer, fixer solutions, and washwater from this laboratory were typically dumped yearly to the stormwater treatment system (ESE, 1981).

Maintenance. General automotive maintenance and repair was performed in Building B-9. Waste motor oils and other waste fluids (brake fluid, transmission fluid, etc.) were collected in sumps, which were pumped daily and transferred to the waste oil tanks in the former Oil House Tank Farm for later recycling. Vehicle motors were overhauled in Building B-12. According to personnel interviews, the maintenance department at SAEP was responsible for blending oils in Building B-15, dispensing them to workstations, and collecting waste oils (W-C, 1991).

Materials and Waste Storage. Underground and aboveground tanks, chemical storage areas, container storage areas, satellite accumulation areas, and open storage areas were used on-site. Many of the areas have been identified as Areas of Concern (AOCs) or other potential release areas in Section 7.0, which discusses the potential for release from these areas.

Plant operations required the use of materials that eventually became waste, either by degradation of their performance characteristics or by mixing with other materials. A summary of the manufacturing operations after 1951, including the chemicals used, a description of waste generated and the waste's fate, is provided in Table 1-2.

A summary of the use and storage of materials used in support of facility operations, including pesticides and herbicides, explosives, and radiological materials is also included in the following paragraphs.

Storage Tanks. Storage tanks were used to store fuels, oils, solvents, water, and waste materials at the site. A summary of the status of current and former aboveground and underground storage tanks is provided in Table 1-3.

Most recently, underground storage tanks were used at the facility for storage of small volumes of fuel for engine testing (near Building B-19) and vehicle maintenance (near Building B-9), and as waste storage pits (near Building B-13). Historically, underground tank use was more widespread at the facility. Many of the tanks were removed between 1987 and 1993.

Aboveground storage tank use at the site generally was associated with storage of larger quantities of fuel for engine testing, with the facility boilers, and with fire suppression. Numerous tanks were contained at the former Jet Fuel Tank Farm and the former Oil House Tank Farm; large tanks were also present adjacent to testings facilities at Buildings B-6, B-16, and B-19, and within the main boiler storage area, adjacent to Buildings B-2, B-10, and B-44. Fire suppression tanks were located adjacent to Buildings B-2 and B-19. Based on a review of historic photographs and site plans, the majority of aboveground storage tanks at SAEP were installed on concrete cradles or pads, and tanks were usually surrounded by a containment berm (W-C, 1991). It is not known if the bottom surfaces of the containment areas were sealed and maintained to prevent leakage.

Chemical Storage Areas. The primary chemical storeroom identified in the 1991 PAS and the Environmental Baseline Survey (EBS) (ABB-ES, 1996), was located adjacent to Building B-15. This storage area had concrete floors and walls, and no floor drains. Oils, solvents, and cleaners were stored inside Building B-15, segregated by chemical type (ABB-ES, 1996). Raw chemicals formerly were stored in Building B-13, although the specific types and quantities of chemicals are unknown (ABB-ES, 1996).

Sulfuric acid, sodium hydroxide, and sodium hypochlorite were formerly stored outside Building B-70 in 55-gallon drums. These drums were apparently stored in an uncovered and unbermed area (ABB-ES, 1996). Another chemical storage area was located next to the former B-2 Chromium Plating Facility in Building B-2. Chemicals in this storage area were segregated into three types: acids, cyanides, and alkalines. The acids occupied two storage bays and the cyanides and alkalines occupied a third. The containers were situated on wood pallets on concrete floors (ABB-ES, 1996).

Building B-7 was used for paint and oil storage prior to use as a testing facility (FIA, 1956). Flammable materials were stored in Building B-8 on wood pallets or on metal shelves and barrel racks contained within the building were surrounded by a concrete containment dike. The floor in this area was covered with absorbent material and no floor drains were observed in the building.

Batteries, oil, grease, and hydraulic fluid were stored in Building B-9, a maintenance shop area. The floors in this building are concrete, but there is a hydraulic lift pit and floor drains that lead to the OATP.

Paints, acids, and gases were stored in several storage sheds north of Building B-4, starting in the 1950s (FIA, 1956). Later, machining oils and engine oils were stored in 55-gallon drums on metal racks in this area (W-C, 1991). These oils were used in the engine testing and development areas.

<u>Container Storage Areas.</u> Used from 1980 to 1984, the Container Storage Area, located adjacent to the north and northwestern side of Building B-13, was used accumulate 55-gallon drums of hazardous waste. This area was the original Container Storage Area identified in the site's RCRA Part A Permit Application submitted in November 1980. Reportedly, containers were allowed to accumulate in this area for less than 90 days.

The Container Accumulation Area and Drum Staging Area, located adjacent to and southeast of the former Oil House Tank Farm, were used to collect and prepare containerized waste for disposal off site or into tanks in the Oil House Tank Farm. The western half of the area was used for staging, and the eastern half for waste accumulation. The area served as a staging area for drums that required testing of contents prior to disposal off-site and was reportedly used to store drums for less than 90 days (W-C, 1991). During large rain events, rain water that accumulated in the area was allowed to drain to a gravel area through three drain plugs located on the back wall; the drain plugs were sealed with concrete in 1990 (CDM FPC, 1992).

Container Storage Areas A and B are located along the southern property boundary, south of Building B-18. These areas were used from 1983 to 1986 for less than 90-day hazardous waste accumulation prior to off-site disposal. Containerized liquid and solid wastes, typically in 55-gallon drums, were collected from numerous locations at the facility and brought to these storage areas. The storage areas consisted of sloped concrete slabs, each approximately 12 feet wide and 22 feet long. Each had a separate roofed area, with containment dikes consisting of low concrete block walls. Historically, containers of the following waste materials were accumulated in these areas: waste jet fuel, paint, waste acetone, waste sodium hydroxide, waste 1,1,1-TCA, waste oil, and chromium-contaminated plating wastes (Honeywell, 2000).

Satellite Accumulation Areas. Since 1990, numerous satellite accumulation areas were established in various locations at SAEP. Satellite accumulation areas served as temporary storage areas for small quantities (one to six 55-gallon drums) of waste liquids and/or solids. Several of these satellite areas did not have containment systems, and storm drains were observed near some of them. At the time of the publication of the Final EBS Report in 1996, the number of satellite accumulation areas had been reduced, and all were eliminated from service in by 1997 when facility operations were ceased.

Open Storage Areas. The shoreline area (i.e., area where hydraulic fill was placed) was used to store raw stock such as castings (which were manufactured elsewhere), fuels and oil, scrap metals, and waste fuels, oils, and solvents. Aerial photographs indicate open storage areas historically were located throughout the site; however, the most significant of these was a large open storage area located between Building B-16 and the former Oil House Tank Farm from the early 1950s to the mid-1990s.

Use and Storage of Pesticides, Rodenticides, and Herbicides. Pesticides and rodenticides were used at SAEP to control pest-related structural problems and to prevent health problems. All pest control services were contracted to State of Connecticut certified applicators (ABB-ES, 1996). According to interview information obtained during the EBS, pest control was also accomplished by SAEP maintenance crews. However, no evidence of widespread pesticide or rodenticide use or storage was discovered through records research.

Small quantities of herbicides were formerly stored in Building B-13. Herbicides were mixed outside, generally on paved areas, without containment. No wastes were reportedly generated, because the entire mixed product was used at the time of application.

Use and Storage of Radiological Materials. Thorium alloy was machined at SAEP as part of engine production and required special handling. All machine turnings of the alloy and machining fluids were recovered and recycled. Radiation and contamination surveys were performed and recorded semi-annually, and the SAEP Safety Office ensured compliance with the conditions of the U.S. Nuclear Regulatory Commission (NRC) license and AlliedSignal's Standard Operating Procedures (SOP). Readings reportedly average 0.001 millirem per hour (mrem/hr). This radiation level was well below the 5-mrem/hr level allowable for occupational exposure (ESE, 1981). In accordance with license requirements, radiological inspections of the installation were conducted by the NRC every three years on an unannounced, random basis.

Historical operations conducted at the SAEP utilized instruments and materials containing radioactive nuclear byproduct materials (i.e., cobalt 60, silver 110, cadmium 109, cesium 134, cesium 137, iridium 192, phosphorous 32, krypton 85, strontium 90, and hydrogen 3). The instruments and materials were used in the 1960s and 1970s in the production and testing of missile components, and were regulated under NRC License Numbers 06-08612-01, 06-08612-03, 06-08612-04, and 06-08612-05. Possession and use of the licensed materials was discontinued in the 1970s when the licenses expired. The instruments were used in testing and evaluating material thickness, density, temperature, corrosion, and ablation, and contained sealed sources of radioactive material. A sealed source means that the radioactive source material is shielded to prevent emission of radiation outside of the instrument. Metallic materials (e.g., bearings) containing radioactive byproducts were used in the production and testing of missile components. All licensed radioactive materials and their uses reported in the NRC licenses are presented in the Table 1-4.

Magnesium- and nickel-thorium (thorium) alloys (2 % thorium) were also used as part of the turbine engine intake structures at SAEP. These materials were handled under NRC License Number STB-393. The maximum licensed amount of thorium alloy that could be stored on-site at a given time was 2,300 kilograms (kg). In a letter dated July 14, 1999 from the NRC to AlliedSignal, the NRC indicated that all portions of the facility except the Causeway "…may be released for unrestricted use."

Waste Treatment and Disposal. Since 1951, most of the wastes generated resulted from production operations in Buildings B-2. Wastes were also generated in Building B-10; engine and engine component testing in Buildings B-16 and B-19; research and development in Building B-3; raw materials testing in Buildings B-3, B-6, B-7, and B-58; and vehicle maintenance in Building B-9. Industrial activities at SAEP resulted in the generation of wastewater contaminated with heavy metals, cyanide, caustics, acids, oils, greases, fuels, and solvents.

Quantities of waste generated at SAEP from 1985 to 1990 are listed by waste type in Table 1-5. SAEP-manifested wastes from 1992 to 1994 are listed by waste collection stream in Table 1-6. The information in Tables 1-5 and 1-6 was compiled by SAEP environmental staff based on manifests in their files (ABB-ES, 1996). Generated waste was either treated on site or disposed of off-site by a private contractor.

The following waste treatment and disposal systems were used at the Site and are discussed in the following paragraphs:

- chemical waste treatment system (CWTP and CDF);
- stormwater treatment system (OATP);
- stormwater outfalls;
- sanitary sewer system;
- solid waste management system; and
- Causeway.

Chemical Waste Treatment System (CWTP and CDF). From 1951 (when plating operations began at SAEP) to 1958, wastewater from plating and related operations was neutralized in the individual process tanks and then discharged to the storm water system for disposal (ESE, 1981).

At that time, storm water drained directly to the Housatonic River. The chemical waste treatment system began collecting wastes generated by electroplating and other corrosion resistance operations in May 1958.

As originally designed, the CWTP had an associated bentonite-lined equalization lagoon, which has been closed under RCRA Subtitle C requirements. Treatment processes at the CWTP originally involved alkaline chlorination for cyanide oxidation; hexavalent chromium [Cr(VI)] reduction; and coagulation and sedimentation of metal hydroxide precipitates. Sludges generated by these treatment processes were disposed of in one of three on-site, unlined lagoons. These lagoons were also closed under RCRA Subtitle C requirements. Clarified water from the treatment process was originally discharged to a tidal ditch located east of Building B-18, briefly discharged to the OATP (approximately 1976 to 1979), and subsequently discharged to OF-008 constructed within the tidal ditch (ESE, 1981 and USEPA, 1990).

In the 1980s, various problems associated with operations at the CWTP were recognized, such as the disposal of waste sludge in unlined lagoons, discharges above permitted pH limits, and potentially unsafe mixing of waste streams that could result in generation of hydrogen cyanide gas (Weston, 1982). To address these issues, the plant and piping were upgraded to segregate waste streams, add a separate cyanide destruction facility, add equalization tanks, renovate existing tanks and clarifier, and add a sludge dewatering facility.

After the system upgrade, cyanide-contaminated wastewater was piped to CDF, where it was treated by alkali chlorination and converted to nitrogen and bicarbonate. Effluent from this process was combined with other wastewaters and pumped to equalization tanks at the CWTP. Treatment at the CWTP involved chromium reduction, precipitation of chromium and other heavy metals, and clarification. Effluent from the clarifier passed through sand filters before final discharge at OF-008. A thickener and filter press in Building B-71 dewatered sludge from the clarifier. The filter cake was stored in a roll-off prior to disposal off-site, and the filtrate was returned to CWTP for further treatment.

<u>Stormwater Treatment System (OATP).</u> The OATP began operations in 1976 to remove oil and grease from wastewater in the plant's storm drainage system prior to discharge to the Housatonic River. The OATP operated continuously, with an average flow of 1.8 million gallons per day, and a maximum design flow of about 4,200 gallons per minute (INEL, 1991).

The plant drainage system is currently equipped to convey dry weather flow, including the first flush of storm water, to the OATP for treatment; however, excessive runoff from large storm events is discharged directly to the Housatonic River. Wastewater is delivered to the OATP by six pump stations. Pump stations B-36, B-37, and B-38 serve the northern half of SAEP, and pump stations B-41, B-40, and B-64 serve the southern half. Each half of the plant is served by its own transmission main to convey water to the OATP (ABB-ES, 1996).

Influent to the OATP enters a surge tank for flow equalization. Treatment processes involve coagulation and flocculation by addition of liquid alum in a flash mixer, and dissolved air flotation and skimming in a flotation chamber. Effluent from the OATP is discharged to the Housatonic River through OF-007.

Stormwater Outfalls. SAEP is authorized to discharge from eight OFs under NPDES Permit No. CT0002984. OF-007 currently discharges wastewater treated at OATP to the Housatonic River (see above) and OF-008 discharges wastewater treated at CWTP to a drainage channel that flows to the Marine Basin south of the plant. Outfalls OF-001 through OF-006 discharge intermittently to the Housatonic River when heavy runoff enters the plant's storm water drainage system. Four chemical releases to the intertidal flats have been documented, including chromic acid (1978 and 1987), oil (1979) and fluorescent metal penetrant dye (1981) (ABB-ES, 1996).

Sanitary Sewer System. Sanitary wastewater generated at SAEP is conveyed to the Town of Stratford's treatment plant. Sanitary sewers from plant buildings connect into a main discharge line that runs across the site and connects into a pump station located in the North Parking Lot, owned and maintained by the Town of Stratford. The Town of Stratford has an easement across SAEP for their sewer line and pump station, which was granted in early 1982, consisting of approximately 1,250 linear feet and 0.22 acres of land at SAEP.

The Town of Stratford's treatment system consists of activated sludge secondary treatment; effluent is discharged into the Housatonic River. The discharge outfall is located about 1/2-mile upstream of SAEP.

Solid Waste Management System. Solid wastes generated at SAEP included sludges from treatment at the CWTP (listed hazardous waste not solid waste) and the OATP, scrap metal and wood, waste paper, and small amounts of waste food scraps and medical waste. The majority of these wastes were collected for disposal or reclamation by private contractors. Reportedly, ash and cinders from incineration were disposed of on site (W-C, 1991).

Waste sludges generated by the CWTP were stored in three unlined sludge lagoons. From 1957 to 1968, about 7,000 gallons of sludge were removed bi-annually by a private contractor and disposed of at the Town of Stratford's landfill. This practice ceased in 1968, when the landfill refused to accept sludge. In 1981, approximately 10,000 cubic yards of sludge was removed and disposed in Bridgeport's Seaside Park Landfill (ESE, 1981). These lagoons were closed in 1990.

Waste sludges generated by the OATP were stored on site in the oil-alum tank near Building B-44 and periodically removed for off-site incineration. This tank was previously located at the southwest corner of Building B-13. Since SAEP closed less sludge is produced and is removed directly from OATP by vacuum truck.

Causeway. Fill materials were deposited along the northern edge of the Causeway during the 1950s and 1960s; the source of these materials is not known. Some of the fill is reported to consist of asbestos-containing materials, as well as construction debris (W-C, 1991). The Causeway was also reportedly used as a training area for the plant's fire department; fires were started and extinguished on the Causeway (ESE, 1981).

1.6 PREVIOUS INVESTIGATIONS

This section is divided into subsections that discuss the following previous investigations:

- RCRA Lagoons (1983-Present)

- Preliminary Assessment Screening (1991)
- RCRA Facility Assessment (1992)
- Phase I Remedial Investigation (1992)
- Phase II Remedial Investigation (1994-1995)
- Environmental Baseline Survey (1996)
- Operable Unit (OU) 2 Non-Time Critical Removal Action (1999-2000)
- Causeway and Dike Non-Time Critical Removal Action (1999-2000)
- Phase III Remedial Investigation (1998-2004)

1.6.1 RCRA Lagoons

Wastewater treatment at SAEP began in 1957 with the construction of CWTP. The spent plating bath was discharged to the equalization lagoon then pumped through the CWTP, which produced a metal hydroxide sludge that was pumped to one of three sludge lagoons. The lagoons were replaced 1987 with an equalization tank, sand filter and filter press. The lagoons were closed under RCRA Subtitle C provisions in 1990. As part of the closure, sludge and contaminated soil were excavated to the low tide level and disposed of off-site.

In 1983, a groundwater monitoring program was initiated to determine potential impacts to the subsurface as a result of the lagoons. Since 1983, quarterly and semi-annual groundwater samples have been collected from the LW-series wells and analyzed for volatile organic compounds (VOCs), semi-volatile organic compounds (SVOCs) and metals. In summary, VOC groundwater contamination is present in the vicinity of the former equalization lagoon and sludge-drying beds. VOCs detected include PCE, TCE, 1,1,1-TCA and vinyl chloride. The contamination is generally located within shallow groundwater, but has migrated to deep groundwater downgradient of the former equalization lagoon (LW-5DI). Deep groundwater contamination is also present at LW-10D.

In 1990, C.A. Rich Consultants was contracted to evaluate; the hydrogeological and hydrochemical role of the organic peat layer, evaluate the potential for tidal-induced flow in the surficial aquifer, the cause for groundwater mounding in the “sand channel”, and the hydrogeologic impacts of the landfill cap.

It was determined that the direction of shallow groundwater flow in the upper aquifer is to the east, and is controlled by the organic peat present beneath the former lagoons and tidal ditch. Flow is radial toward the east, south, and west. The hydraulic conductivity of the peat is lower than the corresponding upper and lower glacial sand and gravel stratigraphic units, which results in a partially perched water table in the vicinity of the former lagoons. The groundwater flow direction in the lower aquifer is south-southeast. The peat layer retards, but does not prevent, natural vertical flow of groundwater. The Marine Basin and Long Island Sound regionally influence groundwater flow in the lower aquifer (C.A. Rich, 1991).

C.A. Rich concluded that groundwater quality in the lower aquifer is generally better than the upper aquifer. The potential exists for contaminant migration through the peat layer into the lower aquifer; however, the lower concentrations in the intermediate and deep wells suggest that the peat has been relatively effective in retarding the downward flow of contaminants. One exception is the presence of chlorinated solvents in well LW-10D. This well is located in the lower aquifer upgradient of, and separated vertically from the former lagoons by the peat layer. LW-10D contamination may suggest a secondary source of contamination not associated with the former lagoons. (C.A. Rich, 1991)

1.6.2 Preliminary Assessment Screening

In 1991, the U.S. Army was considering the lease or sale of government-owned property at SAEP to Textron Lycoming, Stratford Operations. U.S. Army Regulation 200-1 (AR 200-1) required that a PAS be completed for any real property for which a transaction was being proposed. The purpose of the PAS was to determine the U.S. Army's potential liabilities associated with the environmental condition of SAEP.

URS Greiner Woodard Clyde (URSGWC) was retained by United States Army Corps of Engineers (USACE) to complete the PAS for SAEP. During the PAS, information was collected to outline the type and extent of the real property transaction being considered, to generally describe the environmental setting, and to identify and evaluate subject areas of concern. The PAS included a review of SAEP records, interviews with SAEP personnel, visual reconnaissance of the facility and surrounding area, and collection of information from other sources.

The PAS investigation identified eight broad areas of potential environmental concern at SAEP: Tidal Flats; Causeway; hydraulic fill area; plating and manufacturing area; most of building B-2 (excluding the plating shops); research and development area; testing area; and drainage ditch. These eight areas of potential concern are discussed briefly below:

The Tidal Flats was identified as an area that may have received hazardous compounds from past discharges and/or runoff from SAEP, as well as from upstream areas of the Housatonic River. Contaminants which may have been deposited in this area include VOCs, SVOCs, polychlorinated biphenyls (PCBs), heavy metals, and cyanide.

The Causeway was used for a fire training area and a landfill area. It was thought that contaminants might be present in the fill material comprising the Causeway, including asbestos, polynuclear aromatic hydrocarbons (PAHs), and metals.

The hydraulic fill area (located immediately landward of the Tidal Flats and Causeway and including Buildings B-16, B-19, B-48, B-65, and numerous other small buildings along the eastern edge of the Site) was used to store, treat, and dispose of materials used at SAEP including metals, fuels, oils, and solvents. Potential contaminants of concern included VOCs, SVOCs, PCBs, and heavy metals.

The plating and manufacturing area is located in the central portion of the Site and includes the easternmost part of Building B-2, the western half of Building B-3, Buildings B-10, B-12, B-77, and other small buildings in the central portion of the Site. The plating and manufacturing area was used for plating, engine cleaning and manufacturing, and for storage of fuels, solvents and paints. Past disposal and/or uncontrolled releases are known to have contaminated soils and groundwater within this area. Potential contaminants of concern included VOCs, SVOCs, heavy metals, and cyanide.

Most of Building B-2 (excluding the plating shops) was used for assembling aircraft and manufacturing gas turbine engines. Past disposal practices and/or uncontrolled releases may have resulted in soil and groundwater contamination. Potential contaminants of concern included VOCs, SVOCs, and heavy metals.

The research and development area encompasses the eastern half of Building B-3 and Buildings B-3A, B-4, B-69 and B-79. This area was used for research and development operations for over 50 years. Metals, fuels, oils, and solvents were handled in this area. Potential contaminants of concern included VOCs, SVOCs, and heavy metals.

The testing area is located in the southeastern corner of the Site, and includes Buildings B-6, B-6A, B-17, B-53, B-61, and B-72. This area contains the steam generating plant and was also used for experimental engine testing and the storage of fuels. During excavation for lagoon closures within the southern portion of this area, petroleum-contaminated soils were reportedly observed. Potential contaminants of concern included VOCs, SVOCs, and heavy metals.

The last area, the drainage ditch, is located along the Housatonic River at the southern boundary of the Site. This area includes a drainage ditch that originates at SAEP and flows into the Marine Basin. No hazardous materials were handled or generated within this area, but it was thought to represent a potential depositional area for contamination from stormwater runoff and effluent from SAEP. Effluent from the CWTP discharged to the ditch under the facility NPDES permit. This discharge occurred since 1958. Potential contaminants of concern primarily included compounds in the CWTP waste stream and, to a lesser extent, chemicals associated with site runoff. These chemicals included VOCs, SVOCs, PCBs, heavy metals, and cyanide.

Conceptual models were developed for each of the eight areas based on information available at that time. These models were developed to identify the types of information needed to conduct a baseline risk assessment. Based on these models, it was determined that additional data were needed to determine if complete exposure pathways for site-specific contaminants existed. The PAS concluded with a recommendation that additional sampling was needed for the eight areas of potential environmental concern.

1.6.3 RCRA Facility Assessment

A RCRA Facility Assessment (RFA) was conducted by the USEPA in 1991/1992 (CDM FPC, 1992). The purpose of the RFA was to identify and gather information on known and potential releases from Solid Waste Management Units and Areas of Concern (AOCs) and to evaluate their potential for future impact to human health and to the environment.

As part of this assessment, the USEPA identified 58 AOCs. These AOCs are areas where the USEPA believed releases of hazardous materials or petroleum may have occurred. These 58 areas are included within the eight areas identified in the PAS. The RFA Report recommended further investigation of each of the 58 AOCs. It was noted in the RFA Report that several of the recommendations for further investigation may be addressed in the Army’s future remediation investigation activities at the Site.

1.6.4 Phase I Remedial Investigation

URSGWC was retained by USACE in 1991 to prepare the work plan and implement the Phase I investigation. The objective was to evaluate the presence or absence of contamination and to recommend further action regarding the disposition of the eight areas of potential environmental concern identified in the PAS. Evaluation of the extent of contamination was not included in the scope of this investigation. The overall approach of the investigation was to sample areas that had, or potentially had, contaminant releases or which were locations for disposal activities. The Phase I investigation included completion of ten soil borings, collection of subsurface soil samples, installation of 15 monitoring wells, groundwater sampling, and the collection of sediment/surface water samples at 13 locations. No samples were collected from locations inside building footprints because the plant was operational at the time.

Elevated levels of chlorinated hydrocarbons and benzene, toluene, ethylbenzene, and xylenes (BTEX) were detected in groundwater. Sediment results indicated concentrations of BTEX (limited aerial extent), PAHs, PCBs, cyanide (limited aerial extent), and metals that were elevated with regard to background. Concentrations of halogenated aliphatics (a class of VOCs), the ketone 2-propanone, cyanide, and some metals in surface water appeared to be associated with effluent from SAEP outfall locations. Additional investigation was recommended to define the extent of PAHs and PCBs in the sediment and to determine the impact on the environment.

No regulatory guidance was available at the time and, for most media, standards did not exist. Therefore, conclusions and recommendations presented in the Phase I report were based on professional judgment and a specific end use for the Site. Results of the sampling program indicated that concentrations of contaminants in soil or groundwater were not of concern in terms of direct contact.

1.6.5 Phase II Remedial Investigation

URSGWC was retained by USACE in 1994 to prepare the work plan and implement the Phase II investigation. Activities of the Phase II investigation included; sediment sampling for contamination delineation, sediment toxicity sampling and benthic community characterization, subsurface soil sampling and analysis, monitoring well installation, and groundwater sampling. No samples were collected from locations inside building footprints because the plant was operational at the time.

The objectives of the Phase II investigation were to 1) define the extent, degree, and rate of migration of the contaminants found in the groundwater and sediments in the Phase I investigation and 2) identify the potential human health and ecological risks pertaining to the contaminants found in the groundwater and sediments in the Phase I investigation. The overall objective of the Phase II investigation at SAEP was to provide results that could help to define the environmental liabilities of the SAEP property for the purpose of real estate transfer.

Sediment samples from 0 to 6 inches and from 24 to 48 inches below surface were collected at 28 locations along six transects in the Tidal Flats and five other locations (three in the Housatonic River and one each in the Marine Basin southeast of the SAEP and OF-008 [in the drainage channel]) and analyzed for Target Compound List (TCL) PAHs, Priority Pollutant Metals (PP Metals), PCBs, Total Organic Carbon (TOC), grain-size distribution, and percent moisture.

Surficial sediment grab samples for benthic macroinvertebrate community analyses and toxicity testing were collected from ten locations in the Tidal Flats, one location each in the drainage channel (Outfall 008) and in the Marine Basin, and at three background sample locations in the Housatonic River.

Oyster tissue samples were collected at five locations in the Tidal Flats and one location from the Housatonic River and analyzed for selected metals (mercury, cadmium, chromium, nickel and lead), PCBs, percent moisture, and lipids.

Eleven monitoring wells were installed at five locations in the presumed upgradient direction of the SAEP. Soil samples from the screened interval of each well were submitted for analysis of grain size distribution and percent moisture. Five soil samples, one from each new monitoring well location, were analyzed for TCL VOCs and PP Metals.

Groundwater from the eleven newly installed wells was sampled and analyzed for TCL VOCs and PP Metals on two separate occasions. In addition, a total of 46 existing wells were sampled once and analyzed for TCL VOCs and PP Metals to confirm analytical data of earlier sampling rounds.

As part of the Phase II investigation, data were compared to CTDEP RSRs that were developed in 1996. A copy of CTDEP RSRs is attached as Appendix A. Data were compared to the PMC, I/C DEC, and SWPC for groundwater. Exceedances were detected across SAEP in various media. Elevated levels compared to background of PAHs, PCBs, and PP Metals (particularly chromium, copper, and mercury) were detected in sediments at some locations of the Tidal Flats. Exceedances of criteria in groundwater monitoring well samples were noted primarily for chromium, copper, lead, arsenic, nickel, and zinc. VOCs detected included the chlorinated compounds PCE, TCE, 1,2-dichloroethene (1,2-DCE), 1,1-dichloroethene (1,1-DCE), 1,1-dichloroethane (1,1-DCA), 1,1,1-TCA, and vinyl chloride and aromatic compounds, including BTEX and chlorobenzene.

The Phase II Report concluded that, based on the distribution of chlorinated compounds found in Phase II monitoring wells, the degradation pathways of the chlorinated compounds found, and inferred groundwater flow directions, contaminants detected at Phase II monitoring well locations WC-20 and WC-21 near Frash Pond are likely the result of off-site sources. In addition, groundwater contour maps suggested that groundwater may be flowing on-site from the vicinity of the airport. In addition, it appeared that dissolved contamination from potential off-site locations south and west of the SAEP may be influenced by this suggested groundwater flow regime. Other PCE and degradation product concentrations found on-site may result from separate on-site and/or off-site sources.

The Phase II Report also included a baseline human health risk assessment and a baseline ecological risk assessment. The risk assessments were used to assess the significance of the exceedances of the CTDEP criteria/standards. The risk assessments assumed an industrial re-use of SAEP. As a result, the risk assessments focused on potential exposures to construction workers on the Site and commercial fishermen and recreators in the tidal flats. Exposures were not considered for occupational workers at SAEP because the entire Site is paved.

The baseline human health risk assessment concluded that, based on information from the Phase I and Phase II investigations, the noncarcinogenic hazard index for all receptors were below 1.0 and the cancer risk for all receptors was within or below USEPA's target risk range. Therefore, no adverse human health effects were expected at SAEP and no remediation to address potential risks to human health was indicated.

The baseline ecological risk assessment indicated population-level risks to indigenous benthic macroinvertebrates that colonize the Tidal Flats in the vicinity of some outfalls. These potential risks may impact localized portions of the study area (in the case of the benthic habitat) or individual birds that occasionally feed at SAEP. Therefore, remediation to address potential ecological risks in the portion of the Tidal Flats in the vicinity of the outfalls in the Tidal Flats and the drainage channel was indicated.

Details on the baseline human health risk assessment and baseline ecological risk assessment are available in Human Health Baseline Risk Assessment (HHBRA) and the Baseline Ecological Risk Assessment (BERA), respectively, both dated April 1996 (W-C, 1996).

1.6.6 Environmental Baseline Survey

SAEP was placed on the BRAC list in 1995. As a result, the focus of the on-site characterization shifted to a BRAC property transfer track. Under BRAC 95, the Environmental Restoration Program begins by conducting an EBS, which describes the environmental condition of the property. This is used to determine the suitability to lease or transfer excess BRAC property. The USAEC contracted the EBS to ABB-ES. The EBS was completed in December 1996.

The activities completed by ABB-ES included:

- Review of SAEP records including reports, engineering drawings, and historical photographs;
- Discussions with SAEP personnel familiar with current and past Site activities;
- Visual reconnaissance of the Site and areas immediately adjacent to the Site;
- Collection of information pertaining to the Site or adjacent areas from public organizations; and
- Review of the Phase I Report and Draft Phase II Report (including the HHBRA and BERA).

The EBS concluded that several areas of the SAEP property have known or potential environmental contamination caused by operational or waste disposal practices. These areas are listed below:

- Tidal Flats;
- Shoreline Fill Area;
- Plating and Manufacturing Area;
- Building B-2, North Parking Lot and West Parking Lot;

- Building B-65;
- Research and Development Areas (Buildings B-3A, B-4, B-69, B-79, and the eastern half of B-3);
- South Parking Lot, Chemical Waste Treatment Plant, and Closed Lagoons; and
- Testing Area (Buildings B-6, B-6A, B-17, B-53, B-60, B-61, and B-72).

The EBS presents a site numbering system for AOCs identified in the RFA, and additional areas of environmental concern identified in the EBS investigation. Site numbers 1 through 58 correspond to RFA AOCs 1 through 58, respectively. Site numbers 59 through 75 correspond to those areas of environmental concern due to storage, release or disposal of hazardous materials or petroleum not associated with RFA AOCs. The AOCs and EBS Sites are included in Table 1-7. Figure 1-5 shows their locations. The EBS concluded with a recommendation that additional monitoring wells and soil borings may be needed to assess source area conditions and distribution of contaminants at SAEP.

1.6.7 OU 2 Non-time Critical Removal Action

Foster Wheeler Environmental Corporation (FWENC) and Harding Lawson Associates (HLA) were contracted through the USACE – New England District (NAE) in 1999 to complete a Non-time Critical Removal Action (NCRA) for chromium and VOC source area groundwater remediation. The objectives were to: 1) complete field activities necessary to provide further characterization of subsurface conditions at SAEP; 2) summarize the results of previous field activities in a report; 3) conduct treatability testing to determine the effectiveness of particular in-situ technologies at reducing contamination in groundwater at the site; and 4) document the decision process for selection of removal actions for OU 2 in an Engineering Evaluation/Cost Analysis (EE/CA).

The goal of the OU 2 NCRA was to address high concentrations of Cr(VI) and chlorinated VOCs in groundwater that may pose a threat to the Housatonic River environment by off-site groundwater migration. As part of the NCRA for OU 2, it was necessary to more fully characterize site conditions, and to provide a recommended removal action to address groundwater contamination. Project activities included:

- Establish the presence or absence, and the concentration of VOC vapors beneath SAEP buildings through a soil vapor survey.
- Evaluate the presence or absence of dense non-aqueous phase liquid (DNAPL) contamination at selected subsurface locations.
- Assess through a geophysical seismic survey the features of the bedrock surface.
- Evaluate the hydraulic properties of the aquifer in the vicinity of the former B-2 Chromium Plating Facility with an aquifer-pumping test.
- Use the results of the aquifer test and the cone penetrometer stratigraphic data, to estimate the hydraulic properties of the aquifer in selected other areas of the site.
- Determine the location and elevation of previous sampling locations with an elevation and location survey.
- Test the effectiveness of selected in-situ technologies at reducing Cr(VI) and VOC contamination in groundwater using treatability testing.
- Document the process used to determine the recommended removal action in an EE/CA.

The OU 2 NCRA process was abandoned after the publication of the Final OU 2 EE/CA in September 2001, in favor of completing the contaminant characterization and remedial alternatives evaluation as part of the site-wide RI/FS process.

1.6.7.1 Pre-design Investigation Report. Data from HLA's groundwater and soil investigations from August 1998 through March 1999 were evaluated and summarized in the Draft Data Package for the former B-2 Chromium Plating Facility Investigation (HLA, 1999). The Pre-Design Investigation Report (PDIR) summarized the data presented in the Draft Data Package and the additional investigation data collected under the OU 2 NCRA. The Final OU 2 PDIR was completed in May 2000 (FWENC/HLA, 2000a).

The PDIR includes results of decontamination of the former B-2 Chromium Plating Facility, investigations at the former B-2 Chromium Plating Facility, Cr(VI) and VOC pre-design investigations, OU 2 NCRA investigations, and indoor air sampling. Based on these investigation activities, the following description of the source, nature, and extent of contamination was developed.

<u>Hexavalent Chromium-contaminated Structures.</u> Wipe sampling completed following decontamination of the former B-2 Chromium Plating Facility in December 1998, identified concentrations of Cr(VI) on the floor, walls, and overhead beams in excess of HLA-developed risk-based cleanup standards. Cr(VI) was visually identified throughout the entire thickness of the concrete floor in the northwestern corner of the plating room.

<u>Hexavalent Chromium-contaminated Soil and Groundwater.</u> Subsurface soil and groundwater analytical data were collected during field investigations completed from January 1999 to April 1999 (FWENC/HLA, 2000a). Total chromium in soil was compared to the CTDEP RSR PMC and DEC. Groundwater results were compared to CTDEP RSR SWPC. Groundwater Protection Criteria are not established for GB classified areas, which is the classification of SAEP groundwater.

The comparison of soil data to the CTDEP RSR criteria identified two locations of subsurface soil exceeding the CTDEP RSR DEC for Cr(VI) (SP-99-11 and SP-99-14). There are also exceedances of the CTDEP RSR PMC for total chromium beneath the footprint of the former Chrome Plating Facility.

Groundwater beneath the northwestern-most corner of the former B-2 Chromium Plating Facility, in an area covering approximately 40,000 square feet, contains Cr(VI) in excess of the CTDEP RSR SWPC (0.11 milligrams per liter [mg/L]). A second area of Cr(VI)-contaminated groundwater at concentrations in excess of the CTDEP RSR SWPC is located beneath the south central portion of the former B-2 Chromium Plating Facility, and covers an area of approximately 10,400 square feet.

<u>VOC-contaminated Groundwater.</u> Groundwater analytical data were collected during field investigations completed from January 1999 to June 1999. Additional data were collected during aquifer testing, bench-scale treatability testing, and pilot-scale treatability testing, as discussed in Subsection 1.6.7.2. Groundwater VOC concentrations were compared to the CTDEP RSR SWPC

for class GB groundwater and I/C VC. Additionally, soil gas concentrations were compared to the CTDEP RSR Soil Vapor I/C VC and indoor air concentrations were compared to CTDEP RSR I/C Indoor Air Target Concentrations (IATC).

Three groundwater VOC source areas were identified during evaluation of the VOC groundwater data. These source areas are summarized below.

Chlorinated VOC Hot-spot No. 1. VOC Hot-spot No. 1 is located beneath the former B-2 Chromium Plating Facility in Building B-2. TCE was detected in groundwater at a maximum concentration of 830 mg/L immediately above the surface of the aquitard identified during investigations (approximately –20 feet mean sea level [MSL]). The estimated horizontal extent of TCE in groundwater at concentrations exceeding 100 mg/L covers the majority of the footprint of the former B-2 Chromium Plating Facility. The source of the TCE is suspected to be from degreasing operations completed as part of facility operations. Visual observation of subsurface soil and groundwater samples, and shake tests using Sudan IV dye, did not reveal the presence of TCE dense non-aqueous phase liquid (DNAPL).

PCE and 1,1-DCE concentrations also exceeded SWPC in the vicinity of VOC Hot-spot No. 1. VOC concentrations in groundwater also exceeded the I/C VC in the vicinity of VOC Hot-spot No. 1

Chlorinated VOC Hot-spot No. 2. VOC Hot-spot No. 2 is located between Buildings B-48 and B-16. The primary VOC detected at high concentrations in this area is TCE. TCE was detected at a maximum concentration of 264 mg/L within the sandy silt zone (approximately –8 feet MSL). The horizontal extent of TCE in groundwater exceeding 100 mg/L is estimated to cover an area roughly 75 feet in diameter; however, it is possible the area of TCE concentrations greater than 100 mg/L may extend under Building B-16. The source of the TCE is suspected to be from disposal on the ground surface, and/or degreasing operations completed in Building B-16. Visual observation of subsurface soil and groundwater samples did not reveal the presence of TCE DNAPL.

PCE and 1,1-DCE concentrations also exceeded SWPC in the vicinity of VOC Hot-spot No. 2. VOC concentrations in groundwater also exceeded the I/C VC in the vicinity of VOC Hot-spot No. 2.

Chlorinated VOC Hot-spot No. 3. VOC Hot-spot No. 3 is located in the center of Building B-2. The primary VOC detected in this area is 1,1,1-TCA. The maximum concentration of 1,1,1-TCA detected in VOC Hot-spot No. 3 was 280 mg/L at a depth of approximately –24 feet MSL. Concentrations of 1,1,1-TCA from deeper samples in this area indicate that 1,1,1-TCA has migrated to the bedrock surface (approximately -152 feet MSL). The estimated horizontal extent of 1,1,1-TCA in groundwater at concentrations exceeding 100 mg/L covers an area of roughly 35,000 square feet; however, it is possible this area of contamination is actually two distinct areas, centered around explorations CP-99-08 and WP-99-48, respectively. The estimated source of the 1,1,1-TCA is suspected to be from degreasing operations formerly conducted in this portion of Building B-2. Visual observation of subsurface soil and groundwater samples did not reveal the presence of 1,1,1-TCA DNAPL.

TCE and 1,1-DCE concentrations also exceeded SWPC in the vicinity of VOC Hot-spot No. 3. VOC concentrations in groundwater also exceeded the I/C VC in the vicinity of VOC Hot-spot No. 3.

VOC-Contaminated Soil Vapor and Indoor Air. Analytical results from the soil vapor survey indicate that soil vapor in the subsurface exceeds I/C VC for vinyl chloride, 1,1-DCE and TCE. VOC soil vapor concentrations beneath the central portion of Building B-2, extending northeast and east toward Buildings B-15, generally exceed CTDEP I/C VC. The highest levels of VOCs in soil vapor were co-located with groundwater contamination. Analytical results of indoor air sampling through April 2000 indicated concentrations of 1,1-DCE and vinyl chloride above the CTDEP I/C IATC in Buildings B-2, B-9, B-12, and B-65. Contaminants detected in indoor air samples were also found in soil vapor samples and groundwater.

1.6.7.2 Pilot-scale Treatability Studies. Two pilot-scale treatability studies were conducted by FWENC/HLA in 1999-2000 to investigate the effectiveness of in-situ treatment technologies at reducing concentrations of Cr (VI) and VOCs, primarily TCE, in groundwater hot-spot areas. Information obtained during operation of the pilot-scale systems was intended to aid in cost estimation and design of full-scale systems for treatment of the hot-spot areas. The selected technologies were proposed possible solutions to address hot-spot contamination only, and were not intended to fully remediate site groundwater.

The pilot tests were conducted in accordance with the methods described in the Pilot-Scale Treatability Study Work Plan for the Chromium and VOC Groundwater OU 2 EE/CA (FWENC/HLA, 1999). The work plan was developed based on the results of bench-scale treatability testing and in accordance with the USEPA's Guide for Conducting Treatability Studies (USEPA, 1992).

Two treatment technologies were investigated in separate test areas during pilot-scale testing at SAEP. The in-situ Cr(VI) reduction test investigated the ability of ferrous sulfate ($FeSO_4$) to reduce concentrations of Cr(VI) in the Chromium Hot Spot. The in-situ chemical oxidation test investigated how well potassium permanganate ($KMnO_4$) reduces concentrations of VOCs, primarily TCE at VOC Hot-spot No. 1. Tests were conducted in two phases. Phase 2 was added after the Phase 1 results indicated additional treatment was required to properly evaluate technologies.

1.6.8 Causeway and Dike Non-time Critical Removal Action

The goal of the Causeway and Dike NCRA was to characterize subsurface conditions and provide a recommended removal action to address identified contamination, if necessary. In order to provide a removal action recommendation for the Causeway and Dike subsurface geologic and analytical data was required to characterize conditions. The Pre-Design Investigation Report for the Causeway and Dike addressed the following goals:

- Evaluate the presence or absence of contamination in the area of the Causeway and Dike through surface and subsurface soil sampling and analysis, and assess the potential for migration of contaminants to groundwater.
- Evaluate using geophysical (electromagnetic induction [EMI] and ground-penetrating

radar [GPR]) surveys, the location of subsurface features and potential drilling obstructions on the Causeway and Dike.
- Determine the location and elevation of sampling locations with an elevation and location survey.
- Document the removal action alternative evaluation process in an EE/CA.

To characterize soil and groundwater conditions at the Causeway the field investigation included the completion of 15 borings, 10 test pits, four monitoring wells and collection of soil samples. Contamination was detected in the soils along the Causway at concentrations above regulatory criteria for residential DEC and GB PMC. VOCs were detected in soils at boring locations CB-99-01, CB-99-02, CB-9-03, CB-99-04, CB-99-08, CB-99-11, and CB-99-13. Other locations where VOCs exceeded criteria included TP-99-06, TP-99-10, TP-DEP-11, and TP-DEP-12. VOCs and SVOCs were concentrated in the northern one-third area of the Causeway. Most detections occurred generally at shallower depths (1-3 feet bgs), whereas the fuel-related compounds were also be found at deeper depths nearer the water table. PCB exceeded criteria at CB-99-02, TP-99-10, and TP-DEP-11.

The extent of inorganic contamination above criteria was also located in the outer one-third area of the Causeway. These were mostly for the shallow soils and are residential DEC exceedances. The only inorganic that exceeded criteria away from the outer end of the Causeway was detected in CB-99-03, where the GB PMC was exceeded for vanadium at 4-6 feet bgs.

To characterize conditions at the Dike, field investigation included completion of 18 borings and collection of soil samples. Comparison of soil boring data to residential DEC and GB PMC indicated only one exceedance at DB-99-08 in subsurface soil samples collected from the 18 Dike borings. Hand auger samples HA-99-03, HA-99-07, and HA-99-08 from the facility-side of the Dike contained VOC, SVOC, and inorganic concentrations exceeding residential DEC and GB PMC.

Using the Pre-Design Investigation data for the Causeway and Dike, an EE/CA was prepared presenting the recommended removal action alternative. The selected remedy for the Causeway was construction of an erosion control cover system. This remedy included: (1) construction of the erosion control cover system preventing receptors from direct exposure to contaminated soil at concentrations in excess of the CTDEP DEC and (2) removal of soil preventing leaching of contaminants in excess of the CTDEP RSR PMC (GB area) or 10-times the Groundwater Protection Criteria.

1.6.9 Phase III Remedial Investigation

In 1998, URSGWC prepared a work plan and implemented the Phase III RI. Activities included the review of historic file information, site-wide subsurface soil sampling and analysis, direct-push groundwater sampling, monitoring well installation, two rounds of monitoring well groundwater sampling, completion of a tidal study to assess impacts on groundwater levels, surface water and sediment sampling and analysis, and aquatic biota sampling and analysis.

The goals of the Phase III RI were to:

- Define site physical features and characteristics;
- Determine the physical, chemical, and toxicological characteristics of the wastes;
- Determine the nature and extent of source areas;
- Evaluate fate and transport pathways; and
- Characterize risk to current and future exposed human/biotic population.

The following paragraphs present a summary of the Phase III RI activities:

- A total of 192 soil borings were advanced to evaluate unsaturated soil quality and to identify potential groundwater contamination source areas.
- Three monitoring well borings (WC2-2D, WC2-3D, and WC5-1D) were advanced to the bedrock and rock core samples were collected to delineate the depth to bedrock and develop information regarding the competence of the bedrock.
- A total of 39 direct-push borings were advanced in four areas (Areas 2, 3, 5, and 6) on the facility. The direct-push study was used for VOC groundwater screening and placement of permanent monitoring wells, assessment of contaminant impacts to the intertidal mudflat areas, groundwater screening of a potential chromium hot-spot at Building B-3, and groundwater screening of on-site and off-site VOC contamination impacting Area 3.
- A total of 24 monitoring wells were installed in the overburden soils in order to determine groundwater elevations and to better define groundwater conditions at SAEP in areas where wells were not present. The 24 monitoring wells (8 shallow wells, 10 intermediate wells, and 6 deep wells) were installed at 15 locations in five areas (Areas 1, 2, 3, 5, and 6) at the facility.
- During the remedial investigation, two rounds of monitoring well groundwater sampling were conducted in June/July 1999 and November 1999 using low flow rate purging and sampling techniques. Samples were collected in order to determine the current constituents, concentration, extent, and migration of the contaminants in the groundwater at the facility. The groundwater sampling program consisted of newly and previously installed monitoring wells throughout the facility in Areas 1, 2, 3, 4, 5, and 6 of the site.
- A tidal study consisting of long-term groundwater level monitoring of 34 monitoring wells and three surface water stilling wells was conducted from July 20 through August 18, 1999. The purpose of the tidal study was to compare tidal variations in surface waters versus groundwater, determine the tidal fluctuations in groundwater, and determine mean groundwater elevations and hydraulic gradients at SAEP.
- Slug tests were conducted from November 15 through November 19, 1999 on a total of 34 monitoring wells. Slug tests are an in-situ measurement of hydraulic conductivity (K) in the area surrounding the well screen and filter pack.
- A seepage meter pilot study was conducted from July 15 through August 2, 1999 in the Tidal Flats. The pilot study included the installation of four seepage meters in the Tidal Flats to measure seepage flux between the groundwater and mudflats. The pilot study was intended to evaluate the feasibility of the use of seepage meters to provide data regarding the groundwater and surface water interaction in the tidal mudflats.
- The locations and elevations of all exploration sites were surveyed by a registered land surveyor.
- Twenty surface water samples were collected within one hour before or after high and low tides from seven stations in the Marine Basin, downgradient of Outfall 008, and the reference location in the wetland drainage across the Housatonic River.

- A total of 24 surface sediment samples (0-6 inches below surface) were collected to better delineate and assess areas that may be impacted by site-related constituents. The samples were collected from the Tidal Flats, the upstream reference area, the Marine Basin, and the OF-008 Drainage Ditch. In addition, subsurface sediment (6-24 inches below surface) samples were collected within the Tidal Flats area for an indication of vertical extent of constituents.
- Ribbed mussel tissue was collected from the Marine Basin area in order to obtain input data for the assessment of potential risk to the human receptors from ingestion of shellfish tissue.
- Fish sampling was performed by seining the shoreline, hook and line, and trapping to qualitatively characterize fish populations occurring in the Tidal Flats adjacent to the Site, Marine Basin, and the Nell's Island reference location.

The Draft RI Report was submitted in September 2000 and contained the following:

- The chemical characterization had delineated the horizontal and vertical extent of groundwater contamination at the site;
- Significant groundwater contamination is present at SAEP. The chlorinated solvent and metals contaminant plumes were delineated on-site in the unconsolidated aquifer;
- VOC contamination in the West Parking Lot was due to an off-site source;
- The impacts to the Tidal Flats had not been completely defined; however, monitoring wells installed on the Causeway did not contain elevated concentrations of chlorinated solvents or metals;
- Hydrodynamic studies indicated that the low permeability of the silt may be limiting potential impacts to the intertidal mud flats;
- The chemical characterization identified three VOC hot-spot areas;
- The groundwater velocity and contaminant transport calculations were conflicting due to the highly variable nature of the hydrostratigraphy.

The Phase III RI Report also included a baseline human health risk assessment and a baseline ecological risk assessment. The risk assessments were used to evaluate risks to potential receptor populations at the Site.

In September 2001, a sampling plan was prepared to address comments by the CTDEP and USEPA on the Draft RI Report submitted in September 2000. The comments indentified the need for further groundwater and soil sampling including:

- Delineation of the extent of deep groundwater VOC contamination, primarily beneath Building B-2, requires additional delineation;
- Collection of monitored natural attenuation (MNA) groundwater data to evaluate this remedial alternative in the FS;
- Collection of another round of groundwater sampling for all monitoring wells sampled in the 1999 to evaluate the current VOC concentrations;
- Collection of VOC and MNA data required to characterize potential discharge of VOCs to groundwater beneath the Tidal Flats; and
- Soil sampling and analysis to address areas that indicate contaminant concentrations exceeding CTDEP DEC and GB PMC.

The additional data was collected and incorporated into a revised draft RI report that was submitted to the USEPA and CTDEP in January 2003. The CTDEP reviewed the RI report and in October 2003 provided written comments on all sections of the report with exception of the ecological risk assessment. The following data needs were identified:

- The final RI needs to draw conclusion on releases of contamination on all AOCs;
- A greater systhesis of existing information is needed between plant history and contamination assessment;
- Determine the extent of light non-aqueous phase liquid (LNAPL) in the vicinity of the monitoring wells MW-3, LW-5S and WC-5S;
- Identify the current concentrations of VOCs, inorganics, and MNA parameters in groundwater monitoring wells at the Site and in the Tidal Flats;
- Identify concentrations of VOCs in soil vapor beneath Building B-2 north, Building B-3, B-3A and B-4, Building B-16 and Building B-19. Identify the source of anomalous soil vapor levels in Building B-2 South and Building B-3A courtyard;
- Identify the groundwater flow directions and gradients taking into consideration the effect of tidal cycle influences; and
- Characterize the upper 20 feet of bedrock beneath the portion of the site adjacent to the Tidal Flats, and evaluate bedrock groundwater quality.

2.0 SOILS INVESTIGATION

2.1 METHODS AND ANALYSES

This section describes the methods and analyses used during the soil investigation portions of the SAEP RI. The soil investigations involved soil boring completion, surface and subsurface soil sample collection and analysis, and geophysical surveys. Soil samples were collected during the Phase I and Phase III RI, during the former B-2 Chromium Plating Facility investigations and during the NCRA field activities. Soils investigations are summarized in Table 2-1.

2.1.1 Remedial Investigation

2.1.1.1 Remedial Investigation 1993. A total of 10 soil borings (BR-1 through BR-10) were completed during the Phase I RI for the purpose of soil sample collection. Soil samples were also collected from monitoring well borings. A total of 28 soil samples were collected for laboratory analysis during the Phase I RI. All borings were drilled using a truck mounted auger rig. Soil samples were collected with a 2-inch outside diameter (OD) stainless steel split spoon and screened with a photo-ionization detector (PID). Samples were analyzed for VOCs, SVOCs, TAL metals and Toxicity Characteristics Leaching Procedure (TCLP) metals (W-C, 1993). Figure 2-1 shows the soil boring locations.

2.1.1.2 Remedial Investigation 1998 - 1999. A total of 192 soil borings were advanced during the investigation. A total of 379 soil samples were collected from the borings. The purpose was to evaluate soil quality at AOCs and to identify potential groundwater contamination. Figure 2-1 shows the soil boring locations.

The soil borings were completed between November 1998 and January 1999 by Connecticut Test Borings of Seymour, CT using a CME 85 truck-mounted drill rig. Borings were advanced by the drill rig using hollow stem augers (HSA) with a 4-1/4 inch minimum inside diameter (ID) to accommodate a 2-inch OD stainless steel split-spoon sampler. In areas that the truck-mounted drill rig could not access, a hand-operated electric coring device was first used to penetrate asphalt and concrete. A jackhammer was then used to advance the split-spoon sampler. Auger borings were completed in accordance with SOPs as described in the Remedial Investigation Work Plan (RIWP) (URSGWC, 1998). Borings were advanced until contact with the water table. The depth of soil borings ranged from approximately 4 feet to 7 feet. If refusal was encountered before reaching the required depth, the boring was re-located approximately 2 feet from the original boring and re-attempted. Additional attempts were made to obtain the remaining sample(s) before drilling operations were terminated at that location.

Soil samples were collected using a 2-inch OD split spoon sampler advanced 24 inches using a 140-pound hammer operating over a drop of approximately 30 inches. Blow counts were recorded and soil samples were classified using the Unified Soil Classification System (USCS). Using this system, soils can be categorized easily into groups based primarily on grain size and soil plasticity. Soils in any particular group exhibit similar characteristics and properties. As each soil sample was collected the geologist recorded a description of the soil and the USCS symbol. In addition, PID readings were recorded as an initial field-screening tool. Boring logs are included in Appendix B.

Soil samples for laboratory analysis were obtained and placed in the appropriate lab-clean glassware using a dedicated sampling spoon. All re-usable sampling equipment was decontaminated between sampling locations by rinsing with Alconox and distilled water. All soil sampling was performed in accordance with the SOP as detailed in the RIWP (URSGWC, 1998). Upon completion of sampling, the sampling jar/vials were placed in coolers with ice and maintained at 4° Celsius. Prior to overnight shipment of the samples to the analytical laboratory, sample labels were completed with a sample identification number, project name/number, date, time, and parameters, then placed on the sample jars/vials. The samples were wrapped with bubble wrap and placed in the coolers with the completed chain of custody and secured with shipping tape and tamper-proof labels. The samples were shipped to Emax Laboratories, Inc. of Torrance, CA.

Soil samples were submitted for laboratory analysis of TCL VOCs via USEPA Method 8260B, SVOCs via USEPA Method 8270, PCBs via USEPA Method 8082, Total Petroleum Hydrocarbons (TPH) via USEPA Method 418.1 modified, Target Analyte List (TAL) Metals via USEPA Method 6010, Cyanide (CN) via USEPA Method 9010B, and Cation Exchange Capacity and TOC. The sample parameters for the soil samples were selected in accordance with the October 1998 RIWP.

In accordance with the RIWP, quality assurance/quality control (QA/QC) samples were collected during the second sampling round. Sixteen soil samples were submitted for analysis as matrix spike/matrix spike duplicate samples. A total of fifteen soil samples were submitted as blind duplicates. In addition, twelve samples were QA splits for the USACE and submitted to Quanterra Environmental Services of West Sacramento, CA. Table 2-2 summarizes the QA/QC samples collected during the soil boring program. Data validation reports for this investigation program are included in Appendix C-1.

2.1.1.3 Remedial Investigation 2002. As a result of CTDEP comments (dated November 21, 2001) on the Draft RI Report, additional soil borings and sample collection for off-site analyses were performed in April 2002. Soil sampling was conducted for the following four scenarios:

1) Soil Re-Sampling – 1998 SVOA Detection Limits > CTDEP Residential DEC and/or GB PMC

CTDEP identified soil analyses for some semivolatile organic constituents that had analytical detection limits above the established RSR criteria. The SAEP geographic information system (GIS) database was reviewed to compare SVOC detection limits to Residential DEC and GB PMC. A review of the SVOC soil detection limits presented in the Draft RI Report indicates that detection limits for select SVOCs are up to one order of magnitude higher than the GB PMC for select SVOA contaminants of concern. The top 10% of those samples with the highest concentrations of detected total SVOCs were re-sampled and sent to the analytical laboratory to be analyzed at detection limits to meet the GB PMC and Residential DEC concentrations. The objective of this approach was to evaluate if the previously sampled soils with the highest SVOC concentrations that were previously below detection limits have concentrations of analytes exceeding either GB PMC or Residential DEC. Table 2-3 lists the soil samples collected for re-analysis.

2) Soil Sampling in Areas Exceeding CTDEP GB PMC

Re-sampling and analysis was requested for soils with mass-analysis VOC and/or SVOC concentrations exceeding the CTDEP GB PMC as reported in the Draft RI Report.

The Revised Draft RI Work Plan (Harding ESE, 2002b) stated that if the re-sampled soils were analyzed for VOCs/SVOCs by SPLP, and the data compared to CTDEP Groundwater Protection Criteria for a GA-classified aquifer (GWPC GA) times (x) 10, as permitted under the RSRs, it is likely that the number of exceedances currently requiring remediation will be significantly reduced (the same approach was used with success for Causeway soils). Review of the SAEP GIS database indicated 33 soil samples from 24 soil borings with sample VOC and/or SVOC mass concentrations exceeding the CTDEP GB PMC. Table 2-3 presents the 33 soil samples collected for re-analysis under this scenario.

3) Soil Sampling – Mass Concentrations Exceed 20x CTDEP GB PMC

CTDEP comments on the Draft Phase III RI Report questioned the rationale for selection of soil samples for SPLP analysis, stating that the RI Report does not establish or present this rationale. The URSGWC RI Work Plan (URSGWC, 1998), Table 3, page 3 of 5, presented the following rationale:

"Samples [soil] for which mass concentrations [i.e., non – SPLP analyses] exceed 20x the appropriate RSR SPLP criteria [i.e., GB PMC] will be analyzed for by SPLP for exceeding parameters (except VOCs and TPH)."

An assessment of the mass concentration soil analytical data contained in the SAEP GIS compared to 20x GB PMC criteria indicated a number of samples, from 187 total borings, were not analyzed for SPLP if the mass concentrations were above 20x the GB PMC. The SAEP GIS database was queried by parameter for the highest 3 exceedances (of 20x the GB PMC) by parameter, and the Revised Draft RI Work Plan (Harding ESE, 2002b) proposed that borings/samples associated with these exceedances be re-sampled (total number of borings is 23; 28 total samples) and analyzed for the associated parameters by SPLP. Table 2-3 presents the soil samples collected for re-analysis under this scenario.

4) AOC-related Soil Investigations

CTDEP comments on the Draft RI Report (January, 2003) requested an evaluation of AOCs against existing explorations be performed to identify any data gaps. Existing explorations were reviewed to evaluate the adequacy for identifying potential sources of contamination from AOCs. The majority of AOCs have been characterized, but additional soil borings and analyses were proposed for a limited number, as identified in the Revised Draft RI Work Plan (Harding ESE, 2002b), and in Table 2-4. A summary of soil borings and analyses for 2002 investigations at AOCs is presented as Table 2-5.

Compiling the recommended soil borings and analyses from Items 1) through 4) above, resulted in the completion of 51 soil borings to a maximum of 7 feet bgs and sample collection for the following analyses:

- 11 samples analyzed for SVOCs;
- 39 samples analyzed for SPLP VOCs;
- 30 samples analyzed for SPLP SVOCs;
- 5 samples analyzed for SPLP PCBs; and
- 23 samples analyzed for SPLP Inorganics.

Soil borings were completed between April 16 and 18, 2002 using a truck-mounted direct-push drill rig. Samples were collected at two-foot intervals at the depths and for the parameters shown in Tables 2-3 and 2-5. Due to physical obstructions encountered during drilling, approximately 12 borings were sampled on May 2 and 3, 2002 by pre-drilling locations with a diamond-tipped core boring rig that drilled through the obstructions and allowed for sample collection. The borings were then sampled with hand-driven tools to the required depths to collect the soil samples for the required parameters. All collected samples were prepared as previously indicated and shipped by overnight courier to CompuChem of Cary, NC for analysis. Data validation reports for this investigation program are included in Appendix C-2.

Samples were collected from the targeted depths using a 1-inch diameter coring tube equipped with single-use liners. When the sampled interval was brought to the surface the liners were opened then the soils were collected for analysis. A geologist logged soil descriptions. This approach for sample collection was different for samples collected for VOC compounds. When samples were collected for VOC analysis, the liners were not opened but were cut at sufficient length to provide adequate soil for analysis. The ends of these sections were immediately capped to prevent volatilization. These undisturbed sections were then sent whole to the off-site laboratory for analysis. Boring logs for the 51 soil borings are included in Appendix B-2.

2.1.2 Former B-2 Chromium Plating Facility Investigations

The chromium-focused soil sampling was conducted by HLA through a contract with AlliedSignal from January 1998 through June 1999. Following preliminary investigations conducted by AJS Environmental Services, Inc. and HLA in June and August 1998, a chromium-focused investigation in the former B-2 Chromium Plating Facility was conducted to delineate the extent of chromium contamination in soil.

In August 1998, soil samples were collected from under the concrete floor of the former B-2 Chromium Plating Facility to determine the presence or absence of Cr(VI) contamination. The samples were collected using a GeoProbe™ by driving a sampling assembly into the subsurface using a hydraulic hammer. Soil samples were collected using a 1-1/4-inch diameter hollow tube equipped with a sampling point and piston stop-screw to prevent soil from entering the tube before the designated interval. The following bullets summarize activities completed during the August 1998 investigations:

- Collection of 12 subsurface soil samples (0-7 feet below ground surface [bgs]) from six boring locations (SP-98-01 through SP-98-06).

- Off-site analysis of these samples for total chromium, Cr(VI), nickel, manganese, cyanide, total organic carbon, TCLP total chromium, and TCLP cyanide.

The locations of the soil samples are shown on Figure 2-2. In addition to the soil samples, two dust samples (CD-98-01 and CD-98-02) were collected from the surface of the concrete floor, and analyzed for total chromium and ferrous iron. These samples were collected from areas where a yellow precipitate was visible on the concrete floor.

In January and February of 1999, subsurface soil samples were collected from 21 locations (SP-99-01 through SP-99-20 and SP-PILOT-01 through SP-PILOT-05) in and around the former B-2 Chromium Plating Facility using a GeoProbe™ and hand-operated tripod rig. The soil samples were collected to characterize the soil quality beneath the former B-2 Chromium Plating Facility. The sample locations are shown on Figure 2-2. Analytical soil samples were generally collected from 0 to 2 and 5 to 7 feet bgs in each boring. Soil samples were analyzed by an off-site laboratory for total chromium, Cr(VI), cyanide and pH. Synthetic precipitate leaching procedure (SPLP) analyses for total chromium were performed on select samples.

Soil borings SP-PILOT-01 through SP-PILOT-05 were installed in early January 1999 to provide subsurface soil analytical data prior to a proposed pilot test for in-situ reduction of Cr(VI). Upon further characterization of the area surrounding the former B-2 Chromium Plating Facility, it was determined that the location of the proposed pilot test area be shifted to the north, where Cr(VI) concentrations in groundwater were higher.

2.1.3 NCRA Investigations

This section describes the methods used and analyses completed during the pre-design activities conducted from August to December 1999 for the OU 2 NCRA and pre-design activities conducted on the Causeway and Dike NCRA from September to October 1999 through a contract with the USACE.

2.1.3.1 Operable Unit 2 NCRA Investigations. The following subsections present discussions of pre-design activities conducted under the OU 2 NCRA. Activities included soil boring installations, and a soil vapor survey.

Subsurface Soil Sampling

Soil borings PZ-99-01 through PZ-99-03 were drilled to collect subsurface soil for chemical analyses, and to provide subsurface soil for bench-scale testing as part of treatability studies. PZ-99-01 was drilled inside the former B-2 Chromium Plating Facility adjacent to the GeoProbe™ groundwater exploration WP-99-33, which contained over 800 mg/L of TCE in groundwater. PZ-99-02 was drilled outside the northern corner of the former B-2 Chromium Plating Facility adjacent to WP-99-15, which contained 950 mg/L of Cr(VI) in groundwater. PZ-99-03 was drilled between Buildings B-2 and B-12 adjacent to WP-99-48, which contained 100 mg/L of 1,1,1-TCA in groundwater. The locations of these soil borings are shown on Figure 2-2.

Subsurface soil samples were collected during the field investigation using split-spoon samplers and HSA drilling. Soil samples were sent to an off-site laboratory for VOCs, SVOCs, metals (including Cr(VI) and cyanide), SPLP, TOC, TPH, and cation exchange capacity. Off-site laboratory grain-size analyses were also completed on some samples. VOC soil samples were preserved in methanol at the time of sample collection and decontamination between borings was performed. At each soil sample location, VOC samples were collected from discrete locations (not composite); all other parameters were sampled from composite and homogenized soils that were collected from the depth of interest.

2.1.3.2 Causeway and Dike. The following subsections describe the pre-design field activities completed as part of the Causeway and Dike NCRA. Activities included the completion of a geophysical survey, soil borings, test pits, and hand auger soil sampling. The field activities were conducted between September 7, 1999 and October 12, 1999. Details of the investigations, and results, are presented in the Final Causeway and Dike Pre-Design Investigation Report (FWENC/HLA, 2000c). In support of the NCRA design for the Causeway, a geotechnical investigation was performed to evaluate the global stability of the proposed Causeway Cover System, as recommended in the Final EE/CA (FWENC/HLA, 2000d). This investigation was performed on the Causeway in the fall of 2000.

Geophysical Survey

EMI and GPR surveys were completed within the Causeway and Dike areas prior to initiation of subsurface exploration. The objective of the geophysical investigation was to detect and characterize potential buried debris, buried riprap, voids, and other potential obstructions and hazards within the survey areas that may have limited access to the subsurface during subsequent investigations.

The geophysical surveys of the Causeway and Dike were conducted to determine suitable locations for subsurface borings. Geophysical data were used to detect and map subsurface obstructions. Boring locations were subsequently adjusted to avoid these obstructions, thereby eliminating the unnecessary time and effort often associated with subsurface refusal. The survey was conducted during a seven-day period from September 7, 1999 through September 15, 1999 using two geophysical methods:

- Time Domain Electromagnetic Induction (TDEMI) using a Geonics EM61 High Sensitivity Metal Detector
- GPR using GSSI SIR-3 Ground Penetrating Radar with 500 megahertz (MHz) and 200 MHz antennas

All accessible areas of the Causeway and Dike were surveyed utilizing each geophysical method. For ease of data collection, the Dike was divided into four separate grids and the Causeway was divided into three separate grids. EM61 data were collected utilizing a Trimble Pathfinder Pro XR/XRS Differential Global Positioning System (DGPS) for navigational control. DGPS positions were collected synchronous with EM61 data, at a rate of 1 sample per second. EM61 data collection lines were approximately parallel and spaced 1 meter apart. Three 500 MHz GPR lines were collected along the entire length of the paved portion of the Dike. The lines were parallel and spaced approximately 1.5 meters apart, running down the middle of the Dike, and along each edge of the Dike. One 200 MHz GPR line was collected along the entire length of the paved portion of the Dike, running down the middle of the Dike. The endpoints of GPR lines along the Dike, as well as waypoints within the GPR lines, were surveyed with the DGPS system to allow for accurate positioning of the GPR data. 500 MHz (partial coverage) and 200 MHz (full coverage) GPR data were acquired along a pre-established, orthogonal grid system on the Causeway, with coverage similar to that displayed for the EM61 survey. However, GPR data were collected at a line spacing of 2 meters. The corners of the GPR data acquisition grids were surveyed with the DGPS system to allow for accurate positioning of the GPR data.

All EM61 data were compiled into a single file containing X (easting) and Y (northing) coordinates in Connecticut State Planar Coordinates (NAD83), and four columns of EM61 response data, annotated Channels 1 through 4 (Top Coil, Bottom Coil, Normalized Differential Channel, and Differential Channel). Data for each channel were interpolated to a regular grid using a minimum curvature gridding algorithm. The data were then displayed as high-resolution color plots. GPR data were also processed and output as color amplitude plots displaying subsurface reflections. Processing included the application of low-pass and high-pass filters, horizontal smoothing, application of gains, and distance normalization. Depth migration was carried out for selected profiles to calculate radar wave velocity. An average velocity was then applied to convert two-way travel time to apparent depth for all profiles.

The EM61 data clearly depicts the location and geometry of surficial and subsurface features including underground pipelines/utilities, and scattered, discrete subsurface debris along the Causeway. Field plots of distance versus two-way travel time were generated for each GPR line collected. These plots were analyzed to isolate diffraction patterns or other characteristic responses associated with subsurface structures. The Causeway is characterized by heavily concentrated subsurface debris, much of which can be interpreted as blocks of reinforced concrete. The locations were selected to obtain sufficient, representative coverage of subsurface conditions on the Causeway, while avoiding subsurface obstructions such as pipelines, debris, or other structures. Borings were placed away from large subsurface debris to avoid potential subsurface refusals.

The EM61 data clearly depicts the location and geometry of surficial and subsurface features including known and unknown underground pipelines/utilities, aboveground pipelines and fences, and a fairly continuous, linear anomaly potentially associated with the preexisting dike. This anomaly is indicated by a blue dot-dash line and is most likely related to a structure such as a fence or retaining wall initially installed prior to or during construction of the original dike. The locations were selected to obtain sufficient, representative coverage of subsurface conditions along the Dike, while avoiding subsurface obstructions such as pipelines, debris, or other structures. The centerline of the Dike generally exhibited quiet EM61 data, indicating no subsurface metallic debris or structures were present. Borings were placed along the centerline of the Dike, away from pipelines and the buried, preexisting structure. This allowed for easy drill-rig access while avoiding subsurface refusals.

Soil Borings

A total of 33 soil borings were completed on the Causeway and Dike in order to assess the extent of soil contamination and install monitoring wells. Boring locations are shown on Figure 2-2. Fifteen soil borings (CB-99-01 though CB-99-15) were completed on the Causeway. The borings were drilled using HSA techniques and using split-spoons for sample collection. Generally, two soil samples were collected from each of the boring locations; one from near the ground surface (either 0-2 or 1-3 feet bgs), and a second from immediately above or at the water table. Samples at selected depths were collected for geologic logging, grain-size analysis, and laboratory chemical analysis, depending on specific data needs. During sample collection Standard Penetration Tests (SPTs) were completed in accordance with the American Society for Testing and Materials (ASTM) method D-1586 to provide information on physical subsurface conditions. Except for two of the borings (CB-99-09 and CB-99-14), all borings were completed to depths generally up to 10 feet bgs. Once completed, all borings (except the two deep borings which had monitoring wells installed in them) were backfilled with the soil from the auger borings. The two deep borings, CB-99-09 and CB-99-14, were intended to extend to bedrock; CB-99-09 was terminated at a depth of 110 feet bgs and did not hit bedrock, but CB-99-09 hit bedrock refusal at a depth of 102 feet bgs. Both deep borings were advanced using HSA drilling techniques to evaluate subsurface geology and to investigate the potential for DNAPLs in those areas of the Causeway. In these two borings, soil samples were collected at 10-foot intervals using the SPT Method.

A total of 18 borings (DB-99-01 through DB-99-18) were completed along the top of the Dike using HSA drilling techniques. Depths of these borings generally extended to 11 feet bgs. Two samples were collected from each of the borings at locations above and/or at the water table using split-spoon samplers. Once completed, all borings were backfilled with the soil from the auger borings and were completed with a surface seal of cement grout.

Radiological monitoring for Health and Safety purposes was performed by both a FWENC health physicist and by CTDEP personnel during subsurface investigations on the Causeway. The CTDEP personnel were on site during the investigation on the Causeway in order to assess elevated radiological readings and to collect soil samples at select locations for radiological analysis only. The CTDEP used a sodium iodide (Gamma Scintillator) instrument and the FWENC physicist used a pancake probe (Gamma-Mueller) to perform screening of the exposed soils in test pits and borings and to screen soil samples so that elevated radiological samples were not collected for the analytical samples. Representative samples of the elevated radiological soils were collected by the CTDEP and sent to their laboratory for characterization.

Analytical soil samples were sent to an off-site laboratory for analysis for VOCs, SVOCs, TAL inorganics, PCBs, inorganics using the SPLP (for samples above the water table), and asbestos. Additionally, some samples were identified for the laboratory to have tentatively identified compounds (TICs) provided. Off-site laboratory grain-size analyses were also completed on select samples. Because the samples collected from the Causeway were potentially radiologically contaminated, all of the Causeway samples were sent to a licensed laboratory capable of receiving and processing such samples.

Construction of the Causeway erosion control cover system proposed in the Final Causeway Design was completed in September 2002. As part of the construction, contaminated soils exceeding CTDEP RSR GB PMC were excavated and disposed of off-site. The erosion control cover system will prevent direct exposure routes for potential human and ecological receptors. Therefore, the Causeway soil analytical results are not presented in this report, but can be found in the Final Causeway PDIR (FWENC/HLA, 2000c).

Test Pits

Eleven test pits (TP-99-06, TP-99-10, TP-99-22 through TP-99-26, TP-DEP-11, TP-DEP-12, TP-DEP-15, and TP-DEP-17) were completed by backhoe at locations on the Causeway that were either suspected, based on the geophysical survey, as not being able to be sampled by HSA methods, or in areas where access by a drill rig was not feasible. Test pit locations are presented on Figure 2-2. At eight of the ten test pit locations, soil samples were collected using stainless steel spatulas and spoons, with samples being collected either at the face of the test pits or from the backhoe bucket whenever sampling was conducted from the deeper portions of the test pits. Two test pits, TP-99-25 and TP-99-26, were excavated to test for magnetic anomalies identified during the geophysical survey, and TP-DEP-15 was terminated after only penetrating a half-foot of soil; analytical samples were not collected from these three test pits. Analytical soil samples were sent to the off-site laboratory for analysis for VOCs, SVOCs, TAL inorganics, PCBs, SPLP inorganics, and asbestos. The analytical results for the Causeway test pit soil samples are presented in the Final Causeway PDIR (FWENC/HLA, 2000c).

Hand Auger Borings

Originally, up to nine hand-auger borings were to be installed on the facility side of the Dike. Based on field observations, collecting soil samples from certain locations would compromise the integrity of the geo-textile liner material. It was decided that the liner should not be penetrated since this could potentially impact the integrity of the Dike. For this reason, and due to other physical constraints (e.g., abutments and paved areas), only five hand-auger borings (HA-99-01 through HA-99-03, HA-99-07, and HA-99-08) were advanced at the locations as shown on Figure 2-2. Analytical soil samples were sent to the off-site laboratory for analysis for VOCs, SVOCs, TAL inorganics, PCBs, and SPLP inorganics.

Geotechnical Investigation

The geotechnical investigation consisted of a first phase that included the installation of five borings with a truck-mounted drill rig on the Causeway, and a second phase that included the installation of nine borings utilizing a barge-mounted drill rig around the perimeter of the Causeway. The drilling activities were designed to gather specific information about the Causeway, including:

- Geologic information: thickness and extent of fill and native material, including physical descriptions; and
- Field testing: shear strength testing and standard penetration testing (relative density) to evaluate the soils physical properties.

3.0 HYDROGEOLOGIC INVESTIGATION

This section describes the groundwater investigations conducted during the RI and NCRA activities. Groundwater investigations are summarized in Table 3-1.

3.1 METHODS AND ANALYSES

The groundwater investigations involved the following activities:

- Direct-push groundwater investigations;
- Monitoring well and piezometer installations;
- Bedrock investigation;
- Groundwater analysis;
- Tidal study;
- Slug testing; and
- Location and elevation survey.

3.1.1 Direct-Push Groundwater Investigations

3.1.1.1 URSGWC Direct-Push Groundwater Investigations. A total of 39 direct-push borings were advanced by URSGWC during the Phase III RI for initial characterization of groundwater. The direct-push study was used for VOC groundwater screening and placement of permanent monitoring wells (DP5-5 through DP5-9 and DP6-1 through DP6-4), assessment of potential contaminant impacts to the tidal flat areas (DP2-1 through DP2-4), groundwater screening of a potential chromium hot-spot at Building 3 (DP5-1 through DP5-11) and groundwater screening of on-site and off-site VOC contamination impacting the area surrounding the West Parking Lot (DP3-1 through DP3-13). Figure 3-1 depicts the locations of all direct-push borings.

The direct-push groundwater sampling program was completed as two separate events. The first direct-push sampling event consisted of advancement of 30 direct-push borings from May 3, to June 2, 1999. The direct-push borings advanced during the first event were installed by Vironex Environmental Services of Newark, DE using a GeoProbe® vehicle-mounted hydraulic hammer. The second event was conducted from November 1 through November 5, 1999 and consisted of the advancement of nine additional direct-push borings in the central portion of the Site and in the area surrounding the West Parking Lot. Direct-push borings installed during the second direct-push groundwater sampling event were advanced by Pine & Swallow Associates, Inc. of Groton, MA using a VibraDrill® all-terrain drilling machine. The direct-push borings were advanced to a maximum depth of the 60 feet, with groundwater samples collected at 10-foot intervals to the bottom of the exploration.

During the first sampling round, a hydraulic hammer advanced a 1-inch diameter direct-push point through the unconsolidated materials to the saturated zone via percussion. The direct-push point used was a blank casing with telescoping screened point, which allowed for vertical profile sampling from the same boring. Once the desired sampling interval was reached, the telescoping section of the direct-push point was lifted exposing the screened portion of the point allowing groundwater to enter the screen. At each sampling level, five well volumes of groundwater were purged prior to sampling using a peristaltic pump. Samples for laboratory analysis were obtained in the appropriate lab-cleaned glassware using the peristaltic pump and dedicated tubing. The sampling was performed using a low flow rate in order to minimize volatilization of possible contaminants by the peristaltic pump. Studies evaluating various sampling methods (peristaltic pump, stainless and Teflon bailers, a VOC trap sampler and double-valve sampler) indicate that peristaltic pump results were 7% to 12% lower than the control VOC concentrations (Baerg et al., 1992). All re-usable sampling equipment was decontaminated between sampling locations by rinsing with methanol and distilled water.

The second round of direct-push groundwater sampling was performed using microwells. Microwells were used during the second event because of the greater penetration power of this technology. The equipment change from the first round was made in order to minimize the number of times refusal was met, which occurred frequently during the first event. Microwells were installed with a high frequency vibratory hammer mounted on a VibraDrill® all-terrain drilling machine. Microwells consisted of 1.3-inch OD steam-cleaned steel pipe whose leading end was fitted with a drive point. Screens, manufactured from the same material, consist of a double row of longitudinal slots 0.015-inch wide. Sequential sampling was performed by driving the well screen to a pre-determined depth and collecting a sample. A section of riser pipe was then connected, the well driven to the next sampling interval, and a subsequent sample collected. At each sampling level, at least three well volumes were removed with a peristaltic pump by pumping from the top of the water. The tubing was then lowered to the screened interval and another three volumes purged before sampling. Samples for VOC analysis were obtained in lab-cleaned 40 milliliter vials with septum screw caps using new polyethylene tubing dedicated to each well and sampling interval. A short segment of silicon rubber tubing that is part of the peristaltic pump head was replaced with new tubing between each sampling location. All re-usable sampling equipment was decontaminated between sampling locations by rinsing with methanol and distilled water. The direct-push groundwater sampling procedures are outlined in the RIWP (URSGWC, 1998). Direct-push boring logs are included in Appendix D.

Groundwater samples collected from the area along the Dike (DP2-1 through DP2-4), to detect contaminants that could potentially impact the tidal flats, were submitted for laboratory analysis of TCL VOCs via USEPA Method 8260B, SVOCs via USEPA Method 8270, PCBs via USEPA Method 8082, TAL Metals via USEPA Method 6010, and CN via USEPA Method 9010B. The sample parameters for the four samples were selected in accordance with the USACE contract modification letter dated April 12, 1999.

During both direct-push groundwater sampling events, a mobile laboratory was used on-site to analyze the VOC samples. A gas chromatograph was utilized to analyze the groundwater samples. In addition, samples collected from the Building B-3 area (DP5-1 through DP-10) were analyzed in the field using field screening kits from the Hach Company for total chromium, Cr(VI), and ferrous iron. For QA/QC purposes, split samples for the USACE were collected from 10% of the groundwater samples and submitted to Quanterra Environmental Services of West Sacramento, California for analysis.

3.1.1.2 Harding ESE Direct-push Groundwater Investigations. The following subsections present summaries of the investigations conducted by Harding ESE from August 1998 through June 1999 through a contract with AlliedSignal. These investigations included Cr(VI) and VOC groundwater sampling with a GeoProbe™, monitoring well installation and sampling, and VOC groundwater sampling with a cone penetrometer. The chlorinated VOC investigation was initiated as a result of the very high concentrations (>100 mg/L) of VOCs detected in GeoProbe™ explorations WP-99-08 and WP-99-09 during the chromium-focused investigations. The Army requested that Harding ESE delineate the extent of VOC contamination in groundwater emanating from the former B-2 Chromium Plating Facility. Investigations were conducted in March 1999 with a GeoProbe™, followed by additional investigations using a cone penetrometer in May 1999, and monitoring well installations in September 1999.

GeoProbe™ Investigations – Chromium

In August 1998, groundwater samples were collected by Harding ESE with a GeoProbe™ from under the concrete floor of the former B-2 Chromium Plating Facility to determine the presence or absence of Cr(VI) contamination in this medium.

The GeoProbe™ collects subsurface samples by driving a sampling assembly into the subsurface using a hydraulic hammer. Groundwater samples are collected using a 4-foot screened section of rod housed in an outer sleeve. A disposable point is placed at the tip of the outer sleeve to protect the screen while the assembly is driven to depth. At depth, the point is pushed off the end, and the outer sleeve retracted to expose the screen. Tubing connected to a peristaltic pump is then placed down the rods and into the screen to collect groundwater from the desired depth interval.

The following bullets summarize activities completed during the August 1998 GeoProbe™ investigations:

- Collection of two shallow groundwater samples (7-9 feet bgs) from two of the six borings (WP-98-01 and WP-98-02); and
- Off-site analysis of these groundwater samples for Cr(VI), nickel, cyanide, ferrous iron, iron and sulfate.

Figure 3-1 identifies the locations of the GeoProbe™ explorations completed during this phase of the investigation.

In January and February of 1999, Harding ESE collected 182 groundwater samples from 32 sampling locations (WP-99-01 through WP-99-32) in and around the former B-2 Chromium Plating Facility using a GeoProbe. The "drag" technique was employed for collection of multiple groundwater samples within each boring. This technique involved driving the GeoProbe™ rods to the total depth at the exploration, and pushing off the disposable stainless steel drive point. The 4-foot, 0.010-inch slotted stainless steel screen was then exposed by retracting the outer rods. Approximately 3 to 5 gallons of groundwater was then purged from each sample interval with a peristaltic pump, followed by sample collection. The screen and rods were then retracted to the next higher sample interval, followed by purging and sample collection. This process was repeated until sampling from the required intervals was completed.

Groundwater samples were analyzed on-site by Harding ESE for Cr(VI) and ferrous iron using Hach™ test kits.

Due to the high concentrations of Cr(VI) present in groundwater, many of the samples required dilution with deionized water prior to analysis by the Hach™ test kits. Collection of field measurements of pH, temperature, dissolved oxygen, specific conductance, turbidity, and oxidation-reduction potential (Eh) was also performed for each sample. Thirty groundwater samples were sent for confirmation analysis to an off-site laboratory for Cr(VI), inorganic compounds, CN, sulfate, and alkalinity. In addition, four samples from exploration locations WP-99-08 and WP-99-09 were sent off-site for analysis for VOCs and SVOCs, due to the presence of a solvent-like odor observed during sample purging activities.

GeoProbe™ Investigations -VOCs

In March 1999, Harding ESE initiated a VOC groundwater investigation using a GeoProbe™ and an on-site field laboratory. The objective of this investigation was to delineate the horizontal and vertical extent of VOC groundwater contamination detected in the vicinity of the former B-2 Chromium Plating Facility. A total of 40 GeoProbe™ (WP-99-33 through WP-99-72) borings were advanced throughout the Site. Figure 3-1 depicts the location of VOC GeoProbe™ borings.

Harding ESE collected 126 groundwater samples from the 40 boring locations. The "drag" technique was employed for collection of multiple groundwater samples within each boring. A mobile laboratory was used on-site to analyze the groundwater samples for select VOCs (PCE, TCE, cis-1,2-DCE and trans-1,2-DCE, vinyl chloride, 1,1-DCE, and 1,1,1-TCA). Groundwater samples were analyzed by modified USEPA Method 8021B, using a HP 5890 GC with an electrolytic conductivity detector (ELCD). In addition, ten groundwater samples were submitted for confirmation analysis at an off-site laboratory for VOCs.

A photoionization detector was run in tandem with the ELCD, for the purpose of screening for fuel patterns defined by the BTEX pattern. No BTEX patterns were observed in any of the on-site samples analyzed. No single component BTEX detections were reported in any of the off-site split data results.

Off-site confirmatory water samples were analyzed for VOCs by USEPA Method 8260B at Averill Environmental Laboratories to provide definitive data for confirmation of the on-site laboratory screening data set. Samples were split in the field at the time of sample collection and submitted to both the on-site and off-site laboratories.

Cone Penetrometer Investigations - VOCs

In April and May 1999, Harding ESE conducted additional VOC groundwater investigations using a cone penetrometer to access depths not attainable by the GeoProbe™ equipment. Nineteen total cone penetrometer (CP-99-01 through CP-99-18) explorations were completed for collection of groundwater samples. The cone penetrometer locations are depicted on Figure 3-1.

The piezometric cone penetrometer testing (PCPT) and groundwater sampling was performed by Stratigraphics, Inc. of Glen Ellyn, Illinois, during the period from May 5 through May 26, 1999. Exploration depths ranged from 45 to more than 160 feet below ground surface (ft bgs). During the event, 87 groundwater samples were collected from various depths ranging from the water table to a maximum depth of 158 ft bgs. Groundwater samples were submitted for laboratory analysis of VOCs by USEPA Method 8260B. In addition, 38 dissipation tests were conducted to obtain estimates of in-situ horizontal hydraulic conductivity.

The groundwater sampler used during sample collection is a shielded wellpoint sampler of heavy construction. The shield prevents sampler contamination while penetrating soils above the sampling depth. After shield retraction, groundwater flows under in-situ pressure conditions, through a 20-inch long screen, into the sample barrel. The sampler is retrieved to pour out the sample and for decontamination. Small diameter hand pumps were used with the sampler to purge the sample interval and acquire sufficient volumes of sample.

A pressure transducer was also placed inside the sampler barrel. This allowed the measurement of sample inflow rate. Analysis of inflow data using rising head slug test methods provided a means of estimating soil hydraulic conductivities.

In addition, the data recorded during the PCPT investigations included tip (cone) resistance, sleeve friction, instantaneous pore pressure, and soil electrical conductivity; these data are presented in Appendix D-2.

GeoProbe™ Investigation – 2001

In November 2001, Harding ESE initiated additional RI field activities designed to address comments by the USEPA and the CTDEP on the SAEP Draft RI Report issued in September 2000 (URSGWC, 2000). Part of the fieldwork included the installation of monitoring wells to further delineate the extent of deep VOC groundwater contamination associated with VOC plumes in groundwater below the facility; however, due to logistical situations, two of the originally planned monitoring wells (HESE-01-011I and HESE-01-11D) could not be installed. Therefore, a single GeoProbe™ boring (HESE-01-GP01) that could be placed in a much more timely and less intrusive manner was installed to collect groundwater samples from the same location that the proposed monitoring wells would have been installed. The location of this GeoProbe™ boring is shown in Figure 3-1.

The “drag” technique was employed for the collection of multiple groundwater samples from this boring. The GeoProbe™ tools were advanced without sampling to a refusal depth of 72 feet bgs. As the drill string was pulled back off the bottom, a 4-foot, 0.010-inch slotted stainless steel screen was exposed behind a sacrificial drill point. A water sample was collected from the bottom zone after approximately 5 gallons of water were purged using disposable tubing and a decontaminated foot valve. The screen and rods were then retracted to the next sample interval, followed by purging and sample collection. This process was repeated at 10-foot intervals, and a total of seven groundwater samples were obtained. All samples were collected in pre-preserved 40-ml vials that were then sent to CompuChem Laboratory in Cary, North Carolina for VOC analysis.

3.1.2 Overburden Monitoring Well Installation

3.1.2.1 URSGWC Monitoring Well Installation. URSGWC installed 60 monitoring wells (location IDs begin with “WC-“) and piezometers (location IDs begin with “PZ-“) during the Phase I, II, and III RI field programs from 1992 through 1999. Figure 3-2 depicts the locations of all monitoring wells and piezometers installed by URSGWC, and Table 3-2 presents a summary of all URSGWC monitoring well and piezometer construction details. Installation and development procedures for Phase I and II RI monitoring wells and piezometers are detailed in the Phase I (W-C, 1993) and Phase II (W-C, 1996) RI Reports. The following paragraphs present a summary of installation and development procedures for the Phase III RI monitoring wells.

A total of 24 monitoring wells were installed by URSGWC in 1999 during the Phase III RI field activities in order to better define groundwater conditions at SAEP in areas where wells were not present. The 24 monitoring wells (8 shallow wells, 10 intermediate wells, and 6 deep wells) were installed at 15 locations on the facility. The 24 single-cased monitoring wells were installed in the overburden soils to determine groundwater elevations and to obtain groundwater samples for field and laboratory analyses.

The Phase III RI monitoring well installation program was completed as two separate events. The first well installation event consisted of installation of seventeen monitoring wells from March 22, to April 16, 1999. The monitoring wells in the first event were drilled and installed by Connecticut Test Borings of Seymour, Connecticut, using a CME 85 truck-mounted drill rig. The second event was conducted from October 26, to November 3, 1999 with an additional seven monitoring wells installed during this event. Monitoring wells for the second event were drilled and installed by New England Boring Contractors of Glastonbury, Connecticut using a CME 75 truck-mounted drill rig.

The borings for the monitoring wells were advanced using HSAs. The drill rig, augers, and other drilling tools that were used down the hole were steam cleaned before beginning a new boring and before leaving the Site. During the advancement of the well boring, soil samples were collected continuously from the surface to a depth of ten feet below grade and every five feet to completion depth thereafter. Soil samples were collected using a 2-inch OD split spoon sampler advanced 24 inches using a 140-pound hammer operating freely over a drop of approximately 30 inches. Blow counts were recorded and soil samples were classified visually by a URSGWC drilling inspector using the USCS. In addition, PID readings were recorded as an initial field screening tool. Boring logs for the wells installed during the RI are contained in Appendix E of this report.

The intermediate and deep monitoring wells were advanced using driven casing due to depth and/or running sands. The monitoring wells were installed using a 6-inch inner diameter steel casing that was driven to the maximum advancement depth of the augers. The augers were then placed within the casing and the borehole advanced through the casing. The casing was removed during the installation of the well.

Boreholes for monitoring wells WC2-2D, WC2-3D, and WC5-1D were cored five feet into the bedrock. The cores were collected for analysis and classification. The methodologies of the rock coring are detailed further in Subsection 2.3.2. The interval from which the core was collected was grouted to the top of the bedrock prior to installation of the monitoring well.

Upon completion of the borehole to the desired depth, the monitoring well was installed using 2-inch OD, flush joint, schedule 40 polyvinyl chloride (PVC). All the monitoring wells were installed with 10 feet of 2-inch OD 0.010-inch (No.10) slot PVC screen with a bottom cap at the base and 2-inch PVC riser to the surface. A filter pack of graded sand was tremied down the annulus of the borehole to an elevation of approximately 3 feet above the top of the screen interval. A 3-foot thick bentonite seal was placed above the sand pack. The remaining annular space was tremie grouted with a cement-bentonite grout to within 3 feet of the ground surface. A steel protective casing was cemented into the remaining annular space and the well was finished with a concrete apron. The completed monitoring well was fitted with an internal expandable locking cap and bolted flush mount cover.

After installation of the monitoring well, the wells were developed in order to improve the hydraulic communication between the formation and the monitoring well. The well development process flushes the interface between the aquifer and the well allowing ground water to flow into the well with minimum retardation. The initial development of each well was performed using a surge block to surge the water in and out of the screen. Following surging, a submersible pump was used to remove water from the well. Temperature, pH, specific conductivity, and turbidity were measured once per well volume pumped. The pumping continued until the turbidity was less than 5 nephelometric turbidity units (NTUs) and four consecutive readings were within 10 percent for the other parameters, or five well volumes were removed from the monitoring well, whichever occurred first.

A summary of well construction details for all monitoring wells included in the RI study are presented on Table 3-2. Well construction logs detailing the installed wells are included in Appendix F of this report. Well development logs for the installed monitoring wells are included in Appendix G. The monitoring well installation SOPs are outlined in the RIWP dated October 1998.

3.1.2.2 Monitoring Well and Piezometer Installation - Chromium-focused Investigation. Harding ESE installed 13 piezometers for water level measurement and analytical sampling in the vicinity of the former B-2 Chromium Plating Facility during the chromium-focused investigation in January and February 1999. Locations of these wells/piezometers are presented in Figure 3-2. Piezometer construction details and a summary of sampling and analyses performed are presented in Tables 3-3 and 3-4, respectively. Piezometer construction diagrams are included in Appendix F.

Monitoring wells and piezometers were installed using a GeoProbe, and are constructed of 1-inch ID schedule 40 PVC. All well/piezometer screens are 5 feet in length with 0.010-inch machined slots. Due to the limited ID of the GeoProbe™ drilling rods, a sand pack was not installed around the well screens.

Operable Unit 2 NCRA Investigations

The following subsections present discussions of pre-design activities conducted during the OU2 NCRA. Activities included piezometer and extraction well installation and aquifer testing.

Piezometers and Extraction Well Installations

In August 1999, seven piezometers and one extraction well were installed in the vicinity of the former B-2 Chromium Plating Facility at SAEP. The objectives of these explorations were to; 1) obtain groundwater chemical data, 2) collect groundwater samples for bench-scale testing of Cr(VI) and VOC treatment technologies, and 3) provide a pumping well and water level observation points for aquifer testing of the subsurface hydraulic conditions. These activities are discussed in the following subsections.

Piezometers

A total of seven piezometers were installed from August 8, 1999 through August 20, 1999 at three separate locations (soil borings PZ-99-01 through PZ-99-03) in the vicinity of the former B-2 Chromium Plating Facility. Two sets of three nested piezometers (PZ-99-01 A, B, C and PZ-99-02 A, B, C), consisting of a shallow, mid-depth and deeper piezometer, were set in two borings, with a single shallow piezometer being placed at PZ-99-03. The borings were drilled using traditional HSA techniques, and using split-spoons for sample collection. The piezometers are depicted on Figure 3-2. Construction details of the piezometers are presented in Table 3-5.

The piezometers were constructed of standard Schedule 40 PVC with five-foot lengths of 0.010-inch slotted PVC well screens. All piezometers are one-inch in diameter. The annulus around each piezometer screen was backfilled with silica sand to a minimum of one-foot above the shallow screens and five feet above the mid-level and deeper screens. Bentonite pellet seals were placed between the individual screened sections in the nested sets to isolate these zones.

The piezometers were developed using a peristaltic pump and surging techniques. Each piezometer was pumped and surged to remove at least as much water as was used during the drilling process, and until the field parameters stabilized. Piezometers were completed with flush-mounted protective steel casings cemented into the ground and equipped with locking covers.

Extraction Well

One extraction well, EW-99-01, was installed on August 25, 1999 in order to perform aquifer testing within the former B-2 Chromium Plating Facility. The well was constructed of schedule 40 PVC with 20-feet of 6-inch ID, 0.020-inch slotted well screen extending from 20- to 40-ft bgs. The location of the extraction well is depicted on Figure 3-2.

The extraction well was developed using a submersible pump at variable rates and at different depths, within the screened interval. Surging of the well with the pump was also performed. Development continued until the pump reached a maximum output of approximately 16 gallons per minute (gpm) (measured by a flow meter), which was sustained over 15 minutes of pumping.

Causeway Investigations

Monitoring Wells

Four monitoring wells (MWCD-99-01A, MWCD-99-01B, MWCD-99-02A, and MWCD-99-02B) were installed in pairs at two locations on the Causeway during the Pre-design field activities (see Figure 3-2). The monitoring wells were installed to assess groundwater quality, and to gather hydrologic information in the vicinity of the tidal flats.

Monitoring wells were constructed of schedule 40 PVC with five-foot lengths of 0.020-inch slotted PVC well screen in the shallow wells (MWCD-99-01A and MWCD-99-02A), and 10-foot lengths of well screen installed in the deep wells (MWCD-99-01B and MWCD-99-02B). The annulus around each screen was backfilled with silica sand to one foot above the shallow well screens and three feet above the deep well screens. Typically, a one to three-foot bentonite pellet seal was installed above the sandpack. A bentonite grout slurry seal was tremied from above the bentonite seal to the surface to eliminate the vertical conduit created by the drilling process.

The monitoring wells were developed using a submersible pump and surging techniques. Each well was pumped and surged for at least one hour, or until the discharge water was clear and the field parameters stabilized. Wells were completed with protective steel casings cemented into the ground and equipped with locking covers.

2001-2002 Monitoring Well and Piezometer Installations

The following subsection presents discussions of groundwater investigations conducted during 2001 and 2002. These RI field activities were designed to address comments by the USEPA and the CTDEP on the SAEP Draft RI Report issued in September 2000 and included the following activities:

- Installed eight new monitoring wells to further delineate the extent of deep VOC groundwater contamination;
- Installed eight new monitoring wells to collect data in support of a VOC MNA remedial alternative evaluation; and
- Installed 16 piezometers in the tidal flats to further delineate the extent of the VOC contamination observed in the existing NAE piezometers (see Subsection 3.3.2.3) along the Dike.

<u>Monitoring Well Installations.</u> A total of 16 monitoring wells were installed; eight wells each were installed for both the RI VOC Plume delineation and the MNA VOC Plume program. Earth Exploration, Inc. of Wrentham, MA installed all of the land-based explorations during this portion of the field program between November 7, and December 17, 2001.

Monitoring wells were drilled and installed primarily as proposed in the Revised Draft RI Work Plan (Harding ESE, 2002b). Table 3-6 summarizes some specifics on drilling methods, screened intervals, and materials drilled, and exploration locations are shown on Figure 3-2. In summary, monitoring wells are constructed of 2-inch OD, flush joint, schedule 40 PVC. All wells were installed with 10 feet of 2-inch OD, 0.010-inch slotted PVC screen, capped at the bottom and with a 2-inch PVC riser extending to the surface. Monitoring wells were completed as shown on the monitoring well construction diagrams (Appendix E-2). As stated in the Revised Draft RI Work Plan (Harding ESE, 2002b), reference soil samples were only collected from HESE-01-12D. As a result, boring logs for the installed wells are not as detailed as they would be if reference samples were collected; however, soil boring logs for the wells (based on driller observations, drilling progress, and identification of soils brought to the surface) are included in Appendix E-2.

Piezometer Installations. In order to obtain chemical and hydrologic information of the tidal sediments as well as the deeper outwash deposits, a series of piezometers were placed in the Tidal Flats as shown on Figure 3-2. The objectives of the piezometer installations were to 1) identify potential off-site migration of VOCs to the tidal flats sediments and underlying glacial outwash, and 2) collect data in support of a groundwater MNA remedial alternative component. New Hampshire Boring, Inc. of Londonderry, New Hampshire installed all of the Tidal Flat piezometers using a barge and drill rig during the period between March 19, and May 8, 2002.

A total of 16 piezometers were installed on the Tidal Flats (see Figure 3-2). Generally, the piezometers were drilled and installed as presented in the Revised Draft RI Work Plan (Harding ESE, 2002b). The piezometer construction details are summarized in Table 3-6, and piezometer well diagrams are included in Appendix F-2. Due to extensive geologic characterization of the tidal sediments obtained during the Causeway Design geotechnical investigation, split-spoon reference samples were not collected during boring advancement. As a result, boring logs were not completed for the piezometers, but notes on the casing advancement and number of blows required per foot drilled are included on Table 3-7.

In addition to the piezometers, two borings were drilled in the approximate locations as shown on Figure 3-2. The purpose of these borings was to evaluate the extent of the tidal sediments as they approach the Housatonic River channel. These two borings were drilled with 4-inch diameter flush-jointed casing to approximately 30 feet below MSL. The borings were advanced using standard drive-and-wash drilling techniques, and both were continuously sampled with split-spoons to characterize the tidal sediments at these locations. Boring logs of these two borings are included in Appendix B-3.

All newly installed monitoring wells and piezometers were developed using a submersible pump and surging techniques. The monitoring wells were considered developed when stabilization of field parameters was achieved, or four hours were expended. For the Tidal Flats piezometers, the amount of development time was one hour. The wells and piezometers were generally developed as prescribed in the Revised Draft RI Work Plan (Harding ESE, 2002b). Development records for the wells and piezometers are included in Appendix G-2.

Typically, all wells and piezometers (except PZ-TF-09A and B) were developed after waiting a minimum of two days following installation. Piezometers PZ-TF-09A and -B were originally installed on April 4, and developed on April 17, 2002. At the time of their scheduled sampling on April 30, 2002, these piezometers were found to be damaged beyond use. Subsequently, on May 8, 2002 both piezometers were replaced and sampled in order to be included in the concurrent sampling episode.

3.1.2.3 NAE Piezometer Installations. Twenty piezometers (D-1 through D-20) were installed in the Tidal Flats along the Dike by NAE in early summer 2001. The objective of the piezometers was to delineate potential shallow (0-2 feet bgs) groundwater contamination at the base of the Dike. The piezometers consist of approximately 2 feet of schedule 40, 2-inch ID PVC, with an open end driven to approximately 1.5 feet bgs. The locations of these hand-driven piezometers are shown on Figure 3-2. Sampling of these piezometers identified VOC contamination exceeding 0.1 mg/L at five of the locations. To further delineate the extent of the VOC contamination into and beneath the Tidal Flats, eight additional piezometers were installed in May 2002 adjacent to the existing piezometers as shown on Figure 3-2. Piezometers were installed by hand driving to equivalent depths as existing piezometers.

3.1.3 Bedrock Investigation

3.1.3.1 1999 Bedrock Coring. Three monitoring well borings (WC2-2D, WC2-3D, and WC5-1D) were advanced to the bedrock and bedrock cores were collected as part of the URSGWC RI. The purpose of the bedrock coring was to collect information regarding the depth to bedrock beneath the Site and to collect some limited information regarding the competence of the bedrock. A total of fifteen feet of core were collected from the borings.

The bedrock coring was conducted during the monitoring well installation event from March 22, to April 16, 1999. Soil borings were drilled by Connecticut Test Borings of Seymour, CT using a CME 85 truck-mounted drill rig. After the overburden had been stabilized using casing, an NX-sized core barrel consisting of a double-tube barrel, a diamond-set reaming shell, a non-rotating inner barrel equipped with a core catcher and shoe, and a diamond cutting bit was attached to the drill pipe and lowered to a point just above the bottom of the hole. After the core barrel had been started, the down pressure and rotational speed were increased to obtain optimum penetration rate. The core barrel was advanced five feet at each coring location. After the core was collected, the core barrel was retracted a short distance to break off the core. The core was removed from the borehole and the core barrel was dismantled. The core barrel was placed on a clean platform and opened. The core was removed from the barrel and placed into segmented compartments of the core boxes. The core was marked with the depth, top and bottom of the core, and any other pertinent information.

Rock cores were logged in the field. An experienced geologist examined each rock core sample and prepared a detailed core log. The log included (but was not limited to) characteristics such as mineral content, degree of weathering, and the orientation of layering and fractures (both described and sketched). Boring logs containing the core descriptions are located in Appendix E-1.

3.1.3.2 2004 Bedrock Coring. Two new bedrock explorations, BRW-04-01 and BRW-04-02, were completed at the Site in February-March 2004 to identify the physical quality of the upper 20 feet of bedrock beneath the portion of the site adjacent to the Tidal Flats, and evaluate if potential groundwater within the bedrock contains VOCs.

Drilling was started for BRW-04-01 on Feb-24-2004 and was completed with the installation of a bedrock monitoring well on Mar-4-2004. Overburden drilling was accomplished using 5-inch ID steel casing that was advanced to the top of bedrock using drive-and-wash drilling techniques. Split-spoon sampling was initiated at 50 feet bgs, and conducted continuously in two-foot intervals to a refusal depth of 61.5 feet bgs. Continuous rock cores were then collected from 61.5 feet to a total depth of 82 feet bgs. Rock core drilling was conducted in two steps: 1) coring five feet of rock from 61.5 to 66.5 feet bgs, then drilling out the core hole for installation of permanent 4-inch ID steel casing, cemented with grout (the 4-inch casing could not be advanced beyond a refusal depth of 62.5 feet); and 2) continuation of rock coring to 82 feet bgs. Once the coring was completed, a 2-inch PVC monitoring well was installed within the bedrock boring below the 4-inch casing.

BRW-04-02 was started on Feb-27-2004 and completed with the installation of a bedrock monitoring well on Mar-9-2004. Overburden was drilled using 5-inch ID steel casing that was advanced to the top of bedrock using drive-and-wash drilling techniques. Split-spoon samples were collected continuously from 110 feet bgs to a refusal depth of 126.3 feet bgs. Rock cores were then collected from 128 feet to a total depth of 148 feet bgs. Again, rock core drilling was conducted in two steps: 1) five feet of bedrock was cored from 128 to 133 feet bgs, then the core hole was drilled out to install permanent 4-inch ID steel casing, cemented with grout and 2) continuation of rock coring to 148 feet bgs. Once the coring was completed, a 2-inch ID PVC monitoring well was installed within the bedrock boring below the 4-inch casing.

Rock cores were logged in the field. An experienced geologist examined each rock core sample and prepared a detailed core log. The log included (but was not limited to) characteristics such as mineral content, degree of weathering, and the orientation of layering and fractures (both described and sketched). Boring logs containing the core descriptions are located in Appendix E-1.

3.1.3.3 Seismic Refraction Survey. A seismic refraction survey was conducted from September 28, through October 1, 1999 with the objective of determining the bedrock depth and configuration beneath the survey area. The survey was performed by Northeast Geophysical Services of Bangor, Maine using a Geometrics ES-2401, 24-channel seismograph, with data being obtained along seven seismic lines totaling 5,593 lineal feet. Seismic data were processed and interpreted using the RIMRock Geophysics SIPT-2 (formerly USGS SIPT-2) seismic interpretation program. Seismic survey line locations are presented on Figure 3-3.

In this type of a survey, an interpretation of subsurface densities can be made based on the travel times of sound waves, measured in milliseconds, traveling through and refracting from subsurface layers with contrasting densities. Bedrock travel velocities are the fastest velocities recorded, with lesser velocities being indicative of differing unconsolidated layers above the bedrock. Seismic data are compared to ground truth data, such as borehole information, to calibrate the seismic model for the Site.

Under favorable conditions, seismic refraction results can be fairly precise, within +/- 10 %. Although the SAEP location is a vacant industrial plant, there are still numerous potential “noise” source areas on site including power transformers, fans, maintenance machinery, off-site road traffic, etc. that potentially affected the quality of the seismic data. In fact, one of the survey lines (seismic Line 4) had such poor data quality due to background noise, as well as to a concentration of buried utilities, that an interpretation of this line was difficult, at best. However, results from the rest of the survey produced data of sufficient quality to make interpretations of the depth and configuration of the bedrock surface.

3.1.4 Monitoring Well Sampling

3.1.4.1 URSGWC Monitoring Well Sampling. Two rounds of monitoring well groundwater sampling were conducted in June/July 1999 and November 1999 during the RI using low flow rate purging and sampling techniques. Groundwater samples were collected in order to determine the current constituents, concentration, and the extent and migration of the contaminants in the groundwater at the facility. The groundwater sampling program included newly and previously installed monitoring wells throughout the Site. A list of monitoring wells sampled during the June/July and November 1999 sampling rounds is presented as Table 3-8. Locations of monitoring wells are depicted on Figure 3-2.

During each sampling round, groundwater purging operations and subsequent groundwater sample collection were conducted in accordance with the USEPA Region 1 Low Stress Method (included in Appendix A of the URSGWC RIWP) using adjustable-rate stainless-steel bladder pumps equipped with dedicated Teflon tubing and a flow-through cell. Twenty-four hours prior to purging the monitoring well, a stainless-steel bladder pump, with Teflon-lined polyethylene tubing and safety line was gently lowered into the well casing, to approximately the mid-point of the saturated screen level (target sampling zone). The bladder pump was placed in the well 24 hours prior to purging in order to allow sediments that were agitated during pump installation to settle. Following the 24-hour stabilization period, a water level measurement was recorded using an electronic water level indicator. These measurements were taken cautiously to the extent practicable, in order to cause minimum disturbance to the static water level. After the water level was recorded, groundwater in each monitoring well was purged. During the purging operations, the pump speed was adjusted to achieve minimal stabilized drawdown, to the extent practicable. In case the drawdown could not be stabilized, the pumping rate was reduced to the minimum allowed by the equipment. The groundwater purging was accompanied by the periodic (average of 3 to 6 minutes) measurement of the field indicator parameters including pH, temperature, specific conductivity, dissolved oxygen, turbidity, and oxidation-reduction potential (Eh) using a flow-through cell attached to the Teflon tubing. Once the field parameters were considered to be stabilized within the limits specified in the USEPA's Low Stress Method (USEPA, 1996a), groundwater samples were collected from respective monitoring wells directly from the Teflon tubing into sampling vials/jars. The groundwater purging/sample data sheets with field parameter results for the first and second sampling rounds are included in Appendix H.

Upon completion of sampling, the sampling jar/vials were placed in coolers with ice and maintained at 4° Celsius. Prior to overnight shipment of the sample to the analytical laboratory, sample labels were completed with a sample identification number, project name/number, date, time and parameters and then placed on the sample jars/vials. The samples were wrapped with bubble wrap and placed in the coolers with the completed chain of custody and secured with shipping tape and tamper-proof labels. The samples were shipped to Emax Laboratories, Inc. of Torrance, California.

Round One – June/July 1999

The first round of monitoring well groundwater sampling was conducted from June 29, through July 13, 1999. A total of 42 monitoring wells were sampled during the first round (see Table 3-8). All samples were collected in accordance with procedures described above and outlined in the URSGWC RIWP. The samples collected from the monitoring wells were submitted for analysis of TCL VOCs via USEPA Method 8260B, SVOCs via USEPA Method 8270, PCBs via USEPA Method 8082, TCL Metals via USEPA Method 6010, CN via USEPA Method 9010B and chloride. All sample parameters and well selections for the first sampling round were in accordance with the URSGWC Groundwater Sampling Plan Addendum to the RIWP dated December 23, 1998.

In addition, samples collected from 10 monitoring wells (WC5-3S, PZ-9D, WC-10S, WC-12S, WC2-3D, WC2-1D, WC2-2I, WC5-1S, WC2-2D and PZ-13D) were analyzed for MNA parameters. The MNA parameters consist of pH, conductivity, salinity, total and dissolved manganese and iron, TOC, chemical oxygen demand, biological oxygen demand, Eh, sulfate, sulfide, nitrate, nitrite, ammonia, alkalinity as bicarbonate, dissolved oxygen, nitrogen, chlorides, and phosphates. The analytical results for these parameters will be used for feasibility study options and natural attenuation calculations.

In accordance with the RIWP dated October 1998, QA/QC samples were collected during the first sampling round. Four samples (from PZ-4D, MW-4, WC2-1S, and WC-14S) were submitted for analysis as matrix spike/matrix spike duplicate samples. Four samples (from PZ-4D, MW-4, WC2-1S, and WC-14S) were submitted as blind duplicates. In addition, four samples (from PZ-4D, MW-4, WC2-1S, and WC-14S) were QA splits for the USACE and submitted to Quanterra Environmental Services of West Sacramento, California.

Round Two – November 1999

The second round of monitoring well groundwater sampling was conducted from November 8, through November 19, 1999. A total of 51 monitoring wells were sampled during the second round (see Table 3-2). Groundwater samples collected from monitoring wells that were sampled during the first round (June/July 1999) of groundwater sampling were submitted for analysis of TCL VOCs, SVOCs, and TAL Metals. Monitoring wells that were not sampled during the first sampling round, and newly installed monitoring wells, were submitted for analysis of TCL VOCs, SVOCs, PCBs, TAL Metals, and CN. All sample parameters and well selections for the second sampling round were in accordance with the Groundwater Sampling Plan Addendum to the RIWP dated December 23, 1998.

In accordance with the RIWP dated October 1998, QA/QC samples were collected during the second sampling round. Three samples (from WC-20D1, WC2-3S, and WC3-2D) were submitted for analysis as matrix spike/matrix spike duplicate samples. Four samples (from WC-09D3, WC-12D3, WC-16D3, and WC-17D3) were submitted as blind duplicates. In addition, four samples (from WC-20D1, PZ-1D, WC-4S, and WC3-1I) were QA splits for the USACE and submitted to Quanterra Environmental Services of West Sacramento, CA.

3.1.4.2 Monitoring Well Sampling May 2002. As part of RI field activities performed in the Fall of 2001 and Spring of 2002, Harding ESE performed groundwater sampling on existing and newly installed wells and piezometers during the period between April 22 through May 8, 2002. Groundwater sampling was performed on the eight newly installed wells as part of the RI VOC Plume Delineation Program, eight newly installed and 20 existing wells included in the MNA-VOC Program, 16 newly installed tidal flat piezometers, and 37 wells sampled by URSGWC in 1999. Additionally, 12 Tidal Flats piezometers were sampled by NAE, including four existing and eight newly installed piezometers. Table 3-9 lists the explorations and sampling parameters collected as part of this additional RI work effort, and Figure 3-2 shows the location of the wells and piezometers.

A total of 108 wells and piezometers were scheduled for sampling as part of the 2002 groundwater sampling program; however, as of this writing, six of these wells (WC-20D1, WC-20D2, WC-20S, WC-21D1, WC-21D2, and WC-21S) have yet to be sampled. These wells are on private property located around the shopping plaza across Main Street from the SAEP facility (see Figure 3-2). Negotiations are underway to allow access to sample these wells. When permission to sample these wells is received, one additional piezometer on site, PZ-7D, will also be sampled; this location was not sampled due to heavy equipment storage over this piezometer location.

During sampling, groundwater purging and sampling was conducted using low stress (low-flow) purging and sampling methods following the USEPA Region 1 SOP, Revision Number 2, July 30, 1996 procedure. Variations to the USEPA SOP were presented in the Revised Draft RI Work Plan (Harding ESE, 2002b). The groundwater low-flow field data records are included in Appendix H. Groundwater samples were sent for analyses to CompuChem Laboratory of Cary, NC. Data validation reports for this investigation program are included in Appendix C.

3.1.4.3 Monitoring Well Sampling – 1999. Twenty monitoring wells and piezometers were sampled in 1999 by Harding ESE around the former B-2 Chromium Plating Facility to assess the potential for groundwater contamination in the area resulting from historic chromium plating activities. Table 3-4 presents a summary of sampling and analyses performed.

3.1.5 Tidal Study

A tidal study consisting of long-term groundwater level monitoring of 34 monitoring wells and three surface water stilling wells was conducted by URSGWC from July 20 through August 18, 1999. The purpose of the tidal study was to compare tidal variations in surface waters versus those in groundwater, determine the tidal fluctuations in groundwater across the Site, and determine mean groundwater elevations and hydraulic gradients at SAEP. A total of seventeen monitoring wells were monitored for a one-week period. The remaining seventeen monitoring wells and three stilling wells were monitored for a four-week period in order to gauge the full range of changes during a tidal cycle.

As part of the tidal study, three stilling wells for the monitoring of surface water levels and tidal variations in comparison to groundwater levels at SAEP were placed in the Housatonic River, Frash Pond, and the Drainage Channel. The stilling wells were used in order to minimize surface water fluctuations from disturbing the pressure transducer. The groundwater measurements were collected from all wells utilizing In-Situ® Troll® pressure transducers and data loggers. The data loggers were programmed to record a time and pressure head reading once a hour during the monitoring time period.

3.1.6 Permeability Tests and Water Level Measurements

3.1.6.1 Slug Testing. Slug tests were conducted from November 15 through November 19, 1999 on a total of 34 monitoring wells. The monitoring wells tested included each of the newly installed monitoring wells including Harding ESE's monitoring wells installed on the Causeway (MWCD-99-01A, MWCD-99-01B, MWCD-99-02A, and MWCD-02B), as well as six existing monitoring wells (LW-3S, LW-5S, LW-2S, PZ-5D, PZ-9D and WC9-D2) as requested by the

USACE. Slug tests are an in-situ measurement of hydraulic conductivity in the area surrounding the well screen and filter pack. Hydraulic conductivity is a measurement of a porous medium's ability to transmit water with units of distance divided by time (volume per unit time per unit area).

The slug tests were performed using a "rising head" test, which is performed by removing a volume of water or equivalent (a solid slug) from a monitoring well and recording the recovery of the water level within the well. The "rising head" method was selected in order to minimize the tidal fluctuations from influencing the slug test data. Following a manual static water level measurement, an In-Situ® Troll® pressure transducer and data logger was placed within the well. The Troll® data logger was programmed to record data on a logarithmic mode. Next, a dedicated, disposable bailer was lowered into the well below the static water level. The bailer remained at this depth until the water level approached the original static level. To start the test, the data logger was started and the bailer was quickly removed in order to simulate instantaneous change in volume in the well. Time and pressure head measurements were then collected until the water level had recovered to within 90% of static conditions.

3.1.6.2 Permeability Testing and Water Levels. Harding ESE conducted permeability testing in support of the 2002 RI field program. The objective of the permeability testing was to provide supplemental hydrogeologic data for the overburden aquifer. Permeability testing was conducted on May 6 and May 7, 2002 in accordance with the Revised Draft RI Work Plan (Harding ESE, 2002b).

A total of 20 monitoring wells and piezometers were visited for permeability testing at SAEP. All but one of these locations was conducive to permeability testing; PZ-TF-03A could not be tested due to physical access issues.

Permeability testing was conducted with an In-situ® Hermit 1000 data logger equipped with a 20 pounds per square inch (psi) transducer, a water level meter, and a 5-foot long, 1-inch diameter, solid high density polyethylene (HDPE) slug. In-situ rising head permeability tests were performed on all of the tested monitoring wells and piezometers. The general procedure for the permeability testing was as follows:

- Open the well head or piezometer;
- Record the water level and other pertinent well characteristics and features;
- Install a 20 psi transducer approximately 10 feet below the water table;
- Lower the entire slug approximately 3 below the static water table;
- Allow the water level to reach equilibrium or 90% recovery (in most instances recovery occurred in less than 2 minutes);
- Start the Hermit 1000 data logger test then immediately remove slug from well;
- Allow the Hermit 1000 data logger to log data to log cycle 3 (approximately 3 minutes);
- Check the water level; if 90% recovery achieved then end test;
- Decontaminate the transducer, wireline, slug, and water level meter; and
- Secure the wellhead or piezometer.

Displacement data was downloaded from the Hermit 1000 using the manufacturer's software. Displacement data was processed and analyzed using Aqtesolv v.2.50.002. Sensitivity analysis was conducted on four wells spanning four different depth intervals within the overburden aquifer. Comparison of hydraulic conductivities calculated under confined and unconfined condition solutions were within 5 feet per day (ft/day). Aquifer properties were estimated using the straight-line method solutions of Hvorslev (1951) and Bouwer and Rice (1976). Computations and curve fit solutions are presented as Appendix I-2.

Since additional monitoring wells were installed in 2001, a round of synoptic water levels was collected from monitoring wells and piezometers across the Site on May 7, 2002 to provide data for an evaluation of general groundwater flow direction. In general, measurements were taken from wells/piezometers in the tidal flats and closest to the Dike between 3 and 5 PM; low tide on the Housatonic River occurred at approximately 3:57 PM. These measurements are included in Appendix I-2, and results are discussed in Subsection 8.3.2.

3.1.7 Aquifer Testing

Stepped-rate and constant-rate discharge tests were performed by Harding ESE on extraction well EW-99-01 located in the former B-2 Chromium Plating Facility in August and September 1999. The stepped-rate test was performed on August 30, 1999. The objectives of the stepped-rate test were to: 1) assess immediate aquifer response to pumping, 2) identify the approximate specific capacity of the pumping well, and 3) determine an appropriate pumping rate for a longer-term constant rate test. The constant-rate discharge test was initiated on September 2, 1999 and continued for 51 hours. The objectives of the constant-rate test were to determine aquifer hydraulic parameters in the vicinity of the pumping well, including transmissivity (T), K, and specific yield (Sy). Additional constant-rate test objectives were to provide observational data on the approximate zone of capture, evaluate the presence of aquifer boundary conditions, and determine the magnitude of tidal influence on groundwater elevations in the area of the test. The tests also provided information on groundwater quality near the areas of known chromium and VOC impact. The locations of these test well EW-99-01, nearby observation wells, and piezometers are shown on Figure 3-2.

The tests were performed using facility electric power, electronic pressure transducers, and battery powered automated data recorders. The data recorders and transducers were products of In-Situ Inc. of Laramie, Wyoming. Electronic data collection was supplemented by periodic hand-measured water levels. Extraction well EW-99-01 was fitted with a 3.5-inch diameter submersible electric pump (manufactured by Grundfos, Inc.). The pump was placed near the bottom of the screened interval, approximately 40 ft bgs.

Two in-line flowmeters were used to measure discharge. An electronic digital flow meter (Flowmeter 1) measured totalized flow and instantaneous discharge rate. A second analog meter (Flowmeter 2) recorded totalized flow. The meters were placed approximately twenty feet from the extraction well and were read periodically throughout the course of the pumping tests.

Water level data were collected prior to pumping to assess head variations or trends that could influence interpretation of well responses to pumping stress. Transducers were installed in five monitoring wells: PZ-99-08I, MWCR-99-02, PZ-99-01A, PZ-99-01B, and PZ-99-01C. A barometer recorded atmospheric pressure coincident with the electronic water level measurements. Measurements were collected at 60-minute intervals beginning on August 26, 1999 at 19:02 hours and ending on August 30, 1999 at 09:02 hours. Water level measurements are recorded as feet below the top of each well casing rim.

Background measurements were also collected from PZ-16D; a piezometer located approximately 600 feet from the pumping well, and assumed to be beyond any anticipated measurable pumping effect and tidal influence. Measurements were collected at 10-minute intervals from 10:00 hours on August 27, 1999 to 14:00 hours on September 02, 1999, a period of over 5 days.

The stepped rate discharge test was initiated at 10:00 AM on August 30, 1999 at EW-99-01. Three steps of increasing discharge rate were completed. Pumping interval durations were 120, 120, and 140 minutes, with corresponding discharge rates of 5, 10, and 17 gpm. Data were collected electronically from piezometers during the test, but were not considered usable due to the marginal observed drawdown response in observation piezometers. The pressure transducer used in EW-99-01 did not record initial water level decline due to installation deeper than its calibrated depth. Manual measurements, collected using a Slope Indicator water level meter, were used to assess drawdown response in the pumping well.

A constant-rate discharge test was initiated 09:37 hours on August 31, 1999 and ending at 12:37 hours on September 2, 1999, a pumping period of 51 hours (3,060 minutes). A discharge rate of 15 gpm was chosen for the test and held constant throughout the pumping interval.

Electronic measurements of pressure head were collected in eight observation piezometers, extraction well EW-99-01, and the background piezometer (PZ-16D). These were supplemented by manual water level meter data throughout the course of the pumping test. Two additional transducers installed in PZ-99-02B and PZ-99-02C did not function properly due to faulty connections. Electronic data collection continued for several days past the end of the pumping interval to observe general head trends.

3.1.8 Seepage Meter Pilot Study

A seepage meter pilot study was conducted by URSGWC from July 15 through August 2, 1999 in the tidal mudflats. The pilot study included the installation of four seepage meters in the tidal mudflats to measure seepage flux between the groundwater and mudflats. In addition, seepage water samples were collected and submitted for laboratory analysis. The pilot study was intended to evaluate the feasibility of the use of seepage meters to provide data regarding the groundwater and surface water interaction in the tidal mudflats.

Seepage flux between the groundwater and the overlying surface water can be measured directly by covering an area of sediment with an open-bottomed container and then measuring the time and change of water volume in a bag connected to the container. The seepage meters were constructed by cutting 15 centimeters (cm) deep, end sections from 55-gallon steel drums. The seepage meters can detect flux rates as low as 0.1 millimeters/day if the bag is left connected for a day or longer. A single tube through the top of the seepage meter serves as both a vent for any gas released from the sediment and as a connection for the measuring bag. Four seepage meters were placed 40 meters apart on a line perpendicular to shore in the direction 42 degrees northeast. To install a meter, the cylinder is turned slowly, open-end down, into the sediment until its top is about 2 cm above the sediment surface. The vent hole is elevated slightly so that any gas from the sediment may freely escape. Then a stopper with tube is inserted into the cylinder hole. To collect a sample, a deflated plastic bag is connected to the tube and left for a specified time or until it contains 0.5 to 3 liters. The volume of seepage water was measured and the water was placed in the appropriate glassware for laboratory analysis. The samples were submitted for laboratory analysis of TCL VOC and TAL metals.

3.1.9 Location and Elevation Survey

3.1.9.1 URSGWC Survey. The locations and elevations of all explorations were surveyed by a registered land surveyor. Horizontal and vertical control points used during the previous RI surveys at SAEP were used during this survey for consistency. The horizontal control is on NAD 83 based upon NGS monuments CONTROL & CONTROL AZ MK. The vertical control is on NGVD 29 based upon USC&GS monument BM F 31 1963. Surveyed points were located by field transverse. Vertical elevation accuracy was established at 0.01-foot and horizontal location accuracy was set at 0.1-foot. A summary of survey data are presented as Appendix J.

3.1.9.2 Harding ESE Survey. The Harding ESE investigations consisted of explorations installed under the various programs conducted at the site and included tasks under the OU 2 investigation, EE/CA activities for the Causeway and Dike, and the additional RI activities. Registered land surveyors surveyed the locations and elevations of all explorations installed during these programs. Horizontal and vertical control points used during the previous RI surveys at SAEP were used during these surveys for consistency. As with the URSGWC surveys, horizontal control was based on the Connecticut State Plane NAD 83, and vertical control on NGVD 29. Meridian Engineering, Inc. of Danvers, MA performed surveying services for Harding ESE for all but the additional RI activities; the Compass Engineering Group, LLC of Rocky Hill, CT, surveyed the additional RI exploration locations. All survey data are presented in Appendix J.

3.1.10 Non-Aqueous Phase Liquids Monitoring

3.1.10.1 LNAPL. During the week of March 22, 2004, a total of 20 LNAPL monitoring wells were installed in three areas on site at which LNAPL has been observed. These areas are: in the vicinity of monitoring well WC-5S; in the vicinity of MW-3, and; in the vicinity of monitoring well LW-5S (Figures 3-4 through 3-6). These wells were installed for the purpose of determining the extent of LNAPL in these areas, and facilitating the evaluation of remedial alternatives.

The area of Buildings B-13, B-15, B-58 and the former Oil House Tank Farm has been used for oil blending, waste reclamation, and waste storage since the 1940s. The 12 monitoring wells placed in the vicinity of monitoring well WC-5S were based on the historical practices, as well as to augment the existing water table monitoring well locations (see Figure 3-4).

LNAPL has been measured in MW-3. This well is near the northeast corner of Building B-16 in the vicinity of former Jet Fuel Tank Farm, where petroleum contaminated soils were excavated during tank removal. Evidence of petroleum has also been found in soil borings in this area (see Figure 3-5).

The CWTP and a former equalization lagoon are located in the vicinity of monitoring well LW-5S. The installation of four LNAPL monitoring wells in this area were based on interpretive groundwater flow direction to the east-northeast and the location of the former equalization lagoon (see Figure 3-6).

Table 3-10 lists the well completion details for the 20 LNAPL wells which were installed at the three areas where LNAPL has been previously detected. As noted on Table 3-10, a total of four proposed wells were not installed because contamination was not noted during the drilling of three borings. Additionally, one well could not be installed at a location because of a shallow refusal depth encountered above the water table.

All borings were installed using a GeoProbe® drilling rig. Where installed, wells consisted of 10- or five-ft screen sections of 1-inch ID Schedule 40 PVC pipe with 0.010-inch slots. Well screens were placed to straddle the water table. At each location, Macro-Core samples in five-foot lengths and 1.5-inch diameter were collected which allowed for borehole logging and the collection of soil samples. The soil core from each boring was characterized and logged, monitored with a PID for VOCs, and were observed visually for signs of LNAPL. The LNAPL monitoring well boring logs and construction diagrams are presented in Appendices B-4 and H-3, respectively.

After the soils from the borings were characterized, representative soil samples from all borings (except from LNAP-04-02) were placed into clear drillers' soil jars to evaluate the soil core for the presence on LNAPL. This was accomplished by adding enough potable water to be able to cover the soil in each jar. The jars were then vigorously agitated and allowed to equilibrate for approximately 48 hours prior to noting the results (see Section 8.2.4.1).

All LNAPL monitoring wells were installed by inserting the well screens to the depths indicated on Table 3-10. Screen lengths were 10-feet except for LNAP-04-11, which had a five-foot screen due to refusal at that depth. Backfill for the wells was mostly natural cave material except for the intervals noted on the logs, where well sand was added if the natural backfill did not fully surround the screens. Appropriate riser lengths were added to complete the well construction as noted on the logs. All wells except those installed in the LW-5S vicinity were completed with flush-to-ground road boxes. The four wells in the LW-5S area were completed with above ground protective casings.

Following the installation of the LNAPL monitoring wells, all wells were developed for approximately 20 minute durations. Development consisted of inserting dedicated tubing to the bottom of the wells which were surged as the wells were pumped with a peristaltic pump. Because at least some of these wells were expected to contain free-phase product, monitoring of the development water with monitoring equipment was not conducted in to prevent the instrumentation from becoming contaminated. All wells, however, produced clear development water after varying amounts of water were removed. Good hydraulic connection to the water table aquifer was noted in that each well freely produced water throughout the development period.

3.1.10.2 DNAPL. DNAPL monitoring was conducted on May 17, 2004 at the following monitoring wells: HESE-01-05D, HESE-01-06D, HESE-01-07D, HESE-01-08D, HESE-01-

09D, HESE-01-10D, HESE-01-12D, HESE-01-17D, and HESE-01-18D. Measurements of DNAPL were conducted with an oil-water interface probe. Results are presented in Section 8.2.4.2.

4.0 SOIL VAPOR AND INDOOR AIR INVESTIGATIONS

4.1 OVERVIEW

This section describes the soil vapor and indoor air investigations conducted by Harding ESE at the facility. The soil vapor investigation consisted of collection of 52 subsurface soil vapor samples across the site for analysis of VOCs. The indoor air investigation has consisted of collection of indoor air samples in selected buildings across the Site since September 1999.

Subsection 4.2 describes the objectives of the soil vapor and indoor air investigations, and Subsection 4.3 reviews the methods and analyses used. The results of the soil vapor and indoor air investigations are presented in Subsection 8.3.

4.2 OBJECTIVES

4.2.1 Soil Vapor Investigation

Two soil vapor investigations were conducted at the SAEP site, the first in 1999 and the second in 2004. The objective of the soil vapor investigation conducted in August 1999 was to determine if concentrations of VOC vapors in the subsurface exceed the applicable CTDEP criteria for soil vapor, and therefore pose a risk to indoor air quality in SAEP buildings. The 2004 investigation was completed to install permanent soil vapor monitoring probes for compliance monitoring purposes in accordance with the CTDEP RSRs and to collect samples for analysis around existing vapor points where previous analytical results suggested possible sources of VOCs in soil.

4.2.2 Indoor Air Monitoring

In August of 1999, Harding ESE was contracted to conduct indoor air monitoring for a potential lease area inside Building B-2. Results of Rounds 1 and 2 of monitoring, both conducted in September 1999, indicated the need for additional monitoring on a frequent (i.e., monthly) basis at additional locations around the SAEP facility. Rounds 3 through 35 of indoor air monitoring were conducted between October 1999 and September 2003. The general objective of the indoor air monitoring was to evaluate concentrations of VOCs inside SAEP buildings relative to the CTDEP I/C IATCs, and thus evaluate potential risks to current SAEP workers. Since the spring of 2001, the monitoring locations have been adjusted on a per-round basis to also include unoccupied buildings having the potential for future re-use.

4.3 METHODS AND ANALYSES

4.3.1 1999 Soil Vapor Survey

A soil vapor survey was conducted in August 1999 to determine if concentrations of VOC vapors in the subsurface exceed the applicable CTDEP criteria for soil vapor, and therefore pose a potential risk to indoor air quality in SAEP buildings. The survey consisted of the collection of soil vapor samples from 52 temporary locations (SG-99-01 through SG-99-52), at depths ranging from 1 to 3 feet bgs using a GeoProbe. The locations of the 52 explorations were based on identified areas of groundwater VOC concentrations exceeding I/C VC. Figure 4-1 presents the locations of the soil vapor samples.

Samples were collected at a minimum depth of at least 1 foot below the bottom of the building floors. Soil vapor samples were collected using a low flow volume pump to purge the entire probe assembly. The pump inlet was then sealed off from the atmosphere at the required depth in the probe, and an in-situ soil vapor sample was drawn into a glass bulb using the pump. Samples were analyzed on-site using a Photovac10S50 portable gas chromatograph (GC). Ten % of samples were collected as duplicates and sent to an off-site laboratory for analysis by Method 8260B to provide a comparison to on-site data.

Soil vapor samples were analyzed on-site for 1,1-DCE, TCE, PCE, and vinyl chloride using a Photovac 10S50 GC. Reporting limits for TCE and PCE were obtained from the USEPA Region I method "Ambient Air Grab Sample Analysis for Volatile Organic Compounds". Reporting limits for vinyl chloride and 1,1-DCE were extrapolated from this method.

4.3.2 2004 Soil Vapor Monitoring

The 2004 soil vapor monitoring program was conducted to provide permanent soil vapor monitoring probes for compliance monitoring purposes in accordance with the CTDEP RSRs. Additionally the program was designed to install soil vapor monitoring points and collect samples for analysis around existing vapor points SG-99-51 (southern portion of Building B-2) and SG-99-22 (courtyard area between Buildings B-3 and B-4) to potentially identify possible sources of VOCs in soil. At these two areas, existing data suggested isolated spills or releases of solvents may have occurred. To this end, monitoring points were installed beneath the northern portion of Buildings B-2, and beneath Buildings B-65, B-3, B-3A and B-4 (see Figure 4-1). Also, as part of this objective, there was the need to determine the levels of VOCs in vapor beneath buildings and along the property boundary where previous soil vapor data were insufficient to determine if compliance monitoring is necessary.

Therefore, permanent soil vapor points were installed beneath Buildings B-19, B-6, and in the parking lot along the east side of Building B-19. As part of this program, vapor points were originally scheduled to be installed beneath Building B-16, but these were placed outside of the building due to asbestos contamination being present within the building.

During the period of March 22 through March 26, 2004, a total of 68 permanent soil vapor probes were installed beneath the floors of buildings and pavement using direct-push techniques (Geoprobe^{TM}). Figure 4-1 presents the locations of the installed soil vapor probes.

A diamond concrete boring drill was used to core through the concrete floor slabs at each of the sampling locations. The core size allowed for the collection of soil cores, the installation of soil gas probes, and for the placement of protective casings and cement to hold them in place. In paved areas, an area of pavement large enough to allow soil gas probe insertion and placement of protective casings and cement surface pads was cut.

At compliance locations, soil cores were collected continuously to a minimum of 30 inches beneath the ground surface. For soil gas probe installations at Buildings B-2 south and Building

B-3A courtyard, soil cores were collected continuously from the surface to 2 feet below the water table. This was accomplished by using a 5-foot Macro-Core sampler that collected a 1.5-inch diameter soil core. At all soil gas locations, soil cores were characterized, logged and screened with a PID. Soil vapor boring logs are presented in Appendix B-5.

Soil vapor probes were installed using a 1.25-inch rod, so that the bottom of the probes were 30-to-31-inches beneath the surface of the floor or pavement. Soil vapor monitoring probes consist of 0.5-inch diameter, woven stainless steel approximately 21 inches in length. The lower end of the probes were screwed into an expendable point and the upper end was fitted with a barbed fitting for connection of 0.25-inch OD/0.17-inch ID low-density polyethylene tubing that can extend above the floor surface for sampling. The annular space around and generally the first few inches above, each sampling probe was filled with 60- to 100-mesh glass beads to create a permeable layer. The annulus above the glass beads for each probe was then filled with a sand and granular bentonite mixture to seal the annulus. Each probe was finished with a flush-mount protective casing installed level with the surface of the surrounding floor and securely cemented in place.

The grout was allowed to set for at least 48 hours, then, prior to sampling of the vapor points, the integrity of the surface seal at each location was tested, as outlined in the Sampling and Analysis Plan (U.S. Army Assistant Chief of Staff for Installation Management [ACSIM], 2004). Surface seal testing was necessary to evaluate the effectiveness of the seal for satisfactory sample representation. Based on the integrity testing, it was determined that the bentonite seal at the road box, polyethylene tubing/silastic tubing connection was not always effective. Therefore, an additional amount of surface sealant consisting of expanding and anchoring cement was placed over the top of the hydrated bentonite. After a minimum twenty four hour cure time, integrity testing confirmed that all vapor points were successfully sealed. Surface seal integrity testing verification is noted at the bottom of all soil vapor boring logs.

Soil vapor samples were analyzed for 1,1,1-TCA, 1,1-DCE, PCE, TCE, cis-1,2-DCE, trans-1,2-DCE, and vinyl chloride using USEPA Method TO-15. Air Toxics Limited (ATL) in Folsom, California completed all soil vapor sample analyses.

4.3.3 Indoor Air Monitoring

Indoor air monitoring activities were performed under two separate work plans:

- Indoor Air Monitoring Plan, Occupied Building Areas, Stratford Army Engine Plant, February 4, 2000 (FWENC/HLA)
- Final Indoor Air Monitoring Work Plan, Stratford Army Engine Plant, March 8, 2002 (Harding ESE)

Both of the work plans were submitted to CTDEP for review and comment before conducting sampling activities. In general, the following procedures were conducted for each round of air monitoring:

- an average of 11 samples per round were collected in evacuated 6-liter Summa canisters using fixed-orifice flow controllers;

- at least one background (outside air) monitoring location was sampled;
- samples were collected over an approximate period of 8 hours;
- a trip blank (consisting of an evacuated Summa canister) accompanied each shipment of sample containers (i.e., one trip blank per round);
- one to two field duplicate samples were collected per monitoring round;
- meteorological data from the Bridgeport, CT Igor Sikorsky Memorial Airport (across the street from SAEP) was recorded for the date of each sampling event;
- samples were analyzed by USEPA Method TO-15 for the VOCs, 1,1,1-TCA, 1,1-DCE, PCE, TCE, and vinyl chloride (in some cases samples were also analyzed for cis-1,2-DCE and trans-1,2-DCE); and
- a technical memorandum was produced for each of the monitoring events, summarizing the air monitoring data, sampling period meteorological data, and presenting a comparison of the data to CTDEP I/C IATCs.

Table 4-1 presents a summary of the locations monitored, including building identification, during each of the 35 rounds, and indoor air monitoring locations are presented on Figure 4-2. Separate technical memoranda were prepared for each of the 35 sampling rounds, presenting the air monitoring data, sampling period meteorological data, and comparison of the data to CTDEP I/C IATCs.

5.0 SEDIMENT, SURFACE WATER, AND BIOTA INVESTIGATIONS

5.1 OVERVIEW

5.1.1 Sediment

The sediment investigations involved collection of surface sediment samples throughout the Tidal Flats, Outfall 008/Marine Basin Area and a reference location to better delineate and assess areas that may be impacted by site-related constituents.

In October 1992, as part of the Phase I RI, sediment samples were collected at 13 locations. Details on the Phase I investigation are available in the Phase I RI Report (W-C, 1993).

In December 1994, as part of the Phase II RI, sediment samples from 0 to 6 inches and 24 to 48 inches below surface were collected at 28 locations along six transects in the Tidal Flats and five other locations (three in the Housatonic River and one each in the Marine Basin southeast of the SAEP and Outfall 008 [in the drainage channel]) and analyzed for TCL PAHs, PP Metals, PCBs, TOC, grain size distribution, and percent moisture. In addition, surficial sediment grab samples for benthic macroinvertebrate community analyses and toxicity testing were collected from ten locations in the Tidal Flats, one location each in the drainage channel (Outfall 008) and in the Marine Basin, and at three background sample locations in the Housatonic River. Details on the Phase II investigation are available in the Phase II RI Report (W-C, 1996).

Subsection 5.2 describes the objectives of the 1999 sediment investigation, and Subsection 5.3 reviews the methods and analyses used. Subsection 8.4 describes the results of the analyses. The results of the sediment investigations are discussed in further detail and incorporated into the HHBRA and BERA in Sections 11.0 and 12.0, respectively.

5.1.2 Surface Water

The surface water investigation involved surface water sampling at several areas, including a reference area to determine relative concentrations of site-related constituents. The investigation permitted comprehensive assessment of the surface water chemistry at the Site.

In October 1992, as part of the Phase I RI, surface water samples were collected at 13 locations. Details on the Phase I investigation are available in the Phase I Report (W-C, 1993).

Subsection 6.2 describes the objectives of the 1999 surface water investigation, and Subsection 6.3 reviews the methods and analyses used. Subsection 8.4 describes the results of the analyses. The results of the surface water investigation are discussed in further detail and incorporated into the HHBRA and BERA in Sections 11.0 and 12.0, respectively.

5.1.3 Biota

The biota investigation consisted of three types of biological collections throughout the Tidal Flats, Outfall 008/Marine Basin Area, and reference (background) locations to obtain relevant data for ecologic and human health risk assessments. The investigation permitted comprehensive assessment of the biota system at the Site.

Subsection 5.2.3 describes the objectives of the biota investigation, and Subsection 5.3.3 reviews the methods and analyses used. Subsection 8.6 describes the results of the analyses. The results of the biota investigation are discussed in further detail and incorporated into the HHBRA and BERA in Sections 11.0 and 12.0, respectively.

5.2 OBJECTIVES

5.2.1 Sediment

The objective of the 1999 sediment investigation was to better delineate and assess areas that may be impacted by site-related constituents in the vicinity of the outfalls and adjacent to the Causeway in the Tidal Flats. Additional sediments were collected from stations between Outfall 008 and Marine Basin for delineation purposes.

The specific goals for the aquatic sediment data collection effort were:

- to better delineate the vertical and horizontal extent of constituent distribution of sediments in the vicinity of the outfalls;
- to identify the potential for elevated site-related constituents and potential human and ecological threats in the vicinity of the Causeway;
- to delineate the extent to which site-related constituents have migrated from the Site, and to characterize the potential for human and ecological threats in the 008/Marine Basin area; and,
- to use sediment data collected in the reference (background) area as a basis to characterize regional sediment chemistry, and as a basis for comparison with data collected on and in the vicinity of the Site.

In addition, extensive data collected in 1994 were supplemented with data collected in 1999. Sediments were also collected from the plant infrastructure as part of the investigation. A variety of tests and analyses were performed on the sediments as described in the following sections. The data collected during the sediment investigation were used to develop portions of the HHBRA and BERA in Sections 11.0 and 12.0, respectively.

5.2.2 Surface Water

The objectives of the 1999 surface water investigation were to:

- characterize the surface water chemistry,
- establish reference locations (background) to serve as a basis of comparison,
- determine if concentrations of site-related constituents are higher relative to the reference location data, and
- determine if the constituents are present at concentrations that could pose a threat to human health or ecological receptors based on comparison to water quality criteria.

The data collected during the surface water investigation were used to develop portions of the HHBRA and BERA in Sections 11.0 and 12.0, respectively.

5.2.3 Biota

The objectives of the biota investigation were to:

- characterize fish communities that use the area;
- obtain input data for the assessment of potential risk to wading and shore birds;
- obtain input data for the assessment of potential risk to the human receptors from ingestion of the shellfish tissue; and
- obtain input data for the assessment of potential risk to the human receptors from ingestion of a recreationally and commercially important species of fish

The data collected during the biota investigation are used to develop portions of the HHBRA and BERA in Sections 11.0 and 12.0, respectively.

5.3 METHODS AND ANALYSES

5.3.1 Sediment

The following tests were performed on the 1999 sediment samples in addition to chemical analyses:

- Physiochemical parameters to characterize the indigenous sediments
- Simultaneously extracted metals/acid volatile sulfide (SEM/AVS) to assess the bioavailability of divalent metals
- Toxicity testing to determine the combined toxicity of constituents present in the sediments and compare these results with reference sediments
- Benthic community analyses to determine whether site-related constituents have impacted benthic communities which inhabit sediments in the vicinity of the Site
- Bioaccumulation testing to determine whether site-related constituents can accumulate in prey organisms.

The data from these tests are used in Sections 11.0 and 12.0 to evaluate potential risk to humans and aquatic receptors from exposure to these sediments.

A total of 24 surface sediment samples (0 to 6 inches) were collected by URSGWC during the 1999 investigation to better delineate and assess areas that may be impacted by site-related constituents. The 24 samples consisted of 14 from the Tidal Flats, three from the upstream reference area, four within the Marine Basin, and three along the Outfall 008 Drainage. In addition, eight subsurface sediment (6 to 24 inches) samples were collected within the Tidal Flats area for an indication of vertical extent of constituents. Sediment collection and analytical methods are briefly discussed in this section. Figure 5-1 shows sampling locations for the 1999, and previous, sediment sampling efforts at the SAEP Site. Table 5-1 provides a summary of the scope of work. The September 1998 RIWP (URSGWC, 1998) provides a detailed description of the sediment sampling methods.

Prior to sampling, geologic and bathymetric observations including surface water and sediment depth, grain-size and color, and stratigraphy, were recorded as part of the characterization of the Tidal Flats area and Outfall 008/Marine Basin area at a total of 12 stations, in addition to stations where sediment samples were collected. Sediment samples were collected using a properly decontaminated hand-held coring device from the surface and at depth at the eight stations located in the Tidal Flats adjacent to the outfalls. Surficial sediment samples reflect conditions in the biologically active zone, while the sediments below this depth reflect historical deposition.

For the remainder of the sediment stations adjacent to the Causeway, 008/Marine Basin area, and the reference location, sediments were collected from the biologically active zone using a decontaminated Petite Ponar bottom grab sampler. All sediments were analyzed for chemistry and physicochemical parameters. In addition, some surficial sediments were analyzed for SEM/AVS, toxicity and bioaccumulation testing, and benthic community analyses. Sufficient sediment was collected from each location and homogenized to perform all tests except benthic analyses.

All aquatic sediment samples were analyzed for TCL VOCs, PAHs, PCBs, TAL metals, CN and physiochemical parameters using analytical methods described in the Quality Assurance Project Plan (QAPP) in the September 1998 RIWP.

All infrastructure residue samples were analyzed for TCL VOCs, SVOCs, PCBs, TAL Metals, and CN or as modified by the Infrastructure Sampling Plan Addendum.

Toxicity testing included the chronic 20-day Neanthes arenaceodentata test using endpoints of mortality and growth following ASTM 1611-94 protocol. Tests also included the 28-day Leptocheirus plumulosus with the endpoints of mortality, growth, and reproduction.

Bioaccumulation testing was performed following ASTM 1688-96 protocol for 28-day Nereis virens. Worm tissues were analyzed for PCBs and TAL Metals.

Benthic community analyses were also performed at all stations where sediments were collected.

5.3.2 Surface Water

The 1999 surface water investigation consisted of one activity by URSGWC: surface water sampling. This section briefly describes the methods and analyses used during the surface water sampling. The surface water investigation at the SAEP Site was limited to evaluation of surface water chemistry in the Outfall 008/Marine Basin Area and a reference area located outside the influence of the Site. The September 1998 RIWP (URSGWC, 1998) includes more detailed information regarding the methods and analyses.

Surface water was sampled at high and low tide in the Outfall 008/Marine Basin area at the same locations where sediments were collected to characterize the sediment chemistry. Additional samples were collected from a reference (background) location across the Housatonic River adjacent to Nell’s Island to serve as a basis for comparison. Samples collected at high tide provide input into regional surface water quality; those collected at low tide represent local contributions, including potential site-related inputs. The surface water results were used to determine if concentrations of Site-related constituents are higher relative to the reference location data, and whether they are present at concentrations that could pose a threat to human health or ecological receptors based on comparison to water quality criteria. Figure 5-2 shows sampling locations for the 1999, and previous, surface water sampling effort at the SAEP Site. Table 5-1 provides a summary of the scope of work.

Twenty surface water samples were collected within one hour before or after high and low tides from seven stations in the Marine Basin, downgradient of Outfall 008, and the reference location in the wetland drainage across the Housatonic River. Samples were collected from the top two feet of water column using the direct fill techniques detailed in SOP No. 11 of the September 1998 RIWP (URSGWC, 1998). In situ field measurements of temperature, specific conductance, pH, dissolved oxygen and turbidity or total suspended solids were also collected at each location during each sampling event, using a properly calibrated water quality meter.

All surface water samples were submitted for laboratory analysis of TCL VOCs, PAHs, PCBs, and total and dissolved TAL metals. Analyses for arsenic, mercury, and chromium included trivalent arsenic, methyl mercury, and Cr(VI). Samples in Marine Basin were also analyzed for CN.

5.3.3 Biota

Three types of biological collections were performed as part of the SAEP biota investigation. These are:

- Qualitative fish community surveys performed in the Tidal Flats area and the Outfall 008/Marine Basin area,
- Collection of whole body tissues of Atlantic silverside and mummichog, a prey/forage species, and fillets of striped bass, a recreationally and commercially important species, and
- Finfish and consumable shellfish tissues (ribbed mussel) collected from the Tidal Flats and Marine Basin.

In addition, benthic macroinvertebrates were collected at all sediment stations in the Tidal Flats, Outfall 008 Drainage, Marine Basin and a reference location across the Housatonic River from SAEP. The objectives and methods of the benthic macroinvertebrate analyses are presented in Section 5.0, and results are presented in Subsection 9.6. Figure 5-3 shows sampling locations for the 1999 biota sampling effort at the SAEP. Table 5-1 provides a summary of the scope of work. Detailed procedures used for the collection of biological tissue are provided in SOP No. 8 of the QAPP from the September 1998 RIWP (URSGWC, 1998). The procedures and analyses are summarized in the following sections.

5.3.3.1 Shellfish. Ribbed mussel tissue was collected from the Marine Basin area in order to obtain input data for the assessment of potential risk to the human receptors from ingestion of the shellfish tissue. Sufficient ribbed mussel tissue was collected in triplicate from three locations in the Tidal Flats, Marine Basin and one reference location at Nell's Island. Station locations corresponded with sediment and surface water stations. Incidental field observations and water quality parameters including temperature, pH, dissolved oxygen, alkalinity and hardness were recorded at each station.

Samples were collected manually. Shellfish were frozen and transported whole to the analytical laboratory where they were shucked. Tissues were analyzed for PCBs, TAL metals, percent lipids and percent moisture using methods described in the QAPP (URSGWC, 1998).

5.3.3.2 Finfish. Fish sampling was performed by seining the shoreline, hook and line, and trapping to qualitatively characterize fish populations occurring on the Tidal Flats adjacent to the SAEP Site, Marine Basin, and the Nell's Island reference location. Based on availability of sufficient numbers of individuals, Atlantic silverside and mummichog were selected as the whole fish species for ecological assessment of potential risk to wading and shore birds. Striped bass were selected for the commercial/recreational species in the assessment of potential risk to human receptors. Striped bass samples were filleted in the laboratory to prevent cross contamination.

Approximately 45 grams (g) (wet weight) of tissue were needed to perform the analyses (i.e., 30g for PCBs, 10g for metals, 3g for lipids) for each replicate of each species. QA/QC samples required approximately 100 grams wet weight. Sufficient tissue was collected for the following samples:

Whole Body Prey Fish	**Commercial/Recreational Fish Fillets**
Tidal Flats: 2 samples	Tidal Flats: 4 samples
Marine Basin: 3 samples	Marine Basin: 2 samples
Nell's Island: 5 samples	Nell's Island: 4 samples

All fish and shellfish specimens were individually wrapped in solvent-rinsed aluminum foil, labeled, and packed on ice for shipment to the laboratory. Fish tissues were analyzed for PCBs, TAL metals, percent lipids, and percent moisture using methods described in the QAPP (URSGWC, 1998).

6.0 PHYSICAL CHARACTERIZATION OF THE STUDY AREA

This section contains a discussion of the results from the geological and hydrogeological investigations conducted at SAEP. The combined data are utilized to develop physical conceptual models for the geologic and hydrogeologic systems.

Subsection 6.1 discusses the results and findings of the geologic investigation; Subsection 6.1.3 presents the geologic conceptual model; Subsection 6.2 discusses the results and findings of the hydrogeologic investigation; and Subsection 6.2.5 presents the hydrogeological conceptual model.

6.1 GEOLOGY

This section discusses the results from the geological investigations conducted by URSGWC, Harding ESE, and Sound Environmental Solutions (SES). Sources of data for interpretation of geologic conditions presented here include the Phase II RI Report (W-C, 1996), boring logs and observations from 1999 Phase III RI investigations, 1998 through 1999 Harding ESE investigations (including soil borings, PCPT investigations, and a seismic survey), and C.A. Rich and SES RCRA investigations at the wastewater treatment plant and former lagoon area. The regional geology, local geology, and site-specific geology are presented in the following subsections.

6.1.1 Regional Geology

SAEP is located in a geological province known as the Connecticut Valley Synclinorium (a composite synclinal structure of regional extent composed of lesser folds) that consists primarily of Ordovician age (about 440-500 million years ago) metamorphosed sedimentary and volcanic rocks (Crowley, 1968; Rodgers, 1985). These sedimentary and volcanic rocks were probably deposited on oceanic crust in a continually subsiding eugeosyncline along the eastern margin of what is now North America. Intense deformation and metamorphism reconstituted and uplifted the rocks in Middle to Late Devonian time (360-390 million years ago). The deformation and metamorphism was followed by pre-Cretaceous (69–144 million years ago) subaerial erosion, Cretaceous deposition, and additional erosion subaerial erosion. Late in geologic time, Pleistocene (10,000 years ago to 1.5 million years ago) glacier ice scraped the thin regolith from the bedrock and deposited a variable thickness of till and stratified drift. The resulting landscape has persisted to the present with little change (Crowley, 1968).

Bedrock exposures do not occur in the Site vicinity, but it is inferred that the Lower Ordovician (470-500 million years ago) Oronoque Schist underlies the area (Rodgers, 1985). The Oronoque Schist consists of gray to silver, medium-to fine-grained, well-layered to laminated schists and granofels, and additional small lenses of amphibolite or greenstone (Rodgers, 1985).

The area including the Site is located on the eastern limb of the northeast-trending, doubly-plunging Bridgeport Syncline. The Bridgeport Syncline is the southeastern-most fold of five major folds that comprise the synclinorium. Foliation is folded, but is typically vertical to steeply southeast- and northwest-dipping (Rodgers, 1985). The bedrock of the Site lies immediately west of the staurolite (toward the east) and the kyanite (toward the west) metamorphic isograd boundary (Fritts, 1965). On the basis of geologic correlations with surrounding areas, Flint (1968) estimated bedrock to occur more than 100 ft below MSL. Based on soil borings and seismic profiling conducted during the investigations, bedrock beneath SAEP occurs at elevations ranging from approximately -50 to -175 ft MSL. Flint (1968) indicates that the Housatonic River in the vicinity of Stratford at one time, prior to glaciation and post glacial sea level rise, did not curve past Crimbo and Stratford Points (see Figure 1-1) as it does today. Instead it may have continued southwestward from Devon, along the trend of the belts of bedrock, passing through what is now the site of the Bridgeport Municipal Airport. Geological maps of the area show this area west of the Site as being covered. The glacial sediment cover could have obliterated a former incised bedrock valley by filling it, thereby offering the river a broad fan-like plain, extending from Rivercliff and Cedar Beach on the east to the center of Bridgeport on the west, over which to flow in a generally southerly direction. Shifting channels of the river probably occupied many positions on the fan-like plain while depositing sand and gravel. The present route, passing between Crimbo and Milford Points, may have originated in the latest such shift (Flint, 1968).

Pleistocene glaciation, ending about 10,000 years ago, affected the southern Connecticut area. It seems likely that at least two glacial movements (possibly several more) modified the topography of the area. Glaciers advanced southward as far as Long Island, scouring the underlying bedrock as they advanced. Striations on local bedrock exposures indicate ice advancement directions between S30°E and S10°W (Flint, 1968). Unconformable deposits of till and stratified sand and gravel were deposited as the glaciers retreated. These deposits are of variable thickness. As the glaciers receded further, meltwater from the ice was locally impounded, creating a freshwater lake in what is today Long Island Sound. Glacial streams entering the lake left deltaic sand deposits along coastal Connecticut, including the Site vicinity (Stone et al., 1985). Locally, the deltaic sand and gravel deposits are referred to as the Stratford Outwash sediments (Flint, 1968). The southern part of the outwash, in which a portion of the Site lies, consists of sand with less than 25% gravel, most of it pebble size (Flint, 1968).

Currently, the alluvium/outwash deposits are being dissected and locally affected by recent erosional and depositional processes. The Housatonic River has eroded much of the glacial sands from its channel while depositing recent alluvium and estuarine and intertidal flat sediments. Locally, estuarine sediments also overlie the bedrock. These gray silts, sands, clay, fine organic matter, bits of plants, and broken shells comprise a mud with a maximum thickness to 60 ft (Flint, 1968). Area swamp deposits of muddy peat and peaty mud form crude wedge-shaped bodies that thicken seaward. These sediments are underlain by estuarine muds and alluvium/outwash deposits as you move seaward and landward, respectively (Flint, 1968). Beach sediments are also being locally deposited over the alluvium/outwash sediments along the coastline of the Long Island Sound.

Based on the subsurface data collected during URSGWC's RI Reports of 1993 and 1996, Harding ESE's OU2 PDIR of February 2000 and Causeway and Dike NCRA Report of January 2000, Harding ESE's additional RI investigations conducted in 2001 and 2002, as well as previous investigations (ESE, 1991; Haley and Aldrich, 1987; Metcalf and Eddy, 1987; and Zecco, Inc., 1990), at least five distinct stratigraphic units can be identified in the local area. These units, from shallow to deep, are artificial fill, peat and silt alluvium, estuarine silt, stratified drift and bedrock.

The bedrock geology underlying SAEP is reported to consist of lower Ordovician age metamorphic schists, phyllites, and paragneisses of the Oronoque Member of the Derby Hill Schist (Fritts, 1965). Flint (1968) identifies these rocks as the Orange Formation. Exposures of bedrock do not occur in the SAEP vicinity. Borings made along the Housatonic River (Flint, 1968) and borings completed on-site (ESE, 1991) encountered bedrock at depths ranging from approximately –50 to –150 feet MSL.

Recent deposition of alluvium, estuarine sediments, tidal marsh sediments, beach sediments, and man-placed artificial fill occur along the Housatonic River. The surficial unconsolidated sediments reported near SAEP are Stratford Outwash, tidal marsh peat, and artificial fill (Flint, 1968; U.S. Department of Agriculture, 1981). The presence of estuarine and tidal marsh sediments have also been observed and documented during RI field investigations. Lordship Outwash sediments are found south of the SAEP (Flint, 1968).

Stratified drift, consisting of sorted sediments deposited in streams formed by the meltwater of glaciers, is the main water-bearing hydrogeologic unit in the Site area. The stratified drift forms two depositional facies, known as ice-contact stratified drift and outwash (Flint, 1968). Ice-contact stratified drift is defined as "sediments deposited in streams and other bodies of water against, upon, beneath, or otherwise in immediate contact with melting glacier ice", and is characteristically poorly sorted, and contains irregular beds with large and abrupt changes in grain sizes ranging from clay to boulders (Flint, 1968). Conversely, outwash is defined as "sediments deposited by streams beyond the glacier, and free of any influence of buried ice", and is generally well-sorted sand to fine gravel with lenticular beds (Flint, 1968).

Stratford Outwash is found along the fringes of the Housatonic River and consists of well-sorted sand with small amounts of gravel. Borings completed for the Washington Bridge (Highway 1, about 2 miles north of SAEP) encountered outwash sand underlying more recent alluvium, tidal marsh and swamp peat, and estuarine sediments to an elevation of about 115 feet below MSL (Flint, 1968). This indicates that the outwash had filled the entire valley of the Housatonic, but after extensive erosion by the river and rise in sea level, only remnants of the deposit remain. In some exposures along the Housatonic River north of the Site, the Stratford Outwash is found overlying ice-contact stratified drift (Flint, 1968).

Borings completed near the mouth of the Housatonic River encountered post-glacial estuarine mud unconformably overlying stratified drift at depths as great as 60 feet below MSL (Flint, 1968). The estuarine sediment is described as a gray mud consisting of silt and clay with organic matter. It has a maximum reported thickness of about 60 feet. The tidal marsh and swamp deposits in the area consist of decayed plant matter, peat, and mixtures of silt and clay with high amounts of peat. These deposits may be as thick as 15 feet. The SAEP area is influenced by tidal marsh sediments deposited at, and upstream from, the mouths of tidal inlets (due to rise in sea level since the last glaciation and daily tides) that discharge to the ocean (Flint, 1968). Tidal marsh sediments consist of peat and very organic silt or clay that form wedge-shaped deposits, which become thicker towards the ocean or mouth of the streams.

A large part of the area, including a portion of SAEP, consists of artificial fill deposited over all older deposits (i.e. swamp, estuarine sediments, alluvium/outwash deposits). Artificial fill of the area is said to include "railroad, road, building construction fill and large accumulations of trash" (Flint, 1968). Mapped artificial fill is typically at least 5 feet thick (Flint, 1968).

6.1.2 Site-Specific Geology

The assessment of the site-specific geology at the SAEP is based on the following:

- Supplementary Hydrogeologic Investigation Report (C.A. Rich, 1991);
- The Phase II RI Report (W-C, 1996);
- The OU2 PDIR (FWENC/HLA, 2000a);
- The results of RI activities conducted by URSGWC and Harding ESE from 1998 through 2002; and
- The installation of two bedrock groundwater monitoring wells in 2004.

Based on material type, five major stratigraphic units have been identified underlying the Site (from top down):

- Fill
- Estuarine silt
- Peat
- Alluvium/outwash deposits
- Bedrock

Generalized geologic cross sections have been prepared from previous engineering reports as well as from boring logs, cone penetrometer logs, and seismic data from Phase I, Phase II, and recent investigations conducted by URSGWC and Harding ESE. The locations of the cross-sections are shown on Figure 6-1. Figure 6-2 presents cross sections A-A' and B-B'. Figure 6-3 presents cross sections C-C' and D-D'. Figure 6-4 presents cross sections E-E', F-F', and G-G', and Figure 6-5 shows cross sections TF A-A’ and TF B-B’. These descriptions are generally accurate with minor variations in the stratigraphy seen throughout SAEP. A description of the five major stratigraphic units and their distribution across SAEP is presented below (from top to bottom):

Fill

Fill material is found throughout most of the SAEP, consisting of sand, gravel, and debris associated with buildings, roads, utilities, site grading, and other structures. The fill is generally about 2 to 5 feet thick, but reaches a thickness of about 20 ft adjacent to the Dike as shown on cross section A-A' (Figure 6-2). The fill is thicker near the Dike due to the placement of fill over tidal sediments to extend the shoreline of the facility in the late 1940s.

Fill was used for road construction, site grading, and as foundation material for buildings. Fill material includes fragments of concrete, brick, asphalt, wood, cinders, copper wire, and rebar. Fill, in areas along the shoreline, is reported to also include materials hydraulically dredged from the Housatonic River. The composition of the fill is variable, but most of it is described as a granular material that was placed to promote drainage.

Fill also consists of glacial material deposited on the surface from cut-and-fill operations made during facility development between 1940 and 1960 (C.A. Rich Consultants, 1991). The glacial fill material generally consists of brown medium to fine sand and gravely sand. This glacial fill material is most pronounced on cross section A-A', but can also be observed on cross sections D-D', E-E', and G-G' as the sand and gravel deposit shown overlying the estuarine silt layer.

The fill appears to thin somewhat with distance away from the river. This is probably the result of more extensive historical filling operations along the river in order to bring the shoreline up to grade with the rest of the Site.

Estuarine Silt

A thick silt deposit underlies the fill at the northern edge of the facility along the Housatonic River, as seen on cross-section A-A' (see Figure 6-2). This deposit typically consists of dark organic fine silts with very fine sands, occasionally clayey and having a hydrogen sulfide smell consistent with intertidal mud-flat deposits. This material is an estuarine sediment deposited by the Housatonic River and subsequently topped with fill (both artificial and glacial fill) during enlargement of the facility property.

Cross section A-A' shows the continuity of this silt, which stretches from CP-99-05 eastward to WC2-5I (see Figure 6-2). The thickness of the estuarine silt layer varies from as much as 30 feet beneath the Causeway and adjacent to the Dike at the north end of the facility (see cross section A-A', Figure 6-2) to nonexistent toward the interior of the facility (see cross sections D-D' through G-G', Figures 6-3 and 6-4). Silt deposits exist beneath the fill from the length of the Dike southwestward 150 to 250 ft to approximately the location of cross-section B-B' (see Figures 6-3 and 6-4). This areal extent is consistent with the area of the former Tidal Flats, which were filled in the 1940s to extend the shoreline of the SAEP property further northward and eastward toward the Housatonic River.

Figure 6-5 depicts the interpretive stratigraphy of the tidal mudflats along two transects (TF A-A' and TF B-B') that include the piezometers and soil borings installed during the additional RI activities in 2002. These cross sections agree with the interpretive stratigraphic sequence of the mudflats as presented in the Geotechnical Investigation Summary Report (Harding ESE, 2000). As shown in Figure 6-5, the upper silt deposits are continuously present within the mudflats at depths of around 40 feet to at least as far as the end of the Causeway, and up to 20 feet throughout the remainder of the investigated area. Towards the deeper portion of the Housatonic River, there is an overall thinning of the organic silts, possibly due to the increased flow velocity and scouring effects of the river. The silts, however, are shown in all explorations to be a continuous unit throughout the tidal mudflats and Housatonic River channel. The borings B-TF-01 and B-TF-02 were both sampled at 5-foot intervals, so abrupt geology changes would only be observed if a particular sample interval happened to straddle a geologic boundary. The soil boring log for B-TF-01 suggests a transitional change from silts to the underlying sands and gravels as seen in the sample from the 23 to 25-foot interval (see Appendix E-2 for boring logs). Here, some silt is observed along with mostly sand, and the blow counts for this interval show the sediments being very soft. However, a more abrupt change is seen in the log for B-TF-02 from the 30 to 32-foot interval. The blow counts for this sample increase dramatically and the stratigraphy also appears to quickly change from silts to sands. On the basis of these two borings, there appear to be both transitional and abrupt stratigraphic changes between the estuarine silts and the alluvium/outwash deposits in the Housatonic River channel area. Borings that show transitional stratigraphic changes, however, demonstrate that the change between the organic silts and the lower sands and gravels occurs over relatively short (1 to 2-foot) changes in depth.

Figure 6-5 further shows a decrease in depth to the contact between the silts and underlying coarser deposits. As also shown in the Geotechnical Investigation Summary Report (Harding ESE, 2000), toward the end of the Causeway the depth of the sand and gravel deposit extends to a depth of at least –102 feet MSL, with the underlying top of bedrock surface dipping towards the north in this portion of the site.

Figure 6-6 shows an interpretation of the elevation of the top of the silt deposit and how this changes, especially in the areas around the Dike and Causeway. The interpretive contours represent the contact between the native estuarine silt deposit and the overlying fill materials that were emplaced throughout the history of the facility. Because of the soft nature of the silt, when the denser fill materials were emplaced, the top of the silt was compressed, distorted, and subsided from its natural depositional surface. Subsequently, the more fill that was placed, the more the silt adjusted downward due to overlying pressure to eventually reach a new equilibrium. This contact is not a sharp one, but transitional, with decreasing amounts of fill observed with depth into the silt unit. The contours of this contact show the silt being distorted throughout the length of the Dike with decreasing subsidence occurring along the eastern portion of the Dike. This may be due to a thinning of the silt deposit in this area with the underlying denser alluvial deposits being closer to the ground surface. The silts are shown as being most distorted under the thick fill of the Causeway. Compression of the silt surface under the Causeway may have changed from a natural elevation of around –1.35 ft MSL as seen in PZ-TF-07B to as deep as –12-ft MSL seen in GB-00-08A.

Peat

A deposit of brown and black peat with some organic silt was encountered in the eastern portion of the SAEP, east of Sniffens Lane, in the vicinity of the former lagoons (impoundment area). This peat deposit is discussed in detail in the C.A. Rich Consultants report (C.A. Rich Consultants, 1991). The presence of this peat apparently results in a partially perched water table in the vicinity of the former lagoons. The near circular shape of this deposit indicates it was probably formed by a marsh or swamp. Older USGS quadrangle maps show that this area was formerly a tidal marsh that was subsequently filled. The organic peat layer is continuous across 10.5 acres of this area, and is greater than 14 feet thick near the lagoon and tidal ditch. The average thickness of the peat is 7.2 feet. The peat lies directly beneath fill material in this area at depths ranging from 3 to 20 ft bgs. Portions of the upper peat layer may have been excavated prior to fill placement.

Peat was also encountered at locations WC-8S and WC-2D at depths of 5 ft bgs and 30 ft bgs and thicknesses of 2 ft and 5 ft respectively. A 10-inch thick layer of peat was also encountered at a depth of 64 ft bgs at MWCD-99-01B (see Figure 6-4, cross section F-F'). These three peat deposits are apparently small, isolated pockets, and are not a part of the continuous deposit in the lagoon area. The presence of peat at MWCD-99-01B occurs at the interface of the estuarine silt and the glacial outwash, thereby representing the earliest post-glacial estuarine deposits (see Figure 6-4, cross section F-F').

Alluvium / Outwash Deposits

A thick zone of alluvium/outwash deposits (sand and gravel) underlies the fill, estuarine silt, and peat deposits at the Site, and unconformably overlies the bedrock. The thickness of the alluvium/outwash deposits ranges from less than 10 feet at HESE-01-18D (Figures 6-2, cross section A-A'), to 28 ft near WC2-2D (see cross sections A-A' and G-G'), to greater than 156 feet at CP-99-04 (Figures 6-3, cross section C-C'). The alluvium/outwash deposits consist of sands, silty sands, and gravely sands with occasional boulders, clay stringers, and varved silt. For the conceptual model for the Site, these alluvium/outwash deposits can be grouped into two broad units: 1) Alluvium below the Fill and Estuarine silts, and 2) Glacial Outwash below the alluvium.

The alluvial deposits consist of sand with trace amounts of coarser materials (sand and gravel) with clay, silt, and cobbles. This unit probably represents alluvium/outwash deposits that have been reworked and sorted by the actions of the meandering Housatonic River. The alluvium is thickest beneath the southwestern part of the Site (along Main Street, see Figure 6-3, cross-section C-C'), and thins toward the Housatonic River (see Figures 6-3 and 6-4, cross-sections D-D' through G-G'). The distinguishing feature of these deposits on the referenced cross-sections is the trace gravel, and loosely cemented gravel zones, although a few gravelly zones also appear within the glacial outwash. Cone penetrometer tests encountered a hard interface at some locations within this unit. This resistance may be due to gravelly zones or loosely cemented aggregate, but the zones encountered in some locations appeared to be relatively thin where penetrated, and likely do not affect horizontal groundwater flow to any great extent. If these are cemented zones, then they may provide local resistance to vertical migration of groundwater flow and contaminant transport. The bottom depth of these deposits varies between approximately –14 and –36 feet MSL.

Glacial outwash beneath the alluvial deposits and above the bedrock surface is a fine to medium sand with some silt. The glacial outwash also contains silt/clay seams and fine silty sand lenses (see cross-sections A-A' through G-G', Figures 6-2 through 6-4). The fine silty sand units likely have little or no impact on horizontal groundwater flow across the site. The glacial outwash, which has a micaceous component, is generally stratified and exhibits a fining-down sequence. Micaceous zones are observed in the northwestern area of the Site as seen in PCPT explorations CP-99-01, -02, -03, -05, and –09. These zones are mostly below –50 feet MSL. Cone penetrometer tests within this unit also encountered a hard interface. At these depths, the contact may correspond to the till encountered overlying bedrock. Here the till unit may be relatively impermeable, and effectively decrease the thickness of the water-bearing unit, as well as limit interaction of bedrock and overburden groundwater.

A varved silt unit had been reported to occur only locally in the vicinity of Building B-65 (Haley and Aldrich, 1987). Varved silt layers were encountered at a depth of 60 ft bgs in boring WC-9D2 and are described as 2-millimeter bands of red clay interlayered with orange sand layers. At other depths in WC-9D2 and in WC-20D2 red clay stringers were observed (see Appendix E-1 for boring logs). It is unclear as to whether the red clay stringers represent a portion of a larger varved sequence; however, the presence of varves is consistent with alluvium/outwash sediments that have not been re-worked.

Figure 6-7 depicts the interpreted top of glacial outwash deposits at the site. As with the top of silt/bottom of fill contact as shown in Figure 6-6, these contours may show a depositional response in the top of this unit to the relative amount of fill emplaced above. This is evident under the Causeway where there may be a 10-ft deeper elevation to the top of the unit than to nearby unfilled areas. The rest of the figure, however, may suggest an erosion surface due to the scouring effects of the ancient Housatonic River. The top of the glacial deposit contours dip toward the river and decrease in slope towards the east, and suggest that this is due to natural processes, rather than to the weight of the fill in the Causeway and Dike areas. The top of the glacial outwash contours around the tidal drainage ditch leading to Marine Basin also suggests a natural rather than man-induced form as seen with the top of silt contours. Taken as a whole, the top of glacial outwash contours show there may be a greater chance for groundwater to discharge to the tidal mud flats more easily in the area east of the Causeway. In this portion of the mud flats, there is a decreased thickness of the overlying estuarine silt deposits.

Bedrock

Bedrock beneath SAEP has been identified as black schist with greenstone. Results of the seismic refraction survey, coupled with soil boring information, indicate bedrock depth ranges from about 49 feet to 184 feet bgs beneath SAEP. These depths translate to elevations of approximately –50 to –175 MSL.

Figure 6-8 presents the interpretive bedrock surface elevation. The seismic survey results show that the bedrock surface elevation is highly variable, even over relatively short horizontal distances. As a result, the contours presented in Figure 6-8 should be considered generally accurate within ±10 feet. Bedrock is deepest to the northwest along seismic Line 7 and the west end of seismic Line 2, and shallowest to the southeast along seismic Lines 3 and 5, and the east end of seismic Line 2. Site-wide, results show that the bedrock surface dips to the northwest and to the southwest, with a bedrock high, or “saddle”, located approximately along Sniffens Lane and below Buildings B-3, B-4, and B-5. The deepest confirmation of bedrock beneath SAEP is observed at HESE-01-05D in the northwest area of Building B-2; depth to bedrock at this location is observed at nearly 174 feet bgs, or at elevation –166 feet MSL. If the locally N35°E trending trace of the Mixville Fault (Flint, 1968) is extrapolated to the southwest (Rodgers, 1985), across the Housatonic River, it would pass directly under the SAEP. However, there is no confirmation of the existence of this fault southwest of its mapped terminus.

Flint (1968) indicates that although the bedrock has been strongly glaciated, the presence of till over much of the nearby bedrock surface indicates that the bedrock had been sculpted essentially to its present surface form before glaciation.

Amphibole-mica schist bedrock was reported at locations LW-5D, LW-9D and LW-10D at depths of 163 ft, 151.5 ft, and 103 ft below grade, respectively (C.A. Rich Consultants, 1991). Bedrock was encountered at three locations during the drilling of Phase II monitoring wells WC-9D2, WC-20D2 and WC-21D2; at respective depths of 156 ft, 150 ft and 148 ft below grade (see cross sections F-F’ and G-G’, Figure 6-4). Bedrock was encountered in Phase III monitoring wells WC2-1D, WC2-2D, WC5-1D, and MWCD-99-01B at respective depths of 85, 62, 85, and 102 ft bgs.

Five feet of NX-size rock core was collected at each location from the bedrock encountered during the drilling of monitoring wells WC2-2D, WC2-3D, and WC5-1D. Bedrock was encountered at 62 feet bgs in WC2-2D, and at 85 feet bgs in WC2-3D and WC5-1D. The bedrock at WC2-3D and WC5-1D is described as a gray, strong, thickly-bedded, fine-grained phyllite with moderate fracturing. The bedrock at WC2-2D is described as gray, strong, thickly-bedded, medium-grained schist with minimal fractures. Core logs are included on the boring logs for these wells included in Appendix B. The core logs for these borings indicate very good rock quality from a drilling perspective; core blockage (indicative of poor rock quality) did not occur during the drilling process. Core recoveries were registered at 92% for WC2-3D, 95% for WC5-1D, and greater than 98% for WC2-2D. Rock Quality Designations (RQDs) were recorded at 75%, 85%, and 89%, respectively, all of which indicate good rock quality.

Two new bedrock explorations were completed at the Site in February-March 2004. Each boring was cored approximately 20 feet into rock, and a bedrock monitoring well was installed at each location. These installations and the geologic conditions encountered are described in detail in the following two paragraphs.

BRW-04-01, located on the north side of Building B-19, encountered rock at 61.5 feet bgs. The RQD ranged from 92 to 100 %, indicating an excellent rock quality with little weathering or fracturing. The core from 61.5-feet to approximately 72.5-feet bgs is dark-colored, fine-grained, schist. A minor amount of white quartz and calcite occurs in thin bands. This upper section of core is softer than the lower portion. There are numerous breaks in this section, but only four are interpreted as being natural, with the remainder of breaks interpreted as being caused by the drilling process and the handling of the core. The natural breaks have slight to moderate oxidation staining on the break surfaces. It was noted that after the core was allowed to dry, the section of core between 68 to 71-feet bgs retained some water, possibly as a result of micro-fractures within the rock. From approximately 72.5-feet to 82-feet the core becomes more crystalline, harder and lighter in color, due mostly to more quartz. This lower portion of rock resembles gneiss. One small vuggy section was noted at 79.7-feet in which partially dissolved calcite produced voids within the rock. The voids have slight oxidation on the surfaces.

BRW-04-02, located just east of Building B-64-1, encountered rock at 126.3 feet bgs. Continuous bedrock core was collected from 128-feet to 148-feet bgs. Coring started at 128-feet bgs rather than at 126.3-feet bgs because of very soft rock. There was no perceptible loss of drilling water throughout the drilling of the rock in this boring. RQD calculations for this boring range from a low 76% (in a 2.1-foot section between 130.9 to 133-feet bgs) to a high of 98%. Except for the small section from 130.9 to 133-feet bgs (which is rated as good), the RQDs for this bedrock core are considered “excellent”. The bedrock from this exploration varied little over the 20-foot length of core, and is described as a dark gray to purple gray phyllite that exhibits a characteristic sheen. Although there are numerous breaks in the core, none of these are interpreted as being naturally caused, and the core has a fresh unweathered appearance. These break faces have ubiquitous pyrite ‘blooms’, as well as exhibiting a ‘greasy’ luster. The core has minor white calcite veining throughout the rock, and the bedding/foliation planes are steeply dipping. Small sections of core at 133.8-feet and at 142-feet bgs retain water after all other core sections have dried; this may be due to micro-fractures being present in the rock. Compared to the rock cored in BRW-04-01, the rock core in this boring is softer.

A dense basal till was noted overlying rock at each location. The till was approximately 3 feet thick at BRW-04-01, and approximately 11 feet at BRW-04-02.

The descriptions of the bedrock encountered beneath the facility are consistent with descriptions of the Oronoque Member of the Derby Hill Schist as described by Fritts (1962), Crowley (1968), and Rodgers (1985).

6.1.3 Geologic Conceptual Model

This section presents a conceptual geological model of the Site based on the investigations conducted by URSGWC and Harding ESE in 1998 and 1999, and Harding ESE in 2001 and 2002. The geologic model is described using subsurface cross-sections A-A' through G-G' and the interpretive bedrock surface elevation presented as Figure 6-8. Cross sections A-A', B-B', and C-C' are roughly parallel to the Housatonic River and the remaining cross sections are perpendicular to the river. The conceptual geologic model has been developed with stratigraphy descriptions on a macroscopic scale. These descriptions are typically accurate with minor variations in the stratigraphy seen throughout SAEP. The following paragraphs present the conceptual geologic model (from top to bottom):

Fill material is found throughout most of the SAEP consisting of sand, gravel, and debris associated with buildings, roads, utilities, site grading, and other structures. The fill is generally about 2 to 5 feet thick, but reaches a thickness of about 20 feet adjacent to the Dike as shown on cross section A-A' (Figure 6-2). The fill is thickest near the Dike and beneath the Causeway. The composition of the fill is variable, but most of it is described as a granular material, generally consisting of brown medium to fine sand and gravely sand. This fill material is most pronounced on cross section A-A', but can also be observed on cross sections D-D', E-E', and G-G' as the sand and gravel deposit shown overlying the estuarine silt layer. The fill decreases in thickness with distance away from the Dike.

A thick silt deposit underlies the fill at the northern edge of the facility along the Housatonic River, as seen on cross section A-A' (Figure 6-2). This deposit typically consists of dark organic fine silts with very fine sands. This material is estuarine sediment deposited by the Housatonic River and subsequently topped with fill during enlargement of the facility property. Cross section A-A' shows the continuity of this silt, which stretches from CP-99-05 eastward to WC2-5I at an elevation of approximately -6 to -14 feet MSL. The thickness of the estuarine silt layer varies from as much as 30 feet beneath the Causeway and adjacent to the Dike at the north end of the facility, to nonexistent toward the interior of the facility (see cross sections A-A' and D-D' through G-G'). Silt deposits exist beneath the fill from the length of the Dike southwestward 150 to 250 feet to the approximate location of cross section B-B' (see Figure 6-3 and 6-4).

A deposit of brown and black peat with some organic silt was encountered in the eastern portion of the SAEP, east of Sniffens Lane, in the vicinity of the former lagoons. The peat, covering approximately 10 acres, forms a continuous concave layer, approximately 7 feet thick, and deepens to the west-northwest. The peat lies directly beneath fill material in this area at depths ranging from 3 to 20 ft bgs. Peat material was also encountered at locations MWCD-99-01B, WC-8S and WC-2D. These three peat deposits are apparently small, isolated pockets, and are not a part of the continuous deposit in the former lagoon area.

A thick deposit of alluvium/outwash sediments (sand and gravel) underlies the fill, estuarine silt, and peat deposits at the Site, and unconformably overlies the bedrock. The thickness of the alluvium/outwash sediments ranges from less than 10 feet at HESE-01-18D (Figure 6-2, cross section A-A'), to greater than 156 feet at CP-99-04 (Figure 6-3, cross section C-C'), 28 feet near WC2-2D (see cross sections A-A' and G-G') to about 156 ft thick at CP-99-08 (see cross-section E-E'). The alluvium/outwash deposits consist of sands, silty sands, and gravelly sands with occasional boulders, clay stringers, and varved silt. For the Site conceptual model, these alluvium/outwash deposits can be grouped into two broad units: 1) alluvium below the fill and estuarine silts, and 2) glacial outwash below the alluvium.

Alluvium consists of sand with trace amounts of coarser materials (sand and gravel) with clay, silt, and cobbles. This unit probably represents alluvium/outwash deposits that have been reworked and sorted by the actions of the meandering Housatonic River. Outwash beneath the alluvium, and above the bedrock surface is a fine to medium sand with some silt. The outwash contains silt/clay seams and fine silty sand lenses (see cross-sections A-A' through G-G'). The outwash is generally stratified, and exhibits a fining-down sequence, which has a micaceous component. In recent bedrock explorations (BRW-04-01 and BRW-04-02 in 2004), the outwash was noted transitioning into a dense basal till above bedrock and at thicknesses of 3 and 11 feet, respectively, at each location.

Bedrock beneath SAEP has been identified as black schist with greenstone. Results of the seismic refraction survey, coupled with soil boring information, indicate bedrock depth ranges from about 49 feet to 184 feet bgs beneath SAEP. These depths translate to elevations of approximately –50 to –175 MSL. The seismic survey results show that the bedrock surface elevation is highly variable, even over relatively short horizontal distances. The rock quality, however, as observed in rock cores obtained from borings on site, is indicative of competent to very competent bedrock.

A graphic presentation of the manner in which the dip of the bedrock surface changes beneath the Site is presented in Figure 6-9. This figure shows the four primary compass views of the site and how the bedrock surface changes dip directions in these three-dimensional interpretive perspectives. The views presented in this figure are based on the borings drilled to, or into, bedrock and on the bedrock information that was obtained during the seismic refraction survey. Also depicted on these perspectives are the buildings, the ground surface, and the top of the estuarine silt and top of alluvium/outwash deposits. Because the alluvium/outwash deposit extends to the top of bedrock, the thickness of this deposit has been omitted from the figure to allow for a better view of the top of bedrock surface. This surface dips fairly steeply towards the southeast in the views shown looking west and north, and steeply to the northeast in the view looking east. In the view looking to the south, the bedrock surface has a less steeply dipping northern direction. The information presented in this figure combined with the plan view of the bedrock surface presented in Figure 6-8 provides a thorough understanding of the bedrock surface at the Site.

6.2 HYDROGEOLOGY

6.2.1 Hydrogeologic Investigation Results

This section discusses the results from the hydrogeological investigations conducted by URSGWC and Harding ESE. Sources of data for interpretation of hydrogeologic conditions presented here include: the Phase II RI Report (W-C, 1996); data from URSGWC 1998-1999 RI investigations; 1998 through 2002 Harding ESE investigations, and data from the 1983 through 2000 C.A. Rich and SES investigations. This section discusses the results of the groundwater investigations in terms of local hydrogeology and site-specific hydrogeology. The hydrostratigraphy, hydrodynamics, and aquifer properties of the Site are described in detail.

Tidal Study Results

A tidal study consisting of long-term groundwater level monitoring of 34 monitoring wells and three surface water stilling wells was conducted by URSGWC from July 20 through August 18, 1999. The purpose of the tidal study was to compare tidal variations in surface waters versus groundwater, determine the tidal fluctuations in groundwater across the Site, and determine mean groundwater elevations and hydraulic gradients at SAEP. A total of seventeen monitoring wells were monitored for a one-week period. The remaining seventeen monitoring wells and the three stilling wells were monitored for a four-week period in order to gauge the full range of changes during a tidal cycle.

Tidal fluctuations in surface water bodies produce progressive pressure waves in adjacent aquifers. As these pressure waves propagate inland, groundwater levels and hydraulic gradients at wells within the zone of tidal influence continuously fluctuate, creating a situation where a single set of water level measurements cannot be used to accurately characterize groundwater flow. Where groundwater is tidally affected, the magnitude and direction of the hydraulic gradient fluctuates about the mean or regional hydraulic gradient. The net effect of these fluctuations on groundwater flow can be eliminated by numerically filtering the groundwater level measurements to obtain mean groundwater elevations. As Table 6-1 indicates, the zone of tidal influence extends upgradient into the facility, with the observed tidal influence decreasing with distance from the river. This decrease is not uniform with distance, probably affected in part by the hydraulic conductivity and degree of connection with the river. However, at distances of 125 feet or greater, for the locations monitored, the apparent amplitude due to tidal effects was consistently less than 0.2 feet (also see Subsection 6.4.3).

The filtering method detailed in Serfes, 1991, is used to effectively remove all diurnal and semidiurnal lunar and solar harmonics from 71 consecutive hourly observations. Using moving averages, it yields a filtered mean level for the median time of the 71 hours. First, a sequence of means is computed for 24 observations, starting with observation one for the first mean and observation 48 for the last, yielding a total of 48 means. Second, a similar series of means is computed for 24 of those means, yielding 25 means. Last, the mean of those 25 means is computed, yielding the mean level for each well at hour 36. The resulting set of filtered mean values for each of the wells represents the best estimate of average groundwater conditions with tidal influences removed. These values represent a best estimate for steady-state flow conditions that would be incorporated into a groundwater model as target head values. These data also provide the most accurate basis for estimating horizontal hydraulic gradients. If nested wells or piezometers have the same tidal influence lag times, then the filtered heads also represent the best values with which to calculate a vertical hydraulic gradient.

The filtering method can be expressed mathematically as:

> Let the consecutive hourly water level values be O(1), O(2), O(3), …, O(71):
> The first sequence of means (X_i) is
>
> $$X_i = \sum_{K=0}^{23} \frac{O(K+i)}{24} \quad \text{where } i = 1, 2, 3, \ldots, 48;$$
>
> the second sequence of means (Y_j) is
>
> $$Y_j = \sum_{i=0}^{23} \frac{X_{(i+j)}}{24} \quad \text{where } j = 1, 2, 3, \ldots, 25;$$
>
> then the mean level (M) at hour 36 is
>
> $$M = \sum_{j=1}^{25} \frac{Y_j}{25}$$

The groundwater level monitoring raw data recorded by the data logger were converted into elevations and five 71-hour time periods were chosen for statistical filtering. The five time periods were July 20 through July 23, July 24 through July 26, July 27 through July 29, August 3 through August 5, and August 10 through August 12, 1999. The first two time periods included all 34 monitoring wells and 3 stilling wells. The remaining three time periods included only the wells monitored for four weeks and the three stilling wells. Table 6-2 summarizes the mean groundwater elevations for the five periods.

The mean groundwater elevations from the first two 71-hour periods (July 20 through July 23 and July 24 through July 26) were used to construct groundwater elevation contour maps. Groundwater elevations for each 71-hour period were divided into two categories, shallow (<50 ft bgs) and deep (>50 ft bgs) and groundwater elevation contours were developed for each category. The shallow and deep aquifer groundwater elevation contour maps are included as Figures 6-9 through 6-12. Appendix K-1 contains graphs which depict the data filtering process for each well included in the tidal study.

Water Level Measurements

Table 6-2 presents the groundwater elevations computed from water level measurement data collected by Harding ESE on May 7, 2002. Water level measurements for wells/piezometers closest to the Housatonic River, and therefore more significantly influenced by tidal fluctuations, were generally collected between 3 and 5 PM; low tide on May 7 was approximately 4 PM.

Groundwater elevations presented in Table 6-3 were divided into two categories, shallow (0 to –50 ft MSL) and deep (–50 to –175 ft MSL), and interpreted isocontours are presented on Figures 6-13 and 6-14. Appendix K-2 presents the water level measurement data.

Hydraulic Conductivity Test Results

The slug tests conducted by URSGWC during the 1999 remedial investigation field program provided in situ data to estimate horizontal hydraulic conductivities surrounding the boreholes. The slug test data were analyzed with the aid of the computer software program AQTESOLV™. The Bouwer and Rice solution (1976) was utilized to determine the hydraulic conductivity of the aquifer in the immediate vicinity of the well screen. Bouwer and Rice (1976) developed an empirical relationship describing the water-level response in an unconfined aquifer to instantaneous withdrawal of water from a well:

$$K = \frac{r_c^2 \ln\left(R_e / r_w\right)}{2d} \frac{1}{t} \ln \frac{h_o}{h_t}$$

where: K = horizontal hydraulic conductivity

$r_c =$ radius of the unscreened part of the well where the head is rising

$r_w =$ horizontal distance from well center to undisturbed aquifer

$R_e =$ radial distance over which the difference in head, h_o, is dissipated in the flow system of the aquifer

$d =$ length of the well screen or open section of the well

$h_o =$ head in well at time $t_o = 0$

$h_t =$ head in well at time $t > t_o$

The assumptions that the Bouwer and Rice solution make are the aquifer has an infinite areal extent, is homogenous and of uniform thickness, the potentiometric surface is initially horizontal, the volume of water is discharged from or into the well instantaneously, the aquifer is confined or unconfined, and ambient flow within the aquifer is at steady-state.

The hydraulic conductivity values obtained from the hydraulic conductivity tests ranged from 0.3 ft/day to 273.6 ft/day. Hydraulic conductivity values for shallow screened wells ranged from a low of 0.3 ft/day at LW-3S to a high of 170.1 ft/day at WC5-3S. The wells installed within the intermediate zone ranged from a low of 0.03 ft/day at WC-18D1 to a high of 270.1 ft/day at WC6-1I. The hydraulic conductivity values for the deep monitoring wells ranged from a low of 3.6 ft/day at WC2-1D to a high of 32.5 ft/day at WC3-2D. A summary of hydraulic conductivity test results is presented in Table 6-4. Hydraulic conductivity test raw data and plots are presented in Appendix I.

Table 6-5 provides a listing of wells and piezometers and corresponding Harding ESE 2002 hydraulic conductivity test results. Table 6-5 also includes pertinent well information and a summary of hydraulic conductivity test results. Hydraulic conductivity test data were analyzed by both the Bouwer and Rice (1976) and Hvorslev (1951) methodologies using AQTESOLV^{TM}. The average of Bouwer and Rice and Hvorslev method hydraulic conductivity test results ranged between 7.7 and 317.9 ft/day.

In addition to slug tests in monitoring wells and piezometers, Harding ESE also recorded estimates of hydraulic conductivity derived from cone penetrometer explorations and derived hydraulic conductivity estimates from correlations to grain-size determinations for a few soil samples of aquifer materials. The cone penetrometer results (included on Table 6-5 for completeness only) were determined to be biased low by at least an order of magnitude. These results are not included in the statistical evaluation of the hydraulic conductivity data. However, estimates of hydraulic conductivity derived from the grain-size data using the Hazen method correlation (Foster Wheeler and HLA, 2000) did appear to be consistent with slug test results, and were included in the analysis.

Seepage Meter Pilot Study Results

A seepage meter pilot study was conducted in the tidal mudflats from July 15 through August 2, 1999. The pilot study was intended to evaluate the feasibility of seepage meters to provide meaningful data regarding the groundwater and surface water interaction in the tidal mudflats. Four seepage meters were installed in the tidal flats for the purpose of determining seepage flux and seepage water quality.

The seepage meters were installed on July 15 and were checked 24 hours later to determine if groundwater had seeped into the sampling bags. The sample bags from two seepage meters were lost during the tidal cycle and the remaining bags were filled with surface water. URSGWC attempted to correct the difficulties with the samples by creating a stronger fastening system utilizing clamps as fasteners and double sealing the sample bags to prevent surface water from entering the sample bags. The seepage meters were checked 72 hours later, and similar problems were detected with the loss of sample bags and presence of surface water in the sample bags. This process was repeated several times during the pilot study. The pilot study was discontinued on August 2 after it was determined that the seepage meters would not be effective in this environment in collecting representative seepage water samples. The failure of the seepage meter study is likely due to several reasons. The method employed during the pilot study is more effective where surface waters lie in high to moderately permeable material. The groundwater velocity may be too low to measure with this technique in fine low permeability sediments such as those found in the tidal mudflats. Also, this method was not designed for use in a tidal zone, which likely explains the continued loss of sampling bags. Therefore, the pilot study indicated that the seepage meters would not effectively provide the seepage water data required for the investigation and a full-scale study was not performed. In order to determine the groundwater and surface water interaction in the tidal mudflats, two well clusters were installed on the Causeway to provide the data required to evaluate the interaction. In addition, Harding ESE installed several piezometer clusters within and beneath the tidal flats in April and May 2002; the interpretation of water level measurements from these piezometers is presented in Subsection 6.2.4.

Aquifer Pumping Test Results

An aquifer pumping test was performed at the former B-2 Chromium Plating Facility by Harding ESE in September 1999. The objective of the tests was to determine aquifer hydraulic parameters in the vicinity of the pumping well. The following paragraphs summarize the findings of the aquifer testing.

Aquifer testing revealed that the EW-99-01 pumping zone (20 to 40 ft bgs) has a high specific capacity that indicates a relatively high aquifer transmissivity. The pumping well responded largely in an unconfined manner, with a quick flattening of drawdown that can be associated with either delayed drainage from an unconfined aquifer or leakage through semi-confining overlying soils. Shallow piezometers close to the pumping well exhibited relatively small drawdown responses, indicating that geologic stratification is likely restricting the vertical propagation of pumping stresses. Similarly, small drawdowns at more distant piezometers screened in the same zone as extraction well EW-99-01 (e.g. PZ-99-01B) suggest that delayed drainage or leakage buffered pumping stresses.

Cyclical tidal influence limited the usefulness of data from any of the observation wells, particularly because the drawdown from pumping in all observation wells was so small. Tidal variation, while only about 0.03 feet in magnitude at these wells, was generally equal to 30 % or more of the apparent drawdown. Attempts to resolve trend and tidal influence within the very small drawdowns observed were considered unsatisfactory and drawdown data were not used to estimate the T and K of the aquifer.

The best estimate of aquifer T is derived from the pumping well specific capacity. Specific capacity was approximately one gpm/foot of drawdown at each of the four test pumping rates (three step test and the constant-rate test discharge rates). The T derived empirically from this specific capacity is 1500 gallons per day (gpd)/ft (200 square feet [ft^2]/day). Lower values of T, estimated using pumping well drawdown data and various Theis-based curve-matching and straight-line techniques, appear to underestimate the T, probably as a result of relatively large amounts of recharge to the aquifer from leakage or delayed drainage. If these lower values were correct, it would be theoretically impossible to maintain the constant-rate test pumping rate of 15 gpm for sustained durations greater than 3 days without de-watering the well.

Using the observational evidence that anisotropy (stratification) limits the flow of water from the upper portion of the saturated zone, the aquifer thickness is best approximated at 20 feet, the length of the EW-99-01 well screen. Using this thickness and the estimated T of 200 ft^2/day produces an estimate of the horizontal hydraulic conductivity for the pumping zone of 10 ft/day. This value is consistent with those observed from slug testing results in the alluvium in the vicinity of the pumping well.

The aquifer storage coefficient could not be calculated from the drawdown data given the small drawdown responses at all observation wells. In addition, due to the relatively low stressing of the aquifer, the pumping well zone of influence did not extend outward far enough to encounter any apparent boundaries, e.g., recharge boundaries or impermeable strata. Complete details of the aquifer pumping tests are presented in the Final OU2 PDIR (HLA, 2000).

6.2.2 Local Hydrogeology

Locally, the hydraulic properties of this region are variable because of the differences in texture, composition, and structural features that result from the different provenance and genesis of fluvial and glaciofluvial units. The presence, extent, and continuity of fractures and lenses or layers of stratified drift are difficult to define, particularly in the deeper parts of thick deposits. This geologic heterogeneity is a consideration at SAEP with respect to the spatial variability of the hydraulic properties and the potential impact on contaminant fate and transport. The horizontal hydraulic conductivities range from 0.004 to 65 ft/day, whereas the vertical hydraulic conductivities range from 0.013 to 96 ft/day. Available data, although sparse, indicate that values of effective porosities are commonly in the range of 20 to 35 % (Melvin et al., 1992).

Flow paths from recharge areas of the groundwater flow system into discharge areas, such as streams and swamps, are likely to range from several hundred to several thousand feet, but much smaller local flow systems with flow paths measuring tens of feet may also be present. Groundwater in recharge areas where hydraulic gradients are vertically downward is likely to flow from the till into underlying bedrock before it ultimately discharges (Melvin et al., 1992).

The climate is humid with an average annual precipitation of about 47 in. In a 30-year period (1951-80), median annual precipitation ranged from about 53 inches in northwestern Connecticut to 40 inches in the southwestern part of the State. Analyses of longer periods of record indicate that precipitation in any given year can be considerably greater or less than the median (Melvin et al., 1992).

Groundwater is recharged mainly from precipitation that percolates from the land surface to the saturated zone. Recharge rates are variable, but the long-term average is estimated to range from about 7 to 20 inches per year. Pumping of wells in areas near streams and lakes can induce large amounts of additional recharge from these surface-water bodies. Most groundwater flow occurs in the upper part of the saturated zone. Because of the shallow depth of the flow system and the moderate topographic relief, the circulation of groundwater is confined to the drainage basin of each perennial stream (Melvin et al., 1992)

Locally, the water table is generally within 10 to 15 ft of the land surface and is rarely deeper than 40 ft. A range of fluctuation of 8 to 17 ft over periods of one or more years is commonly observed. Changes in the position of the water table over time reflect changes in the balance between groundwater recharge and discharge. The magnitude of water table fluctuations also may reflect local differences in topographic and geologic conditions. For example, the range in fluctuation may be smaller near discharge areas where the water table is high and only a limited quantity of recharge can be accepted before the water table or capillary fringe reaches land surface (Melvin et al., 1992).

6.2.3 Site-Specific Hydrogeology

Data from previous investigations and remedial investigation borings and monitoring wells were used to develop a hydrogeological model of the Site. The dynamics of the groundwater system and its interaction with the surface water system are discussed in terms of both the hydrostratigraphy and hydrodynamics of the overburden aquifer. Then the water levels, hydraulic conductivity, and transmissivity results are presented.

Hydrostratigraphy

Stratigraphic borings advanced at the facility has revealed three principal hydrostratigraphic units: fill, estuarine silt, and depositional sediments (alluvium and glacial outwash). The information regarding the shallow bedrock suggests it is not a significant hydrostratigraphic unit. However, the topography of the bedrock surface significantly influences the groundwater flow patterns in the overburden aquifers at the site.

Fill: The fill unit is the uppermost hydrostratigraphic unit and is comprised of sand, gravel, and debris fill. This unit is generally about 2 to 5 feet thick, but locally is approximately 10 to 15 feet thick near the Dike. This zone contains the water table aquifer in areas near the Dike where tidal influences are most apparent on groundwater. In areas further from the Dike, the fill unit is mainly in the unsaturated zone.

Estuarine Silt: The silt is the second hydrostratigraphic unit and is comprised of fine silts with very fine sands, rich in organics. The thickness of the silt unit varies from as much as 30 feet to nonexistent, thinning from the tidal flats toward the interior of the facility. Silt deposits exist beneath the fill along the length of the Dike. The silt unit acts as a semi-confining unit over the depositional sediments in the tidal flats, and between the depositional sediments unit and the fill unit in areas along the Dike where it is sufficiently thick. The low permeability of the silt unit also affects the movement of horizontal groundwater flow in the Dike area. In areas where this unit is absent, there appears to be good hydraulic connection between the fill and depositional sediment units.

Depositional Sediment: This is the lower-most hydrostratigraphic unit within the unconsolidated aquifer and overlies the bedrock. The depositional sediment consists of glaciofluvial and fluvial sediments and can be subdivided into two general units, alluvium and glacial outwash.

> Alluvium: Sand and gravel deposits of glacial origin underlie the fill and silt deposits. The unit consists of fine-grained sand, with trace amounts of coarser sand and gravel with clay, silt, and cobbles. This unit is believed to be glacial deposits that have been reworked and sorted by the actions of the meandering Housatonic River. The alluvium is thickest beneath the southwestern part of the Site along Main Street, and thins toward the Housatonic River. The distinguishing feature of these deposits is the trace gravel and loosely cemented gravel zones. The bottom depth of these deposits varies between approximately 20 and 40 feet bgs.
>
> Glacial Outwash: Beneath the alluvium, and above the bedrock surface, lies a fine to medium sand with some silt, interpreted to be glacial outwash. The glacial outwash

contains silt/clay seams and fine silty sand lenses. The glacial outwash is generally stratified, and exhibits a fining-down sequence. In addition, considerable inter-fingering of slit and sand occurs across the Site within the depositional sediments. The silt layers within the glacial deposits may function as semi-confining layers where they are laterally continuous and sufficiently thick.

Bedrock: The nature of the bedrock encountered in cores taken of the shallow bedrock is indicative of a relatively competent rock (RQDs 75% to 100%) with low primary porosity. Groundwater flow through the shallow bedrock would be expected to be slight, especially when compared to the relatively high transmissivity presented by the overlying alluvium and outwash deposits. Further, as one approaches the river, vertical hydraulic gradients become predominantly upward. Hence plume migration in the shallow rock would be unlikely. The influence of the bedrock is likely greatest where it rises closest to the ground surface and presents a barrier to overburden groundwater flow. Here, one observes local changes in groundwater flow direction and an increase in the local hydraulic gradient.

6.2.4 Hydrodynamics

A total of 186 monitoring wells and piezometers have been installed in the overburden aquifer at SAEP. Twenty-four monitoring wells were installed by URSGWC during the 1999 remedial investigation and 32 monitoring wells were installed during Harding ESE's 2001-2002 investigations; the remaining were installed during previous investigations. Shallow monitoring wells are screened across the water table, intermediate wells are screened at depths from 30 to 50 feet, and deep wells are screened just above the bedrock.

Figures 6-10, 6-11, 6-12, and 6-13 present mean groundwater elevations, and interpretive groundwater isocontours, for two time periods (July 20-23, 1999 and July 24-26, 1999). Mean groundwater elevations on the main portion of the facility range from approximately 2.30 feet MSL beneath Building B-2, to 1.00 feet MSL beneath Building B-19, located near the Dike on the eastern side of the facility.

Figures 6-14 and 6-15 present groundwater elevations from water level measurements obtained on May 7, 2002. Groundwater elevations, for wells screened from 0 to –50 feet MSL, range from approximately 2.83 feet MSL beneath Building B-2, to –3.85 feet MSL in the tidal flats east of the Causeway. For wells screened at elevations less than –50 feet MSL, groundwater elevations range from approximately 2.6 feet MSL beneath Building B-2, to –2.0 feet MSL in the tidal flats east of the Causeway.

Horizontal Groundwater Gradients

Figures 6-10 and 6-11 show the piezometric surface (water table contours) based on mean groundwater elevations recorded in shallow and intermediate monitoring wells for two time periods, July 20-23, 1999 and July 24-26, 1999. The equipotential lines indicate groundwater flow is generally from west to east and southeast toward the Housatonic River across a large portion of the facility. However, groundwater flow in the West Parking Lot and western portion of the study area is to the northwest toward Frash Pond. There is a large area in the center of the Site beneath Building B-2 extending southwest to the west and south parking lots and Sikorsky

Airport of slight groundwater mounding which may act as a groundwater divide. This apparent feature was present during both monitoring periods. However, it is not known if this is a result of the limited data points located in this portion of the study area, a response to Frash Pond as a groundwater discharge area, or whether this apparent mound might reflect utility leaks or simply survey error. More water level data over time may help resolve this uncertainty. Another smaller groundwater divide appears to be present on the north side of Building B-2 where groundwater east of the divide flows toward the Housatonic River and groundwater west of the divide flows toward Frash Pond. Horizontal gradients vary widely across the main portion of the facility (Table 6-7). The horizontal gradient in the vicinity of Building 2 is relatively flat (0.0001 feet/foot [ft/ft]), and slight errors in measured water levels or surveyed reference elevations could lead to the interpretation of mounding beneath Building B-2. In contrast, the gradient increases appreciably near the river to the northeast (0.002 ft/ft). The estuarine silts likely act as a low permeability boundary to groundwater flow in this area resulting in the increased gradient.

Figures 6-12 and 6-13 show the piezometric surface based on mean groundwater elevations recorded in the deep monitoring wells. The equipotential lines indicate that groundwater flow in the deeper portions of the aquifer moves from west to east. However, the groundwater elevations of the deep wells generally are lower than those of the shallow wells, except for the shallow groundwater elevations adjacent to the Dike. The flow pattern indicates the discharge of groundwater toward the Housatonic River.

Additional thickness of fill along the Dike, and fill placed atop the silt for the construction of the Causeway, has likely caused some compression of the organic silts. This is seen in the geologic or hydrogeologic profiles in Figures 6-16 and 6-17, and the interpreted surface of silt elevation contours (Figure 6-6). While the added loading for the Dike and the loading for the Causeway may cause some compression in the silt with accompanying lowering of hydraulic conductivity, there is no evidence in the interpreted water level contours that this has a profound effect on groundwater flow through this zone, i.e., the compressed silt is not acting like a dam, mounding groundwater behind it. In the main facility area, placement of fill with compression of the silt, where present, resulted in much of the shallow groundwater flow occurring in the fill materials. Sands below the silts also have higher conductivity than the silts, and flow would occur through this zone as well. The compression of silts along the Dike, then, would appear to have only a minor impact on groundwater passing by this location. Along the Causeway, the compression of silts might make this zone less permeable to flow, but, again, flow would be mainly in the fill above the silt and the more permeable sand below. Compression of silt along the Causeway would appear to present little resistance to groundwater flow moving from one side to the other (mainly west to east). Existing explorations, few of which are directly in these areas, do, however, measure the total effect of the compressed silt, if any is appreciable. The possible compression of silts, as mentioned above, would appear to have little impact on groundwater flow at the site. Shallow flow off-site appears to be controlled more by the transmissivity of the fill (saturated thickness times hydraulic conductivity) than by any compression of the silt. Due to the age of the placed fills, it is unlikely that any further significant compression/settlement of the silt would occur that would alter this situation.

Steep gradients were observed in the deep aquifer in the area of Building B-19. These steep gradients are attributable to bedrock highs. Bedrock highs effectively reduce the overburden aquifer thickness, constricting flow and causing steeper hydraulic gradients in their vicinity. In contrast, areas with low hydraulic gradients appear to correspond with areas where bedrock elevations are low, as is the case beneath Building B-2.

Due to the nature (i.e., a single round uncorrected for tidal influence) of water level measurements collected by Harding ESE in May 2002, horizontal gradients have not been calculated for this event. However, it is important to note that the trend of increasing horizontal gradients toward the Dike observed in 1999 data appears to further increase in the direction of the tidal flats east of the Causeway (see Figures 6-14 and 6-15).

Vertical Groundwater Gradients

The overall dynamics of the overburden aquifer system are best shown by the cross-sectional flow diagrams (Figures 6-16 and 6-17). The cross-sectional flow diagrams are based on groundwater elevations calculated from tidal study measurements presented earlier. The diagrams clearly show the effects of the bedrock highs and the estuarine silt on the hydrodynamics at the Site. Also note that the equipotential lines tend to flatten horizontally along the berm in the eastern portion of the Site (see cross sections A-A’ and D-D’). Flattened equipotentials indicate layers that have low permeability and act as confining layers. Where these low permeability layers are present, there is a larger head difference between wells penetrating the shallow versus the deep portions of the overburden. The head differences indicate that these low permeability layers are acting as semi-confining units, restricting the movement of groundwater between the shallow and deep portions of the overburden unit. This contributes to an upward gradient in the shallow aquifer and a downward movement in the deeper aquifer (see Table 6-8). Where these low permeability layers are less extensive, the head differences between shallow and deep wells are relatively small and the equipotential lines remain essentially vertical, as shown on cross-section C-C’.

Groundwater elevations from selected well clusters measured on May 7, 2002 have been used to calculate vertical hydraulic gradients (see Table 6-9). The general trend observed in 1999 data of relatively neutral to slightly downward vertical gradients beneath the main portion of the facility, to upward vertical gradients toward the Dike and Housatonic River, is also observed in the vertical gradients computed from the May 7, 2002 water level data (Figure 6-18). However, the May 2002 data indicate neutral to slightly upward vertical gradients beneath Building B-2. The 2002 data allows for computation of vertical gradients between the estuarine silt (PZ-TF-XXA series piezometers) and the underlying alluvium (PZ-TF-XXB series piezometers). The vertical head differentials between these hydrostratographic units indicate a strong upward vertical gradient, up to 0.0579 ft/ft (see Figure 6-18), providing further evidence that the estuarine silts are acting as a confining unit over the alluvium and deeper glacial outwash deposits.

Pore pressure measurements were obtained during cone penetrometer explorations conducted during May 1999 (Foster Wheeler and Harding Lawson, 2000). While these data may provide direct measurement of piezometric heads in sandy soils where pore pressures are rapidly dissipated, the estimated accuracy of these measurements is only on the order of plus/minus 0.5 ft. In paired vertical locations, then, the error in the estimated head differential could be as much as one foot. As seen in Table 6-9, only three of the fifteen locations where vertical gradients were calculated had head differentials that exceeded one foot. The maximum head differential in these readings was 1.97 ft. The pore pressure data, then, are not sufficiently accurate enough to provide reliable estimates of vertical hydraulic gradients at the Site. There are sufficient water level data to provide adequate indications of vertical gradients at the Site.

Groundwater Discharge

Where a river channel is in direct contact with an unconfined aquifer, the river may either recharge the groundwater or receive discharge from the groundwater, depending on the water levels. A losing river recharges the groundwater; a gaining river receives groundwater discharge. As the river stage changes relative to the groundwater elevation, a losing river may become a gaining river, and vice versa.

Water levels to date have not shown the Housatonic River level rise above the water table. Therefore, the equipotential lines of the shallow aquifer near the Housatonic River indicate groundwater flow toward the river. At the center of the Site, the hydraulic gradient is very flat. At this point, the groundwater and surface water systems may interact, with occasional flow from the river to the groundwater.

The amount of groundwater discharging toward the Housatonic River through the overburden aquifer was determined from the cross-sectional diagram A-A' (see Figure 6-16). The discharge rate (Q) can be obtained from Darcy's Law:

$$Q = KiA$$

Where:
Q is the average discharge to the river, ft^3/d
K is the thickness weighted average hydraulic conductivity, ft/d
i is the average hydraulic gradient, ft/ft, and
A is the cross sectional area, width times average thickness, ft^2.

Substituting the relevant values into the equation yields (see calculation sheet included in Appendix L-1) a discharge rate of approximately 10,560 cubic feet (ft^3)/day (79,010 gpd or 55 gpm) through the overburden aquifer near the Dike toward the Housatonic River. This calculation does not include estimates of how that discharge may split to either the tidal flats or to the river. This estimate is also very close to discharge flows in the USACE Engineering Research Development Center (ERDC) site-wide groundwater flow model (submodel2).

Water Level Fluctuations

Unconfined aquifers with water tables near ground surface frequently exhibit diurnal fluctuations which can be attributable to evaporation and/or transpiration. Both processes cause a discharge of groundwater into the atmosphere and have nearly the same diurnal variation because of their high correlation with temperature. Appendix M contains hydrographs for several monitoring well clusters which show overall trends of the water table elevation as a result of changes in storage. These types of fluctuations are relatively small. The hydrographs also indicate the regular cyclic rises and falls caused by the tidal influences on the aquifer.

Tidal fluctuations greater than seven feet were measured in surface water in the Housatonic during the monitoring period. The fluctuation of surface water levels induces the migration of surface water into, and out of, the aquifer. Groundwater level fluctuations were detected in several of the monitoring wells on-site as a result of the introduction of surface water during high tide or loading caused by the introduction of surface water into the aquifer system. Groundwater fluctuations of one to two feet were recorded in monitoring wells located within 100 feet of the Dike. The fluctuation generally diminished proportionally to the distance from the river. At the distance of 300 feet from the river, measurements detected groundwater fluctuations of less than 0.1 feet. Tidal fluctuations may result in the surging of relatively clean river water into and out of the aquifer, creating an effective mixing zone for groundwater. The width of this mixing zone will vary as the tidal amplitude, groundwater flow rates, the hydraulic conductivity of the aquifer, and flow in the river. This mixing zone is likely to be relatively narrow. Plume concentrations may become greatly diluted as while passing through this zone to discharge to the river (see, for example, Yim and Mohsen, 1992).

The hydrographs indicate that the portion of the aquifer with monitoring wells installed within the upper 40 feet of the aquifer appear to be in good hydraulic communication with each other. However, data from well clusters with deep monitoring wells (+50 feet bgs) such as the LW-10 cluster, WC-9 cluster and LW-5 cluster indicate that the deep portion of the aquifer is not hydraulically connected with the upper portion of the unconsolidated aquifer. Groundwater elevation in deep monitoring wells display different responses over the same time period. It appears that layers (i.e., silt) that have low permeability may be acting as semi-confining or confining layers in these areas.

Storativity and Specific Yield

Specific yield is defined as the ratio of the volume of water that drains from a saturated rock (or soil) due to gravity to the total volume of the rock or soil. The specific retention is the ratio of the volume of water a rock (or soil) can continue to store despite drainage to the total volume of the rock or soil. Since specific yield is the volume of water that a rock will yield by gravity drainage, and specific retention is the volume remaining after drainage, the sum of the two is equal to the total porosity. Specific retention increases as soil grain-size decreases.

The groundwater level in an aquifer determines the saturated thickness of the aquifer – as the groundwater level rises or falls, the saturated thickness increases or decreases, respectively. In an unconfined aquifer, water drains from pore spaces when the groundwater level falls. The amount of water that drains is a function of the specific yield of the rock or soil. Conversely, the amount of water that remains stored (i.e., does not drain) is a function of the specific retention of the rock or soil.

The specific storage of an aquifer is defined as the volume of water released due to decline in pressure in the aquifer (i.e., a decrease in the hydraulic head). Two mechanisms produce the release of water from storage when pressure declines: 1) the compaction of the aquifer, and 2) the expansion of water. Storativity relates the specific yield and the specific storage of the rock or soil in an aquifer as a function of the saturated thickness of the aquifer. Specifically, storativity is defined as:

$$S = S_y + hS_s$$

where: S_y = specific yield

h = saturated thickness

S_s = specific storage

The specific yield of an unconfined aquifer can range from 0.01 to 0.35. The geology encountered in borings at the Site is generally fine sand and silty sand. Based on this geology, an initial estimate of the average specific yield at the Site is 0.2.

URSGWC attempted to refine this estimate with a calculation of discharge and site water level decline. However, the method relies on no upgradient contributions to groundwater flow and no delayed yield. Both of these assumptions may be violated as they applied this method to the site. Using this method, URSGWC arrived at an estimate of 0.24 for specific yield (URSGWC, 2000). Applying a similar method with more current data yields a result of 0.18. As URSGWC pointed out, there likely is variability across the site. These estimates are probably no more valid than the 0.2 suggested by the geology and literature.

Aquifer Properties and Testing

The hydraulic conductivity values obtained from the historical hydraulic conductivity tests range from 0.03 ft/day to 318 ft/day. Hydraulic conductivity values for shallow screened wells ranged from a low of 0.3 ft/day at LW-3S to a high of 302 ft/day at PZ-PILOT-06. The wells installed within the intermediate zone ranged from a low of 0.03 ft/day at WC-18D1 to a high of 318 ft/day at PZ-TF-03B. The hydraulic conductivity values for the deep monitoring wells ranged from a low of 0.2 ft/day at WC-9D2 to a high of 59 ft/day at HESE-01-07D. A summary table of results is presented in Table 6-6. The plots for URSGWC 1999 and Harding ESE 2002 hydraulic conductivity tests conducted at the facility during the investigation are presented in Appendix I.

The geology intersected by a well screen may be highly variable, as shown in Table 6-6. This geological variability across the overburden is reflected in the high variability of hydraulic conductivity values determined within the overburden. Notable observations regarding the variability of hydraulic conductivities within the geologic units are as follows:

- Fill: The hydraulic conductivity of the fill ranges from 0.3 to 302 ft/day.
- Estuarine Silt: The hydraulic conductivity of the silt ranges from 0.4 to 80 ft/day. The hydraulic conductivities from tests performed at HESE-01-17I (24 ft/day) and PZ-TF-08A (80 ft/day) likely reflect more permeable sand seams present in the silt which were observed during geotechnical investigations of the tidal flats sediments.
- Alluvium: The hydraulic conductivities of the alluvium range from 0.03 to 318 ft/day. Alluvium hydraulic conductivities are generally lower across the western portion of the site and increase toward the east where the alluvium thickness decreases and bedrock elevation increases.
- Glacial Outwash: The hydraulic conductivity of the glacial outwash ranges from 0.2 to 91 ft/day. The average hydraulic conductivity of the glacial outwash is only slightly less than that of the alluvium; however, the general trend of increasing hydraulic conductivities from east to west in the alluvium also holds true for the hydraulic conductivities of the glacial outwash.

Due to the large range of hydraulic conductivity values (four orders of magnitude), statistical analysis of the in situ hydraulic conductivity values across the overburden aquifer (i.e., without partitioning the data from individual stratigraphic units) was performed by URSGWC for the 2002 Draft RI Report (URSGWC, 2000). The frequency distribution of these hydraulic conductivity estimates derived from the permeability tests suggested that the hydraulic conductivity of the overburden is lognormally distributed. Domenico and Schwartz (1990) reported on the work of several authors who noted that hydraulic values obtained from individual formations were lognormally distributed.

In a comparable exercise, Harding ESE sorted the hydraulic conductivity data in Table 6-6 by unit (fill, estuarine silt, alluvium and glacial outwash) and subjected them to an analysis of their respective distributions using the Shapiro-Wilk test in the USEPA statistical software package, DataQUEST (USEPA, 1997). Data for each of the groups were clearly not normally distributed (failed the test at the 95 % confidence level). When the data were log-transformed, the hypothesis of lognormality could not be rejected at the 95 % confidence level for the estuarine silt. While the logtransformed fill and outwash data did not pass the test at the 95% confidence level, it did at the 99 % confidence level. The log-transformed data for the alluvium did not pass the test for lognormality at either the 95 or 99 % confidence level, but very nearly did (W-statistic of 0.908 vs critical value of 0.912 at the 99 % confidence level). After conducting a Wilcoxon Rank Sum Test on the alluvium and outwash data, it was concluded that the hypothesis of equal medians for these data sets could not be rejected at the 95 % confidence level. Upon pooling these data, a Filliben's test of distribution (similar to the Shapiro-Wilk Test) was conducted. The combined set was shown to just fail the test for lognormality at both the 95 and 99 % confidence levels. However, the sets were deemed approximately lognormally distributed for estimating median values for the hydraulic conductivity using the geometric mean of the values. See Appendix L-1 for further discussion of distributions and statistical measures. The geometric mean for the fill hydraulic conductivity was estimated as 38.6 ft/d, that of the estuarine silt data as 6.3 ft/d, and that of the pooled alluvium and outwash data as 17.8 ft/d. Only one test was conducted in the peat, and this value was 12.8 ft/d.

These data were used to estimate overall groundwater discharge rates to the Housatonic River and tidal flats. These estimates are described in the "Groundwater Discharge" subsection of Subsection 6.4.3, and details of the calculation are contained in Appendix L-1. The estimated discharge rate is approximately 10,560 ft^3/d (79,010 gpd). This rate, of course, would vary seasonally and from year to year depending on the variations in annual precipitation. This estimate is within 5 % of the total groundwater flow in the ERDC groundwater flow model.

Groundwater Flow Velocity

Information on the movement of groundwater is fundamental to understanding the fate and transport of dissolved contaminants. For many contaminants, other information about physical, chemical, or biological processes that affect their fate and transport is also required to make reliable predictions. Groundwater flow velocities are likely to vary widely beneath the facility, based on the range of observed hydraulic conductivities and gradients.

The velocity of groundwater movement through the aquifer can be estimated from the following equation:

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

where K = hydraulic conductivity
i = hydraulic gradient
Φ = porosity

Porosity is a measure of the volume of voids in soil or rock. Literature suggests that unconsolidated deposits like those in the overburden aquifer have porosities ranging from 25% to 70%. In general, fine-grained unconsolidated deposits (e.g., clay and silt) have a higher porosity than do coarse-grained materials (e.g., sand and gravel), yet they have low hydraulic conductivity. In addition, Freeze and Cherry (1979) noted that poorly sorted deposits have a lower porosity than those of well-sorted deposits.

Based on the geological textures and sorting observed, it is estimated that the porosity at the Site ranges from 25% for the higher-permeability deposits to 50% for the lower-permeability deposits. Assuming an average horizontal hydraulic conductivity of 37.5 ft/d as calculated across section A-A', an average porosity of 35%, and an average hydraulic gradient of 0.00163, it is estimated that the groundwater flow velocity averages about 64 ft/year. The groundwater flow velocity is likely variable across the site due to variable horizontal hydraulic gradients and seasonal influences. Higher flow velocities may be expected in fill materials, especially during the spring and fall wet seasons. The higher velocities occur in the more conductive sections of the overburden deposits. For example, using the geometric mean K of the fill material (108.1 ft/d), the maximum hydraulic gradient near the Dike (0.0022), and a porosity of 0.35 would yield a groundwater velocity of approximately 0.7 ft/d and 255 ft/yr. These data emphasize the importance of the more conductive zones of the aquifer, especially because 1) these zones are where the discharge is occurring, and 2) these zones may provide preferential pathways for the transport of contaminants within the aquifer.

6.2.5 Hydrogeologic Conceptual Model

A conceptual model of the Site was developed based on the results from the remedial investigation activities. Three principal overburden hydrostratigraphic units were identified during the investigation: fill, estuarine silts and depositional sediments (alluvium and glacial outwash). In addition, peat is locally present east of Sniffens Lane, but is not considered a significant hydrostratigraphic unit. The fill unit is the uppermost hydrostratigraphic unit and comprises sand, gravel, and debris fill. This zone contains the water table aquifer in areas near the Dike where tidal influences are most apparent on groundwater. In areas further from the Dike the fill unit is in the unsaturated zone. The silt is the second hydrostratigraphic unit and comprises fine silts with very fine sands, rich in organics. This unit acts as a semi-confining unit between the depositional sediments unit and the fill unit in areas along the Dike where the thickness is sufficient. The low permeability of the unit also affects the movement of horizontal groundwater flow in the area from the Dike to the Housatonic River. In areas where this unit is absent, there appears to be good hydraulic connection between the fill and depositional sediment units. The depositional sediment is the lower-most hydrostratigraphic unit within the unconsolidated aquifer and overlies the bedrock. The depositional sediment consists of glaciofluvial and fluvial sediments and geologically can be subdivided into two general units, alluvium and glacial outwash. However, for purposes of the site hydrogeology, they appear to be relatively indistinguishable. For presentation of profiles and modeling purposes, the alluvium and glacial outwash have been represented as a single unit.

Three surface water features influence groundwater flow at the SAEP facility. The primary influence is that of the Tidal Flats and the Housatonic River. Groundwater flow in the northern half of the facility is in the direction of the Tidal Flats at low tide. A second surface water body influencing the groundwater flow at the facility is Frash Pond. Frash Pond appears to be located downgradient of the northwest portion of the SAEP facility. The airport, as well as other off-site properties, is also located upgradient of Frash Pond. The third surface water body influencing the groundwater flow at the facility is the drainage channel located in the southern portion of the SAEP. Groundwater mounds in the shallow portion of the aquifer in this area of the facility indicated that a perched water condition exists on the peat layer found in this area.

The overburden aquifer is complex both geologically and hydrogeologically. The shallow portion of the aquifer is unconfined. The deeper portion of the overburden aquifer appears to range from being unconfined to being semi-confined, and may be fully confined locally based on the presence and thickness of the estuarine silt unit. Groundwater flow within the overburden aquifer is complex and is controlled by the estuarine silt in the shallow portion and bedrock surface elevation in the deep portions of the aquifer. In general, groundwater flows toward and discharges to the Housatonic River. Based on the findings of this investigation, an estimated 10,560 ft^3/d (about 79,000 gallons per day) on average of groundwater is discharged toward the Housatonic River through the length of the Dike area. In addition, a northwesterly flow towards Frash Pond, and flow toward the drainage channel in the southern portion of SAEP have been observed. The steeper hydraulic gradient adjacent to the river along the Dike can be attributed to the estuarine silt unit, which serves as a low permeability impediment to groundwater flow.

A large area of slight groundwater mounding in the center of the Site beneath Building B-2 extending southwest to the west and south parking lots and Sikorsky Airport may act as a groundwater divide for flow at the site. However, it is not known if this is an anomaly caused by the limited data points located in this portion of the study area. Another smaller groundwater divide is present on the north side of Building B-2 where groundwater east of the divide flows toward the Housatonic River, and groundwater west of the divide flows toward Frash Pond. Both groundwater divides change groundwater flow in the western portion of the Site to a northwesterly flow.

Regarding the peat unit present near the RCRA lagoons east of Sniffens Lane, C.A. Rich Consultants determined that the direction of shallow groundwater flow in the upper aquifer is to the east, and is controlled by the organic peat present beneath the former lagoons and tidal ditch. Groundwater flow above the peat unit is radial toward the east, south, and west. The hydraulic conductivity of the peat is lower than the corresponding upper and lower glacial sand and gravel stratigraphic units, which results in a partially perched water table in the vicinity of the former lagoons. The groundwater flow direction in the lower aquifer is south-southeast. The peat layer retards, but does not prevent, natural vertical flow of groundwater.

Downward vertical hydraulic gradients indicate that aquifer recharging conditions exist in the vicinity of the West Parking Lot. Vertical gradients begin to flatten in the center of the Site and reverse (becoming upward) to discharging conditions near the Housatonic River in the shallow and intermediate wells and tidal flats piezometers. Deep wells indicate continued downward gradients, with some discharge likely into the Housatonic River beyond the Tidal Flats. Head differences between the shallow wells and deep wells indicate that the low permeability layers are acting as semi-confining units, restricting the movement of groundwater between the shallow and deep portions of the overburden unit. This allows, simultaneously, an upward gradient in the shallow aquifer and a downward movement in the deeper aquifer. Where these low permeability layers are less extensive, the head differences between shallow and deep wells are relatively small, and the equipotential lines remain essentially vertical.

Tidal fluctuations greater than seven feet were measured in the Housatonic River. Groundwater level fluctuations were detected in several of the monitoring wells on-site as a result of the introduction of surface water during high tide or loading caused by the introduction of surface water into the aquifer system. Groundwater fluctuations of one to two feet were recorded in monitoring wells located within 100 feet of the Dike. The groundwater fluctuation amplitudes diminished proportional to the distance from the river and were generally less than 0.1 feet at a distance of 300 feet from the river.

The hydraulic conductivity of the overburden aquifer spans approximately four orders of magnitude (0.03 to 317.9 ft/day). The data suggest that the hydraulic conductivity is lognormally distributed and skewed right. The large spatial variability in hydraulic conductivity is consistent with the complex geology at the Site. The calculated groundwater velocity ranges from about zero near groundwater divides to about 255 ft/year. The higher velocities occur in the more permeable sections of the overburden deposits. These data emphasize the importance of the more permeable zones of the aquifer, especially because 1) these zones are where the greater portion of discharge is occurring, and 2) these zones may provide preferential pathways for the transport of contaminants within the aquifer.

Information on the bedrock beneath the site suggests that significant groundwater flow through the shallow rock is highly unlikely. High RQDs, probable low primary porosity, increasing upward hydraulic gradients as one approaches the river, and the relatively high transmissivity/hydraulic conductivity of the overlying alluvium and glacial outwash aquifer minimize the significance of the shallow bedrock as a significant contributor to groundwater flow to the river. In addition, the till unit found at several exploration locations may be relatively impermeable, and limit interaction of bedrock and overburden groundwater.

7.0 AOC EVALUATION

This section presents an evaluation of AOCs and potential release areas. The purpose is to determine if a release of contamination has occurred to the environment. AOCs were consolidated into groups according to type and location as shown on Table 7-1. AOC groups include the hazardous waste storage area, the chemical waste treatment system, the manufacturing, testing, research and development areas, the stormwater and wastewater system, and miscellaneous areas.

Baseline Concentration Assessment. An assessment of chlorinated VOCs (cVOCs), PAHs, inorganics, and TPH in soil was performed for the purpose of establishing a baseline concentration, or range of concentrations, that can be used to identify potential releases of contaminants from the five AOC groups (see Appendix N-1). Assessments were also conducted for inorganic concentrations in groundwater (see Appendix N-2) and sediment. Establishing baseline concentrations at SAEP was necessary because the Site lacks a typical background data set representative of soil un-impacted by site activities.

In place of a background data set, Site-wide soil data collected during the RI were used to establish baseline levels for soil. Organic contaminants were found to be present in soil throughout the Site but at many locations that could not be linked to specific areas of chemical release or storage. Establishing baseline concentrations allowed for the distinction between Site-wide organic contamination and areas where chemical releases occurred that require further analysis. Establishing baseline concentrations was also necessary for inorganics since a background soil data was not available to distinguish between naturally occurring levels in soil and possible releases.

The assessment revealed that cVOCs and PAHS have been detected in soils Site-wide, indicating a “baseline” condition, and that inorganic concentrations in soil are very close to the eastern United States background concentrations, with few exceptions. In summary, there is general background soil contamination across the entire site due to the presence of potentially contaminated fill, the use of the property for heavy industry, and naturally occurring inorganics. Concentrations indicated as “detected” within this Section are above the baseline concentrations indicated below. Complete analytical data are presented in the Appendices of this report.

The following paragraphs present the conclusions of the assessment.

Chlorinated VOCs. For purposes of evaluating releases of cVOCs from AOCs, a screening value of 1 mg/kg is used in this section for all individual cVOC analytes. Only concentrations greater than 1 mg/kg will be considered indicative of a release of cVOCs from an AOC.

Polynuclear Aromatic Hydrocarbons. For determination of potential releases of PAHs from AOCs, a screening value of 1 milligram per kilogram (mg/kg) is used in this section for all individual PAH analytes. Only concentrations greater than 1 mg/kg will be considered indicative of a release of PAHs from an AOC.

Inorganics. Detected inorganic compounds may be the result of site activities such as plating, painting, and the use of leaded gasoline; they may have been contained in fill materials placed on-site during historical filling with dredged material from the Housatonic River; or they may be naturally occurring in site soils, groundwater, or sediment as a result of regional geologic conditions. Due to the potential sources for inorganics, detected inorganics concentrations were only evaluated for the AOCs associated with plating activities and dissolved metals solutions. The AOCs identified as having these types of activities are:

- AOC 8 – CWTP Collection System, Pump Station (Building B-63), and associated piping
- AOC 9 – Cyanide Destruction Facility in Building B-70
- AOC 10 – CWTP in Building B-18
- AOC 31 – Building B-6 and associated satellite accumulation areas
- AOC 34 – Building B-3A and associated satellite accumulation areas
- AOC 39 – Building B-4 former Brine UST
- AOC 50 – Building B-2 Plating Area
- AOC 51 – Building B-3 Plating Area
- AOC 56 – Research and Development Area in northern Building B-3, Building B-3A, and Building B-4

Review of historical SAEP documents indicates that cadmium, chromium (total), copper, cyanide, Cr(VI), and nickel were associated with processes at these AOCs. As Cr(VI) and cyanide are not naturally occurring, any detection of these two analytes in soil or shallow groundwater samples associated with any AOC will be considered as evidence of a release. The calculated baseline concentrations for the remaining inorganic compounds include:

- cadmium - 7.2 mg/kg for soil and 0.0044 mg/L for groundwater;
- total chromium - 146 mg/kg for soil and 95 mg/L for groundwater;
- copper - 87.8 mg/kg for soil and 0.085 mg/L for groundwater; and
- nickel - 37.6 mg/kg for soil and 0.063 mg/L for groundwater.

Therefore, detected concentrations of the specific AOC-related inorganics above the estimated baseline concentrations in soil or shallow groundwater will be considered indicative of a potential release of inorganics from the specified AOCs.

An assessment of inorganic concentrations in sediment was performed to establish a baseline value, or range of concentrations, for these analytes to be used in the determination of potential releases of contaminants from AOCs. Due to the potential for upstream and naturally occurring sources of inorganics in sediment only those inorganics related to past industrial practices (e.g., Cr(VI), cyanide, copper, cadmium, nickel, etc.) are considered. Detected inorganics are screened against the maximum concentration from sediment reference locations (see Table 9-11, Sediment Data Summary, Reference Locations in the January 2003 Draft RI Report). Concentrations of sediment greater than the maximum concentration of the reference locations are evaluated in AOC discussions as indicative of a release. All analytes detected, regardless of whether the concentrations exceed this release criteria or not, are still evaluated in the ecological risk assessment.

Total Petroleum Hydrocarbon. For determination of potential releases of TPH from AOCs, a screening value of 872 mg/kg is used for soils. Only concentrations greater than this value will be considered indicative of a release of TPH from an AOC.

Satellite Accumulation Areas. Since 1990, satellite accumulation areas were established in various locations at SAEP. Satellite accumulation areas served as temporary storage areas for small quantities (one to six 55-gallon drums) of waste liquids and solids. Satellite accumulation areas were eliminated from service by 1997 when facility operations ceased. The 55-gallon drums used in the satellite accumulation areas were removed as drums reached capacity. As part of the facility-wide Post-production Environmental cleanup conducted in 1997, all floor surfaces, including satellite accumulation areas, were steam-cleaned to remove any surface contamination. Given the limited duration of use of the satellite accumulation areas, the frequency of removal of accumulated waste, and the Post-production Environmental cleanup, satellite accumulation areas are not considered to have resulted in soil and groundwater contamination. Therefore, satellite accumulation areas were not evaluated as AOCs. The buildings with satellite accumulation areas referenced in the RFA were evaluated for potential releases.

Release Determination Evaluation. Conclusions regarding the presence of a release of contamination from each AOC were drawn based on an evaluation of the AOC, the associated sample and analytical data and file information:

1. A release has occurred.
2. A release has not occurred, or is not suspected.
3. The determination of a release is not possible due to the presence of contamination from other potential source(s).
4. There is insufficient information to determine if a release has occurred.

7.1 HAZARDOUS WASTE AND WASTE OIL AREA

The Hazardous Waste and Waste Oil Area consists of waste storage areas in and around Buildings B-13, B-15, and the former Oil House Tank Farm (Building B-74). This is located in the north-northeastern portion of SAEP and includes Building B-13, Building B-15 and the former Oil House Tank Farm. This group also includes storage areas and three waste transfer systems, which consist of underground tanks, vaults, and transfer piping. The area was used for oil blending, waste reclamation, and waste storage since the mid 1940s (W-C, 1991).

AOCs and other potential release areas in this group include:

- AOC 1 – Waste Oil Accumulation Tanks in the Former Oil House Tank Farm
- AOC 2 – Hazardous Waste Accumulation Tanks in the Former Oil House Tank Farm
- AOC 3 – Hazardous Waste and Waste Oil Transfer Systems between Buildings B-13 and B-15
- AOC 7 – Oil/Alum Tank
- AOC 12 – Container Accumulation Area and Associated Drains (located adjacent to the Oil House Tank Farm)
- AOC 13 – Original Container Storage Area (located east of B-13)
- AOC 16 – Metal Chips Oily Sump
- AOC 28 – Building B-15 and Associated Satellite Accumulation Areas
- AOC 53 – Drum Staging Area Between the Former Oil Tank Farm and Building B-37
- Building B-13 Former Chemical Storage and Scrap Metal Reclamation

- Container Storage Pad and Collection Trench Northeast of Building B-13
- Magnesium-Thorium Scrap Yard Between Buildings B-13 and B-44
- Open Storage Area Between Buildings B-16 and B-74

Evaluation of analytical data in this area is complicated by the presence of fill and general low-level organic contamination in soils. Drilling records indicate soils consist of fill containing fragments of concrete, brick, asphalt, wood, cinders, copper wire, rebar. Sediments dredged from the Housatonic River were used as fill to extend the shoreline. Shoreline filling occurred during the 1940s to provide land area for additional buildings. The fill layer is thickest in the areas closest to the river and thins toward Buildings B-13 and B-15. As shown in Figure 1-5, part of the Hazardous Waste and Waste Oil Area is superimposed over the former shoreline and stormwater outfall.

Figure 7-1 presents the locations and boundaries of the AOCs within this group, as well as the associated explorations. Table 7-2 presents a summary of samples associated with each AOC, the field observations, detected analytes, release determination, and analytes with concentrations exceeding CTDEP RSR criteria.

AOC 1 - Waste Oil Accumulation Tanks and AOC 2 - Hazardous Waste Accumulation Tanks in the Former Oil House Tank Farm

The Oil House Tank Farm contained 13 aboveground tanks used for storage of raw materials and collected waste oils, solvents, and fuels (see Figure 7-1). The tanks were supported on concrete saddles elevated above a concrete slab and surrounded on all four sides by a three-foot high containment berm. All tanks reportedly contained pneumatic level indicators (CDM FPC, 1992). The original tank farm was constructed in the early 1950s approximately 50 feet from the flood-control Dike in the northern portion of the facility. Between 1980 and 1982, the Oil House Tank Farm was relocated 30-50 feet west of the original location, likely to accommodate a covered storage facility at AOC 12 (USEPA, 1990). The Waste Oil and Hazardous Waste Accumulation Tanks and aboveground piping were removed in 1998.

Seven tanks within the Oil House Tank Farm were used for storage of raw materials, including coolants, lubricants, hydraulic oils, and 1,1,1-TCA. The other six tanks received waste liquids from engine testing, maintenance, vapor degreasing, corrosion resistance, and metal working. Waste types included, spent halogenated solvents (e.g., Varsol and 1,1,1-TCA), spent jet fuel, emulsion wastewater, spent slushing oil, and coolant oils. Wastes were transported to the Oil House Tank Farm both directly and via the Waste Transfer System (AOC 3). Liquids were reportedly stored in the Oil House Tank Farm for less than 90 days (W-C, 1991).

The Waste Oil Accumulation Tanks (AOC 1) consisted of aboveground tanks #11, #12 and #13. Each 10,000-gallon horizontal tanks that contained accumulated waste oils, a mixture of lubrication, hydraulic and cutting oils. The tanks were used from 1981 until 1996. Wastewater from the Waste Oil Accumulation Tanks was drawn off and transferred to the OATP for treatment and disposal (INEL, 1991).

The Hazardous Waste Accumulation Tanks (AOC 2) consisted of aboveground tanks #4, #5, #6, #7 and #9 each five 5,000-gallon horizontal tanks used to accumulate waste 1,1,1-TCA waste solvent-oil mixture, and waste jet fuel. Tanks #4 and #5 were in use from 1981 to 1988. The remaining tanks were in use from 1981 to 1996.

Explorations associated with these AOCs include: two soil borings completed inside the containment berm of the former Oil House Tank Farm (EBS01-1 and EBS01-2), two soil borings were completed outside of the berm (BR-6 and HA-99-08), three tidal flats piezometers installed downgradient of the former Oil House Tank Farm (D-3, D-4, and D-4-1), and one monitoring well installed directly west of the former Oil House Tank Farm (WC2-1S) (see Table 7-2). Analytes detected in these soil samples include BTEX, VOCs, and SVOCs (see Appendix O for complete soil boring analytical results).

A release has occurred from these AOCs. The presence of visual and olfactory contamination along with the detection of chlorinated and fuel-related contaminants in soil within the berm of the former Oil House Tank Farm indicates a release has occurred to soil and shallow groundwater, likely due to spills or leaks from tanks or piping. Concentrations of arsenic and dichloromethane in soils exceed RSR criteria (see Figure 7-2). Concentrations of chloroethane, cis-1,2-DCE, TCE, and vinyl chloride in groundwater exceed RSR criteria (see Figure 7-2).

AOC 3 - Hazardous Waste and Waste Oil Transfer Systems Between Buildings B-13 and B-15

The Hazardous Waste and Waste Oil Transfer Systems, installed prior to 1970, consisted of underground tanks located between Building B-13 and Building B-15 and underground transfer piping to the former Oil House Tank Farm (see Figure 7-1). A second transfer system conveyed raw product, waste oil and solvent from the eastern corner of Building B-15 to the Oil House Tank Farm.

AOC 3 consisted of three similar waste transfer systems; one for waste fuel, one for waste solvent and oil mixtures, and one for waste oil. The waste fuel and waste solvent and oil systems each consisted of a 500-gallon underground receiving tank constructed of carbon-steel oriented horizontally in a separate concrete containment vault. The waste oil transfer system consisted of two 400-gallon rectangular steel plate tanks encased in underground concrete vaults. Each tank was connected to a pump inside Building B-13 that transferred the waste from the receiving tank through a covered pipe trench to the appropriate container in the former Oil House Tank Farm. The transfer lines consisted of 2-inch diameter underground carbon steel pipe approximately 250 feet in length.

Liquid wastes from engine testing, maintenance activities, vapor degreasing, and metalworking were transferred to the receiving tanks. These wastes included spent engine fuel, waste 1,1,1-TCA, spent slushing oil, hydraulic oil, coolants, and cutting oils (W-C, 1991). The four receiving tanks were removed in 1993 (ABB-ES, 1996).

Explorations associated with this AOC include: six soil borings, BR-4, WC-5S (completed as a monitoring well), SB08L1-7, SB09B6-1, SB09C1-1, and SB09C2-1, and one soil gas sample, SG-99-47 (see Table 7-2). Analytes detected in these soil sample locations include BTEX, cVOCs, SVOCs, PAHs, TPH, and PCBs (see Appendix O for complete soil boring analytical results). Analytes detected in groundwater samples include BTEX, cVOCs, SVOCs, and PAHs (see Appendix P for complete groundwater analytical results). PCE and TCE were detected in soil vapor samples associated with this AOC.

The determination of release from this AOC is not possible due to the presence of contamination from other potential sources. Materials handled at this AOC include waste fuel, waste solvent/oil mixtures and waste oil. Visual and olfactory observations, elevated PID readings, and analytical results from soil samples taken adjacent to and downgradient of the underground vaults and transfer lines indicate a release of fuels and chlorinated solvents to soil. Possible volatilization of PCE and TCE contamination in vadose zone soil may have impacted soil vapor in the vicinity of Building B-15. However, none of the samples considered during evaluation of this AOC can be uniquely associated with the receiving pits and underground transfer lines comprising AOC 3. Concentrations of BTEX, cVOCs, VOCs, TPH, PCBs, and inorganics in soils exceed RSR criteria (see Figure 7-2). Concentrations of cVOCs and arsenic in groundwater exceed RSR criteria (see Figure 7-2).

AOC 7 - Oil/Alum Tank

The Oil/Alum tank was an aboveground, 10,000-gallon welded carbon steel tank mounted vertically on a concrete pad. The tank was used for temporary storage of recovered cutting oils from the chip sump at Building 13 (see Figure 7-1). Oil that collected in the sump was pumped to the Oil/Alum tank. In addition, flocculent from the OATP was skimmed and piped, with other settled material, to the Oil/Alum tank to allow the liquids to further gravity separate. After a period of time, the heavier water phase was drained from the tank and returned to the OATP, via a nearby catch basin. Accumulated oil was emptied from the tank when several thousand gallons had accumulated, and was then sent off-site for incineration (W-C, 1991). The tank was in operation from 1976 until 1997.

Soil boring SB09B10-1 was completed beneath the former Oil/Alum Tank (see Table 7-2). Analytes detected in these soil sample locations include BTEX (all concentrations less than 1 mg/kg) and acetone (see Appendix O for complete soil boring analytical results).

A release from this AOC has not occurred, or is not suspected. BTEX was detected at concentrations less than 1 mg/kg, no PID readings above background, visual evidence of contamination, or odors were noted during sampling, and the area is covered by 18 inches of concrete. No contaminants were detected above RSR criteria.

AOC 12 - Container Accumulation Area and AOC 53 - Drum Staging Area Between the Former Oil House Tank Farm and Building B-37

The Container Accumulation and Drum Staging Areas, located directly south of the former Oil House Tank Farm (see Figure 7-1), were used to collect and prepare containerized waste for disposal off site or into tanks in the Oil House Tank Farm. The western half of the area (AOC 53) was used for staging, and the eastern half (AOC 12) for waste accumulation. The storage area served as a staging area for drums that required testing of contents prior to disposal off-site and was reportedly used to store drums for less than 90 days (W-C, 1991). Rain water that accumulated in the area was allowed to drain to a gravel area through three drain plugs located on the back wall. The drain plugs were sealed with concrete in 1990 (CDM FPC, 1992).

Use of the area began prior to 1980, when aboveground tanks (See AOC 1 and AOC 2) were located in this area. From 1981 to 1985, the area reportedly was used for temporary storage of solid wastes and recyclable materials. Consent Order HM-215 (circa 1984) identified poor condition and management of drums stored on the facility and drums were not being stored on an impervious bermed surface (CTDEP, 1984). A subsequent CTDEP compliance inspection, conducted in November 1985, identified a new concrete pad, containment berm, and roof being constructed behind the Oil House (Building B-15). At this time, approximately five drums of waste 1,1,1-TCA were being stored near this area, awaiting construction of the pad (CTDEP, 1985). A January 1986 CTDEP compliance inspection found three segregated sections of the storage area containing bagged asbestos (from insulation removal), waste oils, and unidentified drums. Two drums of waste 1,1,1-TCA were identified outside the containment area (CTDEP, 1986). A June 1986 RCRA inspection identified 153 drums in these AOCs, several of them open, unlabeled, or located outside the concrete pad and berm. Around this time, the facility was preparing to transfer the storage of waste oil and 1,1,1-TCA from drums to tanks within the Oil House Tank Farm to limit the amount of drum storage necessary at the facility (USEPA/CTDEP, 1986). An inspection conducted during 1988 found the AOCs surrounded by a 4-foot high fence with a locking gate and containing only thirteen 55-gallon drums (USEPA/CTDEP, 1988); however, during a 1990 CTDEP inspection, AOC 12 contained 118 drums in segregated berms, and AOC 53 contained 14 additional drums and nine portable waste transporters (CDM FPC, 1992).

Two soil borings were completed beneath AOC 12, including EBS12-1 and EBS12-2; soil boring SB09C1-1 has been associated with AOC 53 (see Table 7-2). Analytes detected in soil samples from these locations include BTEX, cVOCs, SVOCs, PAHs, TPH, and PCBs (see Appendix O for complete soil boring analytical results).

The determination of release from these AOCs is not possible due to the presence of contamination from other potential sources. Materials handled at these AOCs included waste oil, solvents, and fuel. Although available information suggests a release of chlorinated and fuel-related VOCs to the soils underlying the AOCs has occurred, it is unknown if this release occurred from handling and storage of drums after designation of the area as a container storage area, or prior to that time when the area contained storage tanks. Concentrations of cis-1,2-DCE and xylenes in soil exceed RSR criteria (see Figure 7-2).

AOC 13 - Original Container Storage Area

Used from 1980 to 1984, the Container Storage Area, located adjacent to the north and northwestern side of Building B-13 (see Figure 7-1) was used accumulate 55-gallon drums of hazardous waste. This area was the original Container Storage Area identified in the site’s RCRA Part A Permit Application submitted in November 1980. Reportedly, containers were allowed to accumulate in this area for less than 90 days.

There is insufficient information to determine if a release has occurred from this AOC. No samples have been collected at AOC 13; however, observations from adjacent and overlapping AOCs indicate there have been releases in this general area.

AOC 16 - Metal Chips Oily Sump

The Metal Chips Oily Sump was located at the northwest corner of Building B-13 (see Figure 7-1), where metal chips were deposited into an outdoor open concrete turnings oil pit (CDM FPC, 1992). Oil drained into the sump and was pumped to a waste oil tank at the Tank Farm (W-C, 1991). The concrete pit was removed in 1993 (ABB-ES, 1996).

SB09B11-1 was taken from within the former limits of the chip collection area, but not adjacent to the sump. Analytes detected in these soil sample locations include BTEX (all concentrations less than 1 mg/kg) and PCB 1260 (see Appendix O for complete soil boring analytical results).

A release from this AOC has not occurred, or is not suspected. Materials handled at this AOC include cutting oils, solvents and metal chips. SB09B11-1 was taken from within the area of the former metal chips bin, but not adjacent to the chip sump. BTEX was detected, but at low concentrations (less than 0.001 mg/kg). No detected concentrations in samples from SB09B11-1 are greater than RSR numerical criteria.

AOC 28 - Building B-15 and Associated Satellite Accumulation Areas

Building B-15 was constructed in 1945 as the Oil, Lubrication, Acid, and Alkali Storage Building (see Figure 7-1). Virgin solvents, cleaners, coolants, lubricants, hydraulics, and other oils and collected waste oils were stored in satellite accumulation areas within the building (CDM FPC, 1992). The storeroom located in the western portion of Building B-15 has historically been used as the primary chemical storage area at SAEP. The storeroom was constructed between 1960 and 1970. Additional chemical storage was conducted in the northern-most portion of Building B-15 (W-C, 1991). This portion of Building B-15 is indicated as the Fire Department in a 1956 Fire Insurance Map. The main portion of Building B-15, known as the Oil House, was used for storage, blending, and dispensing of oils. A pumping station for the waste and product transfer system was located within the eastern portion of Building B-15, where both virgin and waste oils were pumped between Building B-15 and the Oil House Tank Farm via subsurface transfer lines (refer to AOC 3 discussion).

Soil borings BR-3, BR-5, SB09B6-1, SB09B7-1, and SB09B8-1 were all collected within 15 feet of Building B-15. Location BR-3 is east of B-15, locations SB09B6-1 (and co-located SG-99-47), SB09B8-1, and BR-5 are north of B-15, and boring SB09B7-1 is west of B-15. Analytes detected in soil samples from these locations include BTEX, cVOCs, SVOCs, PAHs, TPH, and PCBs (see Appendix O for complete soil boring analytical results).

A release has occurred from this AOC. Although solvent and fuel-related contaminants were identified in soil near this AOC, the presence of the fill from former shoreline filling and an outfall once located beneath Building B-15, complicates the determination of a release. Figure 1-4 depicts the locations of these historical site features. Concentrations of TCE and lead in soil exceed RSR criteria (see Figure 7-2).

Building B-13 Former Chemical Storage and Scrap Metal Reclamation

Building B-13, the Scrap Metal and Material Reclamation Building, was constructed in 1944. Building B-13 was used for raw chemical storage (W-C, 1991), however the specific location of this storage area is not known. A room in northwest portion of was used for Magnesium-Thorium recovery. An oil-water separator in the east corner receives stormwater from the concrete pad with collection trench located to the north, and from the concrete bermed storage area with sump located to the east.

The 1956 Fire Insurance Map indicates three scrap bins along the northern wall. Starting in 1993, when the metal chips concrete sump (see AOC 16) was removed, a roll-off container was reportedly used within B-13 to store cuttings prior to recycling (ABB-ES, 1996). A janitorial storage facility was also located within Building B-13, where cleaning supplies were stored (AlliedSignal, 1995).

Titanium and aluminum cuttings were stored in aboveground dumpsters adjacent to an oil collection sump (see Figure 7-1). The sump, located within the Titanium and Aluminum Chips Collection bin, was used to capture oils associated with titanium and aluminum chips. The system was built in the early 1990s and used until approximately 1996.

Soil borings SB09B3-1 and SB09B9-1, and soil vapor sample locations SG-99-14 and SG-99-33 were collected in Building B-13. SG-99-14 was co-located with SB09B3-1 near the center of the building, and SG-99-33 was collected in the southeastern end of the building. Soil boring SB09B2-1 was completed adjacent to the titanium and aluminum chip sump. Analytes detected in soil samples from these locations include BTEX, cVOCs, SVOCs, PAHs, TPH, and PCBs (see Appendix O for complete soil boring analytical results). PCE and TCE were detected in soil vapor samples associated with this AOC.

A release has occurred from this area. Chemical storage, oily metal chip storage, and product and waste transfer have been historically conducted in and near Building B-13. Chlorinated VOCs, fuel related VOCs, other VOCs, fuel-related SVOCs, PAHs, and TPH have been detected at sample locations associated with this area. Elevated PID readings, petroleum odor, and staining were observed during sampling at these locations. Concentrations of PAHs and TPH in soil exceed RSR criteria. Concentrations of PCE and TCE in soil vapor exceed RES and I/C VC.

Container Storage Pad and Collection Trench Northeast of Building B-13

A concrete pad with collection trench built in 1993 is located northeast of Building B-13. The pad was used to store wheeled roll-offs containing metal scrap from general property maintenance. The collection trench includes a drain line to the oil-water separator in Building B-13 to handle oil and rainwater accumulation.

Soil borings BR-2 and SB09B4-1 were collected beneath the concrete pad. Detected analytes include BTEX, cVOCs, PAHs, TPH, and PCBs (see Appendix O for complete soil boring analytical results).

Debris observed in soil boring BR-2 indicates that filling likely occurred in this area as part of the 1943 expansion of the facility. Drums and debris appear to have been stored within this area since at least 1943 (USEPA, 1990). This area is located inland of the 1943 shoreline but may have received fill material during shoreline expansion.

A release from this area has not occurred, or is not suspected. The concrete pad and collection system were built in 1993 and used for a two-year period to store roll-offs containing scrap metal. It is unlikely that significant quantities of oils and/or solvents were contained on or in the scrap metal, released from the roll–offs, and migrated through the new concrete pad. The source of the contaminants detected in soils is likely from historical usage of this area prior to 1993. Concentrations of TCE, numerous PAHs, TPH, antimony, arsenic, beryllium, and lead in soil exceed RSR criteria (see Figure 7-2).

Magnesium-Thorium Scrap Yard Between B-13 and B-44

A scrap yard containing empty drums and drums full of magnesium thorium chips was identified between Building B-13 and B-44 during a visual site inspection in December 1991 (CDM FPC, 1992). At least 10 empty dumpsters were noted in this area and the concrete was cracked. This area was shown as fenced in a 1993 site plan (Textron-Lycoming, 1993). Discussions with facility personnel indicate that this area was used for storage of off-specification non-thoriated metal parts.

A release from this area has not occurred, or is not suspected. Soil boring SB08L1-8 was completed within the concrete bermed area within the former scrap yard. BTEX, other VOCs, and PCB 1260 were detected in both sample intervals; however, the source of the detected contaminants is likely from historical usage of this area for storage of drums and debris since at least 1943 (USEPA, 1990). No concentrations detected in samples associated with this area were greater than RSR numerical criteria.

Open Storage Area Between Buildings B-16 and B-74

This area consists of open storage conducted between Building B-16 and B-74, which currently houses the OATP and associated treatment buildings and pump stations. Open storage of containers, crates, tanks, and/or drums appears to have occurred since the early 1950s until the 1980s (based upon available aerial photography). Nearly all of this area is located within the portion of the property that was filled in 1943. In 1953, stormwater pump stations B-37 and B-38 (W-C, 1991), and later, in 1976, the OATP, including Building B-64-2, B-64-1, and associated structures were constructed. In 1971, Building B-39 located in the area currently occupied by B-64-1 was demolished. In 1980, Building B-35 located adjacent to the tank farm was also removed.

A 1949 aerial photograph shows dark staining throughout this area. Debris and small areas of dark staining or pooled liquid are present in the central portion of this area on a 1953 aerial photograph. A 1956 Fire Insurance Map indicates the eastern portion of this area was used to stage metal engine containers. The 1953 aerial photograph and a 1956 Fire Insurance Map depict three 1800-gallon propane ASTs at the future location of the OATP (Building B-64-2). Aerial photography from 1960 indicates dark staining or pooling adjacent to Building B-38 and mounded material, potentially debris and/or drums, in the future location of the OATP. A 1970 aerial photograph depicts storage tanks adjacent to Building B-37 and three small buildings located between Building B-38 and the three 1800-gallon propane above ground storage tanks (ASTs).

Soil borings HA-99-07, SB09A1-1, SB09A1-2, SB09A1-3, SB09A1-4, SB09A2-1, SB09C2-1, SB14A2-1, SB14A2-2, SB17A4-1, and WC-5S, and soil vapor sample location SG-99-15 were collected in this open storage area. Many of these borings are located around the area containing Buildings B-37, B-38, and the OATP, near the dike.

Open storage of containers, crates, tanks, and/or drums appears to have occurred since the early 1950s. Various small former buildings, structures, and ASTs have existed within this area. Much of this area is either paved with asphalt or concrete, as thick as 10 inches in places. The OATP, constructed in 1976 to remove oil and grease, as well as 1,1,1-TCA used in Building B-16, (CDM FPC, 1991) from wastewater and stormwater prior to discharge to the Housatonic River, and associated pump stations and underground piping, currently exist in many of the former areas of historical open storage. The highest detected PID readings were from samples SB09A1-3 and SB09A1-4, taken near the dike between pump stations B-37 and B-38. The only noted odors for samples associated with this area, a strong petroleum odor, were at SB09A1-3. Furthermore, the majority of this area was constructed of fill in 1943.

The determination of a release from this potential release area is not possible due to the presence of contamination from other potential sources. Chlorinated VOCs, BTEX, other VOCs, SVOCs, PAHs, PCBs, and TPH were detected in soil samples associated with this area. These samples are spread across the extent of this area, and provide representative data for the majority of the identified former storage areas. Detected analytes in samples associated with this area may have resulted from these storage areas, ASTs historically located within this area, fill used in 1943 to extend the shoreline into the Housatonic, and/or as a result of activities associated with operation of the OATP. Concentrations of BTEX, cVOCs, PAHs, TPH, PCBs, and inorganics in soils exceed RSR criteria (see Figure 7-2). Concentrations of cVOCs and arsenic in groundwater exceed RSR criteria (see Figure 7-2).

7.2 CHEMICAL WASTE TREATMENT SYSTEM

The Chemical Waste Treatment System includes collection sumps, waste lines, pump stations, and treatment facilities associated with the former metal treating and plating operations.). AOCs included in this group include:

- AOC 8 – CWTP Collection System, Pump Station (Building B-63), and Associated Piping
- AOC 9 – Chemical Waste Treatment System: Cyanide Destruction Facility
- AOC 10 – CWTP in Building B-18
- AOC 11 – CWTP Solids Handling Area in Building B-71
- AOC 14 – Container Storage Areas A and B
- AOC 15 – Sludge Roll-off Container Area north of Building B-71
- AOC 18 – Equalization Impoundment (Lagoon #1)
- AOC 19 – Sludge Drying Beds (Lagoons #2, #3, and #4)
- AOC 25 – Outfall-008 (OF-008) and Drainage Ditch
- AOC 43 – Former Underground Storage Tank (UST) at Building B-18

Electroplating, corrosion resistance treatments, heat treating, cleaning, and anodizing processes resulted in the generation of wastewater requiring treatment and disposal. The success processes depends largely on the degree of cleanliness of the receiving metal prior to the application of the coating, the plating process was associated with stripping and cleaning processes. The stripping process was completed with acids such as chromic acid, sulfuric acid, and hydrochloric acid and cleaning agents such as 1,1,1-TCA, Freon-based solvents, Varsol solvent, alcohols, MEK, naphtha, various thinners (e.g., 1,4-dioxane and toluene), and alkaline cleaners (e.g., permanganate salts and sodium hydroxides) historically were used to complete parts cleaning (INEL, 1991).

As originally designed, the CWTP had bentonite-lined equalization lagoon, into which untreated wastewater was discharged. From the equalization lagoon, wastewater was transported to the CWTP building (Building B-18) were the first step of the treatment processes involved alkaline chlorination for cyanide oxidation with the addition of lime slurry and chlorine followed by diffused aeration. The second treatment step included the addition of sulfuric acid (to lower pH) and sodium metabisulfite to reduce Cr(VI) to trivalent chromium. Lime slurry and ferrous sulfate were then added to promote coagulation of metal hydroxide precipitates. Sedimentation of the precipitates occurred in the clarifier (ESE, 1981). Sludge generated by the treatment processes were transferred by underground piping to one of three on-site, unlined lagoons. Clarified water from the treatment process was originally discharged to a tidal ditch located east of Building B-18, briefly discharged to the OATP (approximately 1976 to 1979), and subsequently discharged to OF-008 constructed within the tidal ditch in 1979 (ESE, 1981 and USEPA, 1990).

From 1958 until 1986, wastewaters from chemical tanks and rinse water tanks in the chrome room were poured into floor drains and transferred to the Building B-63 sump and ultimately to the CWTP (ESE, 1981). Wastewaters from Building B-3A were transferred to a tank at the vault and pump station between Building B-6 and Sniffens Lane, where they were combined with wastewaters from Building B-6 and transported to the CWTP.

After the system upgrade, cyanide-contaminated wastewater was piped to CDF, where it was treated by alkali chlorination and converted to nitrogen and bicarbonate. Effluent from this process was combined with other wastewaters at the Building B-63 pump house and transferred to equalization tanks at the CWTP. Treatment at the CWTP involved chromium reduction, precipitation of chromium and other heavy metals, and clarification. Effluent from the clarifier passed through sand filters before final discharge at OF-008. Sludge from the clarifier was dewatered by a sludge thickener and filter press in Building B-71. The filter cake was stored in a roll-off container near Building B-71 prior to disposal off-site, and the filtrate was returned to CWTP for further treatment.

Figure 7-3 presents the locations and boundaries of the AOCs within this group, as well as the explorations which have been associated with the group. Table 7-3 presents a summary of samples associated with each AOC, the field observations, detected analytes, release determination, and analytes with concentrations exceeding CTDEP RSR criteria. Data analysis procedures are discussed in Section 7.0.

AOC 8 - CWTP Collection System, Pump Station (Building B-63), and Associated Piping

The CWTP collection system, received wastewater from electroplating, cleaning, and anodizing processes in Buildings B-2, B-3 B-3A, and B-6. AOC 8 is shown on Figure 7-3 and includes all lines, sumps and pump stations used to transfer wastewater to Building B-18.

Building B-3A cyanide-containing wastewaters were transferred to the cyanide tank and chrome-containing wastewaters were transferred to the chrome tank at the vault and pump station between Building B-6 and Sniffens Lane, where they were combined with the corresponding wastewaters from Building B-6. Cyanide-containing wastewater was pumped back to the CDF for treatment then discharged to Building B-63, to be pumped to the CWTP equalization tanks; chrome-containing wastewater was discharged directly to the CWTP equalization tanks.

Reportedly, the CWTP collection system piping was replaced in the 1980s. No documentation related to the rationale for piping replacement, the procedures followed, or potential sampling conducted during piping replacement has been identified.

Exploration locations SB12B1-1 and SB12B4-1 were completed near Building B-63, the pumping station. Explorations SB19A1-4, SB19A1-5, SB19A1-6, and LW-12 were completed adjacent to the CWTP waste line in the South Parking Lot. Explorations SB12-3, SB24D1-1, and WC-14S were completed near the CWTP waste line adjacent to Building B-6. Explorations WC-10S and SB12A1-1 were completed near the CWTP waste line between Buildings B-2 and B-13. Explorations WC-12S and SB12E1-1 were completed immediately downgradient of the CWTP waste line between Building B-2 and B-12. Analytes detected in soil samples included BTEX, VOCs, PAHs, SVOCs, PCBs, and TPH (see Appendix O for complete soil boring analytical results). Analytes detected in groundwater samples included cVOCs (see Appendix P for complete groundwater sampling results).

The CWTP collection system, and associated pump stations and waste lines handled wastewater resulting from metal finishing, electroplating, cleaning and anodizing processes in Buildings B-2, B-3, B-3A, and B-6 for nearly 40 years. These processes resulted in wastewater containing chlorinated and non-chlorinated solvents, other cleaners (e.g., alcohols, MEK, naphtha, various thinners [including 1,4-dioxane and toluene], and alkaline cleaners), and dissolved heavy metals. Samples associated with this AOC were collected near Building B-63 (the pumping station), the CWTP waste line in the South Parking Lot, the CWTP waste line near Building B-6, and the CWTP waste line between Buildings B-2 and B-13.

The determination of a release from this AOC is not possible due to the presence of contamination from other potential sources. Chlorinated and non-chlorinated solvents, and fuel-related contaminants were detected in these samples. Sampling results indicate releases have occurred; however, evidence suggests there are other potential sources in addition to this AOC. No visual or olfactory observations of contamination or elevated PID readings were observed during sampling of explorations adjacent to the CWTP collection system. Fuel and oil storage in USTs and ASTs and the wide-spread use of solvents in cleaning procedures within Building B-2 are likely contributing sources of contamination. Concentrations of antimony, arsenic, cadmium, copper, lead, and TPH in soils exceed RSR criteria (see Figure 7-4). Concentrations of copper, zinc, cyanide, and cVOCs in groundwater exceed RSR criteria (see Figure 7-4).

AOC 9 - Chemical Waste Treatment System Cyanide Destruction Facility

The CDF was located inside Building B-70 (CDM FPC, 1992). The CDF was constructed in 1986, near the northeast corner of Building B-2, adjacent to the former B-2 Chromium Plating Facility. Cyanide-bearing wastewater was generated as a result of electroplating processes, primarily copper and cadmium plating (ESE, 1981) (see Table 1-2). The CDF, a subsystem of the CWTP, pre-treated cyanide-bearing wastewater in two 5,000-gallon tanks prior to combination with other wastewaters. Cyanide-bearing wastewater piped to the CDF was treated by alkali chlorination in two stages. In the first stage, sodium hypochlorite was used to produce cyanate at a pH of 10 to prevent the formation of toxic cyanide gases. The second stage of treatment included the conversion of cyanate to bicarbonate and nitrogen using a continuation of the alkaline chlorination process. Effluent was pumped to equalization tanks at the CWTP where it was combined with wastewaters from the Building B-63 pumping station (INEL, 1991). The CDF operated from 1986 to 1997. Prior to CDF construction, this area contained an abandoned underground septic tank that reportedly received zinc chromate paint sludge and solvent from 1941 to 1949 (ESE, 1981).

Two soil borings SB12B6-1 and SB12B6-2 were completed adjacent to Building B-70 and Building B-63. Analytes detected in these soil samples included BTEX, VOC, PCBs, and TPH (see Appendix O for compete soil boring analytical results). Both samples were analyzed for total cyanide, but none was detected.

A release from this AOC has not occurred, or is not suspected. Materials handled at the CDF included cyanide-bearing wastewater from copper and cadmium plating operations, as well as sulfuric acid, sodium hydroxide, and sodium hypochlorite used in the treatment process. Cyanide was not detected in samples taken adjacent to the CDF and the upstream waste line, nor was copper or cadmium detected at elevated concentrations. Solvent and fuel-related contaminants detected in soil are likely the results of historical activities in this area, including fuel oil storage in USTs, painting and paint storage, waste paint storage and disposal, and open storage. The concentration of arsenic in soil exceeds RSR criteria at SB12B6-2 (see Figure 7-4).

AOC 10 - CWTP in Building B-18

The CWTP in Building B-18 includes the Chrome Reduction Unit and the Metals Removal Unit (CDM FPC, 1992). The Chrome Reduction Unit consists of six 9,725-gallon tanks located along the southeastern side of Building B-18. The Metals Removal Unit consists of one 240,000-gallon and two 120,000-gallon equalization tanks, and a 60,000-gallon clarifier. The equalization tanks are located west of Building B-18; the clarifier adjacent to the northern end of Building B-18. The CWTP was constructed in 1958, and included the Chrome Reduction Unit and clarifier. In 1986 the equalization tanks were constructed, which replaced the equalization lagoon

The CWTP received wastewater from electroplating, cleaning and anodizing processes in Buildings B-2, B-3, B-3A, and B-6. The wastewaters contained heavy metals, solvents, and acids. Prior to construction of the CDF (AOC 9) in 1986, cyanide-bearing wastewaters were also treated at the CWTP. Chemicals used in treatment processes at the CWTP included sulfuric acid to reduce pH in the equalization tanks, sodium metabisulfite to reduce chromium, and sodium hydroxide.

Soil borings EBS43-1, SB20A1-1, SB20A1-2, and SB20A1-3 were collected from adjacent to the CWTP in Building B-18 and the associated clarifier, equalization tanks, and chrome reduction tanks. Analytes detected in these soil samples include BTEX, cVOCs, VOCs, SVOCs, and PCBs (see Appendix O for complete soil boring analytical data).

A release from this AOC has not occurred, or is not suspected. The CWTP received chemical wastewaters containing heavy metals, solvents, and acids from electroplating, cleaning and anodizing processes in Buildings B-2, B-3, B-3A, and B-6. Low levels of fuel-related contaminants and dichloromethane were detected in soils adjacent to the CWTP and associated structures. Heavy metals were not detected at elevated concentrations. The concentration of dichloromethane in soil exceeds RSR criteria at EBS43-1 (see Figure 7-4).

AOC 11 - CWTP Solids Handling Area in Building B-71

AOC 11 consists of the Solids Handling Area, located in Building B-71, which includes an 8,000-gallon FRP thickening tank and two 1-cubic yard filter presses (CDM FPC, 1992). Operation of the CWTP Solids Handling Area began in 1986. Metal hydroxide sludge precipitated in the clarifier of the CWTP was pumped to the thickening vessel and dewatered with a filter press. Supernatant from this process was routed to the equalization tanks for further treatment and the filter cake was removed from the filter press and transferred to an adjacent roll-off container (AOC 15), located immediately northwest of Building B-71 (CTDEP, 1986).

Soil boring EBS11-1 was completed beneath Building 71. Soil samples from EBS11-1 were analyzed for SPLP inorganics only (see Appendix O for complete soil boring analytical results). Elevated concentrations of inorganics were not identified.

A release from this AOC has not occurred, or is not suspected. Metal hydroxide sludge resulting from the clarification and dewatering of plating wastewater was managed in the CWTP Solids

Handling Area. Elevated concentrations of inorganics were not detected in samples collected from soil boring EBS11-1. No contaminants were detected above RSR criteria in EBS11-1.

AOC 14 - Container Storage Areas A and B

Container Storage Areas A and B are located along the southern property boundary, south of Building B-18. These areas were used from 1983 to 1986 for less than 90-day hazardous waste accumulation prior to off-site disposal. Containerized liquid and solid wastes, typically in 55-gallon drums, were collected from locations at the facility and brought to these storage areas. The containers were reportedly segregated based on waste compatibility. The storage areas consist of sloped concrete slabs, each approximately 12 feet wide and 22 feet long. Container Storage Areas A and B had a combined storage capacity of 2,750 gallons (CDM FPC, 1992). Each has a separate roofed area, with containment dikes consisting of low concrete block walls. Containers of the following waste materials were accumulated in these areas: waste jet fuel, paint, waste acetone, waste sodium hydroxide, waste 1,1,1-TCA, waste oil, and chromium-contaminated plating wastes (Honeywell, 2000).

During a 1985 RCRA Compliance Inspection, a total of 34 drums were located outside the containment areas. Many of these drums were unidentified. Forty-two drums were located within the “general” waste containment pad, at least four of which needed sealing or covers. Several of the drums were unlabelled and some were rusted. At this time the pad was full of water. The “cyanide” waste pad contained seven drums of unlabelled cyanide or alkali waste (CTDEP, 1985). During a 1986 RCRA Compliance Inspection, one of the areas was placarded as “Acid Wastes Storage” and contained 23 drums, seven of them open. The other storage area was labeled “Cyanide Storage Area”. Twenty-one drums were contained in this area; two of them were open (CTDEP, 1986). The storage areas were not used after 1986 (CDM FPC, 1992). It is not known which of these storage areas is designated Container Storage Area A and which is B.

Soil borings SB20A2-1, SB20A2-2, and EBS14-1 were completed adjacent to the storage areas. No samples were collected from beneath the concrete pad. Analytes detected included the SVOC bis(2-ethylhexyl) phthalate and PCBs (see Appendix O for complete soil boring analytical results). Total cyanide was analyzed for in samples taken from SB20A2-1 and SB20A2-2; none was detected.

There is insufficient information to determine if a release has occurred from this AOC. Waste jet fuel, paint, waste acetone, waste sodium hydroxide, waste 1,1,1-TCA, waste oil, and chromium-contaminated plating wastes were handled at this AOC. No solvent or fuel-related contamination or cyanide was detected in soil samples collected from outside the perimeter of the storage area; however, no samples were collected from beneath the concrete pad. The phthalate detected in boring EBS14-1 was not detected in adjacent boring SB20A2-2 and is likely a laboratory artifact.

AOC 15 - Sludge Roll-off Container Area North of Building B-71

AOC 15 consists of a roll-off container located immediately northwest of Building B-71 (CTDEP, 1986). Dewatered sludge from the CWTP Solids Handling Area (AOC 11) was stored in the roll-off for a period of less than 90 days prior to off-site disposal, from 1986 until the facility ceased operation. The roll-off container consisted of a standard, over-the-road 20 cubic yard, steel box covered with a tarp; the container was located on a concrete slab equipped with a concrete containment berm (Honeywell, 2000).

A release from this AOC has not occurred, or is not suspected. No samples were taken from within the area of AOC 15, but the AOC was contained within a concrete berm and sludge material was stored in the roll-off for a period of less than 90 days.

AOC 18 - Equalization Impoundment (Lagoon #1)

This AOC consists of the former Equalization Lagoon (Lagoon #1), located east of Building B-18. The Equalization Lagoon had an approximate capacity of 480,000 gallons (CDM FPC, 1992). The lagoon operated from 1958 to 1986.

The AVCO-Textron Closure/Post-closure Plan for the lagoons was submitted and approved in September 1987 and subsequently amended in September 1987, January 1988, and February 1988 (AVCO-Textron, 1992a). Removal actions were conducted at the former equalization lagoon later that year, consisting of the excavation and removal of contaminated soil to the low tide water level. The lagoon closure was not considered a “clean closure” because residual contamination was left in place. USEPA regulations require post-closure monitoring and reporting for 30 years after closure of a hazardous waste management unit, and annual groundwater monitoring reports have been submitted by the facility since 1983. Closure of the equalization lagoon was completed in the summer of 1989 (W-C, 1991), and RCRA Closure Certification was submitted to the CTDEP and USEPA by AVCO-Textron in 1992 (AVCO-Textron, 1992b).

The AOC has been closed under RCRA Subtitle C, and a post-closure groundwater monitoring program is being conducted. Refer to Subsection 1.6.1 for further information.

A release has occurred from this AOC. Further soil data is necessary to compare contaminant concentrations in soil to RSR criteria. Additionally, LNAPL has been detected in monitoring well LW-5S, and additional investigations are planned for delineation of the extent of the LNAPL.

AOC 19 - Sludge Drying Beds (Lagoons #2, #3, and #4)

This AOC consists of the three former Sludge Drying Lagoons (Lagoons #2, #3, and #4) located east of Building B-6. Lagoon #2 was 8 feet deep with an approximate 547,000-gallon capacity, lagoon #3 was 6.5 feet deep with an approximate 385,000-gallon capacity, and lagoon #4 was 8 feet deep with an approximate 754,000-gallon capacity (CDM FPC, 1992). The lagoons operated from 1958 to 1986.

The AVCO-Textron Closure/Post-closure Plan for the lagoons was submitted and approved in September 1987 and subsequently amended in September 1987, January 1988, and February 1988 (AVCO-Textron, 1992a). The Closure Plan estimated that the lagoons contained 6,600 cubic yards of sludge and 10,300 cubic yard of contaminated soil surrounding and underlying the lagoons (CDM FPC, 1992). Removal actions were conducted at the former sludge drying lagoons later that year, consisting of the excavation and removal of contaminated soil to the low tide water level. Evidence of petroleum contamination was discovered during excavation of the sludge drying lagoons; stained soils were reported near the location of Building B-72, but no action was taken at that time to remove these soils. The lagoon closure was not considered a "clean closure" because residual contamination was left in place. Additionally, USEPA regulations require post-closure monitoring and reporting for 30 years after closure of a hazardous waste management unit, and annual groundwater monitoring reports have been submitted by the facility since 1983. Closure of the equalization lagoon was completed in the summer of 1989 (W-C, 1991), and RCRA Closure Certification was submitted to the CTDEP and USEPA by AVCO-Textron in 1992 (AVCO-Textron, 1992b).

A release has occurred from this AOC, however this AOC has been closed under RCRA Subtitle C, and a post-closure groundwater monitoring program is being conducted. Further soil data is necessary to compare contaminant concentrations in soil to RSR criteria. Refer to Subsection 1.6.1 for further information.

AOC 25 - Outfall-008 (OF-008) and Drainage Ditch

This AOC consists of discharge to the Housatonic River at OF-008 and the associated drainage channel (CDM FPC, 1992). OF-008 was used to discharge supernatant from the CWTP clarifier to the drainage channel immediately northeast of Building B-18 and ultimately to the Housatonic River. The outfall was constructed in 1979. The facility's 1985 NPDES permit allowed a discharge of 190,000 gallons per day of treated finishing wastewater from the outfall, and in 1991, the renewed NPDES permit allowed the facility to discharge 123,840 gallons per day of metal finishing wastewater from cyanide and chromium plating operations via the outfall.

Records indicate that frequent violations of permit limitations (e.g., elevated pH levels, heavy metals concentrations exceeding permitted levels, and discharges exceeding the allowed maximum daily flow) occurred prior to the mid 1980s. Violations occurred after that time with less frequency (W-C, 1991). During a 1984 USEPA inspection, white foam was observed where lime-green colored liquids were being discharged from the CWTP clarifier into the tidal basin (CDM FPC, 1992). A review of the monthly Discharge Monitoring Reports for 1990 identified violations of permit limitations for average daily flow and maximum daily concentration limits for nickel, cyanide, and total toxic organics (CDM FPC, 1992). Elevated levels of chlorinated VOCs, fuel-related VOCs, and other VOCs were detected during required NPDES Permit sampling (W-C, 1991).

Sediment samples OF-008 (SD), 008 (SD), and SD08001A were collected within the tidal channel downgradient of OF-008. Analytes detected in sediment include VOCs, PAHs, SVOCs, PCBs, and inorganics (refer to Appendix Q for complete sediment sample analytical results).

A release has occurred from this AOC. Elevated concentrations of VOCs, PAHs, SVOCs, PCBs, and inorganics were identified in sediment impacted by discharges from OF-008. As there are no RSR criteria for sediment, no comparisons were performed.

AOC 43 - Former UST at Building B-18

This AOC consists of a 1000-gallon #2 Fuel Oil UST, located adjacent to Building B-18 until its removal in 1989 (CDM FPC, 1992). The UST was installed in 1956 when Building B-18 was constructed.

Soil boring location SB20A1-1 was completed immediately adjacent to the former UST. In this boring, a sample was collected from near the surface of the water table. No visual or olfactory contamination was identified. Analytes detected in soil at SB20A1-1 include the SVOCs bis(2-ethylhexyl) phthalate, diethyl phthalate (see Appendix O for complete soil boring analytical results).

A release from this AOC has not occurred, or is not suspected. This AOC handled petroleum products, and fuel-related contaminants were not detected in SB20A1-1. No contaminants were detected above RSR criteria.

7.3 MANUFACTURING, PLATING, RESEARCH AND DEVELOPMENT, AND TESTING

Manufacturing, Plating, Research and Development, and Testing areas include the majority of the operations conducted at the SAEP facility. For purposes of discussion and evaluation, this group has been divided into the Manufacturing and Plating Area, located in Buildings B-2, B-3, and B-10, the Research and Development Area located in Buildings B-3, B-3A, B-4, B-6 and B-6A, and the Testing Area, located in Buildings B-5, B-16, and B-19.

7.3.1 Manufacturing and Plating Areas

The Manufacturing and Plating Areas are located in Buildings B-2, B-3 and B-10. Past manufacturing operations in these buildings included machining, electroplating, corrosion prevention, cleaning, miscellaneous surface treatments, and painting.

These operations used a variety of materials that eventually became waste, either by degradation of their performance characteristics or by mixing with other materials. Although information regarding operations and chemical material usage prior to the late 1970s is limited, it is assumed that many of the types of chemicals used did not change significantly over time. Large quantities of acids, oils, and solvents were used during manufacturing and plating operations, which required the storage, treatment, and disposal of both raw and waste materials. These materials were stored in aboveground and underground tanks, as well as in container storage areas, satellite accumulation areas, and open storage areas. After construction of the CWTP, contents of chemical tanks (e.g., acids) and rinsewater tanks used in the electroplating process were transferred to the CWTP.

AOCs and other potential release areas within this group include:

- AOC 22 - Waste Paint Tank Located Between Buildings B-2 and B-3
- AOC 26 – Former Septic Systems beneath Building B-2 and Potentially Throughout the Facility
- AOC 37 – Building B-10 and Associated Satellite Accumulation Areas

- AOC 38 – Building B-2 Former USTs
- AOC 49 – Building B-2 Manufacturing Areas
- AOC 50 – Building B-2 Plating Area
- AOC 51 – Building B-3 Plating Area
- Former Gasoline USTs near Building B-10

Figure 7-5 presents the locations and boundaries of the AOCs within this group, as well as the associated explorations. Table 7-4 presents a summary of samples associated with each AOC, the field observations, detected analytes, release determination, and analytes with concentrations exceeding CTDEP RSR criteria.

AOC 22 - Waste Paint Tank Located Between Buildings B-2 and B-3

From approximately 1941 to 1949, paint and solvents from the paint shop, located in the northeast corner of Building B-2, reportedly were piped to a septic tank located between Buildings B-2 and B-3. Most of the paint reportedly used during this period was zinc chromate primer (ESE, 1981).

A shop drawing dating from approximately 1950 describes a septic tank receiving overflow from the paint spray shop through a 12-inch tile pipe. The tank is shown near the current site of Building B-70 (ESE, 1981). This tank coincides with a utilities drawing from 1998 that shows a septic tank northeast of the south addition to Building B-2 (Allied Signal, 1998). The 1943 fire insurance map indicates “three 2-inch lines to paint pits” in this area but no pits are shown. (AFM FIC, 1943)

During the 1998 RI soil boring SB12B5-1 was placed between Building B-2 and B-3 to investigate the potential impacts from this tank (W-C, 1998). This boring was not installed in the location of the septic tank.

There is insufficient evidence to determine if a release has occurred from this AOC. No borings were collected at the suspected tank location.

AOC 26 - Former Septic Systems

AOC 26 consists of the former septic systems located beneath Building B-2 and potentially all over the facility The RFA locates this AOC within the central portion of Building B-2 (CDM FPC, 1992), but no documentation has been found that verifies the presence of a septic tank or system in this area.

The PAS indicates a former septic system was removed from beneath the southeastern portion of Building B-2 in 1969 (W-C, 1991). Another septic tank is identified in an undated site plan adjacent to the southeast corner of Building B-10, in close proximity to two former gasoline USTs identified in this area (AVCO, undated). Two additional septic tanks are identified on fire insurance maps from as early as 1943 along the northern side of Building B-2. The first of these tanks is shown adjacent to the current boiler room; the other northwest of the 400,000 gallon fire-water tank, in the area of the later constructed iron and steel center of Building B-2 (see Figure 1-4) (AFM FIC, 1943). Other identified septic tanks include one between the northernmost corner of Building B-3 and Building B-7A and one adjacent to the clarifier at the CWTP (Allied Signal, 1998).

Soil boring SB13I1-1 is located adjacent to the former septic tank identified near Building B-10, SB12E2-1 is located adjacent to the former septic tank identified adjacent to the boiler room, and SB08G1-1 was taken to characterize a former septic system beneath the central portion of Building B-2 (see Table 7-4). Analytes detected in soil at these soil borings included BTEX, PCBs, and VOCs (see Appendix O for complete soil boring analytical results).

A release from this AOC has not occurred, or is not suspected. Benzene, toluene, and methyl isobutyl ketone were detected in soil samples near identified septic tanks at the facility; however, fuel storage was conducted adjacent to sample location SB13I1-1 where benzene was detected and SB12E2-1 where toluene was detected. Maximum BTEX concentrations were less than 1 mg/kg. Methyl isobutyl ketone was detected at all three soil boring locations near identified septic tanks; detected concentrations were 2 mg/kg or less. No contaminants were detected above RSR criteria.

AOC 37 - Building B-10 and Associated Satellite Accumulation Areas

The eastern portion of Building B-10 was constructed as the Service Building in 1929, and later re-named as Building B-10. The western portion of this building was built in 1940 for storage and later re-designated Building B-11. In 1985, Buildings B-10 and B-11 were joined to form one building, resulting in the current footprint of Building B-10 (W-C, 1991). Most recently, the building was used for the production and maintenance of tank engine exhaust gas recuperators, including machining and surface treatment (e.g., plating).

Three soil borings SB13F1-1, SB13G1-1, and SB13H1-1 were completed in Building B-10 adjacent to pits, drain lines and trenches (Table 7-4). Detected analytes include VOCs, SVOCs, and PCBs (see Appendix O for complete soil boring analytical results).

Monitoring wells WP-99-27, WP-99-31, and WP-99-43 were completed beneath Building B-10. Detected analytes included cVOCs and hexavalent chromium. However, concentrations of these analytes are present beneath Building B-10 as a result of the migration of the contaminants from beneath Building B-2 and the former B-2 Chromium Plating Facility.

A release from this AOC has not occurred, or is not suspected. Acetone, methyl isobutyl ketone and methyl n-butyl ketone were the only VOCs detected in soil from samples taken beneath Building B-10. There were no elevated PID readings or visual or olfactory observations of contamination during soil sampling at these locations. The concentration of arsenic in soil at SB13G1-1 exceeds RSR criteria by several orders of magnitude (see Figure 7-6). Concentrations of cVOCs, chromium, and hexavalent chromium exceed RSR criteria beneath Building B-10.

AOC 38 - Building B-2 Former USTs

AOC 38 is comprised of the Building B-2 former USTs, including two 2,500-gallon oil USTs and a 1,500-gallon sanitary UST. The RFA indicates the dates of use for the oil USTs to be until 1955, when the tanks were abandoned in place underneath Building B-2. The 1,500-gallon septic tank was reportedly filled with sand and abandoned in place in 1969 (CDM FPC, 1992). The exact location of these tanks is not known. In addition, five other fuel storage tanks have been identified at Building B-2.

Two 5,000-gallon fuel oil USTs are identified on a 1931 fire map (AFM FIC, 1931). The tanks were located near the southern end of current Building B-70. These tanks are also depicted on fire maps from 1937, 1941, and 1943 (AFM FIC, 1937, 1941, and 1943). The current status of these tanks is unknown.

A fire map from 1941 depicts a "Fuel Oil Tank House", consisting of a 10,000-gallon and a 15,000-gallon fuel oil tank, located in the area between Buildings B-2 and B-12 (AFM FIC, 1941). A 1943 fire map still depicts these tanks, but indicates they were not in use, which is supported by the depiction of a coal silo immediately northwest of these tanks, indicating a change in fuel source for facility heating (AFM FIC, 1943). On a 1956 fire map, a boiler room is shown in the area of the southern-most tank; the other tank is identified as having been buried in sand (FIA, 1956). On a 1986 fire map a 10,000-gallon diesel tank is identified in the area of the remaining tank (FIA, 1986). The current status of this tank is not known.

A fire map from 1943 depicts a 500-gallon gasoline tank in Building B-42 (AFM FIC, 1943). The tank is no longer depicted on a 1956 fire map (FIA, 1956).

There is insufficient information to determine if a release has occurred from this AOC. The exact location of this AOC is unknown. Furthermore, no sample locations were taken proximal to other identified fuel storage tanks at Building B-2.

AOC 49 - Building B-2 Manufacturing Areas

The central portion of Building B-2 was constructed for use as a Factory Building in 1929 as part of the original site development. In the early 1940s, the building was quadrupled in size with an expansion to the west, the addition of employee facilities along South Main Street, and an addition to the east. Also during this time, three buildings were constructed adjacent to the north side of the Factory Building, including a Painting and Sand Blast Building, the Hammer Shop, and the Deflector Shop (currently the former B-2 Chromium Plating Facility). In 1944, another expansion of the building was completed to the north and west and modifications to the building enclosed various abutting buildings under one roof. The result was the current footprint for Building B-2, which is approximately 20 acres in size.

Engine manufacturing and plating operations began after government purchase of the facility in 1951. Raw materials used during manufacturing operations were received at loading docks along the northwest end of the building and finished products were shipped from the southeast end. In general, machining and assembly were completed in the west-central portions of the building and surface finishing (e.g., plating and painting) was completed in the east-central portions of the building. A boiler room, located along the northeastern building wall, contained three boilers and provided heat for the complex. The building also housed a cafeteria, mail room, security and guard house, x-ray room, and on-site incinerator. Ash and cinders were reportedly observed during foundation excavations under the north end of Building B-2 (W-C, 1991).

The following paragraphs summarize the primary operations conducted at Building B-2 during facility operations. Subsection 1.5 provides additional information of facility operations.

<u>General Machining.</u> Machining processes included broaching, drilling, grinding, lathing, stamping, and forging, for the fabrication of gears, shafts, headers, discs, blades, and other metal parts. These processes involved high-pressure, metal-on-metal contact between tools and work pieces. Most of these processes were associated with production work done in the central portion of Building B-2.

During the machining process, a variety of cutting oils and coolants were used to reduce surface friction, cool the tools and work pieces, and remove metal chips from the work surface. Lubricating oils, greases, and hydraulic fluids were used for lubricating internal machine parts and transferring hydraulic energy, and a fluorescent metal penetrant was used to aid in the inspection of finished pieces for hairline fractures and other defects.

Prior to the early- to mid-1980s, waste coolants, oils, and penetrants were disposed of into the stormwater treatment system, likely via floor drains in machining areas of Building B-2. These lines drained waste materials directly to pumphouse B-37 and the OATP for treatment and disposal (CDM FPC, 1992). After this time, waste coolants, oils, and penetrants were removed by vacuum cart and disposed of as hazardous waste oil or as halogenated solvent waste if the material was mixed with a solvent, which was historically used for cleaning in the machining process. These wastes were dumped into concrete pits west of Building B-13, and transferred via subsurface piping (CDM FPC, 1992) to the former Oil House Tank Farm for storage (W-C, 1991).

<u>Corrosion Resistance Processes.</u> The corrosion of metals is a major cause of failure for metallic parts, due to a loss of mechanical strength or ductility. Various surface treatment technologies were used at SAEP to impart corrosion resistance to metallic surfaces. These processes were mostly completed in the northeastern portion of Building B-2 near the former B-2 Chromium Plating Facility.

Corrosion resistance operations included anodizing, chemical conversions coatings, HAE anodic finishing for magnesium, black coating, steel passivation, painting, plasma spray, and slushing oil use. Most of the operations involved cleaning using a solvent such as 1,1,1-TCA, alkaline cleaners, MEK, acetone, or toluene. The operations then involved electrical placement of the coating or immersion of the receiving part in acidic solutions. Slushing oil was used for temporary corrosion protection, providing a thin coating on the metal that could be easily removed for further processing at a later time.

Prior to construction of the chemical waste treatment system in 1958, acid and chemical wastes were neutralized in processing tanks and discharged to the storm water system, which ultimately drained, without further treatment, to the Housatonic River. After 1958, chemical tanks (e.g., acids) as well as water rinsing tanks were routed to the CWTP or the CDF for treatment, via the chemical waste collection system.

Waste solvents were removed by vacuum cart and pumped directly into a waste solvent tank at the former Oil House Tank Farm.

Cleaning. Cleaning was a necessary part of electroplating, corrosion prevention, and heat treatment processes and was widely conducted in Buildings B-2 and B-3 (INEL, 1991). Cleaners used throughout SAEP have included:

- chlorinated solvents – 1,1,1-TCA was the most predominantly used; however, Freon-based solvents, carbon tetrachloride, TCE, and other solvents historically may have been used.
- non-chlorinated solvents - alcohols, MEK, acetone, paint thinners (e.g., 1,1-dioxane and toluene), and mineral spirits (e.g., Varsol).
- alkaline cleaners - composed of alkaline salts such as potassium or sodium hydroxides and sequestering agents, dispersants, and wetting agents, were used to remove greasy materials.
- acids - Five acids were most widely used, including hydrochloric, sulfuric, nitric, chromic, and hydrofluoric acids.
- abrasives - sand, glass beads, and grit.
- miscellaneous cleaners - included vapor blast cleaning, anodic cleaning, periodic reverse cleaning, etching, and general facility maintenance.

Spent chlorinated solvents were removed from degreasing tanks using a vacuum cart and then pumped into a waste solvent tank at the former Oil House Tank Farm (AOCs 1 and 2). Although this disposal process was established, chlorine contamination was still a major problem for the facility; machining oils and coolants were often contaminated with chlorine (INEL, 1991). It is unknown how solvent wastes were handled prior to construction of the waste transfer system (AOC 3).

Spent non-chlorinated, alkaline, and acid cleaners were discharged to floor drains that connected to transfer piping for the CWTP. Non-liquid wastes from the abrasive cleaning process were typically collected by container and entered the waste collection stream as non-hazardous solid waste. In cases where contamination by solvents or oils was possible, the waste material was tested before being disposed off-site. Miscellaneous cleaning processes utilized cleaners that were used during other cleaning processes, such as alkaline cleaners and acids, or cleaners that were not considered hazardous (INEL, 1991).

Miscellaneous Processes. Miscellaneous processes associated with manufacturing at SAEP included:

- Coating of metal parts with solid film lubricants, which were much like paint in content and application. Solid film lubricants contained toluene, xylene, alcohols, and ketones.
- Masking of items on which plating was to be completed only on a specific area. Typical maskants included wax, lacquer, tape, rubber, and plastic.
- Shot peening, which used the projection of ceramic and glass beads to smooth sharp points or edges.
- Heat treatment of metal parts to improve their metallurgical properties. Metal parts were cleaned, heated in a furnace, and quenched in oil or water.
- Printed circuit board manufacturing, involving the etching of copper boards with ferric chloride solution.

- Photographic processing of black and white and color photographs.

Limited information is available regarding the disposal of wastes generated as a result of these processes. Shot peening wastes were containerized and generally considered non-hazardous waste. Waste ferric chloride from printed circuit board manufacturing was collected into a 55-gallon drum and sent off-site for disposal. Developer, fixer solutions, and washwater from the photographic laboratory were typically discharged to the stormwater treatment system (ESE, 1981 and INEL, 1991).

AOC 49 includes degreasers, dip tanks, satellite accumulation areas, press pits, oily machines, paint spray booths, associated with manufacturing in Building B-2. The RFA estimates the size of this AOC to be 18.8 acres (CDM FPC, 1992).

Thirty-five soil borings were completed in Building B-2 adjacent to machine pits, degreaser pits, sumps and other potential sources of contamination (see Table 7-4). Analytes detected in soil included BTEX, cVOCs, VOCs, PAHs, SVOCs, PCBs, and TPH (see Appendix O for complete soil boring analytical results).

The majority of cVOCs in soil beneath Building B-2 are at concentrations below baseline levels (refer to Appendix B). However, several cVOCs were detected in soil beneath Building B-2 at up to 17 mg/kg (1,1,1-TCA), which was detected beneath the heat treat area of Building B-2. BTEX concentrations in soil were less than 0.02 mg/kg, and were generally detected in soil from 0 to 4 feet bgs throughout Building B-2.

Monitoring wells HESE-01-06D, HESE-01-06I, HESE-01-07D, HESE-01-07I, HESE-01-12D, HESE-01-12I, and HESE-01-14I were completed within various manufacturing centers of Building B-2 (see Table 7-4). Analytes detected in groundwater included cVOCs and VOCs (see Appendix P for complete groundwater analytical results).

A release has occurred from this AOC. Chlorinated and non-chlorinated solvent, paints, and other surface treatments were used in manufacturing activities conducted in Building B-2. Chlorinated solvent and fuel-related contaminants were detected in soil samples taken from the manufacturing areas of Building B-2, and chlorinated solvent-related contaminants were detected in groundwater. Elevated PID readings and petroleum odor were noted for several sample locations, and free product was noted at one location during sampling. Concentrations of TPH, carbon tetrachloride, dichloromethane, TCE, PAHs, arsenic, vanadium, and cadmium in soil exceed RSR criteria (see Figure 7-6). Concentrations of cVOCs in groundwater exceed RSR criteria (see Figure 7-6).

AOC 50 - Building B-2 Plating Area

AOC 50 includes the plating and anodizing rooms, including degreasers and satellite accumulation areas associated with the plating areas. The size of this AOC is estimated to be 8.7 acres (CDM FPC, 1992).

Plating operations were generally conducted in the northeastern corner of Building B-2, in the area known as the former B-2 Chromium Plating Facility. The following paragraphs summarize the electroplating operations and Subsection 1.5 provides additional information on this process.

Electroplating. Electroplating is a process whereby metallic coatings are ionically deposited on other metal surfaces, typically by the induction of an electrical current to a solution. Electroplating at SAEP historically included chromium, nickel, copper, and cadmium plating. Copper and cadmium plating were completed using cyanide-bearing plating solutions (ESE, 1981).

Significant electroplating operations began in 1951, primarily in the northeast corner of Building B-2 (e.g., the former B-2 Chromium Plating Facility) and in the southernmost portion of Building B-3 (AOC 51) (ESE, 1981). A fire insurance map from 1956 indicates plating was also conducted in the northernmost portions of Building B-6 (AOC 31) and in southern Building B-3A (AOC 56) (FIA, 1956).

Because the success of a plating operation depends largely on the degree of cleanliness of the receiving metal, the plating process was typically associated with stripping and cleaning processes. The stripping process was completed with acids such as chromic acid, sulfuric acid, and hydrochloric acid and cleaning was conducted with agents such as chlorinated solvents (e.g., 1,1,1-TCA and Freon-based solvents), non-chlorinated solvents (e.g., Varsol, MEK, and thinners), and alkaline cleaners (e.g., permanganate salts and sodium hydroxides). In the 1980s and 1990s, 1,1,1-TCA was widely used for cleaning; the solvent was contained in large vapor degreasing units within the plating facility (ESE, 1981 and INEL, 1991). It is unknown what stripping and cleaning agents were used at the Stratford site prior to the early 1980s; however, groundwater contamination beneath the former B-2 Chromium Plating Facility suggests TCE was used in significant quantities.

Prior to construction of the chemical waste treatment system in 1958, wastewater from electroplating operations was neutralized within processing tanks and discharged to the storm water system, which ultimately drained, without further treatment, to the Housatonic River. After 1958, chemical tanks (e.g., acids) as well as water rinsing tanks were routed to the CWTP (AOC 10) or the CDF (AOC 9) for treatment, via the chemical waste collection system (AOC 8). After construction of the waste transfer system (AOC 3) in the early- to mid-1980s, chlorinated solvent wastes from the vapor degreasers were removed by vacuum cart and pumped directly into waste solvent tanks at the former Oil House Tank Farm (INEL, 1991). It is unknown how solvent wastes were handled prior to this time.

During dewatering operations conducted as part of Building B-70 and Building B-10 foundation construction, groundwater in these areas was observed to be greenish-blue in color. Additionally, analytical results from soil samples collected near these buildings in 1986 indicated total chromium as high as 0.64 mg/L, Cr(VI) as high as 0.42 mg/L, copper as high as 0.34 mg/L, and nickel as high as 0.48 mg/L (CDM FPC, 1992).

For purposes of simplifying this evaluation, only soil boring locations completed after 1998 have been included. The soil boring locations associated with this AOC include SB12B1-1, SB12B2-1, SP-98-01 through SP-98-06, SP-99-01 through SP-99-05, SP-99-07 through SP-99-09, SP-99-

11 through SP-99-14, and SP-99-17 through SP-99-20 (see Table 7-4). Groundwater samples taken within this AOC include WP-99-04, WP-99-08, WP-99-12, and WP-99-15 (see Table 7-4). Analytes detected in soil include VOCs, SVOCs, cyanide, Cr(VI), and nickel (see Appendix O for complete soil boring analytical results). Analytes detected in groundwater include cadmium, chromium, copper, cyanide, Cr(VI), nickel, and cVOCs (see Appendix P for complete groundwater analytical results).

Materials handled at this AOC included chlorinated solvents, lacquers, xylene, toluene, acetone, MEK, cadmium, chromium, copper, cyanide, nickel (electrolytic), zinc phosphate, magnesium phosphate, chromium, aluminum, phosphoric acid, ammoniated stripper, hydrogen fluoride (for cleaning of nickel parts), muriatic acid, permanganates, sodium hydroxide, sulfuric acid, TPC, Varsol, TCE, and carbon tetrachloride. Chlorinated and non-chlorinated solvents, cyanide, Cr(VI), and nickel were detected in soil samples associated with this AOC, and cadmium, chromium, copper, cyanide, Cr(VI), nickel, and chlorinated solvents were detected in groundwater. This information suggests a release to the subsurface from the former B-2 Chromium Plating Facility has impacted soil and groundwater in the area.

A release has occurred from this AOC. Further information regarding the extent of contamination associated with this AOC can be found in Subsections 8.1 and 8.2. Concentrations of chromium and Cr(VI) in soil exceed RSR criteria (see Figure 7-6). Concentrations of TCE, cadmium, chromium, copper, cyanide, Cr(VI), and nickel in groundwater exceed RSR criteria (see Figure 7-6).

AOC 51 - Building B-3 Plating Area

Building B-3 was constructed in 1930 as the Engineering Building during initial site development. In the early 1940s, a Commercial Hangar (currently northern Building B-3) was constructed north of the existing Engineering Building, and two drafting buildings were added, one to the south of the Engineering Building (currently southern Building B-3) and one to the north of the Commercial Hangar (currently Building B-3A). The Engineering Building underwent modifications during 1943 and 1944 to include its' various expansions and the southern Drafting Building under one roof, creating the current footprint of Building B-3, approximately three acres.

AOC 51 includes the plating and anodizing rooms, including degreasers and satellite accumulation areas associated with the plating areas, located in the center of Building B-3. The area was used for these operations from approximately 1951 to the mid-1970s. Building B-3 plating operations included the use of degreasers and satellite accumulation areas. Two vapor degreasers were in use in Building B-3 in the early 1990s (CDM FPC, 1992).

Five direct push groundwater sample locations DP5-6 through DP5-10 were completed in the former Building B-3 plating area (see Table 7-4). At each direct push location groundwater samples were collected at multiple depth intervals between 5 and 40 feet. Analytes detected in groundwater include cVOCs and Cr(VI) (see Appendix R for complete groundwater analytical results).

A release has occurred from this AOC. Elevated concentrations of cVOCs and Cr(VI) were identified in groundwater where chromium plating was conducted. Cr(VI) was detected at a concentration of 0.1 mg/L (SWPC for Cr(VI) is 0.11 mg/L) in at location DP5-9 at eight feet bgs, beneath the southeastern portion of Building B-3. Concentrations of cVOCs in groundwater exceed RSR criteria (see Figure 7-6).

Former Gasoline USTs near Building B-10

Two 1,000-gallon gasoline USTs are identified on a 1931 fire map (AFM FIC, 1931). The tanks were located near the southeastern corner of what is currently Building B-10. These tanks are also depicted on fire maps from 1937, 1941, and 1943 (AFM FIC, 1937, 1941, and 1943). The current status of these tanks is unknown. A review of available aerial photographs indicates open storage was conducted in this area in the 1960s and 1970s (USEPA, 1990). The open storage appears to have consisted of numerous drums and/or containers along the southeastern side of Building B-10.

Soil boring SB13I1-1 was completed near the approximate location of the former USTs (see Table 7-4). Analytes detected in soil at this location include benzene and methyl isobutyl ketone (see Appendix O for complete soil boring analytical results).

A release from this area has not occurred, or is not suspected. Available information indicates the USTs were located adjacent to the southeastern corner of Building B-10 as early as 1931 until 1943. Low levels of benzene (less than 1 mg/kg) and methyl isobutyl ketone (2 mg/kg) were detected in soil at SB13I1-1 but no visual or olfactory contamination was observed during soil sampling. No detected concentrations in soil boring SB13I1-1 are greater than RSR numerical criteria.

7.3.2 Research and Development Area

Table 7-1 provides a summary of the AOCs associated with the Research and Development Area, the associated sampling locations, the sample analyses, and detected analyte groups. AOCs within this group include:

- AOC 31 – Building B-6 and Associated Satellite Accumulation Areas
- AOC 34 – Building B-3A and Associated Satellite Accumulation Areas
- AOC 39 – Building B-4 Former Brine UST
- AOC 40 – Building B-6 Former USTs
- AOC 55 – Building B-72 and Associated Petroleum Storage Tanks
- AOC 56 – Research and Development Area in Northern Building B-3, Building B-3A, and Building B-4
- AOC 59 – Building B-4 Drum Storage Area
- AOC 60 – Building B-6A Waste Oil Rags (Satellite Accumulation Area)
- AOC 61 – Building B-6A Waste TPC and Oil (Satellite Accumulation Area)
- Building B-53 and Associated Fuel Storage Areas

The areas containing these AOCs generally are located in the eastern portion of the SAEP site. Research and development of experimental aircraft was primarily conducted in the northern portion of Building B-3 and in Buildings B-3A, B-4, B-6, and B-6A. Building B-72 was a fuel pumping station used in support of experimental engine testing conducted in Buildings B-6 and B-6A.

Figure 7-7 presents the locations and boundaries of the AOCs within this group, as well as the explorations which have been associated with the group. Table 7-5 presents a summary of samples associated with each AOC, the field observations, detected analytes, release determination, and analytes with concentrations exceeding CTDEP RSR criteria. Data analysis procedures are discussed in Section 7.0.

AOC 31 - Building B-6 and Associated Satellite Accumulation Areas

Building B-6 was constructed in 1944 for engine testing, parts storage, and for use as an experimental hangar. During the 1940s and 1950s, the building was also used for painting (FIA, 1956 and W-C, 1991). Since the 1950s Building B-6 was primarily used for experimental, high altitude testing of aircraft engines; however, limited amounts of machining and plating also were conducted in this building (FIA, 1956). The building includes a high bay area with stand-alone test chambers, a steel framed mezzanine level, machine shop, and older test cells and office areas (AlliedSignal, 1995).

AOC 31 consists of the waste calibration fuel, waste jet fuel, and waste oil that reportedly were stored in 55-gallon drums within various satellite accumulation areas located throughout the building (CDM FPC, 1992).

Soil borings SB24A1-1, SB24A1-2, SB24A2-1, SB24A2-2, SB24A3-1, SB24A4-1, SB24A4-2, and SB24A4-4 are associated with this AOC (see Table 7-5). These soil borings were completed within the footprint of Building B-6, including areas of former plating operations, test cells, and sumps. Analytes detected in these samples include BTEX, VOCs, and SVOCs (see Appendix O for complete soil boring analytical results).

A release has occurred from this AOC. Materials handled at this AOC included fuel and oil; BTEX was detected in soil. Elevated PID readings and a strong petroleum odor were noted for soil sample location SB24A4-4, near the southeast corner of the building. No releases from satellite accumulation areas at Building B-6 are suspected (see discussion of satellite accumulation areas in subsection 7.0); however, analytical results indicate activities conducted within the building have released VOCs to underlying soil. Concentrations of arsenic in soil exceed RSR criteria (see Figure 7-8) at sample location SB24A1-1.

AOC 34 – Building B-3A and Associated Satellite Accumulation Areas

Building B-3A was constructed in 1942 as a drafting building (AFM FIC, 1943 and W-C, 1991). Engineering and chemical laboratories, a machine shop, a heat treatment area, and office space were located in this building (FIA, 1956 and 1986). The RFA indicates waste solvents (1,1,1-TCA), waste jet fuel, waste oil, and acid wastes were stored in 55-gallon drums in satellite accumulation areas within the building (CDM FPC, 1992).

Soil borings SB27A1-1, SB27B1-1, and SB27C1-1 were completed within the footprint of Building B-3A, near potential routes of contamination to the subsurface, including sumps and cracked flooring (see Table 7-5). Analytes detected in these samples include BTEX, VOCs, and TPH (see Appendix O for complete soil boring analytical results).

A release has occurred from this AOC. Historically, Building B-3A was used for operations ranging from drafting and storage to heat treatment and plating for research and development purposes. Analytical results from soil samples collected beneath the building floor indicate fuel-related contamination. No releases from satellite accumulation areas at Building B-6 are suspected (see discussion of satellite accumulation areas in subsection 7.0); however, activities conducted within the building have released VOCs to underlying soil. Detected concentrations of BTEX were less than 1 mg/kg. Concentrations of TPH in soil exceed RSR criteria (see Figure 7-8).

AOC 39 - Building B-4 Former Brine UST

This AOC includes a 20,000-gallon brine storage tank located beneath the northernmost corner of Building B-4, and used during the ECM process. ECM is a process whereby a metal work piece is placed into a salt brine bath and then cut using an arc welder. In this process the metal is removed by dissolution into the ECM brine, a process that results in more rapid cutting of parts. The process generated sludge that was disposed of off-site, and wastewater that was discharged to the OATP (ESE, 1981). The ECM process was decommissioned in 1987 (ABB-ES, 1996), and the tank was removed in 1989 (CDM FPC, 1992).

Soil borings SB27E6-1 and SB27E7-1 were completed in the northern corner of Building B-4, in the vicinity of the ECM area and former brine tank (see Table 7-5). Analytes detected in these samples include BTEX and VOCs (see Appendix O for complete soil boring analytical results).

A release from this AOC has not occurred, or is not suspected. Operations in this portion of Building B-4 were related to the ECM process, where a brine solution was used and metal sludge was generated. Analysis of samples collected from the area near the brine tank contained low levels of fuel-related contaminants and methyl isobutyl ketone. The detected analytes suggest a release has occurred in this area; however this AOC is not likely the source. No concentrations of detected analytes at sample locations associated with this AOC are greater than the RSR numerical criteria.

AOC 40 - Building B-6 Former USTs

AOC 40 consists of the area containing four former USTs, used to store fuel and oil for operations conducted in Building B-6 (see Figure 7-9). There were two 550-gallon fuel USTs, a 5,000-gallon fuel UST, and a 5,000-gallon oil UST (CDM FPC, 1992). The two 5,000-gallon tanks were located near the easternmost corner of Building B-6 (FIA, 1956); the location of the two 550-gallon tanks is unknown. The two 550-gallon tanks were reportedly removed in 1989, the 5,000-gallon tanks, were initially sand filled and abandoned in place in 1979, and were removed in 1998 (CDM FPC, 1992). During the 1998 removal, petroleum-contaminated soil was visually identified.

Other storage tanks were identified in the vicinity of Building B-6. A 1986 fire map (FIA, 1986) depicts two 250-gallon diesel oil tanks west of the central portion of Building B-6, and a 250-gallon gasoline tank in the southeastern-most corner of Building B-6 (see Figure 1-5). The gasoline tank appears to have been used to run an emergency fire protection system, as a 3-inch deluge system is shown in this area.

A release has occurred from this AOC. During removal of one 5,000-gallon tank in 1998, petroleum-contamination was identified in surrounding soil.

AOC 55 - Building B-72 and Associated Petroleum Storage Tanks

Building B-72 was constructed between 1980 and 1986, to serve as a pumping station, replacing the existing pumping station, for fuel storage tanks located in a concrete bermed area immediately to the east. Building B-72 is a metal module building located between the southern corner of Building B-6 and the former sludge drying lagoons. The building serviced two 10,000 and four 20,000-gallon diesel and jet fuel ASTs, which were removed in 1998 (see Table 1-4). Two 20,000-gallon tanks were installed in approximately 1965; the other four tanks were installed in the early 1980s (W-C, 1991). Petroleum-contaminated soils were identified during closure of the adjacent sludge drying lagoons; the contaminated soils were not removed (CDM FPC, 1992).

Soil borings BR-9, BR-10, SB23A1-1, SB23A1-2, and SB23A1-3 are associated with this AOC (see Table 7-5). Soil borings BR-9 and BR-10 were completed during the Phase I RI. Soil borings SB23A1-1, SB23A1-2, and SB23A1-3 were collected to confirm the extent of contamination encountered in soil borings BR-9 and BR-10 (W-C, 1998). Analytes detected in these samples include BTEX, VOCs, PAHs, and SVOCs (see Appendix O for complete soil boring analytical results).

A release has occurred from this AOC. Petroleum contamination was identified during closure of the adjacent sludge drying lagoon in 1986, fuel-related contaminants were detected in soil, and visual and olfactory contamination were noted during the completion of soil borings at this AOC. Concentrations of PAHs in soil exceed RSR criteria (see Figure 7-8).

AOC 56 - Research and Development Area in Northern Building B-3, Building B-3A, and Building B-4

The RFA defines this AOC as the area housing the plant's research and development activities, including a materials testing laboratory. Activities were primarily conducted in the northernmost portion of Building B-3, in Building B-3A, and in Building B-4, a total area of approximately 3.9 acres. According to the PAS, this area was a disposal and uncontrolled release area. As part of research and development operations, heavy metals, fuels, oils, and solvents were used and/or generated as waste (CDM FPC, 1992).

Building B-3 was constructed in 1930 as the Engineering Building during initial site development. In the early 1940s, a commercial hangar and a drafting building were added to the northeast and a second drafting building was added to the southwest of the existing Engineering Building. The Engineering Building underwent modifications during 1943 and 1944 to include its' various expansions and the southern Drafting Building under one roof, creating the current footprint of Building B-3, approximately three acres. Initially, the building was used primarily for engineering and storage. A 1956 fire insurance map indicates the northernmost portion of Building B-3 was used for machining, heat treatment, and parts inspection with liquid penetrants (FIA, 1956). A 1986 map indicates the northernmost area consisted of a storage area (FIA, 1986).

Building B-3A was constructed in 1942 as a drafting building (AFM FIC, 1943 and W-C, 1991). Historically, engineering and chemical laboratories, a machine shop, a heat treatment area, and office space were located in this building (FIA, 1956 and 1986). In April of 1989 a spill of cleaning solvents occurred north of Building B-3A (ABB-ES, 1996).

Building B-4 was constructed in 1945 for parts storage, and covers nearly two-thirds of an acre (FIA, 1956). The building is interconnected with Building B-3A and historically also has been used for tank engine repair, and ECM (W-C, 1991 and FIA, 1986). The ECM process was discontinued at SAEP in 1987, and the former 20,000-gallon brine UST used during this process was removed in 1989 (see AOC 39 discussion).

A spill of liquid cleaning solvents to the ground surface immediately north of B-3A reportedly occurred in April 1989 (AlliedSignal, 1995).

Soil borings SB12D1-1, SB12D1-2, SB12D1-3, SB16A1-1, SB16A1-2, SB16A1-3, SB16A1-4, SB16D1-1, SB16D1-2, SB16D1-3, SB27A1-1, SB27B1-1, SB27C1-1, SB27D1-1, SB27E2-1, SB27E3-1, SB27E4-1, SB27E6-1, SB27E7-1, SB27E8-1, and EBS73-1 were completed within Buildings B-3, B-3A, and B-4, as well as the alcove located along the western side of these buildings (see Table 7-5). Analytes detected in soil from these locations include BTEX, VOCs, SVOCs, PCBs, TPH, and nickel (see Appendix O for complete soil boring analytical results).

Monitoring well PZ-13D was installed within the paved area located within the alcove west of Building B-3, B-3A, and B-4, direct-push groundwater sample location DP2-5 was located in the eastern portion of Building B-4, and locations DP5-2 through DP5-5 were completed within the northern portion of Building B-3. Analytes detected in groundwater at this location include BTEX, cVOCs, and VOCs (see Appendix P for complete groundwater analytical results).

Soil gas sampling locations SG-99-22, SG-99-23, SG-99-38, and SG-99-39 are located within and adjacent to Buildings B-3, B-3A, and B-3A. Analytes detected in these samples include cVOCs (see Appendix O for complete soil boring analytical results).

A release has occurred from this AOC. Materials handled at this AOC include heavy metals, fuels, oils, and solvents. No elevated PID readings or visual observations of contamination during collection of soil samples; however, petroleum odor was noted at one sample, SB16D1-1, located along the ECM line, within the eastern side of B-3A. Fuel-related contaminants and nickel were detected in soil, chlorinated solvent and fuel-related contaminants were detected in groundwater and chlorinated solvents were detected in soil vapor. Concentrations of arsenic, nickel, and TPH in soil, cVOCs in groundwater, and TCE and PCE in soil vapor exceed RSR criteria (see Figure 7-8).

AOC 59 - Building B-4 Drum Storage Area

This AOC was defined in the EBS to include an attached portion of Building B-4 that was used to store 55-gallon drums of machining oil and engine oils used in engine testing and development at the facility (ABB-ES, 1996). The Drum storage area had a cinderblock foundation and metal racks used for storing the 55-gallon drums since approximately 1981 (ABB-ES, 1996).

SB27E2-1 is located within the B-4 Drum storage area (see Table 7-5). Analytes detected in these samples include BTEX, VOCs, SVOCs, and PCBs (see Appendix O for complete soil boring analytical results).

A release from this AOC has not occurred, or is not suspected. Materials handled at this AOC include machining oil and engine oils. Analytical results indicate relatively low concentrations of benzene (less than 1 mg/kg). No concentrations of contaminants exceed RSR criteria in samples from SB27E2-1.

AOC 60 - Building B-6A Waste Oil Rags (Satellite Accumulation Area) and AOC 61 - Building B-6A Waste TPC and Oil (Satellite Accumulation Area)

Building B-6A was built in 1966, and included four individual engine component testing cells. Waste oil rags (AOC 60), and waste TPC and oil (AOC 61) were stored in 55-gallon drums in satellite accumulation areas located throughout the building since 1991 (ABB-ES, 1996).

Soil borings SB24B1-1 and SB24B1-2 were completed adjacent and within Building B-6A (see Table 7-5). Analytes detected in these samples include BTEX, cVOCs, VOCs, and PCBs (see Appendix O for complete soil boring analytical results).

A release has occurred from these AOCs. Materials handled at these AOCs include waste oil and TPC (an aliphatic hydrocarbon). During soil sampling, tar- and heating oil-like odors were observed, although no visual observations of contamination or elevated PID readings were noted. Fuel and solvent related contamination were detected in soil at Building B-6A. Although no release is suspected from these satellite accumulation areas (see discussion of satellite accumulation areas in subsection 7.0), activities at Building B-6A have resulted in a release. Concentrations of cVOCs in soil exceed RSR criteria (see Figure 7-8).

Building B-53 and Associated Fuel Storage Areas

Building B-53 was constructed in 1961. A facility drawing from 1962 (AVCO Lycoming, 1962) depicts two fuel USTs beneath the southern end of Building B-53. Additionally, a plan from 1964 (AVCO Lycoming, 1964) depicts four temporary mobile tankers immediately south of Building B-53. Aerial photographs indicate open storage since at least 1943 in this area, and following the construction of Building B-6 in 1944, stains and/or tanks are identified in aerial photographs (USEPA, 1990).

There is insufficient evidence to determine if a release has occurred from this area because there are no associated samples.

7.3.3 Testing Areas

The Testing Areas consist of the buildings and associated areas where engine and engine systems testing were conducted, including Buildings B-5, B-16, B-19, B-7/7A, B-8, and B-9. The AOCs and other potential release areas within this group include:

- AOC 23 – Building B-19 Dry Well
- AOC 30 – Building B-34 and Associated Satellite Accumulation Areas
- AOC 32 – Building B-5 and Associated Satellite Accumulation Areas
- AOC 33 – Building B-19 and Associated Satellite Accumulation Areas
- AOC 35 – Building B-43 and Associated Satellite Accumulation Areas
- AOC 44 – Building B-19 Former USTs
- AOC 45 – Jet Fuel Tank Farm Former USTs (near Building B-34)
- AOC 48 – Building B-16 and Associated Satellite Accumulation Areas
- AOC 57 – Drum Storage Area East (North) of Building B-19
- AOC 58 – Scrap Metal Yard North of Building B-16
- AOC 62 – Building B-7 Waste Oil Satellite Accumulation Area
- AOC 63 – Building B-8 Flammable Storage Area (Paints and Solvents)
- AOC 64 – Building B-8 Waste Paint Satellite Accumulation Area
- AOC 65 – Building B-19 Diesel 1,000 Gallon AST (north of building)
- AOC 66 – Building B-19 Diesel 1,000 Gallon AST (north of building)
- AOC 67 – Building B-19 JP-5 1,600 Gallon AST (north of building)
- AOC 68 – Building B-19 JP-4 60,000 Gallon AST (northwest of building)
- AOC 69 – Building B-19 Jet-A 60,000 Gallon AST (northwest of building)
- Building B-7/7A Drains
- Building B-9 Floor Drain
- ASTs Southeast of Building B-33

Figure 7-9 presents the locations and boundaries of the AOCs within this group and the associated explorations. Table 7-6 presents a summary of sample locations, the field observations, detected analytes, release determination, and analytes with concentrations exceeding CTDEP RSR criteria.

AOC 23 Building B-19 Dry Well

This AOC includes a dry well reportedly located in Building B-19 where waste fuels, oils, and solvents generated during operations in Building B-19 were discharged. The RFA inspection report indicates the former dry well was inside a building door and was constructed of concrete with a gravel bottom. Disposal to the dry well reportedly ceased in 1987; it is unknown when disposal to the dry well may have begun (CDM FPC, 1992). At the time of the RFA inspection, facility personnel indicated no contamination in the area of the dry well had been found; however, there is no information to suggest an investigation of the dry well has ever been conducted (CDM FPC, 1992).

There is insufficient information to determine if a release has occurred from this AOC. The exact location or existence of the dry well could not be determined based on a review of records and a site inspection conducted in October 2003.

AOC 30 - Building B-34 and Associated Satellite Accumulation Areas

Building B-34 was constructed in 1953. It is located near the eastern end of Building B-16, and served as the pumphouse for the Former Jet Fuel Tank Farm (AOC 45), which supplied fuel for engine testing conducted in Building B-16. Accumulation areas at Building B-34 contained 55-gallon drums of waste oil, filters, and jet fuel (CDM FPC, 1992).

There is insufficient information to determine if a release has occurred from this AOC. There are no borings completed within the footprint of Building B-34 or within 20 feet of the perimeter of the building. Determination of a release from this AOC would not likely be possible due to the presence of contamination from the Former Jet Fuel Tank Farm (AOC 45).

AOC 32 – Building B-5 and Associated Satellite Accumulation Areas

Building B-5 was built in 1944 for use as a testing facility (W-C, 1991). The building historically served as the fire headquarters and a small component test facility, which included velocity testing (FIA, 1956). Reportedly, waste jet fuel was stored within 55-gallon drums in satellite accumulation areas located throughout the building (CDM FPC, 1992). A 1986 fire insurance map identifies a 600-gallon fuel oil tank located in Building B-5A, immediately to the northwest of Building B-5 (FIA, 1986). Building B-5 was demolished in 2001 to provide access during completion of the Causeway removal action. The building foundation was left in place and subsurface building utilities were capped at the surface (Harding ESE, 2003).

Soil boring SB27E9-1 was completed to investigate potential contamination from degreasers located near a drain along the northeast wall of the building (W-C, 1998) (see Table 7-6). Analytes detected in soil from sample location SB27E9-1 included the VOCs acetone and methyl isobutyl ketone (see Appendix O for complete soil boring analytical results).

A release from this AOC has not occurred, or is not suspected, based upon the historical use of this area and analytical results from the associated boring. This AOC handled waste jet fuel in 55-gallon drums. Only the VOCs acetone and methyl isobutyl ketone were detected in soil. No concentrations of analytes were greater than RSR criteria.

AOC 33 – Building B-19 and Associated Satellite Accumulation Areas

This AOC includes Building B-19 and the satellite accumulation areas identified within Building B-19 that contained 55-gallon drums of waste filters (CDM FPC, 1992). The locations of the satellite accumulation areas are unknown and likely changed over time. Building B-19 was used for jet engine testing and turbine engine research and development. The building, consisting of several smaller buildings, was built on fill materials that were hydraulically dredged from the Housatonic River.

Soil borings SB28A1-1, SB28A1-2, SB28A2-1, and SB28A2-2, and soil gas sample locations SG-99-28 and SG-99-52, were completed adjacent to sumps, pits, and concrete trenches within Building B-19 (see Table 7-6). Analytes detected in soil samples included BTEX, VOCs, and PCBs (see Appendix O for complete soil boring analytical results). No analytes were detected in soil gas samples (see Appendix U for complete soil vapor analysis results).

A release has occurred from this AOC. Satellite accumulation areas within this AOC handled 55-gallon drums of waste filters; and although no specific release from the satellite accumulation areas is suspected (see discussion of satellite accumulation areas in subsection 7.0), activities in Building B-19 have resulted in a release. No concentrations of analytes were greater than RSR criteria.

AOC 35 – Building B-43 and Associated Satellite Accumulation Areas

Building B-43 was constructed in the early 1940s to serve as a pumping station for a fire suppression tank located adjacent to the building (AFM FIC, 1943). Building B-43 was modified in approximately 1986 to serve as the fuel pumping station for two 60,000 gallon ASTs that supplied the Building B-19 jet engine testing and turbine research (FIA, 1986). The RFA indicates that waste fuel and filters were stored in 55-gallon drums located in satellite accumulation areas in the building (CDM FPC, 1992).

There is insufficient information to determine if a release has occurred from this AOC. No samples were collected near this AOC.

AOC 44 - Building B-19 Former USTs

The RFA indicates four former fuel USTs, located in the vicinity of Building B-19, were used in support of testing activities within the building. The USTs included two 550-gallon tanks, a 1,000-gallon tank, and a 2,000-gallon tank (CDM FPC, 1992). Fire insurance maps depict the location of the 1,000-gallon and 2,000-gallon tanks as immediately east of the original Building B-19 within the current footprint of the building (see Figure 7-9) (FIA, 1986). The location of the two 550-gallon tanks is unknown. Reportedly, all four USTs were removed in 1987 (W-C, 1991).

There is insufficient information to determine if a release has occurred from this AOC. No samples were collected at this AOC.

AOC 45 - Jet Fuel Tank Farm Former USTs

This AOC consists of the former USTs located at the Former Jet Fuel Tank near Building B-34. It includes five 20,000 gallon jet fuel tanks and one 20,000 gallon diesel tank, a 4,000 gallon waste fuel tank, a 5,000 gallon Varsol tank, a 1,000 gallon fuel tank, and nine 300 gallon fuel tanks (CDM FPC, 1992). The total number of USTs associated with this AOC identified in the RFA is 18.

The tanks were in use from approximately 1953 to 1989. The Former Jet Fuel Tank Farm supplied Building B-16 with fuel for engine testing. The original tank farm consisted of partially buried tanks containing jet fuel, diesel fuel, gasoline, and cleaning agents. The original tanks were removed in 1989, and replaced with ASTs on concrete cradles. During tank removal, approximately 2,000 cubic yards of fuel-contaminated soil, containing levels of toluene and xylenes up to 5,500 ppm (CDM FPC, 1992) was excavated and placed into a soil pile in the South Parking Lot for remediation and ultimate use as backfill within the parking lot (W-C, 1991). The ASTs were removed in 1998.

Fire insurance maps depict the locations of six 20,000 gallon tanks and one 5,000 gallon tank within the bermed area next to Building B-34, southeast of Building B-16 (FIA, 1986). This information suggests the ten tanks were located within the Building B-34 berm and removed and replaced with ASTs in 1989. No additional documentation can be found pertaining to the location of the remaining nine 300-gallon tanks associated with this AOC.

Soil borings SB27E1-1, SB27E1-2, SB27E1-3, and SB27E1-4 were completed at AOC 45 to evaluate soil contamination. Location SB27E1-1 is located within the former berm. The remaining soil boring locations are located no more than 15 feet outside of the former berm. Soil boring SB27E10-1 has also been included in the evaluation for AOC 45. Analytes detected in soil samples at AOC 45 include BTEX, VOCs, PCBs, and TPH (see Appendix O for complete soil boring analytical results).

Direct-push sampling location DP2-7 and DP2-8 were installed at AOC 45. Analytes detected in groundwater at these direct-push locations include BTEX and cVOCs (see Appendix R for complete direct-push groundwater analytical results).

A release has occurred from this AOC. There is documentation of a release of fuel-related contaminants and contaminated soils were excavated in 1989. Soil samples taken following excavation of contaminated soil indicates residual fuel and chlorinated solvent-related contamination. Additionally, visual and olfactory contamination was identified in several of the sampling locations, including elevated PID readings, black-stained soil, an oil sheen, and fuel-like odors. Concentrations of arsenic, benzene, and TPH in soil exceed RSR criteria (see Figure 7-10). Concentrations of vinyl chloride in groundwater exceed RSR criteria (see Figure 7-10).

AOC 48 - Building B-16 and Associated Satellite Accumulation Areas

AOC 48 includes Building B-16, used for production and development of engines in test cells, and various satellite accumulation areas that previously stored waste oil, fuel, filters, and oily rags in 55-gallons drums. A discussion of activities conducted and materials handled at Building B-16, documented releases to the Building B-16 floor drains and sumps (AOC 4), and analytical results for soil, groundwater, and soil vapor at Building B-16 is provided in subsection 7.4.

In addition to sample locations included in the discussion of AOC 4 (subsection 7.4), two additional soil borings, SB17A3-6 and SB17A3-7, are located adjacent to Building B-16 and are associated with this AOC (see Table 7-6). These soil borings cannot be uniquely associated with the floor drains and sumps of Building B-16, and were intended to provide characterization of potential mercury spills north of Building B-16 resulting from activities at Building B-16. Analytes detected in these soil borings include BTEX, VOCs, PCBs, and TPH.

A release has occurred from this AOC. No specific release from the satellite accumulation areas is suspected (see discussion of satellite accumulation areas in subsection 7.0); however, activities in Building B-16 included jet engine testing, assembly, and cleaning and materials handled at this AOC included raw and waste fuels and oils. See the discussion of AOC 4 - Building B-16 Floor Drains and Sumps, in Subsection 7.4 for more information. Fuel-related contamination was detected in soil north of the central portion of Building B-16, along the Dike. Elevated PID readings, strong petroleum odors, and high percent lower explosive limit readings were observed during soil sampling at these locations. Concentrations of TPH exceed RSR criteria (see Figure 7-10).

AOC 57 - Drum Storage Area East (North) of Building B-19

A drum storage area was located north of Building B-19 and held drums containing 1,1,1-TCA, PCE, and other solvents. The drum storage area was bounded by the Dike to the north, the Building B-19 ASTs to the east, Building B-19 to the south, and an air compressor pad to the west. During a 1990 housekeeping tour, the drum storage area was identified as requiring a new bermed concrete pad to meet CTDEP regulations regarding drum storage. During removal of the concrete pad, a grate covered trench containing a soil bottom was discovered. Sampling of the trench and surrounding area indicated the presence of 1,1,1-TCA, PCE, and fuel-related VOCs. An additional spill of diesel fuel into this area from overfilling of the adjacent ASTs resulted in the ultimate removal of approximately 120 cubic yards of material that was sent off-site for disposal. It is unclear if the excavated area was lined with a geosynthetic liner prior to placement of backfill, or if backfill material only was used (Textron Lycoming Memos, 1990).

Soil borings SB31A1-1, SB31A1-2, and SB31A1-3 are located at AOC 57, within the area of the drum storage area and the soil removal (see Table 7-6). Analytes detected in soil at these locations included BTEX, cVOCs, VOCs, SVOCs, PCBs, and cyanide (see Appendix O for complete soil boring analytical results).

Monitoring well WC-1S was installed near the southeastern boundary of the soil removal area. Analytes detected in groundwater at WC-1S included BTEX and cVOCs (see Appendix P for complete groundwater analytical results).

A release has occurred from this AOC. There is documentation of a release of chlorinated solvents and fuel and contaminated soil was removed in 1990. Results of soil and groundwater samples indicate residual fuel and chlorinated solvent contamination in soil and groundwater and cyanide in soil at the drum storage area. Concentrations of 1,1,2,2-tetrachloroethane (1,1,2,2-TCA) in soil exceed RSR criteria. Concentrations of cVOCs and arsenic in groundwater exceed RSR criteria.

AOC 58 - Scrap Metal Yard North of B-16

An area north of Building B-16 was formerly used as a scrap metal yard. Scrap metals that were reportedly covered with oils and greases were stored in an area between Building B-16 and Building B-64-1. It is likely that the oils and greases from the scrap metals eventually were released to the ground (W-C, 1991). Aerial photographs indicate that this area previously was used for open storage of drums and crates (USEPA, 1990).

Soil borings SB14A2-1, SB17A3-4, and SB17A4-1 are located at this AOC (see Table 7-6). Analytes detected in soil at these locations included BTEX, VOCs, and PCBs (see Appendix O for complete soil boring analytical results).

A release has occurred from this AOC. Scrap metals reportedly covered with oils and greases were stored at this AOC. Fuel-related contaminants and PCBs were detected in soil at this AOC at concentrations of 5 mg/kg or less, and black, tar-like material was noted during soil sampling at one of the soil boring locations at this AOC. Concentrations of PCBs in soil at SB17A3-4 exceed RSR criteria (see Figure 7-10).

AOC 62 - Building B-7 Waste Oil Satellite Accumulation Area and Building B-7/7A Drains

Building B-7 was constructed in 1943 for use as a paint and solvent storage facility. Building B-7A was added to the south wall of B-7 in 1962, when engine fuel system testing began in the building (W-C, 1992). Aerial photographs and fire insurance maps from the 1940s indicate the eastern portion of Building B-7 was the location of a barrel rack and that the location of Building

B-7A served as a drum storage area during this time. The content of barrels and drums located in these areas is unknown (AFM FIC, 1943 and USEPA, 1990).

Buildings B-7 and B-7A were most recently used for engine fuel system testing (W-C, 1992). The interior of the building contains perimeter drains that are connected to the stormwater collection system through Building B-64-1. The EBS indicates waste oil was stored at Building B-7 in small quantities (no more than 1 gallon) since approximately 1991 (ABB-ES, 1996).

Soil borings SB13C1-1, SB13D1-1, SB13D1-2, and SB13D1-3 were completed adjacent to the perimeter of Buildings B-7 and B-7A (see Table 7-6). Analytes detected in soil at these locations include BTEX, cVOCs, VOCs, PCBs, and cyanide (see Appendix O for complete soil boring analytical results).

A release has occurred from this AOC. The drains associated with Buildings B-7/B-7A handled waste petroleum product. Fuel and solvent-related contaminants and cyanide were detected in soil near the buildings. Concentrations of antimony, arsenic, cadmium, lead, and cVOCs in soil exceed RSR criteria.

AOC 63 - Building B-8 Flammable Storage Area (Paints and Solvents) and AOC 64 - Building B-8 Waste Paint Satellite Accumulation Area

Building B-8 was constructed in 1943 for use as an auto storage facility. Flammable paints and solvents were stored in quantities of approximately 50-70 gallons from 1943 to the early 1990s (ABB-ES, 1996). These materials were stored on wooden pallets or on metal shelves. The interior of the building contains barrel racks surrounded by a concrete containment dike. The floor was historically covered with absorbent materials, and no floor drains have been observed (W-C, 1991).

Soil boring SB13B1-1 was completed adjacent to the northwest corner of Building B-8 (see Table 7-6). Analytes detected in soil from location SB13B1-1 include BTEX and VOCs (see Appendix O for complete soil boring analytical results). During installation of monitoring well WC2-3D, located adjacent to SB13B1-1, soil samples were collected for observation purposes. The log for this location did not indicate visual or olfactory contamination in any of the collected samples and PID readings were all 0 ppm.

A release from this AOC has not occurred, or is not suspected. Flammable paints and solvents were stored in Building B-8 from 1943 until the 1990s. The interior of the building contained a concrete containment dike and no floor drains. Analytical results of samples collected from a boring located adjacent to the northernmost corner of this building identified relatively low concentrations (less than 1 mg/kg) of toluene and xylenes in soil and no concentrations of detected analytes exceed RSR numerical criteria.

AOCs 65-67 – Building B-19 ASTs North of Building

Three ASTs, including a 2,000-gallon diesel fuel #2 tank (AOC 65), a 1,000-gallon diesel fuel #2 tank (AOC 66), and a 1,000-gallon JP-5 tank (AOC 67) were located to the north of Building B-19, on a bermed concrete pad, since 1953 (ABB-ES, 1996). Fuel in these tanks was used during testing operations in Building B-19, and the tanks were reportedly occasionally overfilled (W-C, 1991). In June 1990, one of the diesel tanks was accidentally overfilled and 150 gallons of fuel was spilled to the ground surface. Approximately 100 gallons of this was collected by facility personnel and the remaining 50 gallons was removed along with contaminated soils from an open excavation west of the concrete pad, in the area of the drum storage area (AOC 57). The excavated soils were stockpiled in the bermed tank area, sampled, and sent off-site for disposal (Textron Lycoming Memo, July 2, 1990). The tanks were removed in 1998 (ABB-ES, 1996).

Soil borings SB31A2-2 and SB31A2-1 are located within the footprint of the concrete pad (see Table 7-6). Analytes detected in soil at these locations included BTEX, VOCs, and PCBs (see Appendix O for complete soil boring analytical results).

A release has occurred from these AOCs. The three ASTs included in these AOCs, located north of Building B-19, historically contained diesel fuel and jet fuel used during testing operations in Building B-19. These tanks were contained in a bermed concrete pad; however, occasional overfilling of the tanks reportedly occurred, including a documented spill in 1990. Analytical results from samples collected underlying and south of the concrete pad indicate fuel-related contaminants in soils. Additionally, observations of samples from the boring underlying the concrete pad indicate petroleum odors, black staining, and elevated PID readings. No concentrations of contaminants exceed RSR criteria.

AOCs 68-69 Building B-19 ASTs Northwest of Building

Two 60,000-gallon jet fuel ASTs were located northwest of Building B-19 since 1986, and were removed in 1998. The northernmost tank contained Jet-A jet fuel (AOC 69), the southern tank JP-4 jet fuel (AOC 68), both used for engine testing conducted in Building B-19 (Textron Lycoming, 1993)). These tanks were located on a concrete pad within a bermed area providing secondary containment. Prior to installation of the fuel tanks, a 400,000 gallon fire suppression tank was located in the area (FIA, 1956 and USEPA, 1990).

Soil boring BR-1 was completed adjacent to the AST containment berm (see Table 7-6). Analytes detected in soil at BR-1 included the SVOCs benzoic acid and diethyl phthalate (see Appendix O for complete soil boring analytical results).

A release from this AOC has not occurred, or is not suspected. Jet fuel was handled at this AOC, inside ASTs located on a bermed concrete pad. Evidence collected during field sample collection at boring BR-1, located approximately 10 feet northwest of these tanks, did not indicate staining, odors, or elevated PID readings. Analytical results from this boring indicated several SVOCs; however, no concentrations of detected analytes exceed RSR numerical criteria.

Building B-9 Floor Drain

Building B-9 was constructed in 1943 for use as a fire truck and truck repair facility (AFM FIC, 1943). Most recently, the building served as a maintenance shop and repair shop (FIA, 1986). Batteries, oil, grease, and hydraulic fluid were stored in 55-gallon drums in Building B-9. The floor in this building is concrete, but there is a hydraulic lift pit and floor drains that lead to the OATP via pump station B-64-1 (W-C, 1991).

The determination of a release from this potential release area is not possible due to the presence of contamination from other potential sources. Soil boring SB13E1-1 was completed adjacent to the storm drain line from Building B-9 (see Table 7-6). BTEX and VOCs were detected in soil at this location (see Appendix O for complete soil boring analytical results). Additional analytical information from the nearby UST area (see AOC 41 and 42 presented in subsection 6.7) suggests the contamination in this boring may be related to a release from the USTs. No concentrations of detected analytes at sample location SB13E1-1 are greater than RSR criteria.

ASTs Southeast of Building B-16

Operations in Building B-16 utilized petroleum product stored in ASTs located adjacent to the Building B-33 pump station and cooling tower near the southeastern end of Building B-16. Four 3,000-gallon engine oil tanks were originally located in this area, likely since construction of Building B-16 in 1953 (FIA, 1956). These tanks were removed between 1980 and 1984, and replaced by two 40,000-gallon #2 Diesel ASTs (W-C, 1991). The diesel tanks were removed in 1998 (IPM, 2002).

There is insufficient information to determine if a release has occurred from this potential release area. No samples were collected near the ASTs located southeast of Building B-16.

7.4 STORMWATER AND WASTEWATER SYSTEM

The stormwater and wastewater system consists of the catch basins, floor drains, roof drains, and underground lines used to collect and convey stormwater and industrial wastewater; the associated pump stations; the Oil Abatement Treatment Plant (OATP); and OF-001 through OF-007. The system collected stormwater from roof drains and catch basins throughout the facility and wastewater from engine test operations in Building B-16, boiler blowdown, cooling water, laboratory wastes, photographic processing wastes, soluble and insoluble cutting oils, and spent hydraulic fluid.

AOCs within this group include:

- AOC 4 – Building B-16 Floor Drains, Sumps, and Piping
- AOC 5 – Stormwater and Wastewater Collection System for OATP
- AOC 6 – OATP in Building B-64-2
- AOC 24 – Discharge to the Housatonic River at Outfall-007
- AOC 52 – Outfalls 001 through 006 and the Intertidal Flats.

Figure 7-11 presents the locations and boundaries of the AOCs within this group, as well as the explorations which have been associated with the group. Table 7-7 presents a summary of samples associated with each AOC, the field observations, detected analytes, release determination, and analytes with concentrations exceeding CTDEP RSR criteria. Data analysis procedures are discussed in Section 7.0.

AOC 4 - Building B-16 Floor Drains, Sumps, and Piping

AOC 4 consists of the gravity drain system that was used to collect and convey accumulated liquids from the floor of Building B-16 (see Figure 7-11). The drainage system was used from 1953 until 1991. There were 52 drains and a drainage system constructed of cast iron. Building B-16 is the former production engine testing facility constructed in 1953 with at least 20 test cells (CDM FPC, 1992).

Solvents were used for cleaning engines and test cells until the late 1980s. Solvents used included carbon tetrachloride, TCE, Varsol prior to 1985, and 1,1,1-TCA. It was estimated that eight to ten drums per month of 1,1,1-TCA were used for engine cleaning (CDM FPC, 1992). From 1953 until 1976, liquid waste that included solvents, water and fuels was gravity fed from test cells through the drainage system to the oil/water separator south of the building prior to discharge to the Housatonic River. After 1976 when the OATP was constructed, liquid wastes were pumped from Building 16 drainage system to the pump station at Building 64 then through the OATP (W-C, 1991). In the early 1990s the drainage system was plugged and no longer used. At this time solvents were replaced with steam cleaning and sumps were installed and piped to a membrane cleaning system to remove fuels and oils.

Waste solvents not discharged to the floor drains were reportedly contained in drums or carts that were stored on small outside docks adjacent to the engine test cells located on the north side of Building B-16, between the building and the Dike. Reportedly some spills occurred in this area (W-C, 1991). These containers were periodically transferred to the waste solvent tanks at the former Oil House Tank Farm.

Emulsion cleaning replaced solvent cleaning of test engines in 1990. Emulsion cleaning utilizes an organic solvent, detergent, and water mixture. Emulsion cleaning wastes were removed by vacuum cart and ultimately placed in tanks at the former Oil House Tank Farm.

Mercury manometers were used in Building B-16 to measure engine system pressure. Occasionally, a high pressure surge would blow the mercury out of the manometer tube; it is possible that mercury entered floor drains.

Seven soil borings, including SB17A2-1 through SB17A2-4, SB17A2-6, SB17B1-1, and SB17C1-1, six monitoring wells, WP-99-57, WC2-3S, DP2-3, WC-3S, WP-99-60, and MW-3, and two soil vapor samples, SG-99-32 and SG-99-34, were completed in and around Building 16 (Figure 7-11). Samples were collected adjacent to the floor drains and sumps. Analytes detected in these soil samples included BTEX, VOCs, SVOCs, PCBs, and TPH (see Appendix O for complete soil boring analytical results). Analytes detected in groundwater included BTEX, cVOCs, VOCs, SVOCs, PCBs, TPH, and cyanide. Analytes detected in soil vapor samples included the cVOCs PCE and TCE (see Appendix U for complete soil vapor analytical results).

A release has occurred from this AOC. Documentation indicates that VOCs and fuels were released to the drainage system in Building B-16 and detections of VOCs in soils collected along the drainage system, and downgradient groundwater, suggest that a release has occurred. Petroleum odors and elevated PID readings were noted for several soil sample locations at this AOC. Other potential contributing sources of VOCs and fuel include prior usage of this area in the 1940s for open storage of containers and documented releases from the Building B-34 Jet Fuel Tank Farm (see AOC 45). Concentrations of TPH and lead in soils at SB17A2-6 and PCBs at SB17A2-1 exceed RSR criteria (see Figure 7-12). No concentrations of analytes detected in groundwater exceed RSR criteria. In soil vapor, TCE was detected slightly above RSR criteria in SG-99-32 (see Figure 7-12).

AOC 5 - Stormwater and Wastewater Collection System for OATP

The stormwater and wastewater collection system includes catch basins, roof drains, sumps, underground lines, and six pump stations (Buildings B-36, B-37, B-38, B-40, B-41 and B-64-1) used to convey stormwater and wastewater. Pump stations B-37 and B-40, and associated catch basins and piping, were the only portions of the stormwater system to have conveyed industrial wastewater (CDM FPC, 1992).

In 1953, following construction of the dike, six stormwater pump stations were built and equipped with pumps for flood control (Buildings B-36, B-37, B-38, B-39 [demolished in 1971], B-40, and B-41) because of the low elevation of the facility and its' proximity to the Housatonic River (W-C, 1991). It is assumed that existing stormwater and wastewater collection lines were routed to these stormwater pump stations during the construction. The pump stations discharged directly to the Housatonic River via plant outfalls until 1976, when the OATP (Building B-64-2) and an associated pump station (Building B-64-1) were constructed to remove oil and grease from influent wastewater in the collection system. Outlet piping was reconfigured for the existing pump houses, such that dry flow and the first flush of stormwater would be routed to the OATP for treatment prior to discharge to the river via OF-007. During large storm events excessive runoff would be discharged directly from each of the pump stations to the river via OF-001 through OF-006 (W-C, 1991).

Paint-contaminated wastewater was generated during use of water-curtain-type paint booths in Building B-2. Approximately 760 liters of paint-contaminated wastewater was discharge to the system every four months. Available information indicates a zinc-chromate primer, epoxy, and non-leaded paints were used in painting processes (ESE, 1981).

Cutting oils and penetrant dyes were used during machining and testing of machined parts. Machining operations were conducted in Buildings B-2, B-3, B-6, and B-10. Waste cutting oils resulting from machining operations in Building B-2 were discharged to the OATP (ESE, 1981), via former catch basins connected to the lines running to the Building B-37 pump station (see Figure 7-11).

Building B-16 industrial wastewater was discharged to Building B-40. ECM was conducted in Building B-4, generating a sludge that was disposed of off-site and heavy metal contaminated brine wastewater that was discharged to the OATP, via the pump house in Building B-40. This pump house also received fluids from a hydraulic lift pit and floor drains in Building B-9 (see Section 6.4.3, Building B-9 Floor Drains), where batteries, oil, grease, and hydraulic fluid were stored (W-C, 1991).

Floor grates in the western portion of Building B-10 were connected to the lines running to Building B-64-1, and floor drains in central and northern Building B-6 and in southern B-3 were connected to lines running to the Building B-41 pump station.

Line to Building B-36.

The Building B-36 pump station receives stormwater flow from roof drains located on the far western end of Building B-2 and on Buildings B-52, B-58 and B-65, and from the catch basins west of Buildings B-15 and B-2 (see Figure 7-11).

Samples from soil boring locations SB05A2-1 and BR-8 were collected along lines to Building B-36, near the northwest end of Building B-65. Analytes detected in these soil samples include BTEX, VOCs, SVOCs, PCBs, and TPH (see Appendix O for complete soil boring analytical results).

The determination of a release from the Building B-36 stormwater lines is not possible due to the presence of contamination from other potential sources. The Building B-36 pump station receives stormwater flow primarily from roof drains (see Figure 7-11). Catch basins associated with this line are located within the far western portion and west of Building B-2, and between Buildings B-15 and B-52. Documented contamination exists throughout much of this area, including soil containing petroleum hydrocarbons, chromium, and lead removed during construction of B-65 (see AOC 21), as well as oil, reportedly observed during pile driving for construction of Building B-58 (see AOC 27). No contaminants were detected above RSR criteria.

Lines to Building B-37.

The Building B-37 pump station receives stormwater flow from the roof drains located on the west-central portion of Building B-2 and on Buildings B-13 and B-15, from catch basins located between Buildings B-15, B-13, and B-37, and a catch basin within Building B-13 (see Figure 7-11).

Soil borings SB08L1-6, SB08L1-7, and SB09C2-1 were completed along the line to Building B-37. Analytes detected in these soil samples included BTEX, cVOCs, VOCs, PCBs, and TPH (see Appendix O for complete soil boring analytical results).

The determination of a release from the Building B-37 stormwater lines is not possible due to the presence of contamination from other potential sources. Strong petroleum odor and elevated PID readings were noted for soil sample location SB08L1-7 and elevated PID readings for location SB08L1-6. Location SB08L1-7 is located within close proximity to the Hazardous Waste and Waste Oil Transfer Systems (see discussion of AOC 3), a source of contamination in this area. Location SB08L1-6 is located the furthest upstream of these three soil boring locations, between Buildings B-2 and B-15. Concentrations of PCBs, BTEX and TPH in soils exceed RSR criteria (see Figure 7-12).

Lines to Building B-38.

The Building B-38 pump station receives stormwater flow from the roof drains located on the central portion of Building B-2 and on Building B-12, and from catch basins located between Buildings B-2, B-13, B-38, and B-44.

Soil borings SB08L1-9 and SB09A2-1 were completed along the lines to Building B-38. Analytes detected in these soil samples included BTEX, cVOCs, VOCs, SVOCs, and PCBs (see Appendix O for complete soil boring analytical results).

The determination of a release from the Building B-38 stormwater lines is not possible due to the presence of contamination from other potential sources. Petroleum odor and elevated PID readings were noted for soil sample location SB08L1-9, located adjacent to a catch basin within the Shot Peening area of Building B-2, adjacent to the Boiler Room. Elevated PID readings (up to 636 ppm) were note for sample location SB09A2-1, located adjacent to Building B-38. This area consists primarily of fill, and open storage and staining or pooled liquid adjacent to Building B-38 are identified in available historical photographs (refer to discussion of Open Storage Area between B-16 and B-74 in subsection 7.1). Concentrations of BTEX and cVOCs in soil exceed RSR criteria (see Figure 7-12).

Lines to Building B-40.

The Building B-40 pump station receives stormwater flow from the roof drains located on the eastern portion of Building B-2 and roof drains and catch basins located near the majority of buildings between Sniffens Lane, Building B-10, and Building B-40.

Soil boring locations SB17A1-1, SB17A3-1, SB17A3-5, SB17A5-1, SB08L1-2, SB08L1-3, SB08L1-4, and SB08L1-5 have been associated with the lines to Building B-40. Analytes detected in these soil samples included BTEX, VOCs, SVOCs, PCBs, and TPH (see Appendix O for complete soil boring analytical results).

The determination of a release from the Building B-40 stormwater lines is not possible due to the presence of contamination from other potential sources. Strong odor, free product, and elevated PID readings at SB08L1-2, and strong odor and elevated PID readings were noted for location SB17A3-1 (see Table 7-7). SB08L1-2 is located immediately downgradient of Building B-33 and SB17A3-1 is located along the Dike, north of Building B-16. Documented releases associated with operations in Building B-16 (refer to discussion of AOC 4 and AOC 48) and the Building B-34 Jet Fuel Tank Farm (AOC 45), and petroleum product stored in ASTs located adjacent to Building B-33, are potential sources of contamination in this area. Furthermore, historical aerial photography and fire insurance maps have been used to identify open storage and the presence of USTs containing oil and gasoline proximal to other portions of the stormwater lines to Building B-40. Concentrations of TPH in soil exceed RSR Criteria (see Figure 7-12).

Lines to Building B-41.

The Building B-41 pump station receives stormwater flow from roof drains located on the southern portion of Building B-3 and the majority of the buildings located north and east of Sniffens Lane, as well as the catch basins within this area.

Soil boring locations SB27E11-1, SB27E11-2, and SB27E11-3 have been associated with the lines to Building B-41. These sample locations are all located to the north and west of Sniffens Lane. Analytes detected in these soil samples included VOCs and TPH (see Appendix O for complete soil boring analytical results).

A release from the Building B-41 stormwater lines has not occurred, or is not suspected. No elevated PID readings or visual or olfactory observations of contamination were noted during sampling at associated soil boring locations. Only two VOCs were detected in the soil samples, acetone and methyl isobutyl ketone. TPH was detected at SB27E11-3 adjacent to the western corner of Building B-19. No contaminants were detected above RSR criteria.

<u>Lines to Building B-64-1.</u>

The Building B-64-1 pump station receives flow from roof drains and/or floor drains/sumps in Buildings B-16, B-10 (western end), B-48, B-7, B-7A, B-9, B-2 (north-central), and B-12 (northeastern), as well as from catch basins in this area.

Soil boring locations SB17A3-2, SB17A3-3, SB17A3-4, SB17A3-8, are associated with the lines to Building B-64-1. Analytes detected in these soil samples included BTEX, VOCs, and PCBs (see Appendix O for complete soil boring analytical results).

The determination of a release from the Building B-64-1 stormwater lines is not possible due to the presence of contamination from other potential sources. Elevated PID readings and petroleum odors were noted during soil sampling at the borings along this line located to the north of Building B-16. Releases to soil in the area along the Dike north of Building B-16 have been documented. Concentrations of lead, arsenic, and PCBs in soil exceed RSR Criteria (see Figure 7-12).

AOC 6 - OATP in Building B-64-2

AOC 6 consists of the oil skimmer in Building B-64-2, the 200,000-gallon surge tank adjacent to B64-2 and the 10,000-gallon sodium hydroxide (NaOH)/Alum tank at B64-2. The OATP was constructed in 1976 to remove oil and grease from influent stormwater and wastewater in the collection system (CDM FPC, 1992).

Prior to plant closure, the OATP operated continuously with an average flow of 1.8 million gallons per day, and a maximum design flow of about 4,200 gpm (INEL, 1991). Influent to the OATP entered the surge tank for flow equalization. Treatment involved coagulation and flocculation by addition of liquid alum in a flash mixer, and dissolved air flotation and skimming in a flotation chamber. Effluent from the OATP is discharged to the Housatonic River through OF-007 and waste sludge and oils are shipped off-site.

The OATP received industrial wastewaters including boiler blowdown, cooling water, laboratory wastes, photographic processing wastes, paint-contaminated wastewater, soluble and insoluble cutting oils, spent hydraulic fluid, penetrant dyes, brine, and emulsion cleaning detergents (ESE, 1981; W-C, 1991).

Following the construction of the waste transfer system (AOC 3) and closure of the wastewater collection lines in the early- to mid-1980s, the OATP continued to receive wastewater in the form of supernatant pumped from waste oil tanks at the former Oil House Tank Farm. The continuous or intermittent presence of oil, copper, 1,1,1-TCA, and ammonia discharge to the OATP was noted during sampling in the early 1990s (W-C, 1991).

Accidental releases to the stormwater system have been documented. The first included the discharge of 25 to 30 pounds of chromic acid in May of 1978, when a tank containing residual chromic acid was accidentally overturned and the acid discharged to a storm drain (CDM FPC, 1992). The second involved a spill of approximately 20 gallons of “Zyglo” penetrant dye into a storm drain in October of 1981 (W-C, 1991).

There is insufficient information to determine if a release has occurred. No samples have been collected at AOC 6.

AOC 24 - Discharge to the Housatonic River at Outfall-007

Treated stormwater from the OATP discharges through Outfall 007. The OATP receives and treats stormwater and dry weather flow, including the first flush of stormwater, from the six storm pump stations. Industrial wastewater discharged to the OATP included boiler blowdown, cooling water, laboratory wastes, photographic processing wastes, paint-contaminated wastewater, soluble and insoluble cutting oils, spent hydraulic fluid, penetrant dyes, brine, and emulsion cleaning detergents (ESE, 1981; W-C, 1991).

Four chemical releases to the intertidal flats have been documented. These releases involved:

- In May 1978, a spill of 25 to 30 pounds of chromic acid was discharged into the OATP and into the river via OF-007 (W-C, 1991).
- In August 1978, CTDEP was advised that a yellow plume of Cr(VI) was extending approximately 200 yards from OF-007 (CDM FPC, 1992). This release occurred during a period when it is suspected that effluent from the CWTP was routed to the OATP for discharge via OF-007.
- Approximately 75 gallons of oil sludge from the OATP bypassed clogged skimmers and discharged from OF-007 in July 1979 (W-C, 1991).
- In October 1981, approximately 20 gallons of “Zyglo”, a fluorescent metal penetrant dye was spilled into a storm drain and discharged from OF-007 (W-C, 1991).

Sediment sample location OF-007 (SD) was taken at Outfall 007. Analytes detected in sediment included cVOCs, VOCs, PAHs, SVOCs, and PCBs (see Appendix Q for complete sediment analytical results).

A release has occurred from this AOC. Several documented releases have occurred at this AOC. As there are no RSR criteria for sediment, no comparisons were performed.

AOC 52 - Facility Outfalls-001 through -006 and the Intertidal Flats

In 1953, six storm pump stations were built (Buildings B-36, B-37, B-38, B-39 [demolished in 1971], B-40, and B-41) that discharged to the Housatonic River via associated outfalls (OF-001 through OF-006) (W-C, 1991). Also, the outfalls received surface runoff, which may have contacted wastes potentially spilled on the site grounds (W-C, 1991).

In 1976, the OATP (Building B-64-2), an associated pump station (Building B-64-1), and OF-007 were constructed to address oil and grease from influent wastewater in the collection system, in order to meet NPDES requirements. Outlet piping was reconfigured for the existing pump houses, such that dry flow and the first flush of stormwater would be routed to the OATP for treatment prior to discharge to the river via OF-007. The result was that discharge from OF-001 through OF-006 would occur only during large storm events when excessive runoff was present (W-C, 1991).

Specific amounts or constituents of materials/wastes that may have been discharged from OF-001 through OF-006 in the past is not known; however, any material or waste discharged or spilled into storm drains prior to construction of OATP was potentially released to the intertidal flats through one of the outfalls. Industrial wastewaters produced at SAEP have included boiler blowdown, cooling water, laboratory wastes, photographic processing wastes, paint-contaminated wastewater, soluble and insoluble cutting oils, spent hydraulic fluid, penetrant dyes, brine, and emulsion cleaning detergents (ESE, 1981; W-C, 1991). These wastestreams likely contained waste fuels and solvents in addition to documented compounds. Further information regarding the wastestreams potentially handled by the outfalls is provided in the previous discussions for AOCs 4 and 5 presented in this section.

Sediment sample locations OF-001 (SD) through OF-006 (SD), TA1, TC1, TD1, TE1, TF1, and SDTA001A are located adjacent to the six facility outfalls associated with the stormwater system. Analytes detected in these sediment samples include BTEX, cVOCs, VOCs, SVOCs, PAHs, and PCBs (see Appendix Q for complete sediment sample analytical results).

A release has occurred from this AOC. Historically, waste oils, fuels, solvents, and paints likely have been released to the storm and wastewater lines which lead to Outfalls OF-001 through OF-006. Solvent, PCBs, and fuel-related contaminants were detected in sediment samples located adjacent to the six facility outfalls associated with the stormwater system. It should be noted however, that these samples are located off the main portion of the SAEP property within the tidal flats, in an area of the Housatonic River that likely has been contaminated as a result of the numerous industrial operations upstream. Furthermore, the current SAEP shoreline is a result of several expansions, most notably in 1943, which utilized both river sediments and fill from off-site. As there are no RSR criteria for sediment, no comparisons were performed.

7.5 MISCELLANEOUS AOCS

The Miscellaneous AOCs include:

- AOC 17 – Soil Pile, South Parking Lot
- AOC 20 – Causeway
- AOC 21 – Building B-65 Area; Previous Area of Buildings B-54 and B-55
- AOC 27 – Building B-58 and Associated Satellite Accumulation Areas
- AOC 29 – Building B-48 and Associated Satellite Accumulation Areas
- AOC 36 – Building B-12 and Associated Satellite Accumulation Areas
- AOC 41 – Building B-9 Former USTs
- AOC 42 – Building B-9 USTs
- AOC 46 – Building B-52 Former UST
- AOC 47 – Building B-73 Radioactive Waste Storage Area
- AOC 54 – Building B-17
- AOC 70-72 – ASTs near Building B-44
- AOC 73 – Fuel, Lubricating, and Hydraulic Oils near Building B-69
- AOC 74 – Building B-2 PCB Transformers
- AOC 75 – Building B-3 PCB Transformers
- Former Pits or Lagoons near the North Parking Lot
- Shed North of Building B-12 Used to Store Cuttings

Figure 7-13 presents the locations and boundaries of the AOCs and the associated explorations. Table 7-8 presents a summary of samples associated with each AOC, the field observations, detected analytes, release determination, and analytes with concentrations exceeding CTDEP RSR criteria.

AOC 17 - Soil Pile, South Parking Lot

In September 1989, an estimated 3,000 cubic yards of contaminated soil, discovered during removal of USTs at the Jet Fuel Tank Farm near Building B-34, were excavated and stockpiled at the South Parking Lot. Toluene and xylene were detected at levels up to 5,500 mg/kg in these soils (CDM FPC, 1992). Additional samples collected just outside the area of removal identified soil containing TPH at concentrations up to 5,500 mg/kg (Zecco, 1990).

In 1990, Buildings B-52 and B-55 were demolished in order to construct Building B-65 for materials receiving and storage. Building B-52 and B-55 had previously been used for production material warehousing. During excavation for the Building B-65 foundation, contamination, which included dark oily liquids and yellow, blue and reddish colored soils, was discovered. Analytical results indicated that contaminated soils contained petroleum hydrocarbons and inorganics including cadmium, chromium, lead and copper distributed throughout much of the Building B-65 area (Textron 1991). An estimated 12,000 cubic yards of contaminated soil was excavated to the low-tide water level and added to the soil pile at the South Parking Lot (W-C, 1991).

SB19A1-1 is located within the area of the former soil pile. SB22A1-1, SB22A1-2 and SB22A1-3 are located within the area of final placement of this soil (see Table 7-8). Analytes detected in soil at these locations include BTEX, VOCs, PAHs, SVOCs, and PCBs.

A release has occurred from this AOC. Remedial actions have been completed for this AOC. Low levels (less than 2 mg/kg) of fuel-related contamination were detected in soil borings from the former area of the Soil Pile and from the final placement location of this soil. Concentrations of 1,1,2,2-TCA, PAHs, SVOCs, and cadmium exceed RSR criteria for samples from borings completed in the final placement location of this soil (see Figure 7-14).

AOC 20 - Causeway

The Causeway was constructed in 1929 for access to the river channel to launch seaplanes. Fill of unknown origin was placed along the northern edge of the Causeway during the 1950s and 1960s, at a time when the Causeway was used for fire training exercises. The fill contained soil, cobbles, and construction debris including, wood, concrete, brick, asphalt, and rebar) (W-C, 1991).

A removal action was completed on the Causeway in 2001, to eliminate the potential for contaminant migration and receptor contact with unacceptable levels of contamination. The Basis of Design (Harding ESE, 2001) and the “Final Non-Time Critical Removal Action Certification Report” document the actions completed on the Causeway. For further details on the Causeway, see Section 1.6.8 of this document.

AOC 21 - Building B-65 Area; Previous Location of Buildings B-52 and B-55

In 1990, Buildings B-52 and B-55 were demolished in order to construct Building B-65 for materials receiving and storage. Building B-52 and B-55 had previously been used for production material warehousing. During excavation for the Building B-65 foundation, contamination, which included dark oily liquids and yellow, blue and reddish colored soils, was discovered. Analytical results indicated that contaminated soils contained petroleum hydrocarbons and inorganics including cadmium, chromium, lead and copper distributed throughout much of the Building B-65 area (Textron, 1991).

Groundwater samples collected from pits during the soil excavation contained cVOCs, fuel-related VOCs, and inorganics. The chlorinated VOCs detected included TCE, TCA, and degradation products. Fuel-related VOCs were mostly below detection limits with toluene and xylene at low levels in a few samples. The highest levels of cVOCs in groundwater were found along the Building B-2 side of Building B-65. Chromium was detected in groundwater where yellow-stained soils were identified (Textron 1991).

This contamination was believed to partially be the result of disposal of zinc-chromate undercoat used in aircraft painting processes conducted in Building B-2 in the 1940s, and/or from fill obtained from contaminated river sediments. An estimated 12,000 cubic yards of paint- and petroleum-contaminated soil was excavated to the low tide water level and placed in a soil pile in the South Parking Lot (AOC 17) (W-C, 1991).

Soil boring SB03A1-1 was completed between the northwestern corner of Building B-65 and the northernmost corner of Building B-2 to confirm the presence of contaminated soil outside the Building B-65 footprint where soils were excavated. Soil borings SB06A2-2 and SB06A2-3, located north of Building B-65, and SB09B7-1, located southeast of Building B-65 indicate low concentrations of BTEX (0.0008 mg/kg) and PCBs.

A release has occurred from this AOC. Analytes detected in soil include benzene and toluene (at less than 1 mg/kg) and PCB 1254 and 1260 (see Appendix O for complete soil boring analytical results). The concentration of TPH in soil at SB06A2-2 exceeds RSR criteria.

AOC 27 – Building B-58 and Associated Satellite Accumulation Areas

Building B-58, built in 1967, was used for missile assembly and as a standards laboratory. Waste 1,1,1-TCA and waste jet fuels were stored in satellite accumulation areas located in the building. Oil was reportedly observed in subsurface soil during pile driving for construction of the building (W-C, 1991). This area was used for open storage in the 1950s and 1960s. Additionally, this area is located next to the Former Oil House Tank Farm.

A release from this AOC has not occurred, or is not suspected. Although visual soil contamination was identified during building construction, this contamination is not associated with activities conducted in the building. It is not believed that activities within the building were associated with a release. No analytical samples are associated with this AOC.

AOC 29 - Building B-48 and Associated Satellite Accumulation Areas

Building B-48, built in 1961, is a Quonset hut that was used as a warehouse to store tooling and equipment. Paint cans and waste paint were stored in satellite accumulation areas located in the building (CDM FPC, 1992).

Soil boring EBS29-1 is located within the northern portion of Building B-48 (see Table 7-8). Analytes detected in soil from EBS29-1 include toluene (see Appendix O for complete soil boring analytical results).

The determination of a release from this AOC is not possible due to the presence of contamination from other potential sources. Building B-48 was a storage facility for tools and equipment. Paint storage cans and waste paint were stored in 55-gallon drums in satellite accumulation areas located throughout the building. Prior to construction of Building B-48, aerial photographs indicated that this area was used for open storage (USEPA, 1990), which likely impacted contaminant concentrations in this area. Analytical results from the sample collected from beneath this building indicate low levels (2 mg/kg) of toluene. The concentration of dichloromethane in soil at this AOC exceeds RSR criteria (see Figure 7-14).

AOC 36 - Building B-12 and Associated Satellite Accumulation Areas

Building B-12, built in 1942 as the Tool, Die, Jig and Maintenance Building, was used for tool storage and maintenance activities. Waste filters were stored in accumulation areas located in this building. A 1943 fire insurance map depicts a machine oil storage area adjacent to the western end of Building B-12 (AFM FIC, 1943). This area is not identified on available aerial photographs, and a 1956 map shows three 1,000 gallon anhydrous ammonia tanks (FIA, 1956) in this area.

There is insufficient information to determine if a release has occurred from this AOC. No soil samples were collected from this AOC.

AOC 41 - Building B-9 Former USTs and AOC 42 - Building B-9 USTs

Fire maps indicate gasoline USTs in the area southeast of Building B-9 and north of Building B-10 as early as 1931 (see Figures 1-4 and 1-5)(AFM FIC, 1931). The PAS indicates that various numbers and sizes of leaded and unleaded gasoline USTs were located in this area (W-C, 1991).

Soil boring SB33A1-1 was completed southeast of Building B-9 in the location of the former USTs (see Table 7-8). Oily stains and a petroleum odor were observed during soil sampling; however, historical information indicates a hydraulic cylinder in Building B-9 leaked oil and was removed. Therefore, it is not possible to determine if the visual contamination identified is a result of the USTs or other activities in the area. No PID was available during sampling. Analytes detected in soil near the former USTs included low levels BTEX and VOCs (see Appendix O for complete soil boring analytical data).

A release has occurred from these AOCs. The USTs associated with the AOCs handled leaded and unleaded gasoline. Low levels (1 mg/kg or less) of toluene and xylenes were detected in soil, and oily stains and odors were observed during sampling. No concentrations of contaminants exceed RSR criteria.

AOC 46 - Building B-52 Former UST

A 1,000-gallon oil UST was located beneath Building B-52 until it was sand filled and abandoned in 1969 (W-C, 1991). The location of the UST is shown on Figure 7-14.

Soil boring location SB08J1-1 was completed adjacent to the UST to document post-closure conditions (see Table 7-14). Elevated PID readings (maximum of 5.1 ppm) were observed during boring completion, and analytes detected in soil at this location included BTEX, VOCs, SVOCs, and PCBs (see Appendix O for complete soil boring analytical results).

A release has occurred from this AOC. Analytical results from the boring completed indicate low levels (less than 1 mg/kg) of fuel-related contaminants. No concentrations of detected analytes in soil at SB08J1-1 are greater than the RSR numerical criteria (see Figure 7-14).

AOC 47 - Building B-73 Radioactive Waste Storage Area

Building B-73 was used to store radioactive magnesium thorium waste in 55-gallon drums on pallets. Waste was stored for periods of up to one year prior to shipment to England for disposal (CDM FPC, 1992). Magnesium thorium waste was generated during machining done during engine production. The use of magnesium thorium at SAEP was handled under NRC License Number STB-393. Following completion of several radiological surveys, the NRC terminated License Number STB-393 on September 29, 2000, releasing former radioactive waste storage areas at SAEP for unrestricted use.

AOC 54 - Building B-17

AOC 54 consisted of a 10,000-gallon aboveground storage tank that contained No. 4 fuel oil. This tank was used to supply fuel to a boiler located in this building that served as a steam generating plant (W-C, 1991). A 1956 fire map depicts a 10,000-gallon fuel oil tank in the southern portion of Building B-17 and the tank is also identified on a fire map from 1986 (FIA, 1986). Building B-17 was constructed in 1952, and the tank was likely installed at that time. The steam plant was no longer in use by the mid 1980s and the tank and boiler were removed in the late 1980s. The building was then used for storage of test equipment.

In 1988, stained soils, apparently caused by petroleum, were observed in this area but not removed (CDM FPC, 1992). 1,1,1-TCA was stored in quantities of up to 55 gallons in Building B-17 until approximately 1993 (ABB-ES, 1996).

There is no evidence or record of a release of fuel oil to the concrete floor or to the floor drain. If a release of fuel oil from the Building B-17 tank had occurred, it would have flowed to the drain that is connected to the stormwater collection system. Soil boring SB24C1-1 was completed adjacent to a drain in center of B-17 (Table 7-8). Analytes detected in soil include BTEX, VOCs, and one phthalate.

A release from this AOC has not occurred, or is not suspected. No concentrations of detected analytes in soil at this AOC are greater than the RSR numerical criteria.

AOC 70-72 - ASTs near Building B-44

AOCs 70 through 72 have been grouped based upon their proximity to each other adjacent to Building B-44. AOC 70 consists of a 10,000-gallon oil-alum tank, transferred from its location near Building B-13 in 1988 (AOC 7); AOC 71 consists of a 5,000-gallon methanol AST; and AOC 72 consists of 400,000-gallon Fuel Oil #6 AST.

There is insufficient evidence to determine if a release has occurred from these AOCs. No samples are associated with these AOCs.

AOC 73 - Fuel, Lubricating, and Hydraulic Oils near Building B-69

This AOC consist of fuel and lubricating and hydraulic oil storage near former Building B-69 from approximately 1980-1991. Reportedly, less than 13,750 gallons (at any given time) of these fluids were stored in 55-gallon drums in this area (ABB-ES, 1996).

There is insufficient information to determine if a release has occurred from this AOC. No samples were collected near this AOC.

AOC 74 – Building B-2 PCB Transformers

This AOC includes 11 PCB (pyranol) transformers that are located in roof vaults at Building B-2. Pyranol oil contains PCBs at concentrations greater than 500 ppm.

A release from this AOC has not occurred, or is not suspected. Although no sample locations are associated with this AOC, the location of the transformers on the building roof, visual observations, and regular inspections indicate no releases have occurred.

AOC 75 – Building B-3 PCB Transformers

Two PCB (pyranol) transformers are located in Building B-3. Pyranol oil contains PCBs at concentrations greater than 500 ppm. Two additional PCB-containing transformers are located at Building B-3A. These transformers are located in floor vaults in Buildings B-3 and B-3A.

A release from this AOC has not occurred, or is not suspected. Although no sample locations are associated with this AOC, visual observations and regular inspections indicate no releases have occurred.

Former Pits or Lagoons - North Parking Lot

Aerial photography from 1943 (USEPA, 1990) indicates the presence of possible pits or small lagoons in the North Parking Lot. Currently, this area is located west of Building B-80, near the town of Stratford sanitary sewer pumping station. In 1944, Building B-2 was expanded to the northwest, and during the expansion this area was likely filled. A 1949 aerial photograph (USEPA, 1990) does not indicate pits or lagoons in this area. No disposal history for this area is available.

Soil borings SB01A1-1 and SB01A1-2 were completed in the former location of the possible pits or small lagoons (see Table 7-8). Analytes detected in soil include BTEX, VOCs, and TPH.

A release has occurred from this area. Analytical results from samples collected in this area indicate fuel-related contaminants. Concentrations of arsenic and TPH in soil exceed RSR criteria (see Figure 7-14).

Shed North of Building B-12 Used to Store Cuttings

A shed located to the northwest of Building B-12 was used to store metal cuttings in the 1990s. The shed was constructed between 1990 and 1995, and still exists, though it is no longer used to store metal cuttings. Metal cuttings were stored in a dumpster within the shed, which consists of a concrete bermed area and corrugated metal sides and roof. A former building was located in this area during the 1940s, and was used as a test house (AFM FIC, 1941) and as a mould shop (AFM FIC, 1943). This building was demolished or removed between 1949 and 1953. Machining oils were stored in an adjacent portion of Building B-12 (AFM FIC, 1943). Aerial photography from 1970 and 1980 show open storage in this area (USEPA, 1990).

Soil boring SB13A1-1 was taken within the metal cuttings shed (see Table 7-8). No analytes were detected in soil from this sample location (see Appendix O for complete soil boring analytical results).

A release from this area has not occurred, or is not suspected. No analytes were detected in soil at the shed located north of Building B-12. No concentrations of detected analytes at sample locations associated with this AOC are greater than the RSR numerical criteria.

8.0 NATURE AND DISTRIBUTION OF CONTAMINATION

This section provides the nature and distribution of contamination in soil, groundwater, soil vapor, indoor air, surface water, and sediment followed by comparison to CTDEP RSR criteria.

8.1 SOIL

8.1.1 Distribution of Analytes in Soil

The following subsection describes the nature and distribution of soil contamination. The discussion is first organized by chemical then into three depth intervals; 0-2 feet bgs, 2-4 feet bgs, and 4-15 feet bgs to evaluate the vertical distribution of chemicals and the possible relationship of soil contamination to fill.

Chlorinated Volatile Organic Compounds. Twenty-two cVOCs were detected in site soils (see Table 8-1). Figures 8.1-1 through 8.1-23 present the distribution and concentrations of cVOCs in soils. Concentration ranges are based upon CTDEP RSR criteria, and baseline soil conditions (Appendix N). As discussed in Appendix N, the observed locations of cVOCs greater than 1 mg/kg correlate well with the areas of solvent use and releases. In general, cVOCs less than 1 mg/kg have been detected in soils across the entire Site, including those areas where no industrial activity or chemical storage was known to have occurred.

The highest detected concentrations (greater than 10 mg/kg) of cVOCs in soils were detected in the vicinity of Building B-37 and B-38 pump stations. An exception is 1,1,2,2-TCA detected near Building B-7A in the 4-15 feet bgs depth interval. Several cVOCs greater than 1 mg/kg were also detected in the vicinity of Buildings B-6A, B-7/B-7A, B-15, the B-74 Tank Farm, and beneath the central and southern part of Building B-2. In general the number of locations where cVOCs were detected and the detected concentrations decrease with depth (see Table 8-1).

Fuel-related Volatile Organic Compounds. The fuel-related constituents, benzene, toluene, ethylbenzene, and xylenes (BTEX), were detected in Site soils (see Table 8-1). Figures 8.1-24 through 8.1-26 present the distribution of BTEX. For each soil boring location where BTEX was detected greater than 0.2 mg/kg (GB PMC for benzene), the figures present the associated soil boring sample location and the constituent with the maximum detected concentration at that location for that depth interval. Toluene was the most frequently detected BTEX constituent. In general, toluene and xylene were detected at concentrations greater than 10.1 mg/kg (GB PMC for ethylbenzene) within the area of Buildings B-37, B-38, and B-74. Concentrations ranging from 0.2 to 10.1 mg/kg were detected within the area encompassed by Buildings B-13, B-15, B-38, and B-74, as well as near Buildings B-6A, B-7, and B-16. Concentrations less than 0.2 mg/kg were detected across the majority of the Site in the 0-2 feet bgs interval, with the number of locations where BTEX constituents were detected decreasing with depth. The number of BTEX constituents at concentrations greater than 10 mg/kg is greatest in the 2-4 feet bgs interval.

Other Volatile Organic Compounds. Eight other (non-chlorinated and non-BTEX related) VOCs were detected in Site soils (see Table 8-1). Figures 8.1-27 through 8.1-29 present these analytes and their detected concentrations. For each soil boring location where other VOCs were detected at concentrations greater than 1 mg/kg, the figures present the associated soil boring sample location and the VOC constituent with the maximum detected concentration at that location for that depth interval. VOC analytes detected at the maximum concentration at sample locations with VOC concentrations greater than 1 mg/kg include acetone, methyl ethyl ketone (2-butanone), and methyl isobutyl ketone. Spatially, soil sample locations with VOC analyte concentrations greater than 1 mg/kg were limited to an area near Buildings B-37, B-38, and B-74, areas both north and west of Building B-19, and a sample location near Building B-33. In general, other VOC concentrations greater than 1 mg/kg were limited to the 0-2 feet bgs interval; no concentrations greater than 1 mg/kg were detected in the 2-4 feet bgs interval.

SPLP VOCs in Soil. SPLP VOC analytical results yielded detections of dichloromethane, toluene, total xylenes, ethylbenzene, TCE and cis-1,2-DCE (see Table 8-2).

Polynuclear Aromatic Hydrocarbons. Eighteen PAHs were detected in site soils (see Table 8-1). Figures 8.1-30 through 8.1-32 present the distribution of PAHs in soils. For each soil boring location where PAHs were detected, the figures present the associated soil boring sample location and the PAH with the maximum concentration at that location for that depth interval. Concentration ranges presented on these figures are based upon CTDEP RSR criteria for select PAHs (see Figures 8.1-30 through 8.1-32).

PAHs were detected in soils across the entire Site at concentrations between 0.001 and 0.5 mg/kg, including areas where no industrial activity was known to have occurred such as the West and South Parking Lots. The distribution of PAHs with concentrations ranging from 0.5 to 1.0 mg/kg is more limited, and suggests that at least some of the detected PAHs within this range may be present as a result of potential releases. However, there are detections within this range in areas where no industrial activity is known to have occurred (e.g., South Parking Lot and the extreme northern end of the Dike), suggesting that concentrations within this range may also be reflective of PAHs used in fill or leached from asphalt surfaces;

The areas of PAHs with concentrations greater than 1.0 mg/kg correlate reasonably well with areas of fuel usage and identified releases. These areas include the Building B-13 and B-15 area and the area underneath the South Parking Lot filled with soils from B-34 and B-65 excavations.. In general, the number of locations where PAHs were detected, and the detected concentrations, decrease with depth. Several anomalies with respect to identified historical activities include PAH concentrations greater than 1 mg/kg detected southwest of Building B-2 along Main Street, and concentrations between 0.1 and 1 mg/kg in the South Parking Lot.

Other Semi-Volatile Organics. Thirty-seven other SVOCs were detected in soil (see Table 8-1). Figures 8.1-33 through 8.1-35 present the distribution of SVOCs. For each soil boring location where SVOCs were detected at concentrations greater than 1 mg/kg, the figures present the associated soil boring sample location and the SVOC(s) with the maximum detected concentration at that location for that depth interval. Concentration ranges presented on these figures are based upon CTDEP RSR criteria for select SVOCs (see Figures 8.1-33 through 8.1-35). In general, non-PAH-related SVOCs were less than 1 mg/kg. Concentrations greater than 1 mg/kg were generally detected in fill, both adjacent to the Dike, and in the portion of the South Parking Lot that received fill from the Building B-65 area and the Jet Fuel Tank Farm.

SPLP SVOCs in Soil. SPLP SVOC analytical results indicated detection of PAHs at similar frequencies to those detected in total mass analyses. The most frequently detected SPLP PAHs were phenanthrene, acenaphthene, fluorene, 2-methylnaphthalene, carbazole, dibenzofuran, and naphthalene. The highest concentration of PAHs detected was for naphthalene, at a concentration of 0.096 mg/L, in soil boring EBS12-2, located at Building B-74.

Polychlorinated Biphenyls. The PCB Aroclors 1016, 1248, 1254, and 1260 were detected in soils; however Aroclor 1260 constitutes the majority (see Table 8-1). Figures 8.1-36 and 8.1-37 present the distribution of PCBs in soil. The majority of PCBs were in the 0-2 feet bgs interval. Concentrations greater than 1 mg/kg were generally limited to areas adjacent to the Dike in fill material. No PCBs were detected in the 4-15 feet bgs range.

Total Petroleum Hydrocarbons. TPH was detected in approximately two-thirds of all samples submitted for TPH analysis (see Table 8-1). TPH was detected in areas of past fuel storage and use, but also in areas where fuels were not used (see Figures 8.1-38 and 8.1-39). There are several possible sources of TPH, including fuel releases at tank farms near Building B-34 and Building B-74 and other UST or AST locations, fill material, and dirt parking lots that were once oiled to reduce dust. The baseline soils evaluation indicated that TPH levels greater than 872 mg/kg potentially indicate areas of potential releases of fuels (Appendix N). With the exception of a single detection in soil from the 2-4 feet bgs interval, TPH is limited to the 0-2 feet bgs interval.

Inorganics. Twenty six inorganics were detected in site soils, seventeen of which have CTDEP RSR criteria (see Table 8-1). Figures 8.1-40 through 8.1-75 present the distribution of the inorganics with applicable CTDEP RSR criteria in soils. Appendix N provides discussion of the estimation of baseline concentrations for inorganics in site soils. There are several possible sources for the inorganics detected in soils, including metal plating/finishing operations, fill materials brought onto the Site or those dredged from the Housatonic River, and the natural occurrence of these analytes. The inorganics selected for evaluation in Appendix N include only those analytes with numerical RSR criteria. Deflection points were determined by review of the concentration distribution plots for each of these analytes, and represent the minimum concentration which supports potential contaminant release. Deflection points have been used as breakpoints, were applicable, on figures presented in this subsection. Hexavalent chromium and cyanide are not naturally occurring, and historically were used in metal plating/finishing operations at the Site.

Hexavalent chromium was detected in soils beneath the former B-2 Chromium Plating Facility at concentrations up to 640 mg/kg (see Figures 8.1-58 through 8.1-60). With the exception of several detections in the 2-4 feet bgs interval, the majority were limited to the 0-2 and 4-15 feet bgs intervals. Concentrations greater than 100 mg/kg were detected at both these intervals, at locations along the northwestern end of the former B-2 Chromium Plating Facility. The highest concentrations were detected near the western end of the former B-2 Chromium Plating Facility.

Detection of cyanide in soil was most frequent, and concentrations the highest, in soil from the 0-2 feet bgs interval (see Figures 8.1-61 through 8.1-63). In general, cyanide was detected from beneath the former B-2 Chromium Plating Facility, as well as from the area of Buildings B-37 and B-38. Open storage of drums was historically conducted in the area near Buildings B-37 and B-38, and cyanide detected in soil in this area potentially resulted from storage of cyanide waste.

Cadmium was detected in soils across the entire facility at concentrations less than the deflection point (7.2 mg/kg). Cadmium was generally detected in soil at concentrations greater than the deflection point within areas that received fill (see Figures 8.1-52 through 8.1-54). Concentrations greater than RSR criteria were detected near Buildings B-7 and B-8, as well as beneath northern Building B-2.

Total chromium was detected in soils across the facility at concentrations less than the deflection point (146 mg/kg). In general concentrations greater than the deflection point were primarily limited to beneath the former B-2 Chromium Plating Facility, but were also detected near the Hazardous Waste Storage Area between Buildings B-74, B-13, and the OATP (see Figures 8.1-40 through 8.1-42).

Copper was detected in soils across the entire facility at concentrations less than the deflection point (87.8 mg/kg). Concentrations greater than the deflection point were generally detected within areas that received fill material, including area near Buildings B-13, B-15, B-19, and B-38 (see Figures 8.1- 43 through 8.1-45).

Lead was detected in soils across the entire facility at concentrations less than the deflection point (65.7 mg/kg). Concentrations greater than the deflection point were limited to fill soil adjacent to the Dike (see Figures 8.1-46 through 8.1-48).

Mercury was detected in soil across the majority of the Site; concentrations greater than deflection point (0.405 mg/kg) were limited to fill soil adjacent to the Dike (see Figures 8.1-55 through 8.1-57).

Nickel was detected in soils across the entire facility at concentrations less than the deflection point (37.6 mg/kg). Concentrations greater than the deflection point were generally detected in the 0-2 and 2-4 feet bgs intervals, beneath the former Chrome Plating Facility, between Buildings B-15 and B-38, near the Causeway, beneath Buildings B-4 and B-3 (see Figures 8.1-49 through 8.1-51).

Arsenic was detected in soils across the entire facility at concentrations less than the deflection point (8.7 mg/kg) at all depth intervals (8.1-64 through 8.1-66). Locations with concentrations greater than the deflection point are limited to fill soil adjacent to the Dike.

Beryllium, selenium, silver, and thallium were detected at concentrations less than 2 mg/kg, the Residential and I/C DEC for beryllium, throughout much of the facility (see Figures 8.1-67 through 8.1-69). Concentrations greater than 2 mg/kg were generally detected in fill soils adjacent to the Dike.

Zinc was detected in soils across the entire facility at concentrations less than the deflection point (175 mg/kg). Concentrations greater than the deflection point are generally located within areas that received fill (8.1-70 to 8.1-72). In the two upper depth intervals (0-2 feet and 2-4 feet, bgs), concentrations greater than the deflection point are relatively numerous in the area of Buildings B-13 and B-15.

Antimony, barium, and vanadium were detected in soils across the entire facility at concentrations less than the 27 mg/kg RSR criterion value for antimony (see Figures 8.1-73 through 8.1-75). Concentrations greater than 27 mg/kg were generally limited to areas of fill adjacent to the Dike. The highest detected concentrations, concentrations of barium greater than 470 mg/kg, were detected in the area between Buildings B-13, B-15 and B-2.

SPLP Inorganics. Analytical results for SPLP inorganics correlate fairly well with the trends observed for total inorganics results (Tables 8-1 and 8-2). The majority of site-related inorganics detected in site soils are located beneath the northern portion of Building B-2 and along the Dike. Detections are also noted for samples from the West Parking Lot, South Parking Lot, CWTP and other locations within the main portion of the facility.

Field Observations. Figure 8-1 presents a summary of field observations as interpreted from available soil boring logs, including observations of free product, staining, solvent or petroleum odors (olfactory), and elevated PID readings. Areas with a high density of noted observations include the area encompassing Buildings B-13, B-15, B-38, and the B-74 Tank Farm, the central portion of Building B-2, and along the Dike, north of Buildings B-16 and B-19. Free product was noted at six locations, consistent with areas that have the highest densities of field observations made. Observed odors were also frequently noted for the areas listed above. Observations of staining were generally limited to areas of fuel and solvent storage. Elevated soil PID readings were observed across the Site in areas associated with fuel and solvent storage and manufacturing.

8.1.2 Comparison to Criteria

Figures 8.1-76 through 8.1-94 present a comparison of soil analyte concentrations to RSR criteria. The figures are organized by chemical class as well as by depth interval. Analytical data were compared to Residential DEC, I/C DEC, and GB PMC (Table 8-3). SPLP organics were compared to 10 x GWPC GA Criteria (Figure 8.1-94).

Chlorinated Volatile Organic Compounds. Eleven individual cVOC were detected in soil at concentrations greater than RSR criteria (Table 8-3). TCE was the cVOC that most frequently exceeded criteria. In general, cVOC concentrations greater than I/C DEC and GB PMC criteria were detected in soil within the 0-2 and 2-4 feet bgs intervals, in the area of Buildings B-13, B-15, B-74, and B-38 (see Figures 8.1-76 through 8.1-78).

Fuel-related Volatile Organic Compounds. BTEX was detected in soil at concentrations greater than RSR criteria values (see Table 8-3). Detected concentrations exceeded GB PMC criteria only. Xylene most frequently exceeded criteria. In general, BTEX concentrations greater than RSR criteria were detected in soil from the 0-2 feet bgs interval in the area of Building B-37, B-38, and B-74 (see Figures 8.1-79 through 8.1-81).

Other Volatile Organic Compounds. No individual non-chlorinated VOC analytes were detected in Site-wide soil at concentrations greater than RSR criteria values (see Table 8-3).

Polynuclear Aromatic Hydrocarbons. Eleven individual PAHs were detected in soil at concentrations greater than RSR criteria (see Table 8-3). The majority were from the 0-2 feet bgs range and exceed the I/C DEC and GB PMC (8.1-82 through 8.1-84). Concentrations that are greater than CTDEP RSR criteria in this depth interval are generally located beneath the northeast corner of Building B-2 and the South Parking Lot. Concentrations of PAHs greater than RSR criteria were detected in soil from 2-4 ft bgs beneath Building B-13 and 4-15 ft bgs near Building B-72 and beneath the northeast corner of Building B-2.

Other Semi-Volatile Organics. Three individual SVOC (non-PAH) analytes were detected in Site-wide soil at concentrations greater than RSR criteria values (see Table 8-3). Detected concentrations greater than RSR criteria were detected only in the 0-2 feet bgs interval, and did not exceed I/C DEC. In general, SVOC concentrations greater than RSR criteria were detected in fill soils adjacent the Dike and from a portion of the South Parking Lot where fill resulting from material excavated during the construction of Building B-65 was used in grading (see Figures 8.1-85 through 8.1-87).

Inorganics. Ten inorganics were detected in soil at concentrations greater than RSR criteria (see Table 8-3). The majority were from the 0-2 feet bgs (see Figures 8.1-88 through 8.1-90). In general concentrations greater than RSR criteria for inorganics are within areas constructed of fill, including areas within several hundred feet of the Dike and the South Parking Lot, and beneath the former B-2 Chromium Plating Facility.

Polychlorinated Biphenyls. Two PCB Aroclors were detected in soil at concentrations greater than RSR criteria (see Table 8-3 and Figures 8.1-91 through 8.1-93). Only one sample location from the 2-4 feet bgs interval had concentration greater than CTDEP RSR criteria, while no concentrations of PCBs at the one sample location from the 4-15 feet bgs were greater than CTDEP RSR criteria. Concentrations of PCBs greater than CTDEP RSR criteria were generally detected in fill soil near the Dike, from Building B-37 to beneath Building B-16.

Total Petroleum Hydrocarbons. TPH was detected in at concentrations greater than CTDEP RSR criteria from the 0-2 feet bgs range only (see Table 8-3). Figure 8.1-38 presents the distribution of TPH in the 0-2 feet bgs interval, including those exceeding RES DEC, I/C DEC, and GB PMC criteria. Concentrations of TPH greater than CTDEP RSR criteria were detected in fill soils near the Dike, including near and beneath Building B-16, near the B-34 Tank Farm, west of Building B-80, near and beneath eastern Building B-2 and between Buildings B-13 and B-37.

SPLP Organic Concentrations Exceeding 10x the GA Groundwater Protection Criteria. Sampling and analysis was conducted where soils with mass-analysis VOC and/or SVOC concentrations exceeding the CTDEP GB PMC reported in the Draft RI Report (URSGWC, 2000). The Revised Draft RI Work Plan (Harding ESE, 2002b) stated that if the re-sampled soils were analyzed for VOCs/SVOCs by SPLP, and the data compared to CTDEP Groundwater Protection Criteria for a GA-classified aquifer (GWPC GA) x10, as allowed under the RSRs, it is likely that the number of exceedances would be significantly reduced.

Query of the SAEP GIS database indicated there were 32 soil samples from 25 soil borings with VOC and/or SVOC mass concentrations exceeding the CTDEP GB PMC. In 2002, a re-sampling of locations with organic contaminants exceeding GB PMC was conducted. Additional locations were also sampled. Samples were submitted for SPLP analysis for the purpose of comparison to 10 x the GA GWPC.

Figure 8.1-94 presents the locations of samples with SPLP organics that exceed 10x GA GWPC. Comparisons with these criteria resulted in fewer exceedances than comparison to the GB PMC (see Table 8-4). The contaminants which exceed 10x GA GWPC include benzene, TCE, cis-1,2-DCE, 1,2-DCA, vinyl chloride, dichloromethane, bis(2-ethylhexyl)phthalate. 1,2-DCA and bis(2-ethylhexyl)phthalate did not exceed the mass analysis GB PMC. Numerous PAHs, the remaining BTEX contaminants, PCE, and 1,1-DCE do not exceed the 10x GA GWPC as they do the mass analysis GB PMC. VOC and SVOC SPLP concentrations greater than 10x GA GWPC are generally limited to fill material adjacent to the western portion of the Dike and the South Parking Lot area; these are areas of releases of chlorinated and fuel-related VOCs.

8.2 GROUNDWATER

8.2.1 Distribution of Analytes in Groundwater

This subsection describes the nature and distribution of analytes detected in groundwater. The discussion has been organized by chemical, and the associated figures present the distribution in groundwater. CTDEP RES and I/C VC were compared to VOCs within 30 feet of ground surface. VOCs and BTEX distributions have been presented as plan views of interpretive contaminant concentrations at elevations -5, -25, and -75 feet MSL, respectively. These depths are approximately equivalent to 10, 30, and 80 feet bgs, and represent the distribution of contamination observed at the water table, in shallow groundwater, and in deep groundwater, respectively. Hexavalent chromium distributions have been presented as plan views of interpretive contaminant concentrations at elevations -5, -25, and -40 feet MSL (approximately 45 ft bgs), respectively.

CTDEP RSR and WQS values have been provided on analyte distribution figures for reference, as applicable (see subsection 8.2.2).

Chlorinated Volatile Organic Compounds. Twenty-one cVOCs were detected in site groundwater (see Table 8-5). The following paragraphs provide discussion of cVOCs detected in groundwater.

PCE. PCE was detected primarily in shallow (-5 and -25 feet MSL) groundwater. PCE greater than the I/C GW VC of 0.810 mg/L was detected beneath the West Parking Lot and in the area between Buildings B-48 and B-16 (Figures 8.2-7 and 8.2-8). Smaller PCE plumes exist beneath or near Buildings B-7, B-34, B-3, B-3A and B-19. These buildings were used for engine testing and research and development. PCE in deep groundwater is relatively limited, but exists beneath several of the areas of contamination in shallow groundwater, particularly beneath the West Parking Lot (see Figure 8.2-9).

TCE. TCE was detected throughout shallow groundwater (-5 and -25 feet MSL) west and north of Sniffens Lane (Figures 8.2-1 and 8.2-2). Elevated concentrations were detected beneath Building B-2, but also extend under Buildings B-10, B-12 and B-44. Separate TCE plumes exist southwest of the West Parking Lot and beneath the Dike north of the B-34 Tank Farm exceed

2.34 mg/L. There are five areas where TCE exceeds 0.1 mg/L. These areas include beneath the Building B-34 Tank Farm, beneath the Dike north of Building B-19, southeast of Building B-19, beneath the West Parking Lot, and beneath Buildings B-3, B-3A, and B-4.

In deep (-75 feet MSL) groundwater (see Figure 8.2-3), the distribution of TCE is generally limited to the area beneath Building B-2 but at elevated concentrations (greater than 2.34 mg/L). Distinctly separate areas of contamination also exist below many of the areas of elevated TCE concentrations in shallow groundwater, most notably south of the West Parking Lot, and beneath the western portion of Building B-16.

Cis-1,2-DCE. Cis-1,2-DCE was detected where TCE was detected in shallow groundwater (see Figures 8.2-4 and 8.2-5). In deep groundwater, however, cis-1,2-DCE was absent from the areas of TCE contamination south of the West Parking Lot and beneath western Building B-16 (see Figure 8.2-6). The highest concentrations of cis-1,2-DCE were detected north of Building B-74 (see Figure 8.2-4). Cis-1,2-DCE is a breakdown product of both TCE and PCE, however a comparison of the distribution of cis-1,2-DCE to both TCE and PCE distributions indicates that cis-1,2-DCE is spatially more consistent with TCE. It should be noted that the areas of elevated cis-1,2-DCE are spatially located closer to the Dike, along the groundwater flow path, than PCE and TCE.

Vinyl Chloride. Vinyl chloride typically results from the breakdown of PCE and TCE, to cis-1,2-DCE which degrades to vinyl chloride. In general, the distribution of vinyl chloride in shallow groundwater correlates with the combined distribution of PCE, TCE, and cis-1,2-DCE. The distribution of cis-1,2-DCE relative to PCE and TCE, indicates degradation of PCE and TCE and natural attenuation of cVOCs.

1,1,1-TCA. The distribution of 1,1,1-TCA in groundwater correlates well with that of 1,1-DCE. (see Figures 8.2-13 and 8.2-14). The distribution of 1,1,1-TCA in deep groundwater is generally limited to the area beneath central Building B-2, and Buildings B-12, B-15, and B-65 (see Figure 8.2-15)

1,1-DCE. 1,1-DCE was detected in shallow groundwater beneath most of the western half of the Site (see Figures 8.2-10 and 8.2-11), which correlates well with the distribution of 1,1,1-TCA. Elevated concentrations (greater than the I/C GW VC of 0.92 mg/L) beneath central Building B-2 appear to extend northeast to beneath western Building B-16 from several distinct areas of concentrations greater than 20 mg/L. Concentrations also exceed 0.92 mg/L immediately west of Building B-6 and beneath the West Parking Lot. In deep groundwater, concentrations also exceed 0.92 mg/L, in an area immediately west of the highly elevated concentrations in shallow groundwater. 1,1-DCE is typically a breakdown product of 1,1,1-TCA, and the observed distribution of 1,1-DCE in groundwater corresponds well with that for 1,1,1-TCA, as discussed above.

Other cVOCs. Fourteen other cVOCs were detected Site-wide in groundwater, generally at concentrations below 1 mg/L. Detected concentrations greater than 1 mg/L were generally limited to beneath central Building B-2. 1,1-DCA was detected in both shallow and deep groundwater at a single well from beneath central Building B-2 at concentrations greater than 10 mg/L.

Fuel-related Volatile Organic Compounds. The BTEX constituents, benzene, toluene, ethylbenzene, and xylenes, were detected in site groundwater (see Table 8-5). The concentrations of BTEX constituents presented in Figures 8.2-21 through 8.2-23 represent the summed concentrations of these individual analytes. The distribution of BTEX at -5 and -25 feet MSL are similar (see Figures 8.2-21 and 8.2-22), including an area of elevated concentrations (greater than 1 mg/L) beneath central Building B-2. An exception to this is an area of concentration up to 0.050 mg/L at -5 feet MSL near Building B-6 (see Figure 8.2-21). Several other distinct areas of BTEX contamination are noted, including north of Buildings B-13 and B-15; near Buildings B-33 and B-34 east of Building B-16; beneath the West Parking Lot; and south of the West Parking Lot. The distribution of BTEX in deep (-75 feet MSL) groundwater is limited to an area from beneath Building B-2 extending beneath Building B-12. The distribution of BTEX beneath the central portion of Building B-2 is generally coincident with that of 1,1,1-TCA.

Other Volatile Organic Compounds. Four non-chlorinated VOCs, MEK, 2-methoxy-2-methylpropane, acetone, and carbon disulfide, were detected in Site-wide groundwater (see Table 8-5). The majority of these detections are located beneath the Dike, the tidal flats, and Building B-2. With the exception of methyl ethyl ketone, these analytes were detected relatively infrequently at concentrations less than 1 mg/L (see Figures 8.2-24 and 8.2-25). Methyl ethyl ketone was detected at concentrations up to 78 mg/L beneath Building B-2 and the former B-2 Chromium Plating Facility.

Polynuclear Aromatic Hydrocarbons. Significantly fewer PAHs were detected in site-wide groundwater at lower frequencies than in soil (see Table 8-5). PAHs were detected in groundwater from beneath the Dike and near the eastern end of Building B-16. With the exception of several locations near Building B-74, concentrations were less than 0.01 mg/L.

Other Semi-Volatile Organics. Seven non-PAH-related SVOCs were detected in Site-wide groundwater (see Table 8-5). In general, these analytes were detected in groundwater along the Dike and near the former RCRA Lagoons at concentrations less than 0.01 mg/L (see Figure 8.2-27). Concentrations greater than 0.01 mg/L were detected near Building B-74 and the former lagoons. The maximum detected analyte at these locations was generally 1,2-dichlorobenzene.

Inorganics. Appendix N provides discussion of the estimation of baseline concentrations for inorganics in site groundwater. As discussed, there are several possible sources for the inorganics detected in soils, including metal plating/finishing operations, fill materials brought onto the Site or those dredged from the Housatonic River, and the natural occurrence of these analytes. The inorganics selected for evaluation in Appendix N include only those analytes with numerical RSR criteria. Deflection points were determined by review of the concentration distribution plots for each of these analytes, and represent the minimum concentration which supports potential contaminant release. Hexavalent chromium and cyanide are not naturally occurring, and historically were used in metal plating/finishing operations at the Site.

The inorganic analytes discussed in the following paragraphs have been limited to those for which CTDEP RSR criteria exist (see Table 8-5), and only those that are potentially site-related, hexavalent chromium, cyanide, cadmium, chromium (total), copper, lead, mercury, and nickel have been presented individually (see Figures 8.2-28 through 8.2-37); the others are discussed collectively (see Figure 8.2-38).

Hexavalent Chromium. Hexavalent chromium was detected in groundwater beneath the former B-2 Chromium Plating Facility, Building B-10, and Building B-12. The highest detected concentrations appear to be in groundwater at -25 feet MSL (see Figures 8.2-28 through 8.2-30) beneath the northern wall of the former B-2 Chromium Plating Facility.

Nickel. Nickel was generally detected in groundwater from locations outside the main portion of the facility. Exceptions to this include detections near Buildings B-8 and B-12 (see Figure 8.2-31). Nickel was more frequently detected in samples from depths greater than 30 feet bgs.

Cadmium. Cadmium was generally detected in groundwater beneath the former B-2 Chromium Plating Facility and immediately to the north, and from the eastern portion of the facility near and beyond the former RCRA Lagoons (see Figure 8.2-32). The maximum detected concentration of cadmium was detected in groundwater near Building B-12 at 26-30 feet bgs.

Copper. Copper was generally detected in groundwater from locations outside the main portion of the facility (see Figure 8.2-33). Exceptions to this include the maximum detected concentration detected beneath the former B-2 Chromium Plating Facility at 7 to 9 feet bgs, as well as detections near Building B-10 and B-8.

Lead. Lead was generally detected in groundwater from locations near the Dike and the South Parking Lot (see Figure 8.2-34). The maximum detected concentration, greater than the SWPC of 0.013 mg/L, was detected near the Dike, northwest of Building B-16

Cyanide. Cyanide was detected in groundwater between Buildings B-2 and B-10 and B-12, and near the Dike (see Figure 8.2-35). The frequency of detection for cyanide was significantly greater in the 0 to 30 feet bgs range than for greater than 30 feet bgs.

Mercury. Mercury was detected in groundwater at three locations, north and northwest of Building B-16, and near Building B-15 (see Figure 8.2-36).

Chromium. Total chromium was generally detected beneath Buildings B-2, B-10, and B-12 (see Figure 8.2-37), at concentrations greater than SWPC. This area coincides with detected concentrations of hexavalent chromium. Chromium was also detected in other portions of the facility, at concentrations less than 1.2 mg/L (the SWPC value).

Other Inorganics. In addition to the inorganics discussed in the paragraphs above, eight additional inorganics were detected in Site-wide groundwater (see Figure 8.2-38). These eight inorganics include beryllium, selenium, thallium, silver, vanadium, antimony, arsenic, and zinc.

8.2.2 Comparison of Groundwater to Criteria

Groundwater contaminant concentrations have been compared to Connecticut RSR criteria, which include the SWPC, RES GW VC, and I/C GW VC, and the Connecticut WQS aquatic life criteria, SWCHRON and SWACUTE (refer to subsection 1.3).

Residential and Industrial/Commercial Volatilization Criteria

The RES and I/C VC apply only to VOCs, including cVOCs and BTEX. The RSRs require compliance with the residential and I/C VC for groundwater within 30 feet of the ground surface. The following paragraphs discuss the contaminants with concentrations exceeding GW VC. Figures 8.2-39 and 8.2-40 present the distribution of VOCs greater than RES and I/C GW VC, respectively.

1,1,1-TCA exceeds both the RES and I/C VC in groundwater beneath the central portion of Building B-2. The maximum concentration exceeding GW VC is 100 mg/L, located at a depth of 7 feet bgs.

1,1,2-Trichloroethane (1,1,2-TCA) and 1,1-Dichloroethane (1,1-DCA) concentrations are greater than RES VC at CP-99-17, beneath central Building B-2 at a depth of 19 feet bgs.

1,1-DCE exceeds the RES and I/C VC beneath the West Parking Lot, Building B-2, the main portions of the facility, the North Parking Lot, and along the Dike (see Figure 8.2-50). Concentrations of 1,1-DCE exceeding GW VC extend as far as the eastern boundary of SAEP along the Dike.

1,2-Dichlorethane (1,2-DCA) exceeds RES GW VC beneath the western end of Building B-2, near Building B-1, and east of Building B-19.

Chlorobenzene marginally exceeds the RES GW VC (1.8 mg/L) at one location, LW-2, at a concentration of 2 mg/L. LW-2 is located downgradient of the former RCRA Lagoons.

Chloroform exceeds RES and I/C GW VC beneath the former B-2 Chromium Plating Facility and near Building B-48.

Cis-1,2-DCE exceeds RES and I/C GW VC. Concentrations greater than the I/C VC are located beneath the tidal flats immediately off-shore of the Hazardous Waste Storage Area and at Hot-spot No. 2. The maximum concentration, 46 mg/L, was detected at D-4 near the Hazardous Waste Storage Area. Concentrations also exceed RES GW VC beneath Building B-2 and along the Dike east of the Causeway.

PCE exceeds RES and I/C GW VC. Concentrations greater than I/C VC are located beneath the West Parking Lot, the former B-2 Chromium Plating Facility, Building B-12, and at Hot-spot No.2. Additionally, concentrations are greater than RES GW VC near the former Jet Fuel Tank Farm, north of Building B-19, and in the alcove between Buildings B-3, B-3A, and B-4. The highest detected concentration of PCE exceeding GW VC is 4.3 mg/L, at 6.5 feet bgs beneath the West Parking Lot.

TCE concentrations in groundwater exceed the I/C VC and RES VC. Concentrations greater than the I/C VC are located beneath the majority of the Site west of Sniffens Lane. In addition to these areas, TCE concentrations exceed the RES GW VC beneath Building B-3, the southeastern end of Building B-2, and the tidal flats. The maximum detected concentration of TCE exceeding GW VC is 830 mg/L, located 28 feet bgs beneath the former B-2 Chromium Plating Facility.

Vinyl chloride concentrations exceed the I/C VC and RES VC. The highest concentrations (greater than 1 mg/L) greater than GW VC are located beneath the Tidal Flats off-shore of the Hazardous Waste Storage Area, near B-64-1, near the tidal ditch, and beneath Building B-2 near the former B-2 Chromium Plating Facility. Concentrations also exceed GW VC between Buildings B-48 and B-16 and near Buildings B-4, B-6, B-16, and B-19.

The VOC 2-butanone exceeds RES and I/C GW VC at WP-99-34, beneath the former B-2 Chromium Plating Facility, at a depth of 28 feet bgs.

Surface Water Protection Criteria

SWPC apply to both organic and inorganic contaminants, and, as presented in subsection 1.3, apply to groundwater immediately upgradient of the point of discharge to surface water (e.g., the point-of-compliance). Surface water downgradient of the Site includes the Housatonic River and the Tidal Flats. The point of compliance for SWPC is the Housatonic River.

Site-wide organic and inorganic contaminant concentrations have been compared to the SWPC. VOC, PAH, and inorganic concentrations greater than SWPC values are presented in Figures 8.2-41 through 8.2-43, respectively. Concentrations of VOCs are greater than SWPC values throughout the Site, to the north and west of Sniffens Lane (see Figure 8.2-41). Concentrations of VOCs detected in groundwater above SWPC values include 1,1,1-TCA, 1,1,2-TCA, 1,1-DCE, 1,2-DCA, PCE, TCE.

The PAH phenanthrene was detected at a single location above the SWPC value between the Dike and Building B-58 (see Figure 8.2-42).

The majority of inorganic concentrations greater than SWPC values is located beneath the former B-2 Chromium Plating Facility and Buildings B-10 and B-12, and includes hexavalent chromium, chromium, cadmium, copper, and cyanide. This is an area of identified releases resulting from metals plating activities, which included the use of these analytes.

Concentrations of arsenic, lead, silver, and zinc in groundwater beneath the Site are also greater than SWPC values; however the distribution in the aquifer is sporadic and does not appear to be associated with releases.

SWPC are applicable to the groundwater immediately upgradient of the Housatonic River. The majority of shallow groundwater discharges to the Tidal Flats; deep groundwater eventually discharges to the Housatonic River, which is approximately 2,000 feet downgradient from the former B-2 Chromium Plating Facility.

A total of ten groundwater piezometers (PZ-TF-01B through PZ-TF-10B) screened beneath the Tidal Flats sediments in the glacial outwash are located downgradient of the main portions of the Site. Groundwater samples from these locations were analyzed for VOCs, including BTEX and cVOCs. Results indicate at least one BTEX or cVOC analyte was detected in each of the ten deep tidal flat piezometers; however, most detected concentrations were several orders of magnitude less than SWPC. BTEX concentrations range from non-detect to 0.003 mg/L. Concentrations of cVOCs range from non-detect to 3.8 mg/L. PCE and TCE concentrations marginally exceed SWPC at PZ-TF-10B, at 43 feet bgs beneath the Tidal Flats north of Building B-19 adjacent to the Dike.

Deep groundwater monitoring well MWCD-99-01B is located along the Causeway, screened 40-50 feet bgs within the flow path of groundwater from the central portion of the Site. Groundwater samples from this well have been analyzed for VOCs, SVOCs, cyanide, and inorganics. TCE aswere detected at a concentration of 0.003 mg/L compared to the SWPC of 2.34 mg/L. SVOCs and cyanide were non-detect. Nickel was detected at a concentration of 0.00571 mg/L (SWPC is 0.88 mg/L).

Aquatic Life Criteria, Saltwater Acute and Chronic

The Connecticut WQS Aquatic Life Criteria apply to groundwater immediately upgradient of the discharge point to surface water. Concentrations of inorganics greater than SWACUTE and SWCHRON values are presented in Figures 8.2-44 and 8.2-45, respectively.

The majority of inorganics concentrations greater than Aquatic Life Criteria are located in shallow and intermediate (less than 50 ft bgs) groundwater beneath the former B-2 Chromium Plating Facility and Buildings B-10 and B-12. This is an area of identified releases resulting from metals plating, which included the use of hexavalent chromium, cadmium, copper, cyanide, and nickel. Hexavalent chromium concentrations are greater than Aquatic Life Criteria throughout groundwater beneath the former B-2 Chromium Plating Facility, Building B-10, and Building B-12. Cadmium, copper, cyanide, and nickel were also detected at concentrations above Aquatic Life Criteria in groundwater within this area. However, Aquatic Life Criteria are applicable to the groundwater immediately upgradient of the Tidal Flats, which are located approximately 400 feet downgradient from the former Building B-2 Chromium Plating Facility.

Concentrations of cyanide are also greater than Aquatic Life Criteria values near Building B-74, where concentrations were also detected in soil, and north of Building B-16, along the Dike.

There are concentrations of copper and nickel in groundwater outside of the Building B-2, B-10, B-12 area that are greater than Aquatic Life Criteria values, but these concentrations are less than deflection point values for these analytes, with the exception of one location north of Building B-19. This indicates that these detected concentrations are not necessarily release-related, but may be naturally occurring or related to the fill material content.

Concentrations of arsenic, lead, silver, and zinc are greater than Aquatic Life Criteria values; however, the distribution in the aquifer is sporadic and does not appear to be associated with releases.

8.2.3 Natural Attenuation Parameters

Figures 8.2-69 through 8.2-71 present selected natural attenuation indicator parameters, including ORP, ferrous iron, temperature, conductivity, and salinity. Natural attenuation parameter data are presented in Appendix P-4. These figures present the natural attenuation parameters at elevations -5, -25, and -75 feet MSL, portraying the distribution in Site-wide groundwater.

ORP levels are generally in the range of -150 to 150 millivolts (mV). Areas with high ORP levels (>150 mV) are located beneath the central portion of the Site, beneath the central portion of Building B-2 and Buildings B-10 and B-12, in shallow groundwater (-5 and -25 feet MSL). This area is consistent with the extent of elevated hexavalent chromium in groundwater beneath the former B-2 Chromium Plating Facility. ORP levels greater than 150 mV are also located to the southwest and to the northeast of the Site and between Buildings B-16 and B-19. Areas with low ORP levels (<-150 mv) are located beneath the western portion of Building B-2, the eastern portion of Building B-16, west of Building B-16, and beneath the tidal flats offshore of the western end portion of the Dike in shallow groundwater (-5 and -25 feet MSL). In deep groundwater ORP levels less than -150 mV are located beneath western and central Building B-2, the former B-2 Chromium Plating Facility, west of Building B-16, and beneath the tidal flats offshore of the western portion of the Dike. Areas of negative ORP levels correlate well with the cVOC contamination in groundwater.

Ferrous iron levels are generally in the range of 0 to 2 mg/L beneath the main portion of the Site. Concentrations greater than 2 mg/L are generally limited to areas including Buildings located within approximately 100 feet of the Dike (e.g., B-16), to the north, beneath the tidal flats. Several areas of concentrations greater than 7 mg/L are located offshore of Building B-74, and also offshore of the eastern portion of the Dike in deep groundwater (-75 feet MSL).

Groundwater temperature has also been represented as the log to the base 10 of the temperature in degrees Celsius to sharpen the contrasts in the data portrayal. Temperatures are relatively consistent through much of the Site-wide groundwater, around 15+/- degrees celsius. The exception is an area of elevated temperatures located beneath Building B-2, where temperatures range from 20 to 25 degrees celsius. This area of elevated temperature is consistent with cVOC contamination in groundwater, and is likely the result of chemical and/or biological degradation of these cVOCs.

Conductivity levels vary throughout site-wide groundwater. Median levels of conductivity (between 10 and 40 micromhos per centimeter [μmho/cm]) are located beneath Building B-2, the Dike, and beyond the tidal flats. Elevated levels (greater than 40 μmho/cm) are generally located beneath the tidal flats offshore of Building B-74 and eastern Building B-16. An additional area of elevated conductivity is located in deep groundwater (-75 feet MSL) immediately southeast of Building B-2. Low conductivity levels are located in the central portion of the Site, between Buildings B-2, B-16, B-74, and B-6, and to the south of Building B-2.

Salinity concentrations across the Site increase with depth. In shallow groundwater (-5 and -25 feet MSL), concentrations are generally less than 0.5% across the Site. At -25 feet MSL, an area of elevated salinity (greater than 2%) is evident beneath the area of AOC 19 and the eastern end of Building B-6. In deep groundwater (-75 feet MSL), the majority of groundwater beneath the

Site has salinity levels around 1%, and elevated salinity levels (greater than 2%) extend beneath the southeastern portion of the facility, as far inland as the eastern portion of Building B-2.

8.2.4 Non-Aqueous Phase Liquids

LNAPL and DNAPL have been detected at SAEP. LNAPL has been measured at three locations including the Building B-37 and Building B-38 pump houses, the Jet Fuel Tank Farm near Building B-34, and the Former Equalization Lagoon. DNAPL has been detected in one location beneath Building B-2.

8.2.4.1 Light Non-Aqueous Phase Liquids

A trace amount of LNAPL was measured in May 2002 in WC-5S, located on the northeast side of Building B-37. LNAPL is also present in the sump of the Building B-38 pump house. The product is likely jet fuel but may also contain gasoline, kerosene or mineral spirits. The source of LNAPL may be the underground waste oil transfer system between Building B-13 and B-15 and the Oil House Tank Farm. LNAPL is also present adjacent to the Dike between the Oil House Tank Farm and the OATP; residual phase product was noted in soil borings SB06A3-1, EBS01-1, and EBS01-2, located near the Oil House Tank Farm northwest of WC-5S.

LNAPL was detected in May 2002 in MW-3, located along the Dike, just northwest of Building B-34 and in the sump of the nearby pumphouse Building B-40. The estimated thickness was 0.1-foot. No other wells in this area contained product but residual phase product was detected in soil boring SB08L1-2, located 50 feet southwest of Building B-34. The source of fuel in this area is the former Building B-34 fuel tanks.

LNAPL was measured during the November 2002 RCRA groundwater monitoring in well LW-5S, located immediately downgradient of the Former Equalization Lagoon. The thickness was estimated to be 0.5 feet.

In March 2004, LNAPL monitoring wells were installed in the three areas where LNAPL had been observed (see Table 8-7). Soil samples from the monitoring well borings were evaluated for the presence on LNAPL (see Section 3.10.1). LNAPL was observed in soil from borings LNAP-04-12, LNAP-04-15, LNAP-04-16, LNAP-04-21, LNAP-04-23. Table 8-7 lists the soil samples, the resulting PID readings from each interval sampled, and the results of the agitation test. As shown, free-phase oil was observed to occur using this technique in six soil samples collected from five different borings. Also noted is that free-phase conditions do not necessarily correspond with soils that had relatively elevated PID readings. The free-phase conditions were present in soils with both high and low PID readings, and all soils with relatively elevated PID results did not necessarily exhibit a free-phase condition.

In May 2004, LNAPL measurements were taken from the twenty new monitoring wells and from the three monitoring wells were LNAPL was previously measured (WC-5S, MW-3, and LW-5S). The results of the May 2004 LNAPL measurements are presented in Table 8-8. LNAPL was present in monitoring wells WC-5S and LW-5S (MW-3 could not be measured). In addition, LNAPL was measured in monitoring well LNAP-04-24, located between Building B-13 and B-15.

8.2.4.2 Dense Non-Aqueous Phase Liquids

DNAPL, measuring 0.2-foot in thickness, was measured in monitoring well HESE-01-12D screened near the bedrock surface (approximately –150 feet MSL) in April 2002. Based on the VOC analysis the product consists of 1,1,1-TCA and fuel hydrocarbons (alkanes, substituted benzenes and naphthalene). The reported fuel hydrocarbons suggest that the product contains light fuel such as gasoline, kerosene, or jet fuel. The product also contains TCE, 1,1-DCE, and toluene. Based on the data, an estimate of the composition of the product is fuel hydrocarbons (60%), 1,1,1-TCA (35%), TCE (4%), and 1,1-DCE and toluene (<1%). The solubility of 1,1,1-TCA in water is approximately 1,500 mg/L and the highest concentration of 1,1,1-TCA detected in groundwater beneath Building B-2 is 680 mg/L, which is approximately 45% of 1,1,1-TCA's solubility in water. DNAPL measurements were conducted in all monitoring wells beneath Building B-2 screened on top of bedrock on December 24, 2002; DNAPL was not detected in any of the monitoring wells, including HESE-01-12D.

On May 17, 2004 DNAPL monitoring was conducted at the following monitoring wells: HESE-01-05D, HESE-01-06D, HESE-01-07D, HESE-01-08D, HESE-01-09D, HESE-01-10D, HESE-01-12D, HESE-01-17D, and HESE-01-18D. DNAPL was detected in monitoring well HESE-01-12D only, at an apparent thickness of 1.2 feet (see Table 8-9).

8.3 SOIL VAPOR AND INDOOR AIR

8.3.1 Soil Vapor Contaminant Distribution and Comparison to CTDEP RSR Criteria

Soil vapor samples were collected in 1999 and 2004. The 1999 soil vapor sampling event was conducted for field screening purposes. During the 2004 sampling event, permanent, vapor monitoring points were installed and sampled for compliance purposes. The 2004 soil vapor points were installed in the northern and southern parts of Building B-2, B-65, B-3, B-3A, B-4, B-16 and B-19 to determine if these areas were in compliance with RSRs. The 1999 and 2004 soil vapor data are presented separately in the following subsections. Complete analytical results for soil vapor samples are included in Appendix U.

8.3.1.1 1999 Soil Vapor Results

Figure 4-1 presents the locations of the soil vapor samples collected in 1999. A summary of detected soil vapor analytes and concentrations exceeding criteria is presented in Table 8-10. The distribution of 1,1-DCE, PCE, TCE, and vinyl chloride in soil vapor is presented in Figures 8.3-1 through 8.3-4, respectively. Figures 8.3-5 and 8.3-6 present the soil vapor analytical results exceeding RES and I/C VC, respectively.

1,1-DCE was detected in 17 out of 54 soil vapor samples, at concentrations ranging from 0.041 to 5.06 parts per million by volume (ppmv) (see Figure 8.3-1). The highest 1,1-DCE concentrations were found in the central portion of the facility, beneath Buildings B-2, B-10, and B-12. 1,1-DCE concentrations exceeded the RES VC in six of the 54 samples, and did not exceed the I/C VC in any samples.

PCE was detected in 30 out of 54 soil vapor samples, at concentrations ranging from 0.026 to 3.53 ppmv (see Figure 8.3-2). The highest PCE concentrations were found in the central portion of the facility, in an area extending from Building B-13 to B-3A. PCE concentrations exceeded the RES VC in 7 of the 54 samples, and exceeded the I/C VC in 5 of the 54 samples.

TCE was detected in 45 out of 54 soil vapor samples, at concentrations ranging from 0.019 to 210 ppmv (see Figure 8.3-3). The highest TCE concentrations were primarily found in the central portion of the facility, beneath Buildings B-2, B-10, and B-12. TCE concentrations exceeded the RES VC in 34 of the 54 samples, and exceeded the I/C VC in 28 of the 54 samples. TCE concentrations exceeding the I/C VC by greater than a factor of 10 were limited to 12 soil vapor samples, out of a total of 54 samples analyzed for TCE.

Vinyl chloride was detected in 21 out of 54 soil vapor samples, at concentrations ranging from 0.041 to 5.06 ppmv (see Figure 8.3-4). The highest vinyl chloride concentrations were found near Buildings B-7A and B-64-2. Vinyl chloride concentrations exceeded the RES VC in only three of the 54 samples, and the I/C VC in only two samples.

Results of the 1999 soil vapor survey identified TCE above RSR criteria in the central part of the site including Building B-2 (center), Building B-10 and Building B-12. Levels outside of this central area were either at or below volatilization criteria.

8.3.1.2 2004 Soil Vapor Results

In 2004 permanent soil vapor points were installed in the northern and southern parts of Building B-2, B-65, B-3, B-3A, B-4, B-16 and B19 to determine if these areas were in compliance. Figure 4-1 presents the locations of the soil vapor samples collected in 2004. A summary of detected soil vapor analytes and concentrations exceeding criteria is presented in Table 8-11. The distribution of 1,1-DCE, PCE, TCE, and vinyl chloride in soil vapor is presented in Figures 8.3-7 through 8.3-10, respectively. Figures 8.3-11 and 8.3-12 present the soil vapor analytical results exceeding RES and I/C VC, respectively. Complete analytical results for soil vapor samples are included in Appendix U.

1,1,1-TCA was detected in 56 of 68 soil vapor samples, at concentrations ranging from 0.0051 to 6.1 ppmv. The maximum detected concentration was observed at SVM-04-21, located in the southeastern end of Building B-2. 1,1,1-TCA concentrations did not exceed RES VC or I/C VC in any samples.

1,1-DCE was detected in 39 out of 68 soil vapor samples, at concentrations ranging from 0.0053 to 1.6 ppmv (see Figure 8.3-7). The highest 1,1-DCE concentrations were detected at SVM-04-10, located in the northwestern end of Building B-2. 1,1-DCE concentrations did not exceed RES VC or I/C VC in any samples.

Cis-1,2-DCE was detected in 35 out of 68 soil vapor samples, at concentrations ranging from 0.0053 to 18 ppmv. Cis-1,2-DCE concentrations exceeded the RES VC in 2 of the 68 samples, and did not exceed the I/C VC in any samples. The maximum detected concentration was observed at SVM-04-23, located in the southeastern end of Building B-2.

PCE was detected in 61 out of 68 soil vapor samples, at concentrations ranging from 0.006 to 1.9 ppmv (see Figure 8.3-8). The highest PCE concentration was observed at SVM-04-29, located in the southern-most corner of Building B-3. PCE concentrations exceeded the RES VC (0.56 ppmv) in 7 of the 68 samples, and exceeded the I/C VC in 3 of the 68 samples.

Trans-1,2-DCE was detected in 9 out of 68 soil vapor samples, at concentrations ranging from 0.0058 to 0.47 ppmv. Trans-1,2-DCE concentrations did not exceed RES VC or I/C VC in any samples. The maximum detected concentration was observed at SVM-04-23, located in the southeastern end of Building B-2, and also the location of the maximum observed concentration of cis-1,2-DCE.

TCE was detected in 66 out of 68 soil vapor samples, at concentrations ranging from 0.007 to 130 ppmv (see Figure 8.3-9). The highest TCE concentrations were found in the northwest and southeast portions of Building B-2, in the central portion of the Building B-3, B-3A, and B-4 complex, and in one location in Building B-6. TCE concentrations exceeded the RES VC in 37 of the 68 samples, and exceeded the I/C VC in 27 of the 68 samples.

Vinyl chloride was detected in 7 out of 68 soil vapor samples, at concentrations ranging from 0.0063 to 0.71 ppmv (see Figure 8.3-10). The highest vinyl chloride concentration was found in Building B-65, at location SVM-04-18. Vinyl chloride concentrations exceeded the RES VC in only 2 of the 68 samples, and did not exceed the I/C VC in any samples.

Results of the 2004 soil vapor survey identified that TCE concentrations were above soil vapor RES and I/C VC throughout Buildings B-2, B-3 and B-3A, and that PCE was above VC in Buildings B-3, B-3A, and B-4. Soil vapor concentrations exceed RES VC and I/C VC at one or more locations in Buildings B-2, B-3, B-3A, B-4, and B-6.

8.3.2 Indoor Air Contaminant Distribution and Comparison to CTDEP RSR Criteria

The objective of the indoor air monitoring was to evaluate concentrations of VOCs inside SAEP buildings relative to the CTDEP Industrial/Commercial Indoor Air Target Concentrations (I/C IATCs). Rounds 1 and 2 were conducted in September 1999. Rounds 3 through 35 were conducted between October 1999 and September 2003. Indoor air samples were collected using 6-liter SUMMA® vacuum canisters, and were analyzed by USEPA Method TO-15 for the VOCs: 1,1,1-TCA, 1,1-DCE, PCE, TCE, and vinyl chloride (see Subsection 4.3.2). In some cases, samples were also analyzed for cis-1,2-DCE and trans-1,2-DCE.

Table 4-1 presents a summary of the locations monitored during each of the 35 rounds. Indoor air monitoring locations are presented on Figure 4-2. Figures 8.3-13 and 8.3-14 present the indoor air analytical results exceeding RES and I/C VC, respectively. Complete indoor air analytical results are presented in Appendix V. A summary of detected analytes from Rounds 1 through 35 is presented in Table 8-12.

The following general observations have been made regarding the analytical data from Rounds 1 through 35 of air monitoring:

- The VOCs 1,1,1-TCA, 1,1-DCE, PCE, TCE, cis-1,2-DCE, and vinyl chloride have all been detected in indoor air samples at SAEP.
- The highest concentrations of VOCs detected in Rounds 1 through 35 were generally found in the central portions of Building B-2 during monitoring Rounds 1 and 2, which is coincident with the groundwater VOC plume beneath this portion of the building.
- Querying the SAEP database for concentrations exceeding the CTDEP RES IATC results in 186 data points from monitoring Rounds 1 through 35:
    - 144 for TCE (RES IATC = 0.18 parts per billion by volume [ppbv]),
    - 22 for vinyl chloride (RES IATC = 0.055 ppbv), and
    - 19 for PCE (RES IATC = 0.73 ppbv).
- Querying the SAEP database for concentrations exceeding the CTDEP I/C IATC results in 164 data points from monitoring Rounds 1 through 35:
    - 144 for TCE (I/C IATC = 0.18 ppbv), and
    - 19 for PCE (I/C IATC = 0.73 ppbv).
- The maximum detected concentrations of the VOCs were as follows:
    - 1,1,1-TCA: 5.0 ppbv (IA-B6-02)
    - 1,1-DCE: 1.9 ppbv (IA-B12-01)
    - Cis-1,2-DCE: 9.4 ppbv (IA-B6-02)
    - PCE: 20 ppbv (IA-ML-01)
    - TCE: 11 ppbv (IA-99-04)
    - Vinyl chloride: 0.99 ppbv (IA-99-01)
- The highest frequencies of concentrations exceeding the CTDEP I/C IATC were found at the following locations (see Figure 4-2):
    - Building B-2: IA-B2-01 (Boiler Room)
    - Building B-6: IA-B6-01 (former engine test facility)
    - Building B-9: IA-B9-01 (automotive maintenance, shop area)
    - Building B-10: IA-B10-01 (unoccupied, open space)
    - Building B-12: IA-B12-01 (Shop Area), IA-B12-02 (Office Area)

  - Building B-65: IA-B65-01
- Monitoring locations, by building, having infrequent VOC concentrations exceeding CTDEP I/C IATC include:
  - Building B-1: IA-B1-02 (2nd floor Office Area), IA-B1-03 (3rd floor Office Area)
  - Building B-2: IA-ML-01 (2nd floor, office area), IA-ML-02 (1st floor, warehouse area)
  - There are additional locations that have only a few VOC concentrations exceeding CTDEP I/C IATC, however those locations were sampled far less frequently.
- Barometric pressure trends (presented in individual monitoring round technical memoranda) do not appear to correlate with changes in contaminant concentrations at monitoring locations from round to round.

8.4 SEDIMENT RESULTS

In this section, the results of sediment chemical analyses, toxicity testing, bioaccumulation testing and benthic macroinvertebrate analyses conducted in the Phase II and Phase III RIs (see Subsection 1.5) are presented. Figure 5-1 depicts the 1994 (Phase II) and 1999 (Phase III) site sediment sampling locations. Data are intended to provide a general description of site sediment quality. A more comprehensive screening of these data against threshold values and background data are provided in Section 12.0 Baseline Ecological Risk Assessment. More detailed data of detected compounds in sediment is provided in Appendix Q.

8.4.1 Sediment Chemical Characterization

The discussions of sediment chemical characterization presented in the following subsections incorporates data from the following investigations: Phase I RI (1992), Phase II RI (1994), and Phase III RI (1999).

8.4.1.1 Reference Locations. Locations of reference (background) samples SDUS001A, SDUS002A, SDUS003A, U1, U2, and IBG are indicated in Figure 5-1.

VOCs. VOCs were not detected in sediments from the reference location samples.

SVOCs. Sixteen SVOCs, all PAHs, were detected at least once in surficial sediments collected from reference locations. Concentration ranges of the PAHs are presented in Table 8-13.

PCBs. Aroclor-1248 and Aroclor-1260 were the only PCBs detected in reference location samples. Aroclor-1248 was detected in one out of seven reference location samples at a concentration of 0.29 mg/kg (see Table 8-13). Aroclor-1260 was detected in three of seven samples at concentrations ranging from 0.03 to 0.084 mg/kg.

Inorganics. Of the 23 TAL metals, only silver and thallium were not detected in reference location samples (see Table 8-13). CN and hydrogen cyanide were also not detected. Trivalent arsenic and methyl mercury were detected in three out of three samples, and mercury was detected in four out of seven samples. Mercury concentrations ranged from non-detect to 0.22 mg/kg. Methyl mercury concentrations ranged from 0.000071 mg/kg to 0.00164 mg/kg.

8.4.1.2 Tidal Flats. VOCs. Eleven VOCs were detected in sediments in the Tidal Flats, including 1,2-DCE, acetone, carbon disulfide, 2-hexanone, TCE, toluene, and ethyl benzene (see Table 8-14). No compound was detected in more than three samples and concentrations were relatively low (less than 0.2 mg/kg).

Of the detected VOCs, two detections of chlorinated solvents are notable. TCE was detected in sample SDTD006A, and 1,2-DCE was detected in sample SDTD007A. Both of these sampling locations are located on the west side of the Causeway (see Figure 5-1), and are near areas of shallow groundwater contamination by the same compounds (see Subsection 8.2). Contamination of sediments by VOCs in this area may be a result of discharging shallow groundwater.

SVOCs. Twenty-seven SVOCs were detected at least once in surficial sediments collected from the Tidal Flats. All SVOCs detected were PAHs (see Table 8-14).

PCBs. PCB detections were reported in the form of three Aroclor mixtures: Aroclor-1248, Aroclor-1254, and Aroclor-1260. These Aroclors were detected in approximately 50% of samples analyzed for PCBs (see Table 8-14). The PCB maximum concentration was detected for Aroclor-1248 in sample TC1-0-6 at a concentration of 130 mg/kg, which exceeds the reference location maximum detected concentration of 0.29 mg/kg (see Subsection 8.4.1.1). Aroclor-1254 was not detected in the reference location samples. The maximum detected concentration of Aroclor-1260 (0.9 mg/kg) also exceeds the maximum reference location concentration of 0.084 mg/kg (see Subsection 8.4.1.1). The distribution of PCBs in shallow (0-2 feet bgs) Tidal Flats sediments is presented in Figure 8.4-1; the distribution of PCBs in deeper (2-4 feet bgs) sediments is presented in Figure 8.4-2. In shallow Tidal Flats sediment, the highest detected PCB concentrations (greater than 10 mg/kg) were in sediments located off-shore of Building B-38 and the OATP. In deeper sediments, PCBs were only detected at two locations at relatively low concentrations, up to 0.5 mg/kg.

Inorganics. All twenty-three TAL metals, trivalent arsenic and methyl mercury were detected in surface sediment samples. For the majority of analytes, the detection frequency was 100%. Cyanide was detected in one sample, SDTD006B at a concentration of 0.79 mg/kg.

The distribution of chromium in shallow (0-2 feet bgs) Tidal Flats sediments is presented in Figure 8.4-3; the distribution of chromium in deeper (2-4 feet bgs) sediments is presented in Figure 8.4-4. In shallow tidal flats sediments, concentrations of chromium were greater than background at nearly all samples throughout the tidal flats sediments off-shore of the Site. In deeper sediments, concentrations above background were limited to two locations east of the Causeway and a single location near Building B-36.

Concentrations of methyl mercury, the most bioaccumulative form of mercury, ranged from 0.03-0.47% of the total mercury. The distribution of mercury in shallow (0-2 feet bgs) Tidal Flats sediments is presented in Figure 8.4-5; the distribution of mercury in deeper (2-4 feet bgs) sediments is presented in Figure 8.4-6. In both shallow and deep sediment, concentrations greater than 1 mg/kg are generally located west of the Causeway. Five sediment sample locations, extending linearly from between Buildings B-37 and B-38, had the highest (greater than 5 mg/kg) detected concentrations for both depth intervals.

8.4.1.3 Marine Basin. VOCs. Four VOCs, including acetone, methyl ethyl ketone (MEK), methylene chloride, and carbon disulfide were detected in Marine Basin sediments (see Table 8-15). Acetone was detected in five of six samples, and is likely representative of laboratory contamination. Carbon disulfide, methylene chloride, and MEK were detected at relatively low concentrations (less than 0.25 mg/kg).

SVOCs. Of the 17 SVOCs detected in Marine Basin sediments, the majority are PAHs. The maximum concentrations of several PAHs were detected in the zero to 6-inch sample MB-0-6 (see Table 8-15).

PCBs. Three PCB Aroclor mixtures were detected in Marine Basin sediments: Aroclor-1248, Aroclor-1254, and Aroclor-1260 (see Table 8-15). Aroclor-1260 was detected in all samples and was present at the highest concentration (0.18 mg/kg). Detected PCB concentrations are within the range detected in reference locations; however, Aroclor-1254 was not detected in reference locations samples (see Subsection 8.4.1.1).

Inorganics. All twenty-three TAL metals were detected in sediment samples collected from the Marine Basin (see Table 8-15). Trivalent arsenic and methyl mercury were also detected, while hexavalent chromium, silver, and CN were undetected.

Marine Basin sediment samples had relatively low concentrations of selenium and thallium. Thallium was detected at only one station at 0.14 mg/kg. The maximum selenium concentration was only 1.9 mg/kg. Selenium concentrations measured at the reference locations ranged from 0.52-0.76 mg/kg.

Sediment concentrations of cadmium, chromium, copper, lead, nickel, and zinc in the Marine Basin were generally higher than average reference concentrations, while mercury concentrations were lower than the reference concentrations. Methyl mercury samples ranged from 0.5-0.7 % of the total mercury detected.

8.4.1.4 Outfall 008 Drainage. VOCs. Five VOCs were detected in sediments in the Outfall 008 Drainage including total 1,2-DCE, vinyl chloride, acetone, MEK, and chlorobenzene (see Table 8-16). Of these analytes only acetone was detected in all samples at relatively high concentrations. Acetone is known to be a common laboratory contaminant.

SVOCs. Twenty-three SVOCs were detected at least once in surficial sediments collected from the Outfall 008 Drainage (see Table 8-16). All detected PAHs exceeded reference location concentrations except dibenz(a,h)anthracene.

PAHs detected generally had concentrations that were over 10 times higher at sample location SD08003A than at other stations. As mentioned above, the Outfall 008 Drainage is shared with a drainage ditch coming from the highway and Igor Sikorsky Memorial Airport. Drainage from the airport and roadway areas connects with the site outfall drainage and then continues toward the Marine Basin. Station SD08003 is located downstream of the airport/highway input. While these compounds may be site-related, the concentrations of PAHs in sediments do not reflect a trend of discharge from the Outfall 008 Drainage (i.e., higher concentrations at the point of discharge and then decreasing with downstream distance).

PCBs. Aroclor-1260 was detected in five of six samples from the Outfall 008 Drainage. The maximum concentration of 1.5 mg/kg, which exceeds the maximum reference location of 0.084 mg/kg, was detected at sampling station SD08002A. Aroclor-1254 was detected in two out of six samples at concentrations up to 1.1 mg/kg. Aroclor-1254 was not detected in reference location samples.

Inorganics. All 23 of the TAL metals, CN, trivalent arsenic and methyl mercury were detected in Outfall 008 Drainage sediment samples (see Table 8-16).

Sediment mercury levels in the Outfall 008 Drainage were greater than the reference location concentrations (see Subsection 8.6.1.4). Methyl mercury samples ranged from 0.3-1.0% of the total mercury. Sediment methyl mercury levels in Outfall 008 Drainage correspond well with Marine Basin sediment concentrations, indicating that the source of mercury in these areas is probably not related to the outfall.

Arsenic concentrations in Outfall 008 Drainage sediments were similar to reference location concentrations; trivalent arsenic concentrations were low, around 6% of total concentrations (maximum value 0.707 mg/kg).

Concentrations of cadmium, chromium, copper, lead, nickel, silver, and zinc were all elevated, with maximum values from 2 to more than 40 times higher than maximum reference location concentrations (see Subsection 8.4.1.1).

8.4.2 Sediment Toxicity Testing

8.4.2.1 Neanthes arenaceodentata. Measurement endpoints for the *N. arenaceodentata* testing were percent survival and mean dry weight. Results of the *N. arenaceodentata* toxicity tests performed in the 1999 Phase III RI are summarized in Table 8-17. Percent survival in the Laboratory Control was 92. At 96, 92 and 88 %, survival in reference samples (SDUS001A, 2A, and 3A) was not significantly different than the Laboratory Control.

Among the 20 samples analyzed from the Tidal Flats, Marine Basin and Outfall 008 Drainage, only two were significantly different than the Laboratory Control. Transect A-Station 07 and Transect C-Station 06 were both significantly different, with survival percentage for both stations of only 60. Both stations are located relatively close to the outfalls along the berm. Several other stations in the western portion of the mudflats, while not significantly different, also had generally low survival.

Mean dry weight in the Laboratory Control was 1.92±1.20 mg. As with the endpoint of survival, mean dry weight in the Laboratory Control and reference location samples were not statistically different.

8.4.2.2 Leptocheirus plumulosus. Measurement endpoints for the *L. plumulosus* testing were percent survival, mean dry weight of surviving individuals, mean fecundity, and mean offspring production. Results of the *L. plumulosus* toxicity tests performed in the 1999 Phase III RI (URSGWC, 2000) are summarized in Table 8-18. Percent survival in the Laboratory Control was 88. Survival in two of the reference samples was significantly different than the Laboratory Control; data from these stations are not considered suitable as a basis for comparison with site data. Only reference station sample SDUS002A, with a survival percent of 90, will be used.

L. plumulosus survival in all ten samples analyzed from the Tidal Flats was significantly different from the Control. Of the samples with surviving test organisms, mean dry weight for all Tidal Mudflat samples was significantly different than the Laboratory Control.

In Marine Basin, only Station MB-02 (sample SDMB002A) had amphipod survival significantly lower than the Laboratory Control. Fecundity at this station was also significantly lower.

In the Outfall 008 Drainage, only Station 08-02 (sample SD08002A) had amphipod survival significantly lower than the Laboratory Control. Fecundity at this station was also significantly lower.

As with the *N. arenaceodentata* toxicity testing, sediments from the Tidal Flats demonstrated the greatest *L. plumulosus* toxicity of all stations tested.

8.4.3 Sediment Bioaccumulation Testing

Table 8-19 summarizes the results of the bioaccumulation testing for *Nereis virens*. Results are discussed in the following paragraphs and Appendix W contains the biota analytical data.

8.4.3.1 Tidal Flats. Of ten sediment samples tested for bioaccumulation potential, fifteen metals were detected in *N. virens* tissues exposed to sediments from the Tidal Flats. *N. virens* tissue metals concentrations were generally similar throughout the Tidal Flats with the following exceptions. Aluminum was detected in *N. virens* tissues exposed to sediments in two replicates from two locations. Mercury was detected in tissues exposed to sediments from only one replicate with a reporting limit ranging from 0.52-0.69 mg/kg. Chromium was also detected in *N. virens* tissues exposed to sediments from only one location, Station D-06 (sample SDTD006). Silver was detected in *N. virens* tissues exposed to sediments from half of the stations sampled. In general, with the exception of nickel and chromium, *N. virens* tissue metal concentrations were higher in organisms exposed to Tidal Mudflat sediments than all other areas in the study area. *N. virens* tissue concentrations were equivalent to or lower than sediment concentrations for all environmental chemicals of concern (ECOC).

Aroclors -1248 and -1254 were detected in all *N. virens* tissue samples exposed to sediments from the Tidal Flats. These Aroclors were not detected in *N. virens* tissues exposed to sediments collected from reference or any other site area. Total PCB concentrations ranged from undetected at Station A-07 (sample SDTA007), to as high as 0.300 mg/kg at Station D-06 (sample SDTD006).

8.4.3.2 Marine Basin. Twelve metals were detected in *N. virens* tissues exposed to sediments collected from four different Marine Basin stations. In general, with the exception of nickel and chromium, metal concentrations were lower in tissues exposed to Marine Basin (Sample DSMB003) sediments than any other area, including the reference area.

Three metals, silver, aluminum, and mercury were undetected (reporting ranges for mercury were the same as the Tidal Flats). Nickel and chromium were detected in half of the samples; tissue concentrations of nickel were highest in *N. virens* exposed to sediments from Marine Basin than any other area. Marine Basin Station 003 (sample SDMB003) recorded the highest metal concentrations in *N. virens* tissues for most metals. *N. virens* tissue concentrations were equivalent to or lower than sediment concentrations for all ECOC.

No Aroclors were detected in *N. virens* tissues exposed to Marine Basin sediments.

8.4.3.3 Outfall 008 Drainage. Thirteen metals were detected in *N. virens* tissues exposed to Outfall 008 Drainage sediments. Two metals, aluminum and mercury, were undetected (reporting ranges for mercury were the same as the Tidal Flats). Nickel and chromium were detected at the lowest concentrations, while lead was detected at the highest concentration compared to all areas tested. While sediment metal concentrations were the highest, tissue samples analyzed from the Outfall 008 Drainage were not, suggesting metal bioavailability differs among the four areas investigated.

Aroclor-1260 was the only PCB detected in *N. virens* tissues exposed to sediments from all sample locations in Outfall 008 Drainage. This is the only area where this Aroclor was detected. *N. virens* tissue concentrations ranged from 0.016 – 0.033 mg/kg. As discussed above, no PCBs were detected in any reference tissue samples.

8.4.4 Benthic Macroinvertebrate Analysis

Information on indigenous benthic fauna in the vicinity of the SAEP Site is based on extensive data collected in December 1994 as part of the Phase II RI effort and a field reconnaissance performed in May 1995 (W-C, 1996). Additional collections were taken in May 1999 as part of the Phase III RI from the Tidal Flats, Marine Basin, Outfall 008 Drainage, and upstream reference area to further supplement the data and provide site comparisons. Results of the 1999 Phase III RI survey are summarized in Table 8-20 and discussed below.

A total of 46 different macroinvertebrate taxa, from five different groups, and common to estuarine systems, were collected in the Site vicinity. Owing to the nature of the substrate, deposit feeding forms were the most abundant at all stations in all areas.

Oligochaetes (aquatic earthworms) and polychaetes (bristle worms) were present at all stations and dominated every sample. Oligochaete relative abundance ranged from 1 % (Station US-2) to 95 % (Station 08-1), while abundance of polychaetes ranged from 3 % (Station 08-2) to 85 % (Station US-2) of the total number of individuals at each station.

At most stations, *Streblospio benedicti* was the dominant polychaete. Its presence alone accounted for 43 % of all individuals at Station US-3. Amphipods (scuds) were recovered from all stations, with the exception of Station 08-1. The most common amphipod species were *Leptocheirus plumulosus* and *Monoculodes edwardsi*. Molluscs were taken from ten of the twelve stations. *Hydrobia minuta* (a gastropod) and *Gemma gemma* (a bivalve) were the predominant species in most samples. In addition to the infaunal invertebrates described above, *Palaemonetes sp.* (shore shrimp), *Carcinus maenas* (green crab) and *Nassarius obsoletus* (mud dog whelk) were collected in high numbers in the Marine Basin and Tidal Flats.

During the field investigation, benthic macroinvertebrates were observed inhabiting the rocky intertidal zone around the Tidal Flats which has formed as a result of rip-rap placement. The following species were recorded in the rocky intertidal zone: *Mytilus edulis* (blue mussel), *Modiolus demissus* (ribbed mussel), and green crab. In addition, the river channel downstream of the Site supports beds of immature *Crassostrea virginica* (eastern oyster), the primary species of shellfish harvested in Long Island Sound.

8.5 SURFACE WATER

Surface water sampling was conducted as part of the 1992 Phase I RI and 1998-1999 Phase III RI field investigations. Surface water sampling locations are depicted in Figure 5-2. Samples were collected in the following areas: Tidal Flats, Marine Basin, Outfall 008 Drainage, and at reference locations across the Housatonic River from SAEP.

Surface water samples from the 1992 Phase I RI were collected at an unspecified time of the tidal cycle; however, samples from the 1998-1999 Phase III RI were collected at high and low tide. In addition, samples from the 1998-1999 Phase III RI were analyzed for both total and dissolved metals.

For the purposes of the contamination assessment, all surface water data from the Phase I and Phase III RIs were compiled and summarized by area to assess potential contamination impacts. The BERA (Section 12.0) presents summaries of surface water data by area, tidal cycle, and total/dissolved groupings.

Table 8-17 presents a summary, by area, of the surface water data assessed in this section. During preparation of Table 8-17, it was noted that the electronic data set provided by URSGWC contained some null values for non-detected analytes. These null values were eliminated from the data set used to create Table 8-17.

VOCs. Several VOCs were detected in surface water samples. Nine VOCs were detected in the 1992 Phase I RI surface water samples collected from the Tidal Flats area, including 1,1,1,-TCA, 1,1-DCA, 1,1-DCE, 1,2-DCE (total), carbon disulfide, chloroform, toluene, and TCE. Concentrations of these VOCs ranged from 0.001 mg/L to 0.048 mg/L. Total 1,2-DCE was detected at a concentration of 0.048 mg/L at Outfall 007 (see Figure 5-2), which compares reasonably well with concentrations of cis-1,2-DCE in shallow Tidal Flats groundwater in this area (see Subsection 8.2). Ten VOCs were detected in high and low tide water samples in the Outfall 008 Drainage. The highest concentrations were recorded during high tide at sampling station 008-01. Of these ten VOCs, only total 1,2-DCE and acetone were detected in the Marine Basin. Methylene chloride and acetone, both common laboratory contaminants, were detected sporadically. 2-butanone and TCE were also detected during high tide in one sample in the Marine Basin and in two samples in the Outfall 008 Drainage, respectively.

VOCs were not detected in reference location surface water samples (see Table 8-17).

PAHs. PAHs were below detection limits in most samples with the exception of the following:

- Tidal Flats: Five PAHs were detected at Outfall 007, and naphthalene was detected at sampling location LT1-SW. The concentration of all PAHs was 0.001 mg/L.
- Outfall 008 Drainage: 4-Nitrophenol and fluoranthene were detected at relatively low concentrations (<0.002 mg/L).
- Reference Locations: Four PAHs were detected in sample SWUS001L, at concentrations up to 0.00017 mg/L.

PCBs. PCBs were below detection in all surface water samples.

Metals and Cyanide. Twenty-one metals were detected in filtered and unfiltered surface water samples. CN was below detection in all samples; however, hydrogen CN was detected in Tidal Flats at Outfall 007 (sample 07-SW).

Mercury was detected in the following areas:

- Tidal Flats: Mercury was detected in six samples, at concentrations up to 0.0006 mg/L at Outfall 003 (sample 03-SW).
- Outfall 008 Drainage: Mercury was detected in two samples, at concentrations up to 0.0002 mg/L at Outfall 008 (sample 08-SW).

Methyl mercury was detected in the following areas:

- Marine Basin: Methyl mercury was detected in one sample (SWMB003H) at a concentration of 9.3E-08 mg/L. Mercury was not detected in Marine Basin surface water samples.
- Outfall 008 Drainage: Methyl mercury was detected in five samples, at concentrations up to 1.63E-07 mg/L in sample SW08001H.
- Reference Locations: Methyl mercury was detected in one sample (SWUS001L) at a concentration of 1.18E-07 mg/L.

Chromium was detected in surface water samples from the Tidal Flats, Marine Basin, and the Outfall 008 Drainage, but was not detected from samples collected from Reference Locations. The maximum chromium concentration was detected in the Tidal Flats area at a concentration of 0.11 mg/L at Outfall 003 (sample 03-SW).

Of the remaining metals, aluminum, arsenic, barium, calcium, copper, iron, lead, magnesium, manganese, nickel, potassium, sodium, vanadium, and zinc were detected in samples from the Tidal Flats, Marine Basin, and Outfall 008 Drainage areas. Thallium was not detected in any of the surface water samples. Silver was detected only in Marine Basin, and trivalent arsenic was detected in the Marine Basin and Outfall 008 Drainage areas. Of the metals detected in the Tidal

Flats, Marine Basin, and Outfall 008 Drainage areas, only barium, nickel and silver were not detected in reference location samples.

8.6 BIOTA RESULTS

This section discusses the results of the biota investigation for the shellfish and finfish in the Tidal Flats and Marine Basin. The results of PCB, TAL metal, percent lipids and percent moisture analyses of the tissues are summarized below. The results of the fish tissue analyses are presented in further detail in Section 12.0 and summarized in Table 8-18.

8.6.1 Shellfish Tissue

8.6.1.1 Tidal Flats. Seventeen metals were detected in ribbed mussel (shellfish) tissue samples collected in the Tidal Flats. Vanadium, silver, and arsenic were recorded in only one sample each. Mercury was undetected at a reporting range of 0.5 to 0.7 mg/kg. One sample tissue had a lead concentration of 20.2 mg/kg and is an apparent outlier when compared to the rest of the lead detections.

Aroclor-1254 was the only Aroclor detected in shellfish tissue samples in the Tidal Flats. Detection levels ranged from 0.020 to 0.050 mg/kg and were similar to the reference station.

8.6.1.2 Marine Basin. Fifteen metals were detected in ribbed mussel tissues collected from the Marine Basin. Mercury was below detection in all tissue samples.

Aroclor-1254 was the only PCB detected in shellfish tissue samples from the Marine Basin. Detection levels ranged from 0.039 to 0.046 mg/kg and were similar to the reference station.

8.6.2 Finfish

8.6.2.1 Tidal Flats. Fourteen metals were detected in whole body fish tissues collected from the Tidal Flats. Arsenic, silver, and mercury were undetected with reporting ranges from 0.39-0.4, 0.25-0.26, and 0.5-0.7 mg/kg, respectively. Cadmium was detected in one sample at 0.05 mg/kg. All metal concentrations were comparable to reference station values.

Aroclors 1248, 1254, and 1260 were the only Aroclors detected in whole body tissue samples collected from the Tidal Flats. Aroclor-1248 was only detected in one sample from the Tidal Flats; it was not recorded in the reference location or Marine Basin samples. Total PCB concentrations were 0.238 and 0.290 mg/kg. These concentrations were slightly higher than reference results that ranged from 0.017 to 0.210 mg/kg. Corrected for % lipids and expressed as mg of PCB per kg of lipid, maximum concentrations in fish collected in the Tidal Flats were higher than levels of PCBs in reference fish. Site area tissues had similar maximum concentration of 83.5 and 85 mg PCB/kg lipid in the Tidal Flats and Marine Basin.

8.6.2.2 Marine Basin. Thirteen metals were detected in fish tissues in the Marine Basin. Mercury was not detected. Nickel was detected in only one sample and arsenic and silver were also undetected. As with the Tidal Flats, tissue concentrations ranged lower than maximum sediment concentrations.

Aroclor 1254 and 1260 were the only Aroclors detected, and they were measured in all samples analyzed. Values measured were also slightly higher than reference levels. Corrected for % lipids and expressed as mg of PCB per kg of lipid, maximum concentrations in fish collected in the marine basin were higher than levels of PCBs in reference fish. Site area tissues had similar maximum concentrations.

8.7 SUMMARY OF CONTAMINATION

The following paragraphs present a summary of contamination by media, followed by a summary of contamination by analyte group.

Soil

The cVOCs 1,1,1-TCA, PCE, and TCE at concentrations less than 1 mg/kg were detected in soils across the entire Site, including those areas where no industrial activity or chemical storage was known to have occurred. Concentrations of cVOCs greater than 1 mg/kg were detected at and in the vicinity of of the Hazardous Waste Storage Area, and near Buildings B-6A and B-7A. In general, the number of location where cVOCs were detected, and the detected concentrations, decrease with depth. The fuel-related constituents, benzene, toluene, ethylbenzene, and xylenes (BTEX), were detected in Site soils. Concentrations less than 0.2 mg/kg were detected across the majority of the Site in the 0-2 feet bgs interval, with number of locations where BTEX constituents were detected decreasing with depth. The highest concentrations of BTEX were generally in the vicinity of Buildings B-37, B-38, and B-74.

PAHs were detected in soils across the entire Site at concentrations between 0.001 and 0.5 mg/kg, including areas where no industrial activity was known to have occurred such as the West and South Parking Lots. The distribution of PAHs with concentrations ranging from 0.5 to 1.0 mg/kg is more limited, and suggests that at least some of the detected PAHs within this range may be present as a result of fuel releases. However, there are detections within this range in areas where no industrial activity is known to have occurred (South Parking Lot and the extreme northern end of the Dike), suggesting that this range may also be related to fill or asphalt. The areas of PAHs with concentrations greater than 1.0 mg/kg correlate with areas of fuel usage and releases at the Building B-13 and B-15 area and the South Parking Lot soils from B-34 and B-65 excavations.

TPH was detected in approximately two-thirds of all samples submitted for TPH analysis. TPH was detected in areas of past fuel storage and use, but also in areas where fuels were not used. The highest detected concentrations of TPH were detected in the vicinity of the Hazardous Waste Storage Area, Building B-16, and the Jet Fuel Tank Farm.

Hexavalent chromium was detected in soils beneath the former B-2 Chromium Plating Facility only (see Figures 8.1-58 through 8.1-60). With the exception of several detections in the 2-4 feet bgs interval, the majority of detections were limited to the 0-2 and 4-15 feet bgs intervals. Chromium concentrations greater than the deflection point were primarily limited to beneath the former B-2 Chromium Plating Facility (co-located with hexavalent chromium), but were also detected near the Hazardous Waste Storage Area between Buildings B-74, B-13, and the OATP. Cyanide soil concentrations were most frequent and greatest in soil from the 0-2 feet bgs interval from beneath the former B-2 Chromium Plating Facility, as well as from the area of Buildings B-

37 and B-38. Open storage was historically conducted in the area near Buildings B-37 and B-38, and cyanide detected in soil in this area potentially resulted from storage of cyanide waste.

In general, cVOCs concentrations greater than RSR criteria were detected in soil within the 0-2 and 2-4 feet bgs intervals, in the area of Buildings B-13, B-15, B-74, and B-38. TCE was the most frequently detected cVOC exceeding RSR criteria. BTEX concentrations were greater than GB PMC only, and were detected in soil from the 0-2 feet bgs interval in the area of Building B-37, B-38, and B-74. PAHs and SVOCs greater than RSR criteria were generally detected in fill soils near the Dike and in the South Parking Lot. Inorganics greater than RSR criteria were generally detected in fill soils near the Dike and in the South Parking Lot, but were also detected beneath Buildings B-2, B-3, and B-10.

Groundwater

The cVOC TCE was detected throughout groundwater north and west of Sniffens Lane. Other cVOCs, including 1,1,1-TCA and PCE, were detected primarily from beneath Building B-2 extending towards Building B-16, and beneath the West Parking Lot. The distribution of detected concentrations of cVOCs between shallow (less than 30 feet bgs) and deep groundwater is approximately equal. The fuel-related constituents, benzene, toluene, ethylbenzene, and xylenes (BTEX), were detected in groundwater beneath Building B-2 and beneath the area encompassed by Building B-13, Building B-64-1, and the Building B-74 Former Oil House Tank Farm.

Concentrations of cVOCs are greater than RSR criteria throughout the site, north and west of Sniffens Lane. The majority of cVOC concentrations greater than RSR criteria are for the analyte TCE. The distribution of concentrations greater than RSR criteria between shallow and deep groundwater is approximately 5:1, respectively. This is influenced by the applicability of GW VC to shallow groundwater only, and not reflective of relative concentrations between shallow and deep groundwater. BTEX concentrations did not exceed RSR criteria.

Hexavalent chromium is present in groundwater beneath the former B-2 Chromium Plating Facility at concentrations up to 950 mg/L.

Soil Vapor

Soil vapor samples were collected in 1999 and 2004. The 1999 soil vapor sampling event was conducted for field screening purposes. During the 2004 sampling event, permanent, rather than temporary, monitoring probes were sampled. Results of the 1999 soil vapor survey identified TCE concentrations greater than RSR criteria in the central part of the site including Building B-2 (center), Building B-10 and Building B-12. Concentrations outside of this central area were either at, or below, volatilization criteria. The 2004 permanent soil vapor points were installed in the northern and southern parts of Building B-2, B-65, B-3, B-3A, B-4, B-16 and B-19 to determine if these areas were in compliance with RSRs. Results of the 2004 soil vapor survey identified that TCE concentrations exceed soil vapor RES and I/C VC throughout Buildings B-2, B-3 and B-3A, and that PCE concentrations exceed RES and I/C VC in Building B-3, and RES VC in Buildings B-3, B-3A, and B-4.

Sediment

Sediment samples from the Tidal Flats, Marine Basin, and Outfall 008 Drainage were analyzed for VOCs, SVOCs, PCBS, and inorganics.

Tidal Flats. VOCs detected in sediments in the Tidal Flats included 1,2-DCE, acetone, carbon disulfide, 2-hexanone, TCE, toluene, and ethyl benzene. Concentrations were relatively low (less than 0.2 mg/kg). TCE and 1,2-DCE were both detected on the west side of the Causeway near areas of shallow groundwater contamination by the same compounds, indicating VOCs in sediment in this area may be a result of discharging shallow groundwater. Twenty-seven SVOCs, all PAHs, were detected in surficial sediments collected from the Tidal Flats.

Three PCB Aroclor mixtures: Aroclor-1248, Aroclor-1254, and Aroclor-1260 were detected in approximately 50% of samples analyzed for PCBs. The maximum PCB concentration was detected for Aroclor-1248 at a concentration of 130 mg/kg. In shallow Tidal Flats sediment, the highest detected PCB concentrations (greater than 10 mg/kg) were in sediments located off-shore of Building B-38 and the OATP. In deeper sediments, PCBs were only detected at two locations at relatively low concentrations, up to 0.5 mg/kg.

All twenty-three TAL metals, trivalent arsenic and methyl mercury were detected in surface sediment samples. For the majority of analytes, the detection frequency was 100%. Cyanide was detected at a concentration of 0.79 mg/kg.

In shallow tidal flats sediments, concentrations of chromium were greater than background at nearly all samples throughout the tidal flats sediments off-shore of the Site. In deeper sediments, concentrations above background were limited to two locations east of the Causeway and a single location near Building B-36.

In both shallow and deep sediment, concentrations of methyl mercury greater than 1 mg/kg are generally located west of the Causeway. Five sediment sample locations, extending linearly from between Buildings B-37 and B-38, had the highest (greater than 5 mg/kg) detected concentrations for both depth intervals.

Marine Basin. VOCs, including acetone, MEK, methylene chloride, and carbon disulfide were detected in Marine Basin sediments, at relatively low concentrations (less than 0.25 mg/kg).

Seventeen SVOCs detected in Marine Basin sediments, the majority PAHs. The maximum concentrations of several PAHs were detected in the zero to 6-inch sample MB-0-6.

Three PCB Aroclor mixtures were detected in Marine Basin sediments: Aroclor-1248, Aroclor-1254, and Aroclor-1260. Aroclor-1260 was detected in all samples and was present at the highest concentration (0.18 mg/kg).

All twenty-three TAL metals were detected in sediment samples collected from the Marine Basin. Trivalent arsenic and methyl mercury were also detected, while hexavalent chromium, silver, and CN were undetected.

Sediment concentrations of cadmium, chromium, copper, lead, nickel, and zinc in the Marine Basin were generally higher than average reference concentrations.

Outfall 008 Drainage. VOCs detected in sediments in the Outfall 008 Drainage included total 1,2-DCE, vinyl chloride, acetone, MEK, and chlorobenzene, at relatively low concentrations.

Twenty-three SVOCs were detected at least once in surficial sediments collected from the Outfall 008 Drainage.

PAHs detected generally had concentrations that were over 10 times higher at sample location SD08003A than at other stations, but does not reflect a trend of discharge from the Outfall 008 Drainage (i.e., higher concentrations at the point of discharge and then decreasing with downstream distance).

Aroclor-1260 was detected in five of six samples from the Outfall 008 Drainage. The maximum concentration of 1.5 mg/kg was detected at sampling station SD08002A.

All 23 of the TAL metals, CN, trivalent arsenic and methyl mercury were detected in Outfall 008 Drainage sediment samples. Methyl mercury levels in Outfall 008 Drainage correspond well with Marine Basin sediment concentrations, indicating that the source of mercury in these areas is probably not related to the outfall.

Concentrations of cadmium, chromium, copper, lead, nickel, silver, and zinc were all elevated, with maximum values from 2 to more than 40 times higher than maximum reference location concentrations.

8.7.1 Chlorinated VOCs

Detected concentrations of 1,1,1-TCA, PCE, and TCE greater than 1 mg/kg in soil were generally detected in the vicinity of Buildings B-15, B-37, B-38, and the B-74 Tank Farm. The highest detected concentrations (greater than 10 mg/kg) of cVOCs in Site soils are generally detected in the vicinity of Building B-37 and B-38 pump stations. In general, the number of locations where cVOCs were detected, and the detected concentrations, decrease with depth. Several isolated cVOC concentrations which are greater than RSR criteria are noted. These include 1,1,2,2,-TCA and TCE concentrations near Building B-7/B-7A and 1,2-DCA concentrations near Building B-6A, potentially the result of a documented 1,1,1-TCA spill in this area. The cVOCs detected near Building B-7 and B-7A may be related to engine testing activities or solvent storage historically conducted in this area.

Detected 1,1,1-TCA, PCE, and TCE concentrations in shallow soils likely represent localized and relatively recent releases to the ground surface, either directly or from leaking drums and/or storage tanks, in the vicinity of AOCs 1, 2, 12, 53 (Hazardous Waste Accumulation Area), or the Open Storage between Buildings B-16 and B-74. These releases were likely related to the storage of waste and raw solvent at these areas. The distribution of breakdown products 1,1-DCE, cis-1,2-DCE, and vinyl chloride generally detected at concentrations greater than 1 mg/kg correlates well with the distribution of parent products, as discussed above.

Concentrations of 1,1,1-TCA, PCE, and TCE in groundwater are generally greatest in central portions of the facility, extending from beneath Building B-2 to beneath Building B-16. Other distinct areas of elevated cVOC concentrations exist beneath and south of the West Parking Lot, west of Building B-6 (1,1,1-TCA), and beneath the Building B-3, B-3A, and B-4 complex, the area of the B-34 Tank Farm, Building B-19, and Building B-37.

The distribution of elevated groundwater cVOC concentrations suggests areas of multiple releases resulting from Site activities related to the use of solvents for parts and/or engine cleaning, particularly in Building B-2. In addition, documented releases of chlorinated solvents have occurred, including releases near Buildings B-3, B-3A, and B-4, B-6 and B-17 (1,1,1-TCA), B-7, B-16, B-19, and B-34 (Jet Fuel Tank Farm). The multiple release areas at the Site have resulted in overlapping groundwater cVOC plumes. Distribution of breakdown products, including 1,1-DCE, cis-1,2-DCE, and vinyl chloride, in groundwater correlate well with the parent cVOCs 1,1,1-TCA, PCE, and TCE, as discussed in subsection 8.2.1.

Dichloromethane was detected in soil at concentrations greater than GB PMC near Buildings B-37 and B-38. All data has undergone validation, and these detected concentrations were not flagged indicating any blank contamination. Therefore, it is likely that these concentrations are valid.

Soil vapor samples were collected in 1999 and 2004. The 1999 soil vapor sampling event was conducted for field screening purposes. During the 2004 sampling event, permanent, rather than temporary, monitoring probes were sampled. Results of the 1999 soil vapor survey identified TCE above RSR criteria in the central part of the site including Buildings B-2, B-10 and B-12. Results of the 2004 soil vapor survey identified that TCE was above soil vapor VC throughout Buildings B-2, B-3 and B-3A, and that PCE was above VC in Buildings B-3, B-3A, and B-4. Soil vapor cVOC concentrations are highest beneath Building B-2, the Building B-3, B-3A, and B-4 complex, and Buildings B-10 and B-12. With the exception of the southern end of Building B-2, the locations of cVOCs detected in soil vapor are generally coincident with groundwater contamination by the same cVOCs. The other possible source of cVOCs in soil vapor is soil contamination that is suspected beneath the southern end of Building B-2.

Sediment samples taken from several areas were analyzed for VOCs, including the Tidal Flats, the Marine Basin, and Outfall 008 Drainage. Concentrations of cVOCs were detected in sediment from the Tidal Flats and at Outfall 008. TCE and 1,2-DCE were detected tidal flats sediment located on the west side of the Causeway, near areas of shallow groundwater contamination by the same compounds. Contamination of sediments by cVOCs in this area may be a result of discharging shallow groundwater. 1,2-DCE, vinyl chloride, and chlorobenzene were detected at Outfall 008.

8.7.2 BTEX and TPH

Comparison of soil and groundwater BTEX distributions indicates several correlations. Elevated BTEX concentrations in soil are evident near Buildings B-37, B-38, and B-74, and overlie an area of shallow groundwater BTEX contamination located north of Buildings B-13 and B-15, encompassing Buildings B-37, B-38, B-64-2, and B-74, and extending beneath the tidal flats for over 100 feet. The distribution of BTEX concentrations in groundwater beneath Building B-2 is consistent with that of 1,1,1-TCA. As discussed in subsection 8.2.4, DNAPL consisting primarily of 1,1,1-TCA was detected in monitoring well HESE-01-12D, located within the area of highest detected concentrations of 1,1,1-TCA and BTEX beneath Building B-2. Analysis of the DNAPL indicates that the product also includes a small fraction (<1% by weight) of toluene. This would indicate that BTEX contamination in deep groundwater beneath Building B-2 is likely the result of the co-located release of 1,1,1-TCA at concentrations high enough to produce DNAPL.

Sediment samples taken from several areas were analyzed for VOCs, including the Tidal Flats, the Marine Basin, and Outfall 008 Drainage. BTEX, specifically ethylbenzene and toluene, were detected in sediment from the Tidal Flats only.

The distribution of TPH in soil correlates well with that of BTEX and historical usage and storage of fuel-related products at the Site. The highest concentrations of TPH in soil are generally located near and beneath Building B-16, the Jet Fuel Tank Farm, and the Hazardous Waste Storage Area.

8.7.3 Other VOCs

MEK, methyl isobutyl ketone, and acetone were the primary non-chlorinated VOCs detected in soil at the Site. These analytes were generally detected across the majority of the facility, while elevated concentrations of these analytes were generally detected in fill soils adjacent to the Dike. MEK and acetone were also detected in groundwater, with the highest detected concentrations located beneath the former B-2 Chromium Plating Facility and the central portion of Building B-2.

Sediment samples taken from several areas were analyzed for VOCs, including the Tidal Flats, the Marine Basin, and Outfall 008 Drainage. The non-chlorinated VOCs acetone, 2-hexanone, methyl ethyl ketone, and methylene chloride were detected in sediment from various samples at these areas.

MEK and acetone were used to coat metal parts, which were then subsequently cleaned with toluene. The distribution of MEK concentrations in groundwater beneath Building B-2 corresponds well with that of BTEX and with areas where cleaning of metal parts was likely conducted.

8.7.4 PAHs and SVOCs

PAHs were detected across the facility in soil, whereas they were detected in only several locations in groundwater near the Dike. In soil, the highest detected PAH concentrations were observed in the area of the South Parking Lot where fill originating from excavation of material in the area of Building B-65 and the Jet Fuel Tank Farm was used in re-grading. This area was likely one of the most recently paved areas at the facility, with paving a potential source for PAHs.

In general, detected concentrations of non-PAH-related SVOCs in soil were less than 1 mg/kg. Exceptions to this include several phthalates, particularly bis(2-ethylhexyl) phthalate and butylbenzyl phthalate, which were detected at maximum concentrations of 96 and 170 mg/kg, respectively.

Sediment samples from the Tidal Flats, the Marine Basin, and Outfall 008 Drainage were analyzed for SVOCs. Concentrations of SVOCs, primarily PAHs, were detected at each of these areas. It should be noted that the lower portion of Outfall 008 Drainage receives discharge from nearby Igor Sikorsky Memorial Airport, and that detected PAH concentrations are greatest below the point of confluence.

8.7.5 PCBs

PCBs were detected in soil mostly in fill adjacent to the Dike. PCBs were primarily the aroclors 1248, 1254, and 1260. No leachable concentrations of PCBs were detected by SPLP method, and PCBs were not detected in groundwater. The material used to construct the Dike appears to have been contaminated with PCBs.

PCB aroclors 1248, 1254, and 1260 were detected in sediment from the Tidal Flats and the Marine Basin. Aroclors 1254 and 1260 were detected in sediment from Outfall 008 Drainage. Aroclor-1254 was not detected in reference sediment samples for the Site.

8.7.6 Inorganics

The inorganic analytes hexavalent chromium, cadmium, copper, cyanide, and nickel were used in metal plating/finishing operations at the Site, and are not generally naturally occurring.

Hexavalent chromium was detected in both soil and groundwater beneath the former B-2 Chromium Plating Facility. The distribution of chromium in soil was primarily limited to beneath the former B-2 Chromium Plating Facility.

Cadmium was generally detected in soil at concentrations greater than the deflection point within areas that received fill. Cadmium concentrations in groundwater greater than the deflection point were detected beneath the former B-2 Chromium Plating Facility and Buildings B-10 and B-12.

Cyanide was primarily detected in soil and groundwater from beneath the former B-2 Chromium Plating Facility and near Buildings B-37 and B-38, adjacent to the Hazardous Waste Storage Area.

Copper and nickel were generally detected in groundwater at concentrations greater than respective deflection points beneath the former B-2 Chromium Plating Facility.

9.0 CONTAMINANT FATE AND TRANSPORT

9.1 INTRODUCTION

Flow and transport modeling at SAEP has been conducted in two phases. *Phase I* focused on the initial calibration of the flow model, assessment of the long-term effectiveness of Soil Vapor Extraction (SVE) treatment by evaluation of treatment depths relative to potential impacts of VOCs on indoor air, and assessment of natural attenuation of VOCs and Cr(VI). *Phase II* modeling focused on the refinement of the flow model and further development and calibration of the natural attenuation model for VOCs.

9.2 PHASE I MODELING

The *Phase I* Modeling was conducted in 2001. That modeling effort produced calibrated flow simulations and preliminary reactive transport simulations conditioned on the data available at the time. The focus was an assessment of the long-term efficacy of any treatment of solvent plumes beneath SAEP buildings. The high concentration plumes represent potential sources of PCE, TCE, various TCA isomers, and their transformation products that could impact indoor air quality. Concerns had been raised that anything less than complete plume treatment might result in rebound of VOCs at the water table and subsequently to indoor air. Groundwater flow models (FEMWATER, MODFLOW) were calibrated to define steady-state conditions, based on available data and reasonable assumptions regarding constant head boundaries and recharge distribution. Two recharge scenarios were evaluated – the current condition with negligible recharge at SAEP, and a hypothetical condition assuming the complete removal of all SAEP structures, *i.e.*, a more uniform regional recharge. Sequential dechlorination and mineralization of PCE/TCE and TCA solvents were modeled (using RT3D) independently to assess MNA, with/without hot spot treatment to various depths, and with/without the limited presence of DNAPL sources. Preliminary transport modeling of Cr(VI) was also conducted (using MT3DMS).

The *Phase I* modeling results demonstrated that cleanup would be required to a depth of approximately 35 feet, and of wide lateral extent, to avoid contaminant rebound above some critical concentration at the water table beneath the buildings. The upward component of groundwater flow beneath the SAEP adjacent to the Housatonic River appeared to be of sufficient magnitude to merit scrutiny. Water quality data were judged insufficient for rigorous calibration of reactive transport because only one reasonably complete sampling was available (1999). The Chromium transport modeling suggested minimal transport toward the Housatonic – the slow advective transport of chromium was retarded further by adsorption and potentially by precipitation as sparingly soluble trivalent chromium (Cr[III]) solids. Analysis of the recharge scenarios indicated only modest head changes beneath the SAEP and only small, localized increases in flow velocity. The potential introduction of aerated water to the groundwater system via recharge with the buildings removed was determined not to be a significant factor since the zone of aeration would be relatively small and near the surface, while most of the contaminants of interest are located deeper in the aquifer. Additionally, the increased organic content of the recharge water under these conditions would likely serve to counterbalance the effects of increased aeration. Based on these observations, the effect of modified recharge on contaminant transport was not considered further.

9.3 PHASE II MODELING

The purpose of the *Phase II* modeling was to develop a better-calibrated model for both flow and transport, conditioned on a new monitoring round. Reactive transport modeling was used to evaluate long-range impacts to surface water.

Phase II modeling including the following: (1) the association of high concentrations of 1,1-DCE with TCA suggests the possible formation of 1,1-DCE by the dehydrochlorination of TCA; (2) DNAPLs, if present, would be expected to inhibit microbially-mediated processes that directly or indirectly drive the degradation reactions; and (3) uncertainty regarding the topography of the impermeable bedrock and its influence on flow and transport in the overlying strata.

Five tasks, including four modeling tasks, were defined and conducted as follows to address the cleanup concerns:

Task 1. Revise the geologic conceptualization
Task 2. Recalibrate the flow model using post-1999 data
Task 3. Develop a new reaction model to accommodate DNAPL inhibition of bioreactions and the possible dehydrochlorination of TCA to 1,1-DCE within a single RT3D module for both chloroethenes and chloroethanes
Task 4. Calibrate the new reactive transport model, conditioned on the 2002 data
Task 5. Assess the long-term impact of various degrees of treatment

Task 1. Revise Geologic Conceptualization. New borehole and other data were collected to further define the extent of the mudflat sediments, fill material, and the topography of the crystalline bedrock. These data include new wells and slug tests performed on piezometers in the tidal mudflat sediments. Other tests within the Glacial and Reworked Glacial units have revealed local zones of elevated permeability, which appear to be associated with areas of shallow bedrock. Minor revision of the stratigraphy was made to accommodate these new data.

Task 2. Recalibrate the Flow Model. Revision of the hydrogeologic conceptualization required reconsideration of the flow calibration. Hydraulic head data from 2002 were significantly different than the 1999 data used to calibrate flow in *Phase I*. Issues related to changes in recharge rates were revisited. Steady-state flow was calibrated for both the 1999 and 2002 conditions. The relatively wet flow conditions for 2002 were used to drive the reactive transport simulations in all subsequent tasks. Sensitivity analyses for flow parameters – primarily permeabilities and recharge – are reported.

Task 3. Develop a New Reaction Package Development of a new reaction package for RT3D that could address multiple reaction pathways for both chlorinated ethenes and ethanes was necessary to resolve some of the issues raised since the *Phase I* effort. The model had to combine the sequential dechlorination pathways for ethenes and ethanes (modeled independently in *Phase I*). The reason for combining the two pathways was to include a linkage from TCA to 1,1-DCE. The recommendation that reaction inhibition within DNAPL zones be considered in the Phase II modeling was accommodated by inclusion of an inhibition function in the reaction package, conditioned on concentrations of any of the primary solvents (PCE, TCE, TCA).

Task 4. Calibrate the new reactive transport model in light of the 2002 data. New data collected in 2002 showed the same basic trends as the 1999 data, with elevated concentrations near the hot spots. These new data provide the minimum requirement – two reasonably complete snapshots of water quality conditions – upon which a credible calibration of reaction rates could be attempted. In addition to the adjustment of 13 reaction rates, calibration was achieved by varying the inhibition factors for PCE, TCE, and TCA and the definition of constant-concentration cells to approximate the effect of a persistent DNAPL source.

The presence of DNAPL was confirmed below Building B-2 (monitoring well HESE-01-12D), though, as indicated by the very high concentrations observed for primary solvents (PCE, TCE, TCA), DNAPL might be present at other locations beneath SAEP. Model estimates of how long it would take to clean the site by natural or engineered processes will be strongly influenced by the mass and distribution of any DNAPL. In the absence of data that quantitatively delineates the extent of DNAPL, a conservative treatment of the DNAPL source was adopted. Imposing a constant-concentration condition at multiple grid blocks within the existing hot spots approximates the effect of a DNAPL. Some uncertainty remains as to the distribution and longevity of possible DNAPLs as finite sources of solvents.

As with all reactive transport calibration efforts, SAEP predictions are conditioned on the available water quality data. Both the 1999 and the 2002 data sets suffer inadequacies. The 1999 data set, which serves to define the initial conditions here, suffers from the fact that it includes different parts of the plume not represented in the 2002 data, and *vice versa*. The hot spots are well represented only in the 1999 data due to direct, one-time samplings, but no observations were made within the mudflat sediments. The 2002 data represent predominantly monitoring wells at the periphery of the hot spots, and other post-1999 data reveal new areas of contamination that could never be predicted from the 1999 conditions. Although the modest reaction rates proposed here are reasonable, the shortcomings of the 1999 data set do not support high levels of confidence in the calibration.

The presence of strongly reducing mudflat sediments may serve as a natural reactive barrier for solvent dehalogenation and biodegradation. Since the groundwater flow rate is slow and the contaminant transport rates are suppressed further by retardation (adsorption), any contaminant flux to the Housatonic likely would be quite small. Remaining questions regarding the reaction rates can only be addressed with continued monitoring. However, even at slower degradation rates and with DNAPL sources, the primary conclusion remains that solute flux to the mudflat surface is extremely low and dilution by the tidal fluctuation of the Housatonic River flow is overwhelming.

Task 5. Assess the long-term impact of various degrees of treatment. Full and complete removal of the solvent contamination is not a practical or necessary goal. A more practical approach is to identify the requisite level of cleanup – in terms of both the physical extent and the degree of removal efficiency – that must be achieved in order to meet remediation goals.

The *Phase I* modeling assessed the requisite physical extent for cleanup, optimistically assuming 100% removal within the treated zone. Here, the key variable is the efficiency of removal. Several cleanup efficiencies are evaluated – 99%, 90%, 75%, and 50% – with benchmarks defined as 0% treatment and no treatment at all (DNAPL cells still active). Treatment at all levels (including 0%) is assumed to be sufficient to remove DNAPLs, which are deactivated in the top five grid layers. The calibrated reaction rates from *Task 4* are applied in these simulations.

In-situ chemical oxidation and thermally-enhanced extraction have been explored for cleanup of the three, main groundwater VOC hot spots at SAEP. The cleanup process is not modeled explicitly. The universal removal of the parent solvents (but not their transformation products) is imposed by manually editing the initial conditions using the data tools in the DoD Groundwater Modeling System (GMS) interface.

The target predictions are VOC concentrations at the water table beneath SAEP buildings which are compared to threshold values in the groundwater and in the mudflat facies. These data are provided in the form of a functional GMS model, from which data can be extracted at whatever level of detail is required..

No new chromium transport modeling was conducted during *Phase II*. However, questions had been raised regarding the fate of Cr(VI) and possible controls on its solubility – predominantly adsorption (reversible) and chemical reduction followed by precipitation as a sparingly soluble solid phase (effectively irreversible). In *Phase I*, it was hypothesized that the reducing conditions in the mudflat facies would be sufficient to reduce Cr[VI] to the trivalent form, Cr[III], which is much less soluble, stable, and nontoxic.

9.4 HYDROGEOLOGIC CONCEPTUALIZATION

The hydrogeologic characterization of the site is based upon the characterization used in *Phase I modeling* with a few modifications. Sixteen additional borings were available (see Figure 9-1), including 6 pairs of deep and intermediate borings were drilled in May 2002 and incorporated to the existing conceptual stratigraphic model of the site (Figure 9-2). Analysis of the differences between the existing model and the stratigraphy as defined by the new borings indicated that only minor, local adjustments to the stratigraphic model were needed. These changes were incorporated into the updated stratigraphic model and used in generating the numerical modeling domains used for the *Phase II* modeling tasks.

Other changes were made to the stratigraphic conceptual model from the *Phase I* modeling effort. Analysis of the available hydraulic conductivity data for the Reworked Glacial Outwash and Glacial Outwash units was performed and it was determined that there was insufficient difference in the hydraulic conductivity values from these two units to support their being modeled as separate units in the numerical models of the site hydrogeologic system.

The same analysis also led to the determination that the lateral zonation of some model stratigraphic units was warranted given the disparity and localization evident in some hydraulic conductivity values. Hydraulic conductivity tests of the topmost fill material indicate highly variable values mainly in the range of 100-300 feet per day (ft/d) except in the areas near the raised dike that runs along the eastern shoreline of the SAEP. Tests of the hydraulic conductivity of the soils located in this area indicate lower values – generally in the range of 20 ft/d. The fill in this area was placed by hydraulic backfilling during construction of the dike and thus it is not unexpected that it would have different hydrogeologic properties than those of the fill materials located throughout the rest of the model area. Accordingly, a zone of lower hydraulic conductivity fill material located under the dike is included in the stratigraphic model.

Analysis of the Glacial Outwash hydraulic conductivity values indicates a zone of higher hydraulic conductivity in an area trending east-to-west along the southern boundary of the SAEP extending beneath the tidal flat sediments. This zone of higher conductivity corresponds well to an area of high bedrock elevations that also is estimated to extend out underneath the tidal flat sediments and under the river (see Figure 9-3). This "bedrock wall" is a significant feature in the subsurface hydrogeology of the site and impacts several aspects of the stratigraphic model as well as the location of model boundaries and assigned boundary conditions that are discussed below.

Because of the influence the bedrock wall has on the hydraulic conductivity, an evaluation was made to determine if it is continuous along the southern boundary of the site. Available borehole and geophysical data indicate the continuity of the bedrock wall. Contours of observed groundwater elevations collected in November 1999 and May 2002 (Figures 9-4 and 9-5) are consistent with that of a continuous bedrock high. A gap in the bedrock wall would cause significant groundwater flow to pass through, forcing flow to the south of SAEP. The interpreted groundwater head contours in the figures are not indicative of a southerly flow direction in this portion of the aquifer. The contours are consistent in their spacing indicating no appreciable change in the transmissivity of the aquifer along the suspected wall length.

Hydraulic conductivity tests in this area indicate a zone of higher hydraulic conductivity on the order of 10 times that of the other glacial outwash materials. These results support the concept of a bedrock high that is constricting the glacial outwash material in the aquifer. With decreased cross-sectional area through which the groundwater can flow, the result is higher groundwater flow velocities, which is consistent with the higher measured hydraulic conductivities in this portion of the study area. Considering all available data and indicators together, it is assumed that the bedrock wall is competent and continuous across the southern boundary of the SAEP facility and extends into the outwash sediments located under the tidal flats and Housatonic River. A zone of glacial outwash materials with higher hydraulic conductivity was added to the stratigraphic conceptual model (Figure 9-6).

No borings to bedrock exist in the sediments beneath the Housatonic River in this area, thus it is not known how far out under the riverbed sediments the bedrock wall extends. However, model simulations with a higher hydraulic conductivity zone that terminated under the tidal flats resulted in groundwater head contours that were not consistent with either the November 1999 or May 2002 water level measurements. Since the bedrock wall and zone of higher hydraulic conductivity are assumed to be strongly correlated, the difficulty in calibrating the model with a truncated higher glacial outwash hydraulic conductivity zone was taken as an indication that the zone and thus the bedrock wall likely do extend out underneath the tidal flat sediments to at least an area somewhere underneath the river sediments, which forms the eastern boundary of the flow model.

The stratigraphic model consists of five distinct aquifer materials all located on top of the bedrock surface, which is represented by the bottom surface of the materials shown in the figure. By not including bedrock material in the model of the aquifer, the assumption is made that the bedrock material acts as a relatively impermeable barrier to groundwater flow. It is possible that the bedrock is not entirely impermeable and there may be some groundwater exchange between the bedrock and the overlying aquifer. Available field data were inconclusive in determining how much, if any, flow might be exchanged between the bedrock and other aquifer materials. If the amount of exchange were significant, our modeling assumption of an impermeable bedrock surface would cause the calibration of the model to be problematic. Calibrating an impermeable bedrock model to observed groundwater head levels of a permeable bedrock system would require abnormal specified head and/or recharge boundary condition values in the model to compensate for the flow exchanged with the bedrock. The head and recharge boundary conditions assigned to the model in its calibrated state is believed that both the level of calibration and the values of these boundary conditions are reasonable. The assumption of a relatively impermeable bedrock surface is therefore assumed to be valid for the purposes of this modeling study.

The topmost material of the modeled aquifer is the fill material, which is separated into two hydrogeologic units: those materials hydraulically backfilled along the eastern edge of SAEP underneath the dike and those materials found on and near the surface in the rest of the study area. The glacial outwash is likewise separated into two units: a zone of higher hydraulic conductivity located on top of the bedrock wall along the southern boundary of SAEP and extending out under the tidal flats and Housatonic River to the model boundary and the other glacial outwash sediments located throughout the remainder of the SAEP study area. The final modeled material is the estuarine silt, which is found above the glacial outwash materials in the tidal flats and Housatonic River areas of the model. Table 9-1 indicates the range of hydraulic conductivity values that were obtained from the remedial investigation, the ranges of values tested in the model calibration process as well as the final calibrated values used in Phase II. Over 110 test results were used distributed fairly evenly between the different geologic units using various hydraulic conductivity estimation techniques including rising head, slug tests and grain size analysis.

Boundary Locations and Specified Head Conditions

As in the *Phase I* modeling, the variably saturated groundwater model FEMWATER was used to simulate the groundwater flows across the entire study area. The red line in Figure 9-7 delineates the boundaries of the study area which were chosen based on available data, hydrogeologic conditions and relative distance from the areas of interest within the model domain. The red points in the figure indicate locations where specified head values were assigned, denoted by the blue values in the boxes adjacent to each point. Specified head values are interpolated linearly between the red points and assigned to the nodes of the model mesh accordingly.

As indicated in the figure, specified head boundaries are assigned to the boundaries of the model on all sides for the entire depth of the aquifer with the exception of the boundary running along the northeast side of the model, which corresponds to the sediments located under the Housatonic River. Along this boundary, specified heads were assigned to the Estuarine Silt materials only and a no-flow condition exists at the boundary face for the glacial outwash layers located beneath the estuarine silt. This simulates the vertical groundwater flow that most likely exists in the glacial outwash materials underneath the river. All other boundaries in the model are assumed to be hydrostatic and correspond to areas of wetlands (along the west, south and southeast), Frash Pond or areas where monitoring wells provide head readings for use as boundary conditions.

The potential existence of regional deep groundwater flow from the northwest toward the southeast was examined in the selection of boundary conditions for this model. Available data for this study are inconclusive on the existence of such a regional flow pattern. Without any data within the study area or upstream to assign for boundary conditions, testing this hypothesis in the model is problematic. However, the existence of the bedrock wall along the south end of SAEP renders moot the question of a regional deep groundwater flow. This model is designed to address the questions concerning the migration of contaminants located in the subsurface at SAEP and thus this study is only concerned with groundwater flow patterns which could potentially affect the contamination at SAEP. Any deep regional groundwater flow from the northwest toward the southeast that may or may not exist in the deep groundwater system under SAEP would necessarily encounter the bedrock wall feature discussed previously. Flow over this feature would be forced to within approximately 50 feet of the surface and therefore come under influence of the near-surface groundwater flow trends, which are toward the tidal flats and Housatonic River. The issue of whether or not such a flow trend exists within the study area is therefore not relevant to the discussion of this modeling effort.

Frash Pond is approximately 40 feet in depth and does not fully penetrate to the bedrock. However, the model assumes hydrostatic conditions at the southern and eastern shore of the pond, particularly since observed groundwater data suggest a northwest flow of groundwater in the areas immediately southeast of the pond. Along the shore of Frash pond, a constant head of 1.68 feet was assigned which simulates a condition of groundwater head along and under the pond equal to a general average Frash pond level. Frash pond does fluctuate somewhat in pool elevation throughout the year, but it was determined from sensitivity analyses that the specific pond level did not have a significant impact on groundwater heads underneath SAEP, likely due to the existence of a groundwater divide located at the northwest corner of SAEP. Both the November 1999 and May 2002 observed groundwater heads show the existence and consistent location of this divide.

Heads along the boundary running across the narrow strip of land between Frash Pond and the tidal flats were assigned based on readings from two monitoring wells (WC-18D1 and WC1-1S) located along or near the boundary (see Appendix 2 of the ERDC 2003 report for the observed values). Along the shore of the northern tidal flats, head values were estimated based on a gradual decrease in groundwater levels as the boundary approaches the Housatonic River. The northeast boundary of the model runs along the centerline of the Housatonic River and heads along this boundary are assigned as shown in Figure 9-7. These heads are assigned only to the Estuarine Silt material while the glacial outwash materials below are not assigned any boundary condition, simulating a no-flow boundary condition as discussed above.

The specified head value assigned to the boundary along the Housatonic River is 0.87 feet. This same boundary condition is also assigned to the surface of the model in the tidal flats. The Housatonic River is tidally influenced with the tidal flat sediments being exposed at least part of each day. However, since this modeling effort is focused on the migration of contaminants over a period of 20 years or more, tidal influences on groundwater flow are assumed to be steady and can thus be effectively modeled with a specified head boundary condition in the tidal flats and along the river. A sensitivity analysis of the specified head value to assign was conducted with several values ranging from 0 to 1.0 feet. Runs with a specified head value of 0.87 feet yielded the best match in head values and flow directions for both the November 1999 and May 2002 data sets.

Being located very close to the mouth of the Housatonic River, the waters in the tidal flats and river at this location are brackish and thus the issue of the potential influence of brackish water on groundwater flow must also be considered in the model. Density dependent flow and transport modeling is a feature of the FEMWATER code used in the study however, the level of data and effort required to appropriately and accurately model such physical processes are beyond the scope of this study. Accepted practice in such cases is to use an "equivalent head" adjustment to the assigned heads on the salt-water boundaries of a model where both fresh and salt water is being simulated. Given the relatively low concentration of salt found in the water at this site (estimated at approx. 11 parts per thousand based on data collected in the tidal flats), the equivalent head adjustment was found to be less than 0.01 feet – an amount deemed to be negligible.

It is important to note that the assignment of specified head values on the top surface of the model in the tidal flats does not specify the head values throughout the estuarine silt material in the model. In both the FEMWATER model and the smaller, high resolution MODFLOW model used in this effort, there are multiple layers of mesh elements (FEMWATER model) or grid cells (MODFLOW model). While heads at the surface of the estuarine silt material are specified, the model computes heads in the model layers below the surface. The tidal flats are assumed to be a sink for the groundwater flow but the flow must first pass through the relatively low permeability estuarine silt sediments. In this fashion, it is believed that this boundary condition assignment accurately simulates the relevant physical processes found in the tidal flats adjacent to SAEP.

The assignment of hydrostatic heads on the boundaries in the wetland areas to the west and south is an assumption that is made in this model. Based on the experience of those conducting this study and others familiar with the hydrogeologic conditions in the area, this assumption is believed to be reasonable. However, given the available data for this study, one cannot determine conclusively whether or not this assumption is valid. Although unlikely, it is possible that the wetlands in this area are perched and not connected directly to the groundwater. If the wetlands are indeed perched, the actual groundwater elevations along these boundaries may be somewhat different than those used in the model. However, these boundaries are located far away from the area of interest in this model and are separated from the SAEP area by at least one groundwater divide. The relative influence of the head values at these boundaries is minimal as evidenced by sensitivity analysis conducted during calibration of the flow model. Changing the head values at these boundaries had a negligible effect on the computed heads under SAEP. Therefore, the hydrostatic assumption along these boundaries is deemed to be acceptable for the purposes of this study.

Recharge Boundary Conditions

The majority of the water flowing through the hydrogeologic system simulated by the models in this study enters the system as recharge. Based on the observed groundwater elevation data from November 1999 and May 2002, the topographic high in the vicinity of the Sikorsky airport also corresponds to a groundwater high with heads falling in all directions away from the high. Because this system appears to be largely driven by recharge, determining appropriate values to assign to the model for the recharge boundary conditions is a critical step in building a calibrated model capable of accurately simulating the hydrogeologic system at SAEP. Because of the varied types of land use across the model domain, several zones of varied recharge values were used to simulate the actual recharge of water to the system. Figure 9-8 depicts these zones (delineated in blue), which are defined by three different types of land use. The areas around the airfield that are not paved or covered with buildings are open grasslands and thus the highest amounts of recharge occur in these zones. The airfield tarmac and SAEP parking lot are considered a greatly reduced recharge area and are assigned a much lower recharge rate. The airfield runways and SAEP installation itself are completely paved or covered with very little recharge possible due to installed drainage systems. These areas have the smallest values of recharge assigned.

In the Phase I effort, only the November 1999 observed heads were available from which to calibrate the model. In that effort, the open areas of the model were calibrated with a recharge value of 8 inches per year. This amount of recharge was thought to be too low. For the Phase II effort, additional data in the form of another set of groundwater heads (May 2002) and a greater number of monitoring well readings were available from which to calibrate the model. Comparison of the November 1999 and May 2002 data showed a dramatic rise in water level at common monitoring wells with the difference being more pronounced the further the well was located from the tidal flat area. Once this change was noted, historical records of precipitation for the area were reviewed.

The nearest National Oceanographic and Atmospheric Association (NOAA) rain gage is located in Bridgeport, immediately to the west of Stratford, CT. According to their records, the average annual precipitation is 41.6 inches. For the month of November, the average precipitation is 3.76 inches. In November 1999, however, only 3.08 inches of precipitation were recorded at the Bridgeport gage, ranking this particular month in the lower third of recorded precipitation totals for November over the past 50 years. The same gage indicates an average of 3.74 inches of precipitation for the month of May but in the month of May 2002, 6.66 inches of rain were recorded with over 5 inches falling before the 19th. This ranks in the top five precipitation totals for May since 1961. Given the fact that more than 2 times as much rain fell in May 2002 than in November 1999 (and some of the November 1999 precipitation could have fallen as snow that did not immediately melt), it is not surprising that common monitoring wells recorded significantly higher heads in May 2002.

Since the November 1999 and May 2002 datasets were found to represent relative dry and wet conditions respectively, it was determined that a single recharge value for the open areas could not be used to calibrate to both sets of observed data. Rather, it was decided to calibrate the model to both a wet and a dry condition. This provides the model with an added level of assurance in that the model has been calibrated to conditions on either end of the hydrologic spectrum.

The blue values shown in the white boxes in Figure 9-8 depict the recharge values that were calibrated for the May 2002 observed heads. The recharge values for the November 1999 observed heads differ only in the open areas where the recharge values change from 20 inches per year to 12 inches per year.

Numerical Model Domains

As indicated previously, a larger scale flow model was built using FEMWATER to establish boundary conditions for a smaller scale, high resolution MODFLOW model that was used for the contaminant transport simulations of the various tasks of this effort. FEMWATER solves Darcy's law in the saturated zone and the Richards' equation in the vadose zone using an unstructured finite element formulation. The SAEP FEMWATER mesh built for this effort is shown in Figure 9-9 and consists of 31,988 nodes and 55,745 elements in 9 layers of unstructured mesh elements. Two layers are used to represent the fill material, two layers for the estuarine silt and seven layers for the glacial outwash material. The estuarine silt material does not extent over the entire model domain and thus not all layers are mutually exclusive in their material type. In the horizontal dimension, triangular elements range from 300 feet on a side at the boundary to approximately 30 feet on a side in the vicinity of the dike along the eastern edge of SAEP. Vertically, the elements range in thickness from approximately 2.5 feet to 20 feet. Because of the unstructured nature of the mesh, resolution can be locally increased in areas of interest or where added mesh resolution is needed to accurately capture hydraulic gradients.

MODFLOW is a saturated-zone only code that uses a structured finite difference formulation to solve Darcy's law. The SAEP MODFLOW grid built for the transport tasks of this effort is shown in Figure 9-10. It consists of 90 columns, 190 rows and 8 layers of cells, giving a total of 78,480 cells, of which it is estimated that approximately 80% are active in the flow computations. The grid covers a smaller area than that of the FEMWATER mesh with the southwestern and southeastern boundaries being along flow lines from the FEMWATER model results. These boundaries are represented as no-flow boundaries in the MODFLOW model. The northern boundary locations and conditions and recharge values are unchanged from the FEMWATER model. Horizontally the grid cells are approximately 40 feet on a side, and range from approximately 1 foot to over 20 feet of saturated thickness (see Figure 9-11).

While the northern boundary between Frash Pond and the tidal flats is located fairly close to portions of the area of interest at SAEP, it should be noted that in this area both the element and grid cell sizes are small which provides numerous computation points between the boundary and the areas of interest. Numerous computation points allow the model to lessen the impact of an imposed boundary and seek a level that is independent of the user-specified head. However, it should also be noted that located along and near this boundary are three monitoring wells from which the boundary head values are taken which increases the level of confidence in the values used for the boundary conditions in this area.

Flow Path Analysis

ERDC also performed some particle tracking evaluations to depict pathways contaminated groundwater within Hot Spots #1, #2 and #3 would take within the model. Particles were seeded or placed in model layers 4, 6 and 8 within the defined areas of the Hot Spots, and tracked using the USGS program MODPATH (see Figure 9-12). Groundwater within Hot Spot #1 migrated to discharge locations to the east of the Causeway. Particles associated with Hot Spot #2 flowed toward and beyond the Causeway, while those in Hot Spot #3 flowed toward the tidal flats and river northwest of the Causeway.

Figure 9-13 depicts only the layer 8 (deepest) particles from an oblique angle (a dip of about 18 degrees) looking to the northwest. Also depicted is the bedrock surface (maroon) and vertical lines (green) drawn from the center of each hot spot to its corresponding position on the surface where the outlines of the buildings are shown. The particles track along the bedrock surface until they reach the portions of the aquifer underneath the tidal flats where their paths turn sharply upward. The bedrock beneath Hot Spot #3 is much deeper than at Hot Spot #1 and thus in the figure it has the appearance of being further to the south whereas in reality, it is located to the north. Although not shown in the figure, the flow lines for particles in the shallower layers all follow similar patterns, undulating somewhat underneath SAEP as influenced by the shape of the bedrock surface and then turning sharply upward once underneath the tidal flats. All of the shallower layer particles daylight in the tidal flats closer to shore than those of layer 8 as shown in Figure 9-13.

This analysis depicts the model-predicted flow paths for particles placed at known contaminant hot spots at SAEP. However, it is only based on hydraulic flow properties and does not take into consideration other factors which affect contaminant transport, such as dilution, degradation, etc. Analysis of the amount of groundwater flux discharged to the tidal flats verses the northeast boundary of the model, which represents the middle of the Housatonic River channel, was performed by ERDC. Of the groundwater flow that reaches either the tidal flats or the Housatonic River, approximately 96 % of that flow is discharged to the tidal flats, leaving about 4 % that is discharged to the river. This fact, coupled with the flow path analysis, indicates that, based on groundwater flow results alone, the likelihood of contaminants presently located in the aquifer beneath SAEP reaching the Housatonic River is minimal (ERDC, 2003).

Flow Model Conclusions

ERDC presented the following summary conclusions, subject to the model assumptions, in a Powerpoint slide show, “Presentation of Phase-II Groundwater Modeling Results” (21 May, 2003):

- Groundwater velocities beneath SAEP are generally low due to hydraulic properties of materials and low hydraulic gradient;
- Assumption of impermeable bedrock surface is supported by ability to calibrate model to wet and dry conditions with reasonable recharge rates;
- Flow path analysis indicates majority of flow daylights in tidal flats;
- Deep groundwater experiences long travel times to tidal flats; and
- SAEP building removal would not significantly affect flow patterns or travel times of contaminants.

9.5 NATURAL ATTENUATION OF SOLVENTS

Extensive contamination of groundwater by chlorinated solvents has been detected at the SAEP. The contaminants include PCE, TCE, TCA, and their transformation products (Figure 9-14). The question addressed here is whether MNA is a viable option for the long-term compliance with state remediation regulations. Additionally, this effort addresses some of the concerns identified in the first modeling phase. These concerns include the following: (1) reaction calibration would require the collection of at least a second round of groundwater monitoring data; (2) uncertainty remains as to the presence and extent of DNAPLs; (3) the presence of any DNAPL could inhibit microbially-mediated transformations; and (4) the reaction model may need to consider formation of 1,1-DCE from TCA, as seems to suggested by their association.

New data collected in 2002 provided the basis for reaction rate calibration. A new reaction model was developed which included additional transformation products of TCA, including 1,1-DCE. The model included inhibition of all reactions at elevated solvent concentrations. Calibration was achieved by adjusting individual reaction rates and inhibition parameters, and by defining DNAPL cells as constant-concentration internal transport boundaries. These subjects are discussed below.

9.5.1 Phase I Reaction Model Development

Numerical Framework

The numerical grid used for MODFLOW and RT3D is derived from the calibrated FEMWATER flow model developed in a previous analysis of the SAEP flow field (see Section 6.2.6). Additional calibration of the MODFLOW model was conducted as discussed in Section 6.2.6 of this document. MODFLOW calculates the head distribution field, which is used by the transport model to calculate local flow rates for advective transport.

RT3D – Reactive Transport in 3-Dimensions (version 2.0; Clement and Jones, 1998) – was used for all reactive transport simulations. RT3D describes the advective-dispersive transport, including the adsorption (retardation) and transformation of multiple constituents. RT3D includes several standard reaction packages, including the sequential dechlorination package as applied in Phase-I. User-defined reaction packages also can be accommodated, as was prepared for this investigation.

RT3D solves the coupled differential equations describing the three-dimensional, reactive transport of multiple species in saturated groundwater systems. The code utilizes a reaction operator-split (OS) numerical strategy to solve any number of coupled transport equations.

The graphical interface to RT3D incorporated into the DoD Groundwater Modeling System (GMS) versions 3.1 and 4.0 was utilized in the definition of initial conditions, setting model parameters, and visualizing results.

Common Transport Processes

Natural attenuation may be defined as the reduction of contaminant toxicity, mobility, or volume to levels that are protective of human health and the ecosystem (USEPA, 1999a). The processes that may contribute to natural attenuation include advection, dispersion, dilution, sorption, volatilization, abiotic (chemical) transformation, and biodegradation. Among these coupled processes, the biogeochemical reactions that affect any irreversible transformation or immobilization of organic contaminants to form innocuous products are of greatest interest.

The reactive transport model developed and applied here simulates the following processes: (1) advection, (2) hydrodynamic dispersion, (3) equilibrium (reversible), linear adsorption, (4) sequential transformation pathways for PCE/TCE and TCA, partially convergent on 1,1-DCE, and described by first-order kinetics; and (5) inhibition of transformation reactions at elevated solvent concentrations.

Advection. Advective transport is typically the most significant mechanism for solute migration in porous media. The rate of advective transport is directly proportional to the intrinsic permeability of the medium and the local hydraulic gradient. The permeability of sedimentary lithologies ultimately depends on the nature of sediment available in the original depositional system (e.g., gravel, sand, silt, clay) and the local conditions under which they were deposited (e.g., sedimentary structures, degree of heterogeneity in multiple directions). The mineralogical composition of the sediments and the post-depositional history also influence net hydrogeologic behavior. An effective porosity of 0.30 was used for all SAEP hydrogeologic units, as in *Phase I* modeling, based on available reports (RI) for the site. The effective porosity influences the relative speed of advective and diffusive transport.

Hydrodynamic Dispersion. Natural porous media are intrinsically heterogeneous, varying only in the magnitude and structure of heterogeneities. This variability results in a medium with a virtually limitless variety of potential flow paths, resulting in water parcels advancing with a distribution of net flow velocities. Variation in the groundwater flow velocities results in the longitudinal dispersion of the solute along the plume's primary flow direction. Similar dispersion in the lateral and vertical directions further contributes to plume dispersion, though usually at a lesser degree. Vertical dispersion tends to be most severely constrained due to the horizontally layered nature of most sedimentary media. For the present RT3D simulations, longitudinal dispersivity was set at 30 feet for all lithologies. Transverse and vertical dispersivities – entered as ratios relative to the longitudinal value – are defined, respectively, as 0.33 (10 ft.), and 0.10 (3 feet).

Adsorption. The retardation of organic solute transport resulting from adsorption may be an important consideration in simulating the effects of natural attenuation. Partitioning of a solute to the aquifer matrix removes it from solution and, depending on the reaction mechanism, may sequester it from transformation. The pseudo first-order kinetic model used here for degradation applies only to the dissolved concentration, upon which the rate is dependent.

SAEP site-specific characterization of CAH partitioning is unavailable. However, some limited organic carbon content data were available. As a reasonable approximation, a hydrophobic partitioning model, conditioned on site measurements of organic carbon content, is adopted to estimate partitioning coefficients (K_d's). Karickhoff *et al.* (1979) defined a linear relation between the organic carbon partitioning coefficient (K_{oc}; L/kg) and the more readily available octanol/water-partitioning coefficient ($K_{o/w}$) for aromatic and chlorinated hydrocarbons (equation 9.1a). They further suggested that the K_d could be estimated as the product of the K_{oc} and the mass fraction of organic carbon (f_{oc}) in the sediment:

$$K_{oc} = 0.63\, K_{o/w} \qquad \text{(a)} \qquad (9.1)$$
$$K_d = f_{oc}\, K_{oc} \qquad \text{(b)}\ .$$

The K_d model describes a linear isotherm, with the implicit assumptions of equilibrium and reversibility. The $K_{o/w}$, and K_d values for SAEP solvents summarized in Table 9-2 are conditioned on a fraction organic carbon of 0.0012 (source: 2002 data). The bulk density of the medium is a factor in the treatment of adsorption. A representative value of 1.45 g/cm^3 (41,059 g/ft^3) is taken for all SAEP media (ERDC, 2003).

Treatment of DNAPLs

Uncertainty remains as to the nature and extent of DNAPLs containing PCE, TCE, and/or TCA at SAEP. The elevated concentrations of solvents are consistent with the presence of DNAPL. DNAPLs serve as a long-term, though finite, source of the primary solvents. The high solvent concentrations proximal to any DNAPL zone could inhibit the biodegradation process. Simulating the presence of a DNAPL could influence model predictions of the duration and efficacy of MNA.

There is no universal consensus as to the best method to accommodate the presence and inhibition effects of DNAPLs in reactive transport models. In the absence of site-specific data, a reasonably straightforward approach is implemented here. First, the definition of a DNAPL cell is imposed by "locking in" the initial condition defined by the value interpolated from the 1999 data. This would be a conservative treatment in that a true DNAPL is a finite source that would dissipate over the course of a simulation.

The designation of DNAPL cells was essentially a constrained calibration tool. Initially, the cell within each hot spot in each grid layer with the highest concentration of TCE (or TCA in hot spot 3) was defined as a DNAPL cell; the DNAPL condition would also be assigned for the other solvents if present in excess of 1% of the its aqueous solubility. If more parent (or transformation) solute was required to improve the prediction of 2002 conditions, additional cells were designated as constant-concentration DNAPL cells, generally starting with the cells having the highest concentrations within a particular layer and hot spot. PCE, TCE, and TCA were treated independently – that is, defining a constant-concentration cell for one solvent does not require that the others be set as constant, but there are cases where multiple solvents are set constant within a single grid cell. Generally, the DNAPL cells are in close proximity within one or more of the hot spots, at least within the same grid layer.

Contour plots of each solvent (PCE, TCE, TCA) in each of the 8 grid layers after extended simulation are presented in Appendix-4 of the Phase II groundwater model report (ERDC, 2003). These figures serve to document which cells were designated as constant-concentration cells and graphically illustrate the effect of assigning DNAPL cells. As an example, the contour of TCE in grid layer 3 after 20 years (Figure 9-15) indicates that constant-concentration DNAPL cells have been defined in each of the three hot spots.

9.5.2 Phase II Reaction Model Revisions

A new, user-defined, RT3D reaction package was developed for Phase-II transport modeling. The new model combines and links the sequential reductive dechlorination pathways for both PCE/TCE and TCA (modeled independently in Phase-I) and considers the likely inhibition of bioreactions at elevated solvent concentrations. Each component in the new reaction network is observed at SAEP. The main features of the new model include the following, each of which will be discussed more fully below:

1) Sequential, reductive dechlorination of the PCE/TCE-associated chloroethenes;

2) Sequential, reductive dechlorination of the TCA-associated chloroethanes;
3) TCA dehydrochlorination to 1,1-DCE;
4) TCA hydrolysis to acetic acid; and
5) Inhibition of all reactions at elevated concentrations of PCE, TCE, and/or TCA.

Sequential Dechlorination Reactions. The more common approach to modeling solvent degradation is as a series of reductive dechlorination reactions. The PCE/TCE and TCA pathways were modeled separately in Phase-I. Due to the partial convergence of the pathways on 1,1-DCE in the new model, both of these pathways are incorporated into the new model.

The sequential dechlorination pathway for the chloroethenes is modeled with first-order kinetic reactions of PCE to TCE to the three DCE isomers to VC and finally to ethene, the mineralization of which is presumed to be relatively rapid. There are three DCE isomers: 1,1-DCE, *cis*-1,2-DCE, and *trans*-1,2-DCE. Each of these can form from the reductive dechlorination of TCE. The formation of multiple products, even under controlled conditions, is a common phenomenon in organic chemistry. The situation is complicated further in natural systems by the likelihood of multiple, coupled, abiotic and microbially-mediated reactions, each of which may *favor* a particular isomer. Bouwer (1994) suggests that the *cis*- isomer is the prevalent product and 1,1-DCE the least common product in microbially mediated reactions. However, the predominance of any particular DCE isomer is site-specific due to differences in microbial consortia as well as geochemical conditions affecting the production and transformation of DCEs. All three DCE isomers have been reported in both the 1999 and 2002 water quality data, but 1,1-DCE predominates in both frequency and magnitude. Thus, 1,1-DCE was modeled as the only isomer in *Phase-I* with the implicit assumption that it represented all three isomers.

The sequential dechlorination pathway for the chloroethanes likewise is modeled with first-order kinetic reactions of TCA to DCA to CA to ethane, which is presumed to mineralize rapidly. Fortunately, the 1,1,1-TCA isomer is the predominant TCA isomer at SAEP. Further reference to TCA refers to the 1,1,1- isomer. Additional complications are discussed in the next section.

TCA-to-DCE Dehydrochlorination. The available SAEP field data seems to suggest that elevated levels of 1,1-DCE are collocated with elevated concentrations of TCA. There are at least two possible explanations for such an observation – either 1,1-DCE is being formed from 1,1,1-TCA as well as from TCE (see Figure 9-14) and/or the presence of TCA inhibits the further dehalogenation of 1,1-DCE to VC (vinyl chloride), leading to a local accumulation of 1,1-DCE.

Vogel and McCarty (1987) report that 1,1,1-TCA can undergo abiotic degradation to form acetic acid by hydrolysis and 1,1-DCE via an elimination reaction (dehydrochlorination). They observed that the formation of acetic acid is favored (by a factor of about five) over 1,1-DCE. The TCA to 1,1-DCE reaction is incorporated in the new model, as is the favored TCA to acetic acid reaction. May 2004 groundwater analytical data confirm that acetic acid is present in higher concentrations within the 1,1,1-TCA groundwater plumes. However, acetic acid is not modeled explicitly; rather acetic acid is assumed to be mineralized relatively quickly.

The various dechlorination pathways are often characterized as being predominantly abiotic or biotic (microbially mediated). In the absence of attempts to resolve or document abiotic *vs*. biotic mechanisms, the distinction becomes academic. In fact, abiotic processes are coupled intrinsically by biotic influences on environmental chemistry, particularly the redox conditions. The RT3D reaction package presented here utilizes a modified, first-order kinetic model for each transformation of the constituents of interest. Application of such a lumped kinetic model does not differentiate between abiotic and microbially mediated processes. Both processes are presumed to be active at SAEP at low rates.

Reaction Inhibition. Elevated concentrations of solvents can be expected to inhibit microbial processes affecting the direct or indirect transformations of organic solutes. Concentrations can be expected to be elevated in DNAPL zones, particularly with respect to the parent solvents (PCE, TCE, TCA). However, the location of DNAPL cells is not known with certainty. Thus, a more general approach to simulating the effects of solvent toxicity is adopted here.

The new reaction model utilizes a simple inhibition factor approach similar to that defined by Molz, Widdowson, and Benefield (1986) and utilized in advanced bioreaction models such as UTCHEM (CPGE 2000). The inhibition factor (F_{Inhib}) is defined as follows:

$$F_{Inhib} = \frac{I_{half}}{I_{half} + C_{Inhib}} \tag{9.2}$$

where C_{Inhib} is the concentration of the inhibiting solute and I_{half} is the concentration of that inhibiting solute at which the uninhibited rate is halved. Thus, when C_{Inhib} equals I_{half}, the inhibition factor is 0.5. The inhibition factors are always between 0 and 1 and decrease in magnitude as solvent concentrations increase. As C_{Inhib} increases well beyond I_{half} (high concentrations), F_{Inhib} approaches zero, as would the reaction rates. Conversely, as C_{Inhib} approaches zero, F_{Inhib} approaches unity and the reaction rates are essentially uninhibited.

The inhibition factors are implemented as multiplicative in the new reaction package, *i.e.*, PCE, TCE, and TCA each will contribute to a net inhibition of *all* reaction rates. In the absence of site-specific experimental data, the I_{half} values for PCE, TCE, and TCA are set to 10% of their aqueous solubility, which are, respectively, 120, 1,100, and 1,360 mg/L at 20 C. These values appear to work reasonably well.

The self-inhibition curves are plotted for each solvent in Figure 9-16 (solid lines). Note that the uninhibited rates are halved at the I_{half} concentrations – 12 mg/L for PCE, 110 mg/L for TCE, and 136 mg/L for TCA. This inhibition function leaves transformation rates effectively uninhibited at low concentrations while severely inhibiting reactions at high concentrations. The position of the inflection point is defined by the value of I_{half}.

When the inhibition factors (equation 9.2) are applied in a multiplicative fashion, the inhibitory influence of a less soluble solvent will have a more dramatic impact on the more soluble solvents at a given concentration. This point is demonstrated by the cumulative inhibition curves in Figure 9-16. The multiplicative inhibition of TCA (the most soluble of the three solvents) degradation appears more severely impeded, relative to its self-impeded function; TCE is affected slightly less but in similar fashion. PCE (the least soluble of the three) appears minimally affected by the additional inhibition factors. These trends, though real, are magnified dramatically in the figure due to the use of a single concentration scale (*vs.* scaled to the saturation of individual solvents). For field conditions in the presence of multiple DNAPLs, the components would be present at *approximately* the same, usually low, proportion of their solubility (assuming similar mass transfer rates into the aqueous phase).

This inhibition model results in a spatial and temporal variation in all reaction rates. For example, the initial distribution of PCE degradation rates in grid layer 3 (Figure 9-17) reveals the extent and severity of inhibition associated with the three hot spots. Comparison of temporal trends of PCE rates (Figure 9-18) at three points identified in Figure 9-17 reveals distinctly different behaviors – rates within the hot spots are severely impeded initially but approach the unimpeded rate after 20 years, whereas rates in the plume are only weakly impeded. Similar trends are observed for the TCE-DCE11 and TCA-DCA reaction rates which are plotted with PCE-TCE in Figure 9-19. Asymptotic approach to an impeded rate reflects achievement of steady-state conditions under the influence of constant-concentration DNAPL cells, *e.g.* for Hot Spot #3 in Figure 9-19.

Results

The contour plots shown in Figures 9-20 through 9-32 support the general trend toward the reduction of contaminants. Low concentrations of transformation products appear to encroach on the mudflat region, and much of that may be due to extrapolation from the 1999 data set into low-permeability media (mudflat facies).

The RT3D-GMS simulations of the coupled PCE/TCE and TCA natural attenuation suggest that the solvents will be degraded and attenuated during their slow transport toward the mudflat facies. The long travel times allow degradation to be a significant factor in the natural attenuation process. Adsorption and low flow rates contribute to the slow rates of transport. The reaction rates developed here are reasonable estimates conditioned on the available data.

9.6 NATURAL ATTENUATION OF CHROMIUM

The 1999 concentrations as high as 950 mg/L were detected beneath the former B-2 Chromium Plating Facility in Building B-2. As with the solvents, the chromium plume appears to be moving very slowly toward a zone of anoxic, tidal mudflat sediments. MNA for chromium may be possible based on (1) the reducing conditions in the estuarine, tidal mudflat sediments may act as a natural permeable reactive barrier; and (2) the slow rates of advective transport provide sufficient time for MNA, though Cr reduction reactions tend to be quite rapid.; Conversely, MNA may not be a viable alternative if: (1) reducing conditions in the tidal flat sediments are not effective in reducing and immobilizing Cr, and/or (2) if part of the plume manages to circumvent the tidal flat sediments.

MT3DMS Modeling

The reactive transport model MT3DMS (Zheng 1999) was utilized to simulate chromium transport at SAEP. Flow conditions were defined with MODFLOW, using the larger numerical grid that extends into the mudflat sediments (as in Task-2). MT3DMS describes advective-dispersive transport, equilibrium adsorption, and first-order decay of independent constituents. This relatively simple modeling approach was adopted to make long-term simulations (up to 50 years) more practical.

The reduction and precipitation, conceptual model described above is approximated by a simple advective-dispersive transport scheme with spatially variable partitioning. Cr adsorption is modeled with a linear isotherm model. A modest adsorption K_d of 0.5 to 3 L/kg is assigned to all geologic media other than the tidal mudflats. The rapid reduction and precipitation reaction is approximated by very strong adsorption of Cr to the tidal flat sediments, which were assigned a K_d on the order of 3500 L/kg. The net effect of Cr immobilization is the same.

Initial conditions were defined using field observations from 1999. As described in Section 1 of the Phase I report, some subjectivity was involved in reducing multiple data entries for the same point at the same or a proximal date. An averaged or higher value was assigned in the case of multiple 1999 entries. Interpolation of the reduced dataset onto the numerical grid defined the initial conditions (Figure 9-33). Several background level data points were introduced in uncharacterized, peripheral areas to constrain extrapolation estimates of unjustifiably high concentrations in these areas.

Results

MT3DMS-GMS simulations indicate that chromium transport toward the Housatonic River is very slow. Assuming a modest partition coefficient of 3 L/kg, the Cr has moved little after 50 years. Assuming an even lower partition coefficient of 0.5 L/kg, the Cr plume can reach the mudflat sediments, where it is immobilized by strong adsorption (as a surrogate for chemical reduction and precipitation). Pump-and-treat to affect a partial reduction of Cr concentrations in the hot spot, which is under consideration, would reduce the plume concentrations but not change the net behavior predicted for the untreated plume.

Figures 9-34 to 9-36 illustrate Cr transport using the modest K_d value of 3 L/kg, consistent with site geochemical conditions. Cr transport is minimal during the 40 years simulation. Apparently, adsorption in combination with slow flow rates, are sufficient to significantly retard Cr transport.

Figures 9-37 to 9-38 illustrate the effect of reducing the K_d to 0.5 L/kg to permit transport into the mudflat. The simulations represented by Figures 9-37 and 9-38 use the same 1999 initial Cr distribution as in Figures 9-34 to 9-36. The Cr is immobilized quickly upon entry to the mudflat sediments.

The effect of cleanup of the chromium plume to 5 mg/L or less is depicted in Figures 9-39 to 9-42, using the lower K_d value (0.5 L/kg). The same effect of immobilization in the mudflat sediments is observed.

Geochemical and hydrogeologic conditions appear to be amenable to the natural attenuation of Cr(VI) by sequestration in the tidal mudflat facies. A reasonable estimate of a linear adsorption coefficient (3 L/kg), conditioned on site water quality, and the low flow rates effectively inhibits Cr migration toward the river. If a lower K_d is considered (0.5 L/kg), the plume reaches the reducing mudflat sediments, where it is estimated that it will be immobilized by chemical reduction and precipitation. A more thorough characterization of the extent and chemistry of the mudflat facies is warranted. If the mudflat facies extends inland beneath the fill material, Cr immobilization may occur further inland.

The strongly anaerobic mudflat facies located between the plume and the river may act as a natural, permeable reactive barrier to induce chemical reduction to trivalent chromium and precipitation of sparingly soluble mineral phases, effectively immobilizing chromium. Though this hypothesis requires more field documentation and feasibility analysis, the modeling conducted here suggests no impediment to its potential success.

10.0 CONCEPTUAL SITE MODELS

10.1 SOIL CONTAMINATION

Soil samples at SAEP were generally collected from the unsaturated zone, typically located less than 10 feet below ground surface. In general, the soil consists of fill soils, though native soils may potentially exist beneath the southwestern portions of the Site, including portions of Building B-2. Areas at SAEP that have received fill include approximately 10 acres of the Site located along the Dike, the majority of which was hydraulically dredged from the Housatonic, and portions of the South Parking Lot. The fill consists of sand, gravel, and debris associated with construction of buildings, roads, utilities, site grading, and other structures at the Site, is found as the uppermost soil type throughout most of SAEP. The fill is generally about 2 to 5 feet thick, but reaches a thickness of about 20 ft adjacent to the Dike. The fill is thickest near the Dike. The composition of the fill is variable, but most of it is described as a granular material, generally consisting of brown medium to fine sand and gravely sand. The fill decreases in thickness with distance away from the Dike.

The Site lacks a background soil data set representative of soils un-impacted by site activities; therefore, an evaluation of soil data was conducted to develop baseline levels. Evaluation of the distribution of contaminants indicates that cVOCs, BTEX, VOCs, PAHs, other SVOCs, PCBs, TPH and inorganics were detected in soil throughout the facility. Because of the widespread nature of contaminants in soil at the Site, and the potential for multiple sources, including fill and site-related activities, the goal was to establish baseline values, or ranges of concentrations, to be used in the determination of potential releases superimposed over the baseline levels.

The baseline evaluation was conducted on analytical data for cVOCs, PAHs, inorganics, and TPH. The evaluation involved producing concentration distribution plots of the analytical data for analytes detected within these groups. The concentration distribution plots were reviewed to qualitatively identify the deflection points. The deflection point is the value above which there is an obvious increase in the slope of the plot that may be used to distinguish between a subset of the data that may be normally-distributed and consistent with baseline concentrations, and sampling locations that may have been influenced by historical practices at the Site.

For VOCs, levels up to 1 mg/kg were determined to be below baseline and often could not be related to a release. VOCs at and below this level were not used for release determinations and are below the RES and I/C DEC and GB PMC values. VOCs greater than 1 mg/kg were found to have better correlation with release areas. Fuel-related VOCs such as BTEX were also detected in soils, but less frequently then cVOCs. Toluene was the most frequently detected fuel-related compound. Concentrations less than 0.2 mg/kg were detected across the majority of the Site in the 0-2 feet bgs interval and decrease with depth.

For PAHs, concentrations between 0.001 and 0.5 mg/kg were found throughout the site including areas where no fuels were used or industrial activity occurred. PAHs in this range could not be related to specific releases. PAHs ranging from 0.5 to 1.0 mg/kg are also present where no industrial activity occurred and may be due to contaminated fill, asphalt, or releases such as minor spills and leaks. Comparison of the geometric mean (0.7 mg/kg) of the estimated deflection points to the observation that PAHs with concentrations greater than 1.0 mg/kg are potentially attributable to Site activities, suggests that a reasonable threshold or baseline value for PAHs as a group is 1.0 mg/kg. The areas with PAHs greater than 1.0 mg/kg correlate with areas of fuel releases.

Areas where contaminants were released to soil at levels above regulatory criteria include the Hazardous Waste and Waste Oil Area, the Jet Fuel Tank Farm, the former B-2 Chromium Plating Facility, the manufacturing areas of Building B-2 and the Drum Storage Area located north of Building B-19.

Hazardous Waste and Waste Oil Area. The Hazardous Waste and Waste Oil Area consists of waste storage areas in and around Buildings B-13, B-15, and the former Oil House Tank Farm (Building B-74). This area was used to manage solvents, fuels and oils and contained both above and below ground storage tanks and waste transfer piping, and was used for oil blending and waste reclamation and storage since the mid-1940s. Sediments hydraulically dredged from the Housatonic River were used as fill to extend the shoreline during the 1940s to provide land area for buildings in this area. Contaminants detected in soil within this area may be attributable to fill, particularly contaminants that are persistent in the environment, such as PCBs. The cVOCs, TCE, PCE, 1,1,1-TCA, cis-1,2-DCE, 1,1-DCE, vinyl chloride and chlorobenzene were detected in soils. The highest cVOC concentrations were found near Buildings B-37, B-38, and B-74, generally between 0 and 4 feet bgs. The distribution of BTEX compounds in soil is similar to that of cVOCs. PCBs detected at SAEP were detected at their highest concentrations near Buildings B-74. Concentrations of contaminants above RSR criteria were detected within an area encompassed by Buildings B-74, B-64-2, B-13, and B-15. Contaminants exceeding RSR criteria include cVOCs, BTEX, PAHs, other SVOCs, inorganics, PCBs, and TPH. Concentrations of cVOC exceed GB PMC values more frequently than DEC. The majority of concentrations greater than DEC only exceed RES DEC. BTEX concentrations exceed GB PMC only, and were observed between Building B-74 to immediately east of Building B-38. PAHs and other SVOCs exceed GB PMC and DEC at several locations, and were generally limited to the 0-2 feet bgs interval. PAHs exceed I/C DEC near and beneath Building B-13. Inorganic concentrations generally exceed RES and I/C DEC in soil from 0-4 feet bgs throughout this area. No concentrations of inorganics were greater than GB PMC. PCB concentrations are generally greater than RES DEC and I/C DEC near Buildings B-37 in soil from 0-2 feet bgs.

Jet Fuel Tank Farm. Jet fuel, diesel fuel, gasoline, and cleaning agents were stored in USTs at the former Jet Fuel Tank Farm adjacent to Building B-34. The jet fuel was used for engine testing in Building B-16. Leaks from the tank farm USTs and piping resulted in releases of fuel and solvents to soil adjacent to Building B-34. Contaminated soils were excavated when the tank farm was replaced with ASTs in 1989. Soil borings completed during the RI indicate VOCs and BTEX are present in soil outside the excavation area and trace amounts of LNAPL exist in MW-3. PCE, TCE, 1,1,1-TCA, 1,1-DCA, cis-1,2-DCE and BTEX were detected in shallow soil.

Drum Storage Area north of Building B-19. A drum storage area was located north of Building B-19 and held drums containing 1,1,1-TCA, PCE, and other solvents. A release of 1,1,1-TCA, PCE and fuel-related VOCs occurred at the Drum Storage Area north of Building B-19 and possibly from the ASTs also north of Building B-19. In 1990, 120 cubic yards of contaminated soil was excavated and shipped offsite. Contaminants detected in soil include cVOCs, BTEX compounds, other VOCs, PAHs, SVOCs, PCBs, and TPH. Concentrations of 1,1,2,2-TCA in soil from 4.6-6.6 feet bgs exceed GB PMC values.

Former B-2 Chromium Plating Facility. The former B-2 Chromium Plating Facility was located in the northeastern corner of Building B-2 since 1951. Electroplating included chromium, nickel, copper, and cadmium plating. Copper and cadmium plating were completed using cyanide-bearing plating solutions. Chromium-plating solutions were typically composed of chromic acid (dissolved chromium trioxide flakes), which contains hexavalent chromium and sulfuric acid in the ratio of 85-100:1. Chemical were pumped to the CWTP for treatment. VOCs, chromium, cyanide, copper and nickel were detected in soils beneath the floor of the facility. Plating solution infiltrated through cracks in the concrete and leaks in the floor drains and piping system that resulted in concrete and soil contamination. Contamination of soils by chromium is observed beneath the entire footprint of the former B-2 Chromium Plating Facility. The mass concentrations of Cr(VI) in soils are generally less than 100 mg/kg. The concentrations of total chromium from SPLP analyses were detected at concentrations up to 25.5 mg/L, versus the CTDEP GB PMC of 0.5 mg/L. The higher chromium concentrations from SPLP analysis were detected in the northern corner of the former B-2 Chromium Plating Facility. Maximum Cr(VI) concentrations detected in shallow soil are 640 mg/kg, which probably reflects plating bath or rinse tank residuals present with the soil moisture in the sample rather than as a solid. Soil moisture containing only 11,230 mg/L Cr(VI) could cause this result in a soil sample with a bulk density of 1.75 g/cc and 10% soil moisture by volume. Soils above the capillary zone likely contain dried plating solution. Any residual Cr(VI) remaining in the concrete floors or unsaturated zone soils is not likely to act as a continuing source of groundwater contamination, as there is no fluid transfer mechanism due to cessation of plating activities in the late 1990s, and the facility being covered by a roof so that rainwater cannot percolate through contaminated soils. Copper and nickel were also released to soils beneath the facility based on a comparison to site baseline levels but neither was detected above DEC or GB PMC.

Building B-2 Manufacturing. The Building B-2 manufacturing area contained degreasers, heat treatment, dip tanks, press pits, metal working machines and paint spray booths covering an area of about 18.8 acres. Thirty-five soil borings were completed adjacent to potential sources of contamination. Contaminants detected in soil include BTEX, VOCs, PAHs, PCBs and TPH. Concentrations of VOCs were generally detected at concentrations ranging from non-detect to 1 mg/kg, the deflection point value for VOCs. Concentrations of carbon tetrachloride, TCE and 1,1,1-TCA exceed 1 mg/kg. TCE was detected at a concentration of 56 mg/kg, which exceeds GB PMC.

Contaminants in soil were released from parts degreasing which was conducted throughout Building B-2. Soil samples have not been collected beneath all machine pits, in large part due to the fact that they were filled with up to ten feet of concrete, however similar levels of contaminants are likely present in soil beneath much of Building B-2.

South Parking Lot. The South Parking Lot was expanded and re-graded with fill, including contaminated soil excavated during construction of the Building B-65 foundation and storage tank removal at the former Jet Fuel Tank Farm. Prior to placement of this soil, it was stockpiled and remediated. Contaminants detected in soils at the South Parking Lot include BTEX, VOCs, PAHs. Low concentrations (less than 2 mg/kg) of fuel-related contamination were detected in soil borings from the former soil stockpile area and from the final placement location of this soil. Concentrations of 1,1,2,2-TCA, and PAHs exceed RSR criteria for samples from borings completed in the final placement location of this soil.

Soil Contaminant Fate and Transport. The ultimate fate of contaminants found in soil depends primarily on the type of contaminants. The VOCs found in soil will leach to groundwater and/or volatilize over time. Infiltration of stormwater runoff is minimized at SAEP by the fact that the majority of the Site is covered either with pavement or buildings. Without infiltration, leaching of soil contamination above the water table is minimal. The volatilization pathway of VOCs in soil is affected by the pavement and facility structures. As the VOCs in soil diffuse through the unsaturated zone they may enter the overlying buildings through cracks in the foundation or floor slab; even floor slabs in good condition typically present a gap between the floor slab and the foundation walls at the periphery of the floor slab and at utility penetrations. Slight pressure differentials between the indoor air and the underlying soil gas produced by natural variations in atmospheric pressure and operation of ventilation equipment provide a driving force that enhances VOC diffusion into the building via the cracks in the floor or foundation. Other main variables that control the rate of volatilization are the chemical/physical properties of the individual VOCs, the soil properties, the thickness of surface treatments, the temperature, the size of the buildings or paved areas, and air exchange rates. PAHs that are fuel release related will likely partition strongly to any organic carbon in the soil and eventually degrade. Inorganics and PCBs found in the soil are likely to remain immobile due to partitioning to organic carbon.

10.2 GROUNDWATER CONTAMINATION

Groundwater flow is in the direction of the Tidal Flats. Downward vertical hydraulic gradients indicate that aquifer recharging conditions exist in the vicinity of the West Parking Lot. Vertical gradients begin to flatten in the center of the Site and reverse (becoming upward) to discharging conditions near the Housatonic River in the shallow and intermediate wells and tidal flats piezometers. Deep wells indicate continued downward gradients, with some discharge into the Housatonic River beyond the Tidal Flats. Head differences between the shallow wells and deep wells indicate that the low permeability silt layers are acting as semi-confining units, restricting the movement of groundwater between the shallow and deep portions of the overburden unit. This allows, simultaneously, an upward gradient in the shallow aquifer and a downward movement in the deeper aquifer. Where these low permeability layers are less extensive, beneath the main Site, the head differences between shallow and deep wells are relatively small, and the equipotential lines remain essentially vertical.

Tidal fluctuations greater than seven feet were measured in the Housatonic River. Groundwater level fluctuations were detected in several of the monitoring wells on-site as a result of the introduction of surface water during high tide or loading caused by the introduction of surface water into the aquifer system. Groundwater fluctuations of one to two feet were recorded in monitoring wells located within 100 feet of the Dike. The groundwater fluctuation amplitudes diminished proportional to the distance from the river and were generally less than 0.1 feet at a distance of 300 feet from the river.

The hydraulic conductivity of the overburden aquifer spans approximately four orders of magnitude (0.03 to 317.9 ft/day). The calculated groundwater velocity ranges from about zero beneath Building B-2 to about 255 ft/year. The higher velocities occur in the more permeable sections of the overburden deposits, in particular the fill near the Dike. These zones are where the greater portion of discharge is occurring, and may provide preferential pathways for the transport of contaminants within the aquifer.

Contaminants detected in groundwater were generally similar to those detected in soil, and include cVOCs, BTEX, VOCs, PAHs, other SVOCs, and inorganics. PAHs and SVOCs were detected far less frequently in groundwater than they were in soil, and PCBs were not detected at all in groundwater. As previously identified, cVOCs, BTEX, and VOCs detected at SAEP generally reflect Site-related contamination.

The Site lacks a background data set representative of inorganics concentrations in groundwater un-impacted by site activities. The existing site-wide groundwater data were used to determine baseline levels. For the purposes of evaluating potential releases, the inorganics selected for evaluation included only those with RSR criteria: antimony, arsenic, barium, beryllium, cadmium, chromium (total), copper, lead, mercury, nickel, selenium, silver, thallium, vanadium, and zinc. Detection of either hexavalent chromium or cyanide is considered indicative of a release, and neither analyte was included in the baseline analysis.

The evaluation involved producing concentration distribution plots of the analytical data for each of the inorganic analytes. The concentration distribution plots were reviewed to qualitatively identify the deflection points. The deflection point is the value above which there is an obvious increase in the slope of the plot that may be used to distinguish between a subset of the data that may be normally-distributed and consistent with baseline concentrations, and sampling locations that may have been influenced by historical practices at the Site.

In addition to hexavalent chromium and cyanide, cadmium, chromium (total), copper, and nickel were used in plating processes at SAEP. Concentrations of these analytes above deflection points are limited to groundwater beneath the former B-2 Chromium Plating Facility and Buildings B-10 and B-12, and are considered attributable to releases from metals plating activities. Other possible sources of inorganics include leaded gasoline, zinc chromate paint, zinc phosphate, and zincate (zinc coating). However, the distribution of lead above the deflection point is not consistent with release areas. Zinc was detected in the vicinity of the former B-2 Chromium Plating Facility and adjacent to the CWTP transfer line in the South Parking Lot at concentrations greater than the deflection point but concentrations are also above deflection points were also found in the West Parking Lot, near B-19 and the west end of B-16 and dies not appear to be release related.

Releases of contamination to groundwater occurred at the Building B-2 Manufacturing area, the former chrome plating room the Hazardous Waste and Waste Oil Area, the Jet Fuel Tank Farm and the Drum Storage Area located north of Building B-19.

Jet Fuel Tank Farm. In groundwater beneath the former Jet Fuel Tank Farm, VOCs, including PCE, TCE, cis-1,2-DCE, 1,1,1-TCA, 1,1-DCE, vinyl chloride and BTEX were detected at elevations at or above approximately -25 feet, MSL. LNAPL, interpreted to be fuel oil, was detected at a thickness of less than 0.1 feet in monitoring well MW-3, located downgradient and northwest of Building B-34. An area of elevated vinyl chloride concentrations (exceeding I/C VC) is located northeast of Building B-16, in the area of identified LNAPL. BTEX detected in shallow (-5 feet, MSL) groundwater upgradient and downgradient of this contamination may be a result of documented releases from the former Jet Fuel Tank Farm, but also could be related to the area of former ASTs located southeast of Building B-33. LNAPL detected in MW-3 may be the result of leaks in the fuel delivery system used to pump jet fuel to Building B-16. Groundwater flow in this area is to the north, towards the nearby Tidal Flats to the east of the Causeway. Shallow groundwater is believed to discharge to the Tidal Flats in the vicinity of the Dike.

Comparison of groundwater concentrations beneath the Jet Fuel Tank Farm, immediately downgradient, and beneath the Tidal Flats downgradient indicates 1,1,1-TCA, cis-1,2-DCE, TCE, and PCE concentrations generally decrease with distance downgradient of the Jet Fuel Tank Farm, while 1,1-DCE and vinyl chloride concentrations remain relatively stable. This data supports degradation/transformation of PCE, TCE, and 1,1,1-TCA in this area. It is likely that the tidal mixing zone influences groundwater contaminant migration in this area; in general, tidal fluctuations hasten the rate of plume attenuation near the bank of the estuary because of the relatively high advective and dispersive fluxes induced by tides. The ultimate fate of volatile contaminants, including cVOCs and BTEX, is volatilization to the vadose zone and ultimately the atmosphere, and/or volatilization from groundwater discharging to the Tidal Flats.

Former B-2 Chromium Plating Facility. VOCs, cadmium, chromium, copper, cyanide and Cr(VI) were detected in groundwater beneath the former B-2 Chromium Plating Facility.

The cVOCs detected in groundwater include 1,1,1-TCA, 1,1-DCE, PCE, TCE, cis-1,2-DCE, and vinyl chloride. The concentrations of TCE are indicative of the possible presence of a DNAPL (maximum concentration of 830 mg/L); however, visual observation of subsurface soil and groundwater samples did not reveal the presence of TCE DNAPL. The vertical distribution of TCE in groundwater beneath the former Chromium Plating Facility appears to be controlled by the layer of silt and very fine sand, the top of which is at an elevation of approximately –20 feet MSL. TCE has migrated from the plating facility vertically through the unsaturated zone, and into and beneath the water table, to the surface of the silt and very fine sand aquitard. The highest concentration of TCE was detected was 830 mg/L in exploration WP-99-33 at a depth of 26 to 30 feet bgs (approximately –19 to –23 feet MSL), immediately above the surface of the aquitard.

The estimated extent of TCE concentrations in groundwater beneath the former B-2 Chromium Plating Facility exceeding 100 mg/L (I/C VC for TCE is 0.067 mg/L) includes an area greater than 20,000 square feet extending from approximately –13 feet to –28 feet MSL. PCE and 1,1-DCE are also detected at concentrations exceeding CTDEP RSR Criteria. Maximum detected concentrations in this area are 0.120 and 0.240 mg/L, respectively.

The groundwater divide affects shallow groundwater flow beneath the former B-2 Chromium Plating Facility. Groundwater gradients are very flat beneath the building and increase to the north and east. Groundwater beneath the western portion of the former B-2 Chromium Plating Facility flows towards the Tidal Flats and Housatonic River or towards the North Parking Lot, and groundwater beneath the eastern portion flows towards the Housatonic to the east of the Causeway. Migration of TCE has generally occurred towards east of the Causeway; however, limited migration has also occurred southward. The plume associated with cVOCs under the former B-2 Chromium Plating Facility likely joins with the plume located downgradient of Building B-10. Increases in groundwater temperatures observed beneath Building B-2 and the former B-2 Chromium Plating Facility favor natural attenuation. ORP levels rise significantly beneath Building B-10 (likely a result of the oxidized hexavalent chromium), which may account for the lack of significant concentrations of anaerobic degradation daughter products (i.e., 1,2-DCE and vinyl chloride) detected directly downgradient of the former B-2 Chromium Plating Facility. However, ORP levels drop upgradient, beneath Building B-2, where concentrations of cis-1,2-DCE and vinyl chloride have been detected within the zone of elevated groundwater temperature.

There appear to be two distinct Cr(VI) groundwater plumes beneath the former B-2 Chromium Plating Facility. The smaller of the two, located beneath the southeast end, is approximately 80 feet by 130 feet in area (10,400 square feet), with a maximum concentration of 11 mg/L of Cr(VI). The vertical extent is generally limited to between 5 and 40 feet bgs (approximately +2 to –33 feet MSL). The larger plume appears to emanate from the northwestern end of the facility, with a maximum concentration of 950 mg/L of Cr(VI) at a depth of 30 to 34 feet bgs at WP-99-15. This second, larger plume extends 160 feet to the northwest, 135 feet to the northeast, 90 feet to the southeast, and 100 feet to the southwest of this point; covering an area of approximately 40,000 square feet. Vertically, the plume extends from the water table at approximately 2 feet MSL down to -45 feet MSL.

Chromium is found primarily in two oxidation states, Cr(VI) and trivalent chromium (Cr[III]), within the environment. Cr(VI) is relatively mobile, acutely toxic, mutagenic, teratogenic, and carcinogenic. In contrast, Cr(III) has relatively low toxicity and is immobile under moderately alkaline to slightly acidic conditions (Palmer and Puls, 1994).Cr(VI) exists in solution principally as chromate at relatively low concentrations, or as bichromate at high concentrations. Chromate imparts a yellow color to the water when the concentration of Cr(VI) is greater than 1 mg/L. Water that contains high levels of bichromate has an orange color. The Cr(VI) detected in groundwater beneath the former B-2 Chromium Plating Facility imparts a yellow color to the groundwater; this observation, and the concentrations observed, indicate that the Cr(VI) is present primarily as chromate. Cr(III) exists in solution as Cr^{3+} at below pH of 3.5, or as various hydroxyl complexes at increasing pH. At high concentrations, the chromium hydroxyl complex imparts a green color to the solution.

There appears to have been limited migration of Cr(VI) contaminated groundwater at SAEP. The TCE concentrations (greater than 800 mg/L]) may contribute to anaerobic conditions within the groundwater between the two Cr(VI) plumes, reducing the Cr(VI) to the less mobile Cr(III). Migration that has occurred has resulted in the plume extending beneath Buildings B-10 and B-12, in the approximate direction of groundwater flow (north to northeast). The low groundwater flow velocities, coupled with reduction by naturally occurring materials in the soils, limit the horizontal movement of the Cr(VI) groundwater plumes.

Vertical migration of the Cr(VI) plumes appears to have been partially driven by the density of the Cr(VI) plating solution. The extent of vertical migration of the Cr(VI) in groundwater has generally been limited to less than 35 feet bgs (approximately –28 feet MSL). The exception is near the suspected source area in the northern corner of the former B-2 Chromium Plating Facility, where Cr(VI) is detected in groundwater up to a depth of 45 feet bgs (approximately –38 feet MSL, at exploration location WP-99-15). It appears that the assumed relative differences in vertical to horizontal permeability in the fine sands and silts at approximately 30 feet bgs (approximately –23 feet MSL) have impeded vertical movement of the Cr(VI) plume, causing the plume to preferentially spread horizontally.

Modeling of the Cr(VI) plume beneath the former B-2 Chromium Plating Facility indicates that transport toward the Housatonic River is very slow. Assuming a modest partition coefficient of 3 kg/L, the plume has moved little after 50 years. Assuming an even lower partition coefficient of 0.5 kg/L, the plume can reach the mudflat sediments, where it is immobilized by strong adsorption (as a surrogate for chemical reduction and precipitation). Modeling suggests that the strongly anaerobic mudflat facies located between the plume and the river will likely act as a natural, permeable reactive barrier to induce chemical reduction to trivalent chromium and precipitation of sparingly soluble mineral phases, effectively immobilizing chromium.

B-2 Manufacturing Area. Groundwater contaminated by chlorinated solvents has been detected beneath most of Building B-2 Manufacturing Area. Contaminants detected include VOCs, BTEX and Cr(VI), with VOC contamination the most widespread. VOCs in groundwater beneath the Building B-2 Manufacturing Area are likely the result of degreasing operations, which were conducted throughout Building B-2.

The highest concentrations of cVOCs in shallow groundwater extend from the central portion of Building B-2 (location CP-99-08) to an area between Buildings B-2 and B-12 (location WP-99-48). WP-99-48 is within the vicinity of former degreasing pits within Building B-2. Limited explorations exist at the location of the former degreasing pits, but it is likely that this represents a contributing source area. It is possible, due to the limited groundwater velocities beneath Building B-2, that soil stratigraphy has primarily driven source area contaminant migration. The soil stratigraphy beneath Building B-2 dips to the west, likely accounting for the oblong shape of the cVOC plume that also dips to the west.

There is layer of silt, as there is beneath the former B-2 Chromium Plating Facility and between Buildings B-16 and B-48 that would retard the vertical migration of cVOCs. Cone penetrometer logs indicate that the geology is primarily uniform fine sand with some silty sand, and a 10-foot thick gravelly zone is centered vertically at –20 feet MSL. The depth to bedrock in the central portion of Building B-2 varies from approximately –144 feet MSL to –170 feet MSL, dipping from southeast to northwest. Chlorinated VOCs (primarily 1,1,1-TCA) have migrated to the bedrock surface (at approximately –152 feet MSL) as a DNAPL in monitoring well HESE-01-12D. The cVOCs, and other VOCs including BTEX compounds, have migrated relatively unimpeded from the ground surface vertically through the unsaturated zone, into the water table, and to the bedrock surface.

The estimated horizontal extent of 1,1,1-TCA in shallow groundwater at concentrations exceeding 100 mg/L is roughly 350 feet long by 100 feet wide beneath the central part of the building. The vertical extent at exploration WP-99-48 extends from approximately –2 feet to –12 feet MSL. At CP-99-08, the extent is from approximately –22 feet and extends into deep groundwater to –52 feet MSL. The 1,1,1-TCA in deep groundwater extends from –120 feet MSL down to bedrock at –152 feet MSL. The estimated area in deep groundwater exceeding 100 mg/L is 120 feet in diameter.

Also detected in groundwater beneath the B-2 Manufacturing Area are 1,1-DCE, TCE, cis-1,2-DCE, and vinyl chloride, at concentrations exceeding RES and/or I/C VC. The presence of these VOCs is an indication of abiotic degradation of 1,1,1-TCA and the anaerobic degradation of TCE. BTEX was detected in both shallow and deep groundwater beneath the area, and appears to be co-located with 1,1,1-TCA.

Groundwater flow beneath the Building B-2 Manufacturing Area is generally to the north; however, gradients are relatively flat. Horizontal mobility of cVOCs in groundwater at SAEP is governed primarily by advection. Dispersion and retardation work to limit the mobility of a cVOC plume, whereby it is possible that the rate of horizontal migration is less than that predicted by advection. Any sorption of chlorinated VOCs to the soil matrix is typically the result of the presence of organic material. High organic content can retard chlorinated VOC contaminant plume migration.

1,1-DCE in groundwater generally coincides with both the location and relative magnitude of 1,1,1-TCA concentrations. Therefore, the origin of 1,1-DCE in groundwater is presumed to be a degradation product of 1,1,1-TCA, which is capable of degrading abiotically to 1,1-DCE. Migration of 1,1,1-TCA at SAEP appears to be limited by degradation to 1,1-DCE and low groundwater flow velocities.

TCE in groundwater beneath Building B-2 is apparently undergoing anaerobic degradation to the daughter products cis-1,2-DCE and vinyl chloride. The horizontal center of mass of the cis-1,2-DCE plume in deep groundwater appears to be downgradient (toward the Housatonic River) of the center of mass of the TCE plume, and the horizontal center of mass of the vinyl chloride plume appears to be downgradient of the center of mass of the cis-1,2-DCE plume.

The RT3D-GMS simulations of the coupled PCE and TCE natural attenuation suggest that the cVOCs will be degraded and attenuated during their slow transport toward the Housatonic River. The long travel times allow degradation to be a significant factor in the natural attenuation process. Adsorption and low flow rates contribute to the slow rates of transport.

Building B-48 Area. An area of cVOC contamination was identified near the easternmost corner of Building B-48. Contaminants detected in groundwater within this area include the cVOCs PCE, TCE, cis-1,2-DCE, and 1,1-DCE. The concentrations of TCE in this area are indicative of the possible presence of a DNAPL; however, visual observation of subsurface soil and groundwater samples did not reveal the presence of any TCE DNAPL. The estimated extent of TCE in groundwater at concentrations exceeding 100 mg/L includes an area roughly 75 feet in diameter extending from approximately –12 feet to –32 feet MSL. The vertical distribution of TCE in groundwater in this area appears to be controlled by the layer of sandy silt, which extends from approximately –8 feet MSL to –32 feet MSL. The lower vertical permeability of the sandy silt appears to have impeded, to a large extent, the vertical migration of the highest concentrations (>100 mg/L) of TCE in groundwater.

Hazardous Waste and Waste Oil Area. Contaminants detected in groundwater beneath the Hazardous Waste and Waste Oil Area include cVOCs, BTEX compounds, other VOCs, PAHs, SVOCs, and potential Site-related inorganics. VOCs and BTEX were detected in shallow groundwater beneath this area and downgradient in the shallow groundwater of the Tidal Flats adjacent to the Dike. This suggests that VOCs and BTEX detected in soil have leached to the groundwater and are migrating to the Tidal Flats. Concentrations of cis-1,2-DCE, TCE and vinyl chloride exceed RES and/or I/C VC. Vinyl chloride in shallow groundwater at the Tidal Flats likely indicates degradation of TCE in the anaerobic environment of the Tidal Flat sediments. Groundwater flow in this area is to the north, towards the nearby Tidal Flats. It is likely that the tidal mixing zone influences groundwater contaminant migration in this area; in general, tidal fluctuations hasten the rate of plume attenuation near the bank of the estuary because of the relatively high advective and dispersive fluxes induced by tides.

Building B-19. Contamination from this area has migrated to the shallow groundwater of the Tidal Flats adjacent to the Dike; however, the vertical distribution of cVOCs in groundwater in these areas is generally limited to the depths less than -25 feet MSL. Concentrations of PCE, TCE, and 1,1-DCE in groundwater in the vicinity of Building B-19 are greater than SWPC values. Concentrations of TCE, 1,1-DCE, and vinyl chloride in groundwater in the vicinity of Building B-19 exceed their respective Residential and I/C VC.

Groundwater flow in the vicinity of Building B-19 is generally to the east. The presence of cVOCs in groundwater upgradient (west-southwest) from Building B-19 makes the origin of the VOCs in groundwater beneath Building B-19 difficult to determine. However, it is likely that the release north of the building contributed to cVOC groundwater contamination in the vicinity of Building B-19.

The cVOCs present in groundwater east of Building B-19 are likely to continue migrating to the east toward the facility property boundary, but the concentrations of 1,1-DCE, cis-1,2-DCE, and vinyl chloride indicate natural degradation is occurring. 1,1-DCE, cis-1,2-DCE, and vinyl chloride present in groundwater north of Building B-19 and underlying the Tidal Flats, as well as at the eastern end of Building B-19, are indicators of the degradation/transformation of PCE, TCE, and 1,1,1-TCA in that area. It is likely that the tidal mixing zone influences groundwater contaminant migration in this area; in general, tidal fluctuations hasten the rate of plume attenuation near the bank of the estuary because of the relatively high advective and dispersive fluxes induced by tides. The ultimate fate of cVOCs is volatilization to the vadose zone and ultimately the atmosphere, and/or volatilization from groundwater discharging to the Tidal Flats.

Groundwater Contaminant Fate and Transport. Modeling of the coupled PCE/TCE and 1,1,1-TCA attenuation suggests that the cVOCs will be degraded and attenuated during their slow transport toward the Tidal Flats and the Housatonic River. Of the groundwater flow that reaches either the Tidal Flats or the Housatonic River, approximately 96 % of that flow is discharged to the Tidal Flats, leaving about 4 % that is discharged to the river channel. The long travel times allow degradation to be a significant factor in the natural attenuation process. Adsorption and low flow rates contribute to the slow rates of transport.

The SWPC apply to GB classified aquifers that discharge to surface water bodies such as the Housatonic River. VOCs detected in groundwater beneath the site above SWPC include 1,1,1-TCA, 1,1,2-TCA, 1,1-DCE, 1,2-DCA, PCE and TCE. To measure compliance with the SWPC groundwater downgradient of the site was monitored for VOCs in ten piezometers (PZ-TF-01B through PZ-TF-10B). VOC concentrations were several orders of magnitude less than SWPC with concentrations ranging from non-detect to 3.8 mg/L. TCE was detected at a concentration of

0.003 mg/L compared to the SWPC of 2.34 mg/L. BTEX was also below SWPC with concentrations ranging from non-detect to 0.003 mg/L.

A release of hexavalent chromium, cyanide and possibly chromium, cadmium and copper occurred beneath the former B-2 Chromium Plating Facility. The plume extends beneath parts of Buildings B-10 and B-12. This release resulted from metals plating. The Connecticut WQS apply to groundwater that discharges to designated wetlands such as the Tidal Flats. Inorganics in groundwater beneath the former B-2 Chromium Plating Facility and Buildings B-10 and B-12 are found at levels above WQC but the plume is 400 feet upgradient of the tidal flats and there appears to be limited migration of Cr(VI) contaminated groundwater at SAEP. Nickel and Zinc in groundwater are found above Aquatic Life Criteria values, but the concentrations are less than deflection point values indicating these inorganics may not release-related and may be naturally occurring.

10.3 SOIL VAPOR CONTAMINATION

Soil vapor samples were collected at SAEP in 1999 and 2004. The 1999 soil vapor sampling event was conducted for field screening purposes. During the 2004 sampling event, permanent, rather than temporary, monitoring probes were sampled. The 2004 permanent soil vapor points were installed in the northern and southern parts of Building B-2, and in Buildings B-65, B-3, B-3A, B-4, B-16 and B-19 to determine if these areas were in compliance with RSR Soil Vapor criteria.

Results of the 1999 soil vapor survey identified TCE above RSR criteria in the central part of the site including Building B-2 (center), Building B-10 and Building B-12. Concentrations outside of this central area were either at or below volatilization criteria. This correlates with concentrations of TCE that exceed I/C VC in shallow (-5 ft, MSL) groundwater beneath the central portion of the Site, including Buildings B-2, B-10, and B-12.

The results of the 2004 soil vapor survey identified that TCE concentrations were above soil vapor RES and I/C VC throughout Buildings B-2, B-3 and B-3A, and that PCE was above VC in Buildings B-3, B-3A, and B-4. Groundwater concentrations of PCE exceed RES VC in isolated areas beneath Building B-3, B-3A, and B-4, as well as in the alcove between these buildings. Soil vapor concentrations exceed RES VC and I/C VC at one or more locations in Buildings B-2, B-3, B-3A, B-4, and B-6. Groundwater cVOC concentrations exceed RES and I/C VC beneath Buildings B-2, B-3, B-3A, and B-4. There are no monitoring wells beneath Building B-6, however soil vapor data suggests cVOCs are present at elevated concentrations.

Soil vapor cVOC concentrations are highest beneath Building B-2, the Building B-3, B-3A, and B-4 complex, and Buildings B-10 and B-12. With the exception of the southern end of Building B-2, the locations of cVOCs detected in soil vapor are generally coincident with groundwater contamination by the same cVOCs. The other possible source of cVOCs in soil vapor is soil contamination beneath Building B-2.

Soil vapor concentrations are a function of the volatilization of groundwater and soil contamination. Soil vapor contamination resulting from groundwater contamination is governed by the mass transfer of gas (i.e., vapors) from water, driven by a concentration gradient: the difference between the saturation concentration of the gas in liquid and the actual concentration. When the actual concentration is greater than the saturation concentration the gas will volatilize. This mass transfer occurs at the interface between the water, and air, in this case at the water table surface. Therefore, soil vapor concentrations will increase or decrease relative to the groundwater concentrations at the water table.

Measured soil vapor concentrations may vary as a result of soil contaminant levels in the vadose zone, fluctuations of the water table level, soil moisture, soil type and density, soil temperature, this may be effected by utilities such as steam lines present at the Site, surface and subsurface structures, and ambient air pressure. However, if soil vapor data are collected properly, the variability in the measurements (i.e., precision) from day to day is generally less than a factor of two (Hartman, 2002). Therefore, a single soil vapor data set may be misleading relative to determination of soil vapor compliance with RSR criteria. Any determination of soil vapor compliance should rely upon the results of multiple sampling rounds over time.

10.4 Sediment Contamination

Tidal Flats. TCE and 1,2-DCE were detected in Tidal Flats sediments west of the Causeway within 200 feet of the Dike; the source of these cVOCs is likely shallow groundwater discharging to the Tidal Flats from the Dike and the former Hazardous Waste Storage Area. PAHs were detected in Tidal Flats sediments, as well as PCB Aroclor-1248, -1254, and -1260 at concentrations exceeding those in reference locations. Twenty-three TAL metals, trivalent arsenic, and methyl mercury were also detected. However, due to upstream industrial discharges the potential for non-SAEP contaminant sources on the Tidal Flats sediments cannot be dismissed.

Outfall 008. 1,2-DCE and vinyl chloride were detected in sediment samples from the OF-008 Drainage Channel. PAHs were detected; however, the highest concentrations are attributable to a sample collected from a section of the OF-008 Drainage Channel shared with the airport across Main Street. The PCBs Aroclor-1254 and -1260 were detected at concentrations exceeding those in reference location samples. Concentrations of cadmium, chromium, copper, lead, nickel, silver, and zinc detected in sediment samples were all elevated above reference location sample concentrations, indicating that the source of these metals is likely attributable to discharge from OF-008.

11.0 BASELINE HUMAN HEALTH RISK ASSESSMENT

This section provides the technical approach and results of the HHBRA performed in support of the RI for the SAEP in Stratford, Connecticut. The purpose of the risk assessment is to quantify the human health risks associated with potential exposures to site-related constituents under the current and reasonably foreseeable future land use conditions, in the absence of any remedial actions.

This HHBRA is a revision to the Draft RI Report submitted in January 2003 (Harding ESE, 2003). That report was a revision to the Draft Phase III RI/BRA submitted in September 2000 (URSGWC, 2000) and commented on by CTDEP and USEPA. The HHBRA presented in the Draft RI Report submitted in January 2003 (January 2003 Draft HHBRA) incorporated responses to USEPA and CTDEP comments on the September 2000 Draft RI/BRA, analytical data that had been collected during additional RI field activities, and revised assumptions regarding current and potential future land uses of the Site. This revision of the HHBRA incorporates responses to CTDEP (CTDEP 2003; 2004) and Connecticut Department of Public Health (CTDPH) (CTDPH, 2003) comments on the January 2003 Draft HHBRA.

The HHBRA was conducted to characterize health risks at the Site in accordance with USEPA national and Region I CERCLA guidance for risk assessment, using the cumulative excess lifetime cancer risk and hazard index thresholds identified in the NCP and in the CTDEP Remediation Standards Regulations (22a-133k-2(d)(2)). This HHBRA meets both CTDEP Alternative Direct Exposure Criteria and USEPA Superfund risk assessment performance standards. The following USEPA risk assessment guidance and directives were used to complete the HHBRA:

- *Risk Assessment Guidance for Superfund. Volume 1: Human Health Evaluation Manual (Part A)* (RAGS) (USEPA, 1989)
- *Risk Assessment Guidance for Superfund. Volume 1: Human Health Evaluation Manual, Supplemental Guidance: Standard Default Exposure Factors (OSWER Directive 9285.6-03.;* USEPA, 1991a)
- *Guidance for Data Usability in Risk Assessments* (USEPA, 1992a)
- *USEPA Region I Risk Updates, Number 2* (USEPA, 1994b)
- *USEPA Region I Risk Updates, Number 3* (USEPA, 1995)
- *USEPA Region I Risk Updates, Number 4* (USEPA, 1996c)
- *USEPA Region I Risk Updates, Number 5* (USEPA, 1999a)
- *Exposure Factors Handbook* (USEPA, 1997a)
- *Risk Assessment Guidance for Superfund. Volume 1: Human Health Evaluation Manual (Part D)* (RAGS) (USEPA, 2001b)

- *Risk Assessment Guidance for Superfund. Volume 1: Human Health Evaluation Manual (Part E)* (RAGS) (USEPA, 2001c)

- *Calculating Upper Confidence Limits for Exposure Point Concentrations at Hazardous Waste Sites (USEPA, 2002a)*

The HHBRA is organized into four sections (hazard identification, exposure assessment, toxicity assessment, and risk characterization), and has four appendices. The hazard identification presents the site conceptual model, a summary of the analytical data that are used in the HHBRA, and identifies the chemicals selected for evaluation in the risk assessment. The exposure assessment provides information about the activities that may occur under the current and anticipated future land uses of SAEP, identifies the pathways by which people engaged in those activities could be exposed to chemical of potential concern (COPCs) at the Site, and quantifies the exposures associated with those pathways. The toxicity assessment provides information about the potential toxicity and dose-response profiles of the COPCs. The risk characterization combines the dose-response information and quantitative exposure estimates to provide quantitative estimates of risk for cancer and systemic toxic effects. In order to provide additional perspective for risk management decision-making, this section also contains an uncertainty analysis of the variables that lend the greatest uncertainty and have the greatest potential effect on the quantitative risk estimates. Appendix S-1 presents lists of the samples that were included in the various data sets evaluated in the HHBRA, Appendix S-2 contains short toxicity profiles, Appendix S-3 contains information to support the exposure assessment (95% UCL calculations), and Appendix -S-4 provides documentation of the risk estimate calculations for the Site.

<u>USEPA RAGS Part D Reporting Requirements.</u> This HHBRA was prepared consistent with the standardized table formatting requirements provided in RAGS Part D (Final) (USEPA, 2001b). The tables formatted per RAGS Part D are presented in conjunction with non-RAGS Part D tables throughout Section 11 and in Appendix S-4. RAGS Part D standardized tables are presented in this HHBRA as follows:

RAGS D Standardized Table …	***Included in this HHBRA as …***
1 – Selection of Exposure Pathways	11-11 – Selection of Exposure Pathways
2.1 – 2.*n* – Occurrence, Distribution, and Selection of COPCs	11-2 – 11-9 - Occurrence, Distribution, and Selection of COPCs [Various media]
3.1 – 3.*n* – Medium-Specific Exposure Point Concentration Summary	11-17 – 11-29 - Medium-Specific Exposure Point Concentration Summary [Various exposure points]
4 – Values used for Daily Intake Calculations	11-13 – 11-16 – Values Used for Daily Intake Calculations
5.1, 5.2 – Non-Cancer Toxicity Data	11-31 and 11-33 – Non-Cancer Toxicity Data
5.3 – Non-Cancer Toxicity Data – Special Case Chemicals	Not applicable
6.1, 6.2 – Cancer Toxicity Data	11-30 and 11-32 – Cancer Toxicity Data
6.3 – Cancer Toxicity Data – Special Case Chemicals	Not applicable

RAGS D Standardized Table …	*Included in this HHBRA as …*
6.4 – Cancer Toxicity Data – Radionuclides	Not applicable
7.1 – 7.*n* – Calculation of Cancer Risk and Non-Cancer Hazards	Component of Risk Calculations in Appendix S
8.1 – 8.*n* – Calculation of Radiation Risks	Not Applicable
9 – Summary of Receptor Hazards for COPCs	Component of Risk Calculations in Appendix S
10 –Risk Assessment Summary	Tables 11-34 - 11-40

11.1 HAZARD IDENTIFICATION

The hazard identification consists of data evaluation, compilation of data sets for the HHBRA, and selection of COPCs. In addition, the hazard identification provides a summary of the possible adverse health effects that may be associated with exposure to the COPCs. A description of the Site and presentation of the site conceptual model are also presented in this section to provide information fundamental to the risk assessment.

11.1.1 Site Description and Conceptual Model

The SAEP is an industrial manufacturing facility that was used between 1929 and 1998 to manufacture aircraft, turbine engines, and related components. The facility consists of 49 buildings and is located on 126 acres adjacent to the tidally-influenced portion of the Housatonic River, approximately 1 mile upstream of Long Island Sound (Figure 1-1).

The portion of the facility containing all of the buildings (and historically all of the manufacturing operations) and located between Main Street, Sniffens Lane, and the Housatonic River is referred to as the *Main Site*. All portions of the Main Site (with the exception of the landscaped area between Building B-2 and Main Street) are paved and enclosed with security fencing. The former *Chromium Plating Facility*, comprised of an extension to Building B-2 that was formerly used for chromium plating operations, is within the Main Site. The portion of the facility on the south side of Sniffens Lane, containing a large parking lot (South Lot), Building B-6, and the former wastewater treatment building, is referred to as the *South Lot Area*.

A manmade Causeway and Dike are located at the portion of the Main Site that borders the Housatonic River.

As described below, there are three areas where surface water and sediment may have been affected by releases from the facility. These include the *Intertidal Flats, Outfall 008 Drainage,* and the *Marine Basin.* The Intertidal Flats are located in the Housatonic River adjacent to the Dike; the Causeway extends into the Intertidal Flats, essentially dividing it into two areas. Outfall 008 Drainage is the former outfall from the chemical treatment building. Outfall 008 Drainage discharged to a ditch that flowed to the Marine Basin. The Marine Basin is an tidally-influenced basin located on the Housatonic River approximately 1,000 feet southeast of the Main Site.

In 1998, manufacturing operations at SAEP were discontinued, employees were released, and the majority of manufacturing equipment was sold at auction. Consequently, the majority of buildings at SAEP are vacant (i.e., contain no equipment, furniture, etc.). However, minimal maintenance of the facility, including 24-hour security, is on-going and is planned to continue until the facility is transferred to another owner. A small number of employees staff the facility daily. The employees consist of security personnel, maintenance workers, and U.S. Government administrators who are overseeing the environmental Site closure and transfer of the property.

Although use of the Housatonic River for recreational and commercial boating occurs in the vicinity of the Facility, these activities do not occur at the Intertidal Flats or Marine Basin. Commercial and recreational angling and shellfish harvesting historically occurred in the vicinity. The Intertidal Flats have no specific use under the existing land use conditions. The Marine Basin is located next to a public park, and adjacent to a landfill that is owned by the Town of Stratford. Although the Marine Basin is posted with a "No Swimming" sign, it is possible that recreational visitation occurs at the Marine Basin. No activities occur in the vicinity of the Outfall 008 Drainage ditch, and access to the portion of the ditch near the Outfall 008 is difficult due to fencing and vegetation, but becomes less restrictive near the Marine Basin.

The Army plans to transfer the facility to the Town of Stratford for continued commercial/industrial use. Commercial and recreational angling and shellfishing are assumed to occur in the future. Areas of the Site evaluated in this risk assessment are assumed to be accessible in the future to people who use the Housatonic River and surrounding parks and open space for recreational purposes. Future use of the facility for residential development is considered implausible; the HHBRA incorporates the assumption that a land use restriction will be in place to prevent use of the Site for residential development without prior evaluation of such use.

During the manufacturing history, constituents were released to the environment through several mechanisms:

- Chlorinated solvents (e.g., PCE, TCE, and 1,1,1-TCA) used for degreasing and cleaning metal components were released (i.e., spilled or otherwise disposed of) from operations, primarily in Building B-2. Solvents migrated to soil beneath the building floor (e.g., via leaks in floor drain systems) to shallow groundwater. Groundwater flow is toward the Housatonic River; however, the rate of flow is extremely slow. The RI has identified three chlorinated solvent hot-spots, two comprised primarily of TCE and breakdown products and the other comprised primarily of 1,1,1-TCA and breakdown products. Chlorinated solvents have been detected in air samples collected from within buildings that overlie the groundwater plumes, indicating that vapor migration from groundwater to indoor air is occurring. DNAPL and elevated concentrations of chlorinated solvents have also been identified in deep groundwater, indicating that vertical migration of the plumes has occurred.
- Solutions containing chromium, including chromium in the hexavalent oxidation state, used for metal plating were released (i.e., spilled or otherwise disposed of) from the former Chromium Plating Facility (attached to Building B-2). Chromium migrated to soil beneath the building floor (e.g., via leaks in floor drain systems) to shallow groundwater. The RI has identified elevated concentrations of chromium in soil and a Cr(VI) groundwater plume extending approximately 150 feet downgradient from the former Chromium Plating Facility.

- Contaminated soil from the Main Site area was placed in the South Parking Lot..
- Site operations also resulted in waste streams potentially containing nickel, copper, cadmium, aluminum, magnesium, zinc, and cyanide. Additionally, manometers containing mercury were used at the engine testing facilities. These sources could potentially have resulted in releases to environmental media associated with the Site.
- Liquid wastes from various industrial operations were transported to the chemical treatment plant; treated wastes were then released to a drainage ditch that discharged through Outfall 008 Drainage to a ditch that emptied into the Marine Basin. Prior to treatment at the CWTP, wastes were discharged directly to the Intertidal Flats through various outfalls. Some constituents that may have been present in wastes from SAEP (e.g., some inorganics) may bioaccumulate in biota (fish and shellfish) that occur at the Intertidal Flats and Marine Basin areas.
- Stormwater that is collected from the SAEP is discharged to the Intertidal Flats.

In addition, chlorinated solvents have been detected in shallow groundwater beneath and within the sediment pore space of the Intertidal Flats. Since the groundwater plumes originating at Building B-2 are not discharging to surface water, the source of chlorinated solvent contamination at the Intertidal Flats is suspected of being related to discrete releases (historic disposal) of solvent waste in the vicinity of the Dike area.

Based on this summary of the site conceptual model and the results of the RI sampling, the following media are considered to be affected by releases at the SAEP:

- Soil at the Main Site and South Lot Areas
- Groundwater at the Main Site
- Indoor air
- Surface water and sediment at the Intertidal Flats
- Biota at the Intertidal Flats
- Surface water and sediment at the Outfall 008 Drainage Area
- Surface water and sediment at the Marine Basin
- Biota at the Marine Basin

Figure 11-1 provides a graphical representation of the site conceptual model. The potential human exposure pathways to each of the media identified in Figure 11-1 and are summarized below.

Due to the security at SAEP, there are a limited number of people who may gain access to the Facility under the existing land use conditions. These include security personnel, authorized visitors, and employees (maintenance workers and administrative staff). These employees work in buildings that are located over the groundwater plumes, and in which VOCs have been detected in indoor air. Therefore, workers presently at the facility may be exposed by inhalation to VOCs that have migrated from groundwater to indoor air.

Since the Main Site and South Lot areas are paved, and the Dike is covered with pavement and rip-rap, there is no exposed soil within boundaries of the Site (the landscaped area between Building B-2 and Main Street is unpaved; however, no documented releases have occurred in this area and no soil samples have been collected). However, facility maintenance workers perform the majority of repair and maintenance work at the facility, including repair of underground utilities (e.g., underground piping for steam heat). Therefore, under the existing land use conditions, facility maintenance workers may contact soil at the Site. Under future land use conditions, if existing pavement is removed soil would be accessible for exposures by receptors associated with the future commercial/industrial use of the property. Potential exposures to chemicals in soil may occur through dermal contact and incidental ingestion. In addition, soil that is agitated (e.g., during excavation work) may release dust that could be inhaled by persons in close proximity.

The groundwater beneath and in the vicinity of SAEP is not used as a source of potable water; SAEP obtains potable water from the Bridgeport Hydraulics municipal supply. The groundwater is not considered a resource that requires protection as a potable water source (i.e., is classified GB by the State of Connecticut). Therefore, potable use of groundwater is not a potentially complete exposure pathway. However, the groundwater at the facility contains VOCs that may migrate to indoor air. Therefore, inhalation of VOCs that migrate from groundwater by future building occupants is a potentially complete exposure pathway.

Potential exposures to surface water and sediment that have potentially been affected by releases from the Site are unlikely under the existing land use. There are no open space or beach areas at the Intertidal Flats, and the area is in an industrialized setting. However, the portion of the Housatonic River in the vicinity of SAEP is used for recreational boating, and there are no restrictions preventing access to the Intertidal Flats, Marine Basin, or Outfall 008 Drainage. It is assumed that these areas may be accessed by recreational visitors in the future. Under such use, recreational visitors could contact the surface water and sediment during wading or recreational angling or shellfishing. However, the substrate at the Intertidal Flats is deep mud that cannot be walked upon without sinking at least two feet into the mud. Therefore, access to this area by children is highly unlikely because the substrate presents a physical hazard. Exposure pathways related to surface water and sediment are evaluated in this HHBRA.

Some chemicals, particularly PCBs, and some inorganics, may bioaccumulate in fish and shellfish that are exposed to those chemicals in sediment and surface water. Humans who ingest fish and shellfish that have bioaccumulated the chemicals may in turn be exposed. Historically, commercial and recreational angling and shellfishing have occurred in the lower Housatonic River. Commercial finfish angling now occurs (and has always occurred) in Long Island Sound. However, these activities are assumed to potentially occur in the future, and consumption of biota is evaluated as a potentially complete exposure pathway under future land use conditions in this HHBRA.

11.1.2 Data Evaluation

The purpose of this section is to provide a summary of the methods and results of the data collection and evaluation activities that were performed to provide the analytical data used in the quantitative risk assessments. A complete discussion of these activities is presented in Sections 2.0 through 7.0. The product of these activities, discussed in this section, is a series of media- and site-specific data summaries that will be used for identifying COPCs and deriving exposure point concentrations. The activities involved in the data evaluation process include identification of data sources, evaluation of data quality, and compilation of data sets for the quantitative risk assessment.

11.1.2.1 Data Sources The analytical data evaluated in the risk assessments come from remedial investigation activities performed by various contractors during 1992, 1994, 1998, 1999, 2000, 2001, and 2002, and indoor air monitoring from 1999 to 2002. The specific sources of the analytical data used to evaluate the Site are listed in Table 11-1.

The environmental media evaluated in the HHBRA are reviewed in Subsection 11.1.2.3. The sample collection and analytical methods used for each medium are discussed in detail in Sections 2.0 through 7.0.

11.1.2.2 Data Quality. The analytical data evaluated in the RI were reviewed for adequacy for use in risk assessment according to criteria established by USEPA (1992a). The data quality evaluation is discussed in Subsection 9.1; the following paragraphs summarize the results of the data quality evaluation as it applies to the data used in the risk assessment.

Sort Data by Medium. Data from the numerous historical investigations were compiled and sorted by environmental medium. The following table summarizes the media that were analyzed in the RI programs:

Summary of Study Areas and Media Evaluated in RI

Study Area	Sediment	Surface Water	Finfish	Shell-fish	Soil	Ground-water	Soil Gas	Indoor Air
Intertidal Flats	X	X	X	X		X		
Marine Basin	X	X	X	X				
Outfall 008 Drainage	X	X						
Main Site					X	X	X	X
Former Chromium Plating Facility					X	X	X	
South Lot Area					X	X		X

The HHBRA evaluates potential exposures and health risks for exposure areas where soil, indoor air, surface water, sediment, and biota data were collected, as summarized in the following table:

Summary of Exposure Areas / Exposure Media Evaluated in HHBRA

Exposure Area	Sediment	Surface Water	Finfish	Shell-fish	Soil	Ground-water	Indoor Air
Main Site					X		X
Former Chromium Plating Facility					X		
South Lot Area					X		X
Intertidal Flats	X	X	X	X		X	
Marine Basin	X	X	X	X			
Outfall 008 Drainage Area	X	X					

The former Chromium Plating Facility is retained as a separate area within the Main Site (Figure 12-2) because this area is associated with elevated chromium concentrations and has a greater sampling density than the remainder of the Site.

Groundwater at the Intertidal Flats is included in the HHBRA because the groundwater discharges through sediment, and exposure to surface water and sediment at the Intertidal Flats would result in potential exposure to groundwater as well. There are no direct contact exposures to groundwater at other areas of the Site.

However, both groundwater and soil may a source of vapors to indoor air via release of VOCs from soil and groundwater to soil gas, and subsequent migration of soil gas to air within structures at the ground surface. Soil gas is not an exposure medium, but rather a source medium to indoor air. Since directly measured indoor air concentration data are available, it is not necessary to use soil gas, soil, or groundwater data to model potential indoor air concentrations.

Evaluate the Analytical Methods. The analytical data used in the risk assessments are from laboratory analyses performed under the USEPA Contract Laboratory Program (CLP) and QA/QC procedures (Level II or III data). All analytical data were subjected to data validation, as described in Subsection 9.1. The analytical data selected for use in the risk assessment are identified in Subsection 11.1.2.3.

Evaluate Quantitation Limits. For the data used in the risk assessments, as described in Section 2.0 through 7.0, the reporting limit for organic analytes is the sample quantitation limit (SQL), whereas the reporting limit for inorganic analytes is the contract required detection limit (CRDL). The quantitation limits associated with the data sets evaluated in the risk assessment were reviewed to determine if concentrations that could pose a concern from a risk or regulatory criteria perspective could be detected.

In most data sets there is at least one chemical, but often several chemicals, which have SQLs that exceed risk-based values or ARAR/information to be considered (TBC) values (i.e., SQLs that are elevated). This indicates that concentrations that could pose a concern from a risk or regulatory perspective could not be quantified in those samples. However, in developing exposure point concentrations, one-half the non-detect values are used as the concentration representative of the sample location. Therefore, non-detect values that are elevated with respect to risk-based value or ARAR/TBC are still included in the risk assessment data sets and are evaluated for potential exposures and risks. The principal uncertainty associated with elevated detection limits concerns analytes with reported concentrations that are lower than the risk-based value used in the COPC selection process, but SQLs that are greater than the risk-based value. This circumstance results in exclusion of the analyte as a COPC, when it could be present at a concentration that would include it as a COPC. These uncertainties and their potential bearing on the results of the risk assessment are evaluated in the risk assessment uncertainty section.

Evaluate Qualified and Coded Data. Both the laboratory and data validators may assign qualifiers to analytical results. The qualifiers assigned during the data quality review supersede those assigned by the laboratory. The results of the data quality review are discussed in Appendix C and the validated data, with qualifiers, are presented in Appendices O, P, Q, R, and V.

For the risk assessment, all analytical results assigned qualifiers indicating that the analyte is positively detected were retained as detected results in the risk assessment data sets. All results assigned qualifiers indicating that the analyte is not positively detected were retained as non-detected results in the risk assessment data sets.

Compare Concentrations Detected in Samples to Concentrations Detected in Blanks. As discussed in Appendix C, constituent concentrations in samples collected at the Site have been compared to concentrations in associated blanks to distinguish artifacts from the actual presence of the constituents in samples collected from the sites. These comparisons were conducted as part of the data validation process, which has been previously discussed.

The following bullets summarize the analytes detected in field and laboratory blanks by field investigation program and media:

- 1998-1999 Phase III RI Soils: TCE, acetone, 2-butanone, dichloromethane, bis(2-ethylhexyl)phthalate, butylbenzyl phthalate, di-n-octyl phthalate, diethyl phthalate, and di-n-butyl phthalate.
- 2002 Supplemental RI Soils: Acetone, 1,2-dichloroethane, toluene, and xylene (all at concentrations between 0.001 and 0.004 mg/L).
- 1999 Phase III RI Groundwater: Diethyl phthalate, dimethyl phthalate, and bis(2-ethylhexyl)phthalate.
- 2002 Supplemental RI Groundwater: 1,1,1-TCA, 1,1-DCE, acetone, benzene, cis-1,2,DCE, ethyl benzene, m,p-xylene, methylene chloride, styrene, PCE, toluene, trans-1,2-DCE, TCE, and vinyl chloride (concentrations of all blank contaminants were less than 0.002 mg/L – see Appendix C-3).
- 1992-1999 RI Sediment: Dichloromethane.
- 1992-1999 RI Surface Water: Acetone and dichloromethane.

Laboratory analytical validation summary reports for the Phase III RI, NAE Intertidal flats piezometer groundwater sampling, and the Supplemental 2002 RI analytical data are presented in Appendix C. Analytical validation reports for previous investigations can be found in associated reports.

Comparison with Background. Analytical data for samples collected at Site were compared with analytical data for samples collected from locations that represent background, or ambient conditions. The comparison was performed to help interpret which constituents may be present in site media at concentrations that are inconsistent with background concentrations. Following USEPA Region I guidance (USEPA, 1999a), comparison of site data to background data was not used in the COPC selection process for the human health risk assessment. However, site data were compared to background data in the risk characterization to help place possible risks associated with the sites in perspective with possible risks associated with background conditions.

Background data are available for surface water, sediment, biota, and indoor air; no background data are available for soil and groundwater. The samples included in the background data sets are identified in Subsection 11.1.2.3. Background data summaries are provided in Appendix S-1.

11.1.2.3 Data Compilation. The ultimate product of data evaluation and data summarization is a set of analytical data in a form that can be used in the quantitative risk assessment. Each data set developed for the risk assessment was summarized to provide the following statistical descriptors:

- the ratio of the number of samples in which the constituent is detected to the total number of samples (i.e., frequency of detection);
- range of analytical quantitation limits;
- range of detected concentrations;
- data qualifiers associated with the minimum and maximum detected concentrations;
- sample identifier associated with the maximum detected concentration; and
- arithmetic mean concentration.

The following procedures were applied when summarizing the analytical data for the risk assessment:

- Data that were rejected during validation (i.e., data with an “R” qualifier) were not included in the risk assessment data sets.
- Field duplicate samples were considered quality control samples and were not used in the risk assessment. Therefore, only the analytical results for the “original” sample in the duplicate pair were included in the data sets (i.e., samples with the “D” designation in the sample ID were not included in the data sets). Exclusion of the duplicate results from the risk assessment data sets does not have a bearing on the results or conclusions of the risk assessment because duplicate and original sample results must be consistent to comply with quality control standards. Results that show disparity beyond quality control limits are addressed through the data validation process.
- For samples in which analyte concentrations were detected outside the calibration range, the samples were diluted and reanalyzed, and two analytical results were available. For analytes with results that exceeded the calibration range in the original analysis, only the reanalysis results were used in the risk assessment.

- When calculating the arithmetic mean concentrations, one-half the value reported as the non-detect value (usually the analytical quantitation limit) was used for results reported as not-detected.
- Inorganics with data for total and dissolved concentrations were summarized separately.

Soil

Over 550 soil samples were collected from locations throughout the Site during the RI field investigations. As shown in Table 11-1, the soil analyses varied among RI field investigations; target analytes were selected according to each of the investigation objectives. Therefore, analytical parameters do not have equal numbers of analyses.

Due to administrative issues, only analytical results that represent a positive detection (i.e., unqualified and “J” qualified results) are available for soil samples collected during the 1992 Phase I RI field investigation. However, only 22 soil samples were collected during the 1992 Phase I RI field investigation. The majority (greater than 95%) of all soil samples were collected during the 1998 Phase III RI field investigation. For these samples, all data (i.e., hits, non-detects, rejected data) were available. The absence of data for non-detects from the 1992 Phase I RI sampling introduces a high bias to statistics that are calculated from the data set and, therefore, contributes an uncertainty that is likely to overestimate rather than underestimate potential exposure point concentrations.

A number of analytical results were associated with elevated detection limits. For some chemicals, SQLs (i.e., non-detect “U” qualified data) were more than one order of magnitude greater than the maximum detected concentrations. Use of data sets with highly elevated SQLs typically results in calculated arithmetic mean concentrations and 95% upper confidence level (UCL) values that are greater than the maximum detected concentrations. Values that are greater than the maximum detected concentrations cannot be used as exposure point concentrations (EPCs), thus limiting the statistic that may be used as the EPC to the maximum detected concentration. Use of the maximum detected concentration as the EPC adds uncertainty to the risk assessment because the maximum detected concentration is generally not representative of potential exposure conditions at an exposure point. To reduce this uncertainty, sample results reported as non-detect with an SQL more than twice the maximum detected concentration were excluded from the data sets used in the risk assessment. An SQL of twice the maximum-detected concentration was selected as the cutoff value because one-half the non-detect values is used in risk assessment statistics (e.g., arithmetic mean and 95% UCL calculations). Therefore, a result reported as non-detect at two-times the maximum detected concentration will be included in the risk assessment statistics as a value that is equal to (but does not exceed) the maximum detected concentration. For each parameter, the number of analytical results excluded from the risk assessment due to elevated detection limits is summarized in Appendix S-1.

USEPA Region I risk assessment guidance indicates that surface soil is soil that is between 0 and 1 foot bgs (USEPA, 1995). The State of Connecticut regulations consider soil to be “accessible” when it is within 2 feet of ground surface and beneath pavement, or when it is within 4 feet of ground surface and not beneath pavement (CTDEP, 1996). Since all of the soil at the Site is beneath pavement, this risk assessment defines surface soil as soils between 0 and 2 feet bgs (i.e., soil samples with a starting depth interval less than 2 feet bgs). In accordance with USEPA Region I risk assessment guidance, human exposures to soils deeper than 10 feet bgs are not evaluated unless site-specific circumstances indicate that exposures may reasonably be expected. There are no site-specific attributes (e.g., depth to utilities) that would suggest that soil deeper than 10 feet bgs would be contacted. However, the State of Connecticut regulations consider soil within 15 of the ground surface to be "potentially accessible" (CTDEP, 1996). Therefore, this risk assessment defines subsurface soil as soils between 2 and 15 feet bgs (i.e., soil samples with a starting depth interval 2 feet bgs or deeper, but less than 15 ft bgs).

Soil data were segregated into three exposure areas: Main Site; Former Chromium Plating Facility; and South Lot Area. Soil data associated with the Causeway are not used in the HHBRA because the remediation completed at the Causeway prevents soil exposure. The areas, as well as sample locations within each of the areas, are shown in Figure 12-2. The samples included in each of the exposure areas are listed in Appendix S-1. Soil data are summarized in Tables 11-2 (surficial soil) and 11-3 (subsurface soil).

Groundwater

The RI field investigations used a combination of direct-push sampling and monitoring well sampling to identify the nature and extent of the groundwater plumes and monitor concentrations within the plumes. As shown in Table 11-1, the groundwater analyses varied among RI field investigations; target analytes were selected according to each of the investigation objectives. Therefore, analytical parameters do not have equal numbers of analyses. Groundwater samples collected at the Intertidal Flats were analyzed for VOCs only, as the objective of the sampling at those areas was to identify the nature and extent of VOC contamination.

Because groundwater concentrations have changed over the past decade in which RI field investigations have been occurring, the following approach was used to select groundwater data for the risk assessment:

- For VOCs, data from the May 2002 Site-wide monitoring well sampling event and from the direct-push sampling between 1999 and 2001 were selected.

This approach ensures that the risk assessment is based on the data that best describe the existing groundwater conditions at the Site.

Human receptors that may directly contact groundwater would only potentially contact shallow groundwater. Specifically, shellfishers at the Intertidal Flats who may potentially contact groundwater within the sediment pore space would only be exposed to shallow groundwater within two feet of the sediment surface. Therefore, the groundwater depth evaluated in the risk assessment is 0-2 ft bgs beneath the Intertidal Flats. The groundwater sample locations at the Intertidal Flats is shown in Figure 11-2. The samples included in the exposure area are listed in Appendix S-1. Groundwater data are summarized in Table 11-4.

Surface Water and Sediment

Surface water and sediment samples were collected from the Intertidal Flats, Marine Basin, and the Outfall 008 Drainage ditch during RI activities in 1992, 1994, and 1999. As shown in Table 11-1, the analyses varied among the RI field investigations; target analytes were selected according to each of the investigation objectives. Therefore, analytical parameters do not have equal numbers of analyses.

The following data were available and used in the risk assessment:

- 1992 Phase I RI: Data were available for analytical results that represent a positive detection (i.e., unqualified and "J" qualified results) for chemicals previously selected as COPCs in Phase I RI risk assessment (W-C, 1993)
- 1994 Phase II RI: Data were available for analytical results that represent a positive detection (i.e., unqualified and "J" qualified results). However, for transect samples collected from the Intertidal Flats, data for positive detections and non-detects (i.e., "U" qualified data) were available if an analyte was detected in at least one sample in a given transect. If an analyte was not detected from any samples within a given transect, data for that analyte were not available for any of the samples within the transect.
- 1999 Phase III RI: Data were available for all analytical results (i.e.,. positive detections and non-detects).

The absence of data for non-detects from the 1992 Phase I and 1994 Phase II RI sampling introduces a high bias to statistics that are calculated from the data set and, therefore, contributes an uncertainty that is likely to overestimate rather than underestimate potential exposure point concentrations.

Sediment data from 0-2 ft bgs were used in the HHBRA. Although potential exposures to sediment are not likely to include sediment deeper than 0.5 ft bgs, the depth interval of 0-2 ft bgs accounts for the possibility that shallow sediments shift with Intertidal cycles and storm events.

Surface water samples at each location at the Marine Basin and the Outfall 008 Drainage ditch were collected during periods of high tide and low tide. Samples collected at high tide and low tide at each location were not averaged together to represent a single result for the location; each sample was included as a unique data point in the data set.

Surface water and sediment data were segregated into three exposure areas: Intertidal Flats; Marine Basin; and Outfall 008 Drainage area. The areas, as well as sample locations within each of the areas, are shown in Figure 11-3. The samples included in each of the exposure areas are listed in Appendix S-1. Surface water and sediment data are summarized in Tables 11-5 and 11-6, respectively.

Surface water and sediment samples were collected from five background reference locations in the lower Housatonic River, as shown in Figure 4-2. Background samples were analyzed for the same parameters as samples collected at the Site. The background values presented in Tables 11-5 and 11-6 were identified using the same methodology that was applied to identify EPCs for the Site data (see subsection 11.2.3).

Biota

Oysters were collected from the Intertidal Flats during the 1994 Phase II RI, and ribbed mussels and finfish (Striped Bass) were collected from the Intertidal Flats and Marine Basin during the 1999 RI. Shellfish were collected from the same locations as sediment samples. Whole body analyses of shellfish (ribbed mussels and oysters) were performed for PCBs and inorganics. Striped bass were collected by line and lure within the Marine Basin and within the area of the Intertidal Flats where surface water and sediment samples were collected. Fillet samples from finfish were submitted for PCB and inorganics analyses (Table 11-1). Finfish and shellfish were segregated into two exposure areas: Marine Basin and Intertidal Flats. Ribbed mussel and oyster data were evaluated separately due to differences in human exposure potential (i.e., people may eat only one type of shellfish). Finfish data are summarized in Table 11-7 and shellfish data are summarized in Table 11-8.

Four finfish samples, three oyster samples, and five ribbed mussel samples were collected from background reference locations in the lower Housatonic River. Background samples were analyzed for the same parameters as samples collected at the Site. The background values presented in Tables 11-7 and 11-8 were identified using the same methodology that was applied to identify EPCs for the Site data (see subsection 11.2.3).

Indoor Air

Sampling of indoor air has occurred in 10 buildings at SAEP that are above VOC plume areas. ampling was conducted between September, 1999and July 2002 and 27 rounds of indoor air samples were collected. Appendix S-1 includes a summary of the indoor air sampling program at SAEP. Not all locations have been sampled throughout the course of air monitoring program. Reasons for including or excluding samples from specific rounds of sampling have been detailed in monthly air monitoring summary reports, but are generally driven by whether there is a potential for human exposure at a given location/building, and whether there have been any substantial trends in analyte concentrations.

Indoor air samples have been analyzed for six site-related chlorinated VOCs: 1,1,1-TCA, 1,1-DCE, 1,2-DCE (cis and trans isomers), PCE, TCE, and VC.

The data set evaluated in the HHBRA is comprised of all samples collected over the 27 rounds of sampling performed through 2002. Samples included in specific exposure points are identified in Subsection 11.2. The indoor air data are summarized in Table 11-9 and sample locations are shown in Figure 4-2.

The background sample location that has been used to evaluate indoor air quality data (location IA-BKGD-07) was collected outdoors, between Buildings B-12, B-44, and B-48, as shown in Figure 4-2.

11.1.3 Chemicals of Potential Concern

Chemicals of potential concern are chemicals that may pose more than a *de minimus* health risk. USEPA Region I does not favor the use of a comparison of site concentrations to background concentrations to exclude chemicals that are detected at concentrations consistent with ambient conditions (USEPA, 1995; 1999a). Therefore, the COPCs selected in the risk assessment may include chemicals that were detected at concentrations consistent with naturally occurring or ambient concentrations. If any COPCs that are consistent with background are determined to be substantial risk contributors, they are further evaluated in the risk characterization (subsection 11.4).

The procedure used to select COPCs for the HHBRA is summarized as follows, and is consistent with USEPA Region I (USEPA, 1999a) methodology:

1) *Comparison to Available Criteria*

   - Selected as a COPC in soil if the maximum detected concentration exceeds the USEPA Region IX PRG for residential soils (USEPA, 2002b) or the CTDEP RSR RES DEC.
   - Selected as a COPC in groundwater if the maximum detected concentration exceeds the USEPA Region IX PRG for tapwater (USEPA, 2002b) or the CTDEP SWPC (CTDEP, 1996).
   - Selected as a COPC in surface water if the maximum detected concentration exceeds the USEPA Region IX PRG for tapwater (USEPA, 2002b) or the USEPA Ambient Water Quality Criterion (AWQC) for the protection of humans who may ingest surface water and/or organisms (USEPA, 1991b)
   - Selected as a COPC in sediments if the maximum detected concentration exceeds the USEPA Region IX PRG for residential soils (USEPA, 2002b) or the CTDEP RSR RES DEC.
   - Selected as a COPC in biota (finfish and shellfish) if the maximum detected concentration exceeds the USEPA Region III risk-based concentration (RBC) for consumption of fish (USEPA, 2004a).
   - Selected as a COPC in indoor air if the maximum detected concentration exceeds the USEPA Region IX PRG for ambient air (USEPA, 2002b) or the CTDEP I/C IATC.

The preliminary remediation goals (PRGs) are protective for direct contact (ingestion and dermal contact) exposures, as well as for inhalation of constituents that may be released to air. The PRGs and RBCs are derived for a $1x10^{-6}$ cancer risk level or a non-cancer HQ of 1. Per USEPA Region I guidance (USEPA, 1995), the PRGs and RBCs based on noncarcinogenic effects have been adjusted to represent a HQ of 0.1 for the purposes of COPC selection.

Potential exposures to groundwater would be associated with substantially lower contact intensity than is assumed in the tapwater PRGs. Therefore, the use of tapwater PRGs to identify COPCs in groundwater represents an extremely conservative approach because the groundwater is not used as source of potable water and is not considered a water resource that requires protection as potable water. However, PRGs or risk-based screening levels for non-potable use exposures to water are not published. The use of tapwater PRGs to identify COPCs in surface water also represents a very conservative approach, since potential exposures to surface water would involve only incidental ingestion of water (the Housatonic River in the area near the Site is brackish and is not used as a source of potable water). The use of residential PRGs for selection of COPCs in sediments represents a conservative approach, since potential exposures to this medium will not occur at the frequency or intensity that would be associated with residential land use contact with surface soil.

2) *Professional Judgment*

   a) Specific Contamination Source at the Site:

- Despite other criteria, <u>selected</u> as a COPC if there is a known source area for a chemical at the Site. Based on this criterion, chlorinated solvents and their breakdown products and chromium, if detected, were retained as COPCs. Common laboratory chemicals/contaminants, including methylene chloride and chloroform, were not included as "site related" chemicals in the application of this criterion.

b) Essential Nutrients:

- <u>Eliminated</u> as COPCs because they are considered essential human nutrients. The following inorganic analytes are considered essential human nutrients: calcium, magnesium, iron, potassium, and sodium.

c) Low Frequency of Detection:

- Despite other criteria, <u>eliminated</u> as a COPC because the analyte was detected at a very low frequency (fewer than 5% of the samples; USEPA, 1989), and there is no known source or documented release of the chemical at the Site.

The following notes are used to denote the reasons for selection or exclusion of analytes as a COPCs:

ASL: The concentration used for COPC screening (the maximum detected concentration) is greater than the risk-based concentration (PRG) and/or ARAR; the analyte is therefore selected as a COPC. Analytes for which no PRG is available are retained as COPCs.

BSL: The concentration used for COPC screening (the maximum detected concentration) is less than the risk-based concentration (PRG) and/or ARAR; the analyte is therefore not selected as a COPC.

CC: The chemical is known to be related to releases at the Site and, therefore, is selected as a COPC even though the maximum concentration is less than the risk-based concentration (PRG) and/or ARAR.

E: The analyte is an essential nutrient, and is therefore not selected as a COPC.

FOD: The frequency of detection is very low (below 5%) and there is no known source of the chemical at the Site; the chemical is therefore not selected as a COPC.

Although background concentrations were not used in the COPC selection process, background data are provided in the COPC selection tables to evaluate site and ambient conditions. For analytes that are substantial contributors to risk, risks associated with background concentrations are identified and discussed in the risk characterization.

The COPC selection for each medium is presented in Tables 11-2 through 11-9; these tables include the rationale for the selection or exclusion of chemicals as COPCs. Table 11-10 presents a summary of the selected COPCs by exposure area and exposure medium.

Surface Soil

Analytes retained as COPCs included chlorinated VOCs, benzene, 8 PAHs, two PCB congeners (Aroclor-1254 and Aroclor-1260), and several inorganics. Total petroleum hydrocarbons was retained as a COPC at the Main Site and cyanide was retained as a COPC at the Former Chromium Plating Facility.

Subsurface Soil

Analytes retained as COPCs in subsurface soil are similar to analytes retained as COPCs in surface soil.

Groundwater

Analytes retained as COPCs in groundwater at the Intertidal Flats included chlorinated VOCs, benzene, toluene, and xylene.

Surface Water

VOCs were detected infrequently and at low concentrations in surface water, and SVOCs were generally not detected. Chlorinated VOCs (e.g., TCE, 1,2-DCE) were retained as COPCs when detected. 2-Methylnaphthalene and naphthalene were retained as COPCs in Intertidal Flats surface water. Eleven inorganics were retained as COPCs. Of the eleven inorganics, antimony (when retained as a COPC) was present at maximum concentrations below the background value.

Sediment

VOCs were detected infrequently and at low concentrations; only two VOCs (1,2-DCE and TCE) were retained as COPCs. Seven PAHs, three PCB congeners (Aroclor-1248, Aroclor-1254 and Aroclor-1260), and several inorganics were retained as COPCs. All inorganics retained as COPCs were detected at maximum concentrations greater than the background values.

Finfish

Three PCB congeners (Aroclor-1248, Aroclor-1254 and Aroclor-1260), as well as chromium, lead, and selenium, were identified as COPCs in fish fillet data. Lead was retained as a COPC because no screening values were available to evaluate the significance of the detected concentrations. With the exception of Aroclor-1248, which was not detected in background reference fish, the maximum detected concentrations of all analytes retained as COPCs were less than or consistent with background values.

Shellfish

Two PCB congeners (Aroclor-1254 and Aroclor-1260), as well as aluminum, arsenic, cadmium, chromium, lead, mercury, nickel, and selenium were identified as COPCs in fish fillet data. Lead was retained as a COPC because no screening values were available to evaluate the significance of the detected concentrations. At the Marine Basin, the maximum detected concentrations of all analytes retained as COPCs in ribbed mussels were less than or consistent with concentrations in the background reference mussels. At the Intertidal Flats, the maximum detected concentrations of PCBs, arsenic, cadmium, and selenium in ribbed mussels were less than or equal to background values, and the maximum concentrations of chromium, lead, and nickel were greater than concentrations in the background reference mussels. In oysters collected from the Intertidal

Flats, the maximum detected concentrations of each of the COPCs were greater than the concentrations in the background reference oysters.

Indoor Air
All of the detected chlorinated VOCs were retained as COPCs. With the exception of 1,1,1-TCA, the concentration of each compound exceeded a Region IX ambient air PRG in at least one of the samples collected over the 27 rounds of monitoring. 1,1,1-TCA did not exceed the Region IX PRG, but was retained as a COPC because it was considered to be associated with releases at the Site.

Summary
Table 11-10 provides a summary, by medium within each exposure area, of the analytes retained as COPCs.

11.1.4 Toxicity Profiles

Toxicity profiles describing the potential health effects associated with potential exposures to each of the analytes retained as COPCs are provided in Appendix S-2.

11.2 EXPOSURE ASSESSMENT

The exposure assessment is conducted to evaluate the populations of humans that may potentially occur at the site, the mechanisms or exposure pathways by which those humans may be potentially exposed to contamination at the site, and the magnitude of exposure that may occur through the potential exposure pathways. This process involves three steps:

1) Characterization of the exposure setting in terms of physical characteristics, current and future uses of the site, and the populations that may be potentially exposed to COPCs under the current and future land uses;

2) Identification of potential exposure pathways and exposure points to which the populations may be exposed; and

3) Quantification of exposure for each population in terms of the amount of chemical either ingested, inhaled, or absorbed through the skin from all exposure pathways.

11.2.1 Characterization of Exposure Setting and Potentially Exposed Populations

In characterizing the exposure setting, the physical attributes and demographics of the area surrounding the site are identified, as well as the current and reasonably foreseeable future land uses at the site.

The physical setting has been characterized in terms of the following attributes: climate, meteorology, geology, vegetation, soil type, groundwater, sediment, and surface water. This information was gathered from previous investigations and additional information collected during this investigation. The information generated from this analysis aids in defining the physical mechanisms that control or influence how people could be exposed to COPCs at the Site, and the processes that may control the fate and transport of the COPCs.

Demographics are characterized for (1) the populations residing or working near the site, (2) the activity patterns of those populations, and (3) the locations of potentially sensitive subgroups (if any exist). Key to this activity was determining current and foreseeable future land use of SAEP and the surrounding areas (e.g., residential, commercial and industrial, and recreational). Sources for this information included the following: (1) site visits to SAEP, (2) previous investigations, (3) information generated during the RI, (4) maps and photographs, and (5) interviews with SAEP personnel.

Site Description and Demographics.

SAEP is situated on Stratford Point peninsula that extends into Long Island Sound. The peninsula is flat with a slight slope toward the Sound. Nearly all of the land at SAEP is less than 10 feet above mean sea level, and is within the 100-year floodplain of the Housatonic River.

SAEP is bounded by a paved parking lot and wetlands to the north; the Housatonic River to the east; an open field, a drainage channel, and small commercial businesses to the south; and hangar buildings, the Sikorsky Memorial Airport, several small businesses, and Frash Pond to the west.

Historically, land in the SAEP vicinity was used for agricultural and residential purposes. At present, local agricultural activities are minimal. Historically, activity in the area involved growing oysters in shallow waters of the Housatonic River. Oysters were seeded in areas of the Housatonic River in the spring, collected in the fall, and placed in Long Island Sound to mature. Land in the vicinity of SAEP is zoned light industrial, business, commercial, or residential. There are several businesses located west of Main Street, across from SAEP, including a small strip mall, several gas stations, and a restaurant.

SAEP is located about 3/4-mile southeast of Johnson Junior High School and Birdseye School.

Recreational facilities in the area include Short Beach Park and nearby public wildlife areas, including Nells Island and the Great Meadow Salt Marsh. SAEP is located about 1/2-mile northwest of Short Beach Park, which had over 80,000 users reported for the year 1991.

The Greater Bridgeport Regional Planning Agency's population census of Stratford was 49,389 people in 1990. Slow population growth has been a trend in Stratford for nearly two decades, and the Connecticut Office of Policy and Management anticipates a continued slow or declining growth rate for Stratford with a population projection of 48,650 for the year 2000 and 45,800 for the year 2010 (W-C, 1991).

The average age of the population in Stratford is older than the state average. The town's median age in 1980 was 38.2, compared to 32 for the State of Connecticut. The Connecticut Office of Policy and Management anticipates the median age of Stratford to be 45.7 by the year 2010. Nearly 23 percent of Stratford's population had reached age 60 by 1980, compared to the state average of 17 percent.

The population of Stratford represents various races and nationalities. More than 8 percent of the 1980 population in Stratford was nonwhite. This compares closely to a nonwhite population of 9.9 percent for the State of Connecticut (W-C, 1991).

The climate in the vicinity of SAEP is strongly influenced by a land-sea breeze, which is most pronounced from spring to early autumn. The sea breeze promotes air mixing that results in slightly higher amounts of precipitation and slightly cooler temperatures than at inland locations. Precipitation averages 44 inches per year, with about 16 inches of snowfall per year. Average monthly temperatures range from a low of 28 degrees Fahrenheit in January, to a high of about 73 degrees Fahrenheit in July.

Current Land Use: The Intertidal Flats and the Outfall 008 Drainage area have no specific use under the existing land use conditions. The Marine Basin is located next to a public park and a municipal landfill, but is posted with “No Swimming” signs.

Interviews were conducted with local commercial fishermen regarding commercial and sport (recreational) angling and shellfish harvesting activities at the Intertidal Flats and Marine Basin (Based on the interviews, it appears that no commercial or sport fishing is occurring in the lower Housatonic River and, more specifically, at the areas of the Intertidal Flats and Marine basin that have been sampled during the RI activities. Specifically, the interviews indicated that:

- seed oysters are no longer collected from the lower Housatonic River;
- commercial fishing no longer occurs in the lower Housatonic River, and has never been observed at the Marine Basin;
- some commercial fishing for bluefish occurs in Long Island Sound;
- the majority of sport fishing for blue fish and striped bass occurs in Long Island Sound;
- no sport fishing has been observed at the Intertidal Flats or Marine Basin;
- shellfish (e.g., blue or ribbed mussels) in the lower Housatonic River are not of suitable quality for commercial harvesting.

SAEP and the properties surrounding SAEP obtain their water supply from the Bridgeport Hydraulics municipal water supply. Groundwater beneath SAEP is not used as a source of potable or non-potable (e.g., industrial) water, and is not considered to be an aquifer that requires protection as a potable groundwater resource by the State of Connecticut (i.e., is considered to be Class GB groundwater).

Under the current land use conditions, the only populations of people who occupy SAEP are SAEP employees (both for the U.S. Army and for government and private entities). Recreational visitors may access the Marine Basin.

Future Land Use: Future use of the Main Site, Chromium Plating Area, and South Lot is assumed to be commercial/industrial. The HHBRA incorporates the assumption that a land use restriction will be in place to prevent use of the Site for residential development without prior evaluation of such use.

Since commercial shellfish harvesting and commercial angling once occurred in the lower Housatonic River, the HHBRA incorporates the assumption that commercial and recreational angling and shellfish harvesting may occur at some point in the future (e.g., if the shellfish and finfish populations rebounded or angling in this area became popular due to socio-economic factors). Future access to the Intertidal Flats, Outfall 008 Drainage area, and Marine Basin during recreational uses of the surrounding area or in association with recreational angling or shellfishing is evaluated in this HHBRA.

Potentially Exposed Populations: Based on the current land use, the following human populations may potentially use, occupy, or access the Site:

- SAEP employees, including maintenance workers (at the Main Site).
- Visitors (at the Main Site)
- Recreational visitors (at the Marine Basin)

Based on the anticipated future land use, the following human populations may potentially use, occupy, or access the Site:

- Commercial/industrial employees, including commercial workers (e.g., office work), and industrial workers (e.g., manufacturing work).
- Commercial anglers and shellfishermen
- Construction workers
- Utility workers
- Recreational visitors
- Recreational anglers and shellfishermen

11.2.2 Identification of Exposure Pathways and Exposure Points

The purpose of this step in the exposure assessment is to identify all pathways through which people may be exposed to site-related COPCs during current and foreseeable future land use. A complete exposure pathway requires four elements: 1) a source or mechanism of chemical release; 2) a transport or retention medium; 3) a point of potential human contact with the contaminated medium; and 4) a route of exposure at the point of contact (USEPA, 1989). In some cases, the source of release may be the point of contact, such as direct contact with a spill area. The potentially exposed populations are the human receptors that are present at the exposure point.

Potential exposure pathways were determined by first identifying all sources of contamination and the receiving media. Once sources were identified, relevant fate and transport mechanisms were evaluated to identify potential exposure media. Exposure points and exposure routes were then identified by determining the areas where individuals may potentially come in contact with contaminated media (i.e., the exposure points), and the likely mechanisms of exposure (i.e., exposure routes). All exposure pathways that have these four elements (i.e., a source or mechanism of release, a transport or retention medium, an exposure point where contact can occur, and an exposure route at the point of contact) were considered complete pathways (USEPA, 1989), and were evaluated in the risk assessment.

The site conceptual site model presented in Subsection 11.1 and Figure 11-1, in conjunction with information concerning the potential receptors associated with the current and foreseeable land uses, were used to identify the potentially complete exposure pathways. The specific exposure routes evaluated for each receptor depend upon the characteristics of the COPCs identified in the exposure media (e.g., volatile vs. non-volatile), the physical characteristics of the site (e.g., soil moist and vegetated vs. dry and barren vs. asphalt covered), and the activities that the various potentially exposed populations may engage in (e.g., active sports vs. passive leisure vs. active industrial).

Table 11-11 provides rationale for the selection or exclusion of exposure pathways, by medium, which are evaluated in this risk assessment. Only potentially complete exposure pathways that are relevant to the exposure areas evaluated in this HHBRA (Main Site, South Lot Area, Intertidal Flats, Marine Basin, and Outfall 008) are identified and discussed. Table 11-12 provides a summary of the potentially complete exposure pathways that are evaluated in the risk assessment, and is used as the basis of the exposure scenarios described in Subsection 11.2.3.1.

11.2.2.1 Main Site. The Main Site includes the area of the facility that is located between Main Street, Sniffens Lane, and the Housatonic River. This portion of the facility contains 49 buildings and is completely paved (with the exception of the landscaped area between Building 2 and Main Street) and enclosed with security fencing.

The facility is idle under the existing conditions, and is being maintained and secured until transfer of the property has been completed. A small number of SAEP maintenance workers and U.S. Government employees occupy the facility daily. The foreseeable use of the facility is for commercial/industrial use. This area includes all of the facility buildings within the Main Site, as well as the former Chromium Plating Facility.

The site conceptual model and results of the COPC selection indicate that COPCs are present in the surface and subsurface soils (primarily VOCs, PAHs, PCBs, and inorganics), and air within several of the facility buildings (VOCs). In consideration of the current and foreseeable site uses and conceptual site model, several separate exposure points are evaluated within the Main Site:

- surface soil and subsurface soil at the Main Site (two exposure points)
- surface soil and subsurface soil at the former Chromium Plating Facility (two exposure points)
- indoor air at each of the facility buildings in the air monitoring program (10 buildings)

Some of the soil samples within the Main Site and Chromium Plating Facility exposure areas were collected from locations beneath buildings (i.e., soil samples collected from borings advanced through the building floors). Soil from all sample locations within each exposure area are evaluated together to account for the possibility that buildings are removed in the future, thereby making soil that is beneath buildings accessible. The chromium plating facility is evaluated as a separate exposure point because the area is associated with elevated concentrations of chromium and other inorganics and has a higher sampling density than other areas of the site (i.e., is a “hot spot”). These attributes could bias the EPCs if this area was combined with the other portions of the Main Site. In addition, the chromium plating facility may undergo remediation on a separate (more immediate) time line than other portions of the facility.

Under current land conditions, SAEP maintenance workers perform most of the facility maintenance and repair needs, including repair of facility-owned subsurface utility lines (e.g., steam heat supply distribution lines). Therefore, SAEP maintenance workers could be exposed to soil that may be excavated during utility maintenance and repair. These activities, which occur very infrequently, occur in the portion of the Main Site that contains the majority of the facility buildings. In addition, maintenance workers spend a portion of their time at the facility indoors, where potential exposures to volatiles in indoor air can occur. SAEP maintenance workers do not excavate soil beneath the chromium plating facility because access to that area is restricted by health and safety regulations due to known contamination. The potentially complete exposure pathways to soil include incidental ingestion, dermal contact, and dust vapor inhalation. Potential exposures and health risks associated with current land use conditions are conservatively evaluated through characterization of a future full-time commercial/industrial worker.

Under future commercial/industrial use of the facility, the most likely exposure pathway to site-related contamination is via inhalation of vapors that have migrated from soil or groundwater to indoor air at the facility buildings. It is likely that the ground will remain paved. However, the HHBRA incorporates the assumption that pavement could be removed and replaced by grass and landscaping. Under those conditions, commercial and industrial workers could potentially be exposed to COPCs in soils via incidental ingestion, dermal contact, and dust inhalation. Potential exposures to COPCs are evaluated for the following additional exposure media:

- surface soil and subsurface soil at the Main Site (2 exposure points)
- surface soil and subsurface soil at the Former Chromium Plating Area (2 exposure points)
- indoor air in facility buildings at the Main Site (10 buildings)

Construction workers employed in support of site redevelopment could be exposed to surface soil and/or subsurface soil by direct contact and dust inhalation, and to indoor air, during demolition/construction activities. Groundwater throughout the Main Site is at least 7 feet below ground surface, and only becomes shallow on the river-side of the Dike, where development is unlikely. It is unlikely that construction excavations would extend to depths of 7 feet or more, especially for commercial/industrial developments that typically use slab-on-grade construction techniques with footings that extend no more than 4 feet below ground surface. Therefore, the principal exposure pathways for construction workers are surface and subsurface soil at the Main Site and Former Chromium Plating area by incidental ingestion, dermal contact, and dust inhalation, and to indoor air in buildings at the Main Site.

11.2.2.2 South Lot. The South Lot is the area of the Site located on the south side of Sniffens Lane. This area contains a large parking lot (South Parking Lot), Building B-6, and the former wastewater treatment building. The media at this area that have been affected by site-related contamination include surface soil, subsurface soil, groundwater, and indoor air at Building B-6. The COPCs identified in these media primarily include VOCs (all media), PAHs (soil), and inorganics (soil).

Under current land use, there is no use of the South Lot area, and there are no potentially complete exposure pathways to site-related contamination due to pavement and vacancy of Building B-6. Under future land use conditions, the South Lot area (including Building B-6) is assumed to be used for commercial/industrial purposes. It is assumed that office workers or commercial/industrial workers would use or occupy Building B-6 and potentially be exposed to vapors in the indoor air. In addition, it is assumed that pavement could be removed and replaced by grass and landscaping. Under this assumption, commercial/industrial workers could be exposed to contamination in surface soil by incidental ingestion, dermal contact, and dust inhalation. Finally, to provide information to support risk management decision-making, potential direct contact and dust inhalation exposures to COPCs in subsurface soil are evaluated. Construction workers employed in support of site redevelopment could be exposed to surface soil and/or subsurface soil by direct contact and dust inhalation, and to indoor air, during demolition/construction activities. Groundwater throughout the South Lot is at least 7 feet below ground surface. It is unlikely that construction excavations would extend to depths of 7 feet or more, especially for commercial/industrial developments that typically use slab-on-grade construction techniques with footings that extend no more than 4 feet below ground surface. Therefore construction workers could potentially be exposed to COPCs in surface and subsurface soil at the South Lot area by incidental ingestion, dermal contact, and dust inhalation, and to indoor air at B-6.

11.2.2.3 Intertidal Flats. The Intertidal Flats are the portions of the Housatonic River that extend from the edge of the river channel to the shoreline. The Intertidal Flats are characterized by a thick layer of soft sediments with varying amounts of surface water cover (i.e., from one-half a foot to several feet), depending on tidal influences and river stage. The portion of the Intertidal Flats that has been sampled and is included in this HHBRA as part of the Site is an area that extends approximately 2000 feet parallel the shoreline and approximately 1300 feet into toward the river channel, as shown in Figure 11-3.

There is little, if any, use of the aquatic environment in the vicinity of the Intertidal Flats. The portion of the Housatonic River where SAEP is located is industrialized. Although a marina with leisure boats is located upstream of SAEP (approximately 1 mile from SAEP), no beach areas or docks used for recreational boating are located near the area that has been sampled and is included as part of the Site. In the area sampled, the Intertidal Flats are covered with a thick layer of very soft sediment that makes walking on the flats treacherous. Commercial angling and shellfishing, as well as recreational angling, shellfishing, and aquatic recreational use are not known to occur in the area sampled. The depth of the water at the portion of the Intertidal Flats included as part of the Site is generally not deep enough to permit swimming. The only uses of the Intertidal Flats that are within the realm of “reasonably foreseeable” are associated with recreational and commercial shellfishing and angling. Angling and shellfishing are uses that do not occur under the existing conditions, but are consistent with uses of this portion of the Housatonic River that have historically occurred and could occur in the future.

The site conceptual model and results of the COPC selection indicate that COPCs are present in the shallow sediments (primarily PAHs, PCBs, and inorganics), surface water (primarily inorganics), and shallow groundwater that is discharging through the sediment (primarily chlorinated VOCs).

Recreational visitors who boat or fish recreationally in the vicinity of the Intertidal Flats in the future could be exposed to surface water and sediment through incidental contact. Recreational angling for shellfish could involve digging in submerged sediments whereupon dermal contact and incidental ingestion of surface water and sediment could occur, as well as dermal contact with shallow groundwater that is present in the pore-spaces of the sediment. Since the sediment layer in the Intertidal Flats is thick, it is possible that access of the Intertidal Flats could result in exposures to sediments and shallow groundwater that include the feet and lower legs (e.g., a person may “sink” into the sediment up to their knees). Incidental ingestion of the shallow groundwater would not occur because the groundwater is present within the interstitial space of the sediment, and it is rapidly diluted into surface when it discharges. Likewise inhalation of particulates (dust) from sediment or vapors from surface water and sediment are not complete exposure pathways because the sediment is submerged, thereby prohibiting dust generation and substantially attenuating vapor release. Any vapors that may be released from the surface water or sediment would be significantly diluted into ambient air and would not represent an exposure concern.

If commercial angling occurs in the future, commercial fishermen could contact surface water through wind spray or when hauling fishing gear into a boat. Angling for shellfish during warmer seasons could involve walking through the Intertidal Flats and manually digging in the submerged sediments. Although commercial fishermen would wear protective gear (e.g., waders and boots), dermal contact and incidental ingestion of surface water and sediment is assumed to occur during these activities.

PCBs and inorganics were retained as COPCs in finfish and shellfish collected from the Intertidal Flats. People who catch and eat the fish may be exposed to the COPCs. Visitors who angle recreationally for fish and shellfish are assumed to ingest the fish that they catch. Similarly, it is assumed that commercial fishermen hold back a portion of their catch for their own consumption.

The surface water and sediment samples were collected from the Intertidal Flats on both sides of the Causeway. There is no difference in exposure potential at the Intertidal Flats on either side of the Causeway. Therefore, the sediment, surface water, and shallow groundwater data collected throughout the Intertidal Flats are used to represent a single exposure point for each medium. Finfish, ribbed mussels, and oysters are each evaluated as separate exposure media because humans may preferentially eat one type of biota over another.

11.2.2.4 Marine Basin. The Marine Basin is located adjacent to a public park and recreational facility owned by the Town of Stratford. A landfill that possibly received industrial waste abuts the basin on one side; open space and the Housatonic River abut the other sides of the Marine Basin. The basin is tidally-influenced, and water ranges in depth from two to three feet during low tide, to five to six feet during high tide. Unlike the Intertidal Flats, the sediment at the Marine Basin is more sandy and firmer and, therefore, can be walked upon. Although the Marine Basin is posted with “No Swimming” signs, the location of the basin is easily accessible to visitors to the public park.. Therefore, access to the Marine Basin by recreational visitors who use the adjacent recreational facilities is possible. Commercial and recreational angling and shellfishing are not known to occur in the Marine Basin.

The site conceptual model and results of the COPC selection indicate that COPCs are present in the shallow sediments (primarily PAHs and inorganics) and surface water (primarily inorganics).

In the future, recreational visitors who boat or fish recreationally at the Marine Basin could be exposed to surface water and sediment through incidental contact. Recreational shellfishing would involve digging in submerged sediments whereupon dermal contact and incidental ingestion of surface water and sediment would occur. If recreational visitors wade or swim in the

Marine Basin in the future, direct contact with surface water and sediment would occur. Inhalation of particulates (dust) from sediment is not a complete exposure pathway because the sediment is submerged, thereby prohibiting dust generation and substantially attenuating vapor release.

If commercial fishing were to occur at the Marine Basin in the future, commercial fishermen could contact surface water through wind spray or when hauling fishing gear into a boat. Shellfishing during warmer seasons could involve walking through the Marine Basin and manually digging in the submerged sediments. Although commercial fishermen would wear protective gear (e.g., waders and boots), dermal contact and incidental ingestion of surface water and sediment is assumed to occur during these activities.

PCBs and inorganics were retained as COPCs in finfish and shellfish collected from the Marine Basin. People who catch and eat the fish may be exposed to the COPCs. Visitors who angle recreationally for fish and shellfish are assumed to ingest the fish that they catch. Similarly, it is assumed that commercial fishermen hold back a portion of their catch for their own consumption.

The sediment and surface water data collected throughout the Marine Basin are used to represent a single exposure point for each medium as there is no indication that exposures would preferentially occur at one are of the Marine Basin versus another. Finfish and ribbed mussels are each evaluated as separate exposure media because humans may preferentially eat one type of biota over another.

11.2.2.5 Outfall 008 Drainage Area. The surface water and sediment samples collected to investigate potential contamination from releases at Outfall 008 Drainage were taken from a ditch that leads from the South Parking Lot to the Marine Basin. The ditch is relatively narrow (e.g., less than 8 feet wide) and shallow (less than 4 feet) and contains varying depths of standing or very slowly-flowing water (water depth is usually less than two feet). Recreational use of this area does not occur under the existing land use because access is prevented by fencing near South Lot, distance from an entry point (from the public park area), and very dense vegetation, although access to the ditch is not as difficult near the Marine Basin. Recreational use of the area in the future is not likely, but is evaluated in this HHBRA to provide information for risk management decision-making. If recreational use of this area occurred in the future, the most likely use would probably be a passive use such as bird-watching.

The site conceptual model and results of the COPC selection indicate that COPCs are present in the sediments (primarily PAHs, PCBs, and inorganics) and surface water (primarily VOCs and inorganics).

Potential exposures to surface water and sediment could involve incidental ingestion and dermal contact during walking/accessing the ditch. Inhalation of particulates (dust) from sediment or vapors from surface water are not complete exposure pathways because the sediment is submerged, thereby prohibiting dust generation, and any vapors that may be released from the surface water would be significantly diluted into ambient air and would not represent an exposure concern.

The sediment and surface water data collected throughout the Outfall 008 Drainage ditch are used to represent a single exposure point for each medium, as there is no indication that exposures would preferentially occur in one area versus another.

11.2.3 Quantification of Exposure

Once complete exposure pathways are selected for evaluation, the final step of the exposure assessment is to quantify exposure (i.e., intake) for each pathway. This quantification process involves developing assumed exposure conditions (scenarios), developing exposure point concentrations (EPCs), and then calculating COPC intakes. The COPC intakes are subsequently combined with dose-response data in the risk characterization to calculate estimates of cancer and non-cancer health risk.

11.2.3.1 Exposure Scenarios. Exposure scenarios are used to quantitatively describe the COPC exposures that could theoretically occur for each land use and exposure pathway evaluated. The exposure scenarios are used in conjunction with EPCs to derive quantitative estimates of COPC intake. The ultimate goal of developing exposure scenarios, as defined in USEPA guidance, is to identify the combination of exposure parameters that results in the most intense level of exposure that may "reasonably" be expected to occur under the current and future site conditions (USEPA, 1989, 1994b). This involves two activities: 1) identifying the receptor subpopulations that incur the most exposure to COPCs among the receptors associated with a given land use, and 2) identifying the quantitative parameters that are used to model COPC exposure to that receptor population. Therefore, a single exposure scenario is often selected to provide a conservative evaluation for the range of possible receptors and populations that could be exposed at the site. The Reasonable Maximum Exposure (RME) for each exposure pathway is evaluated using exposure parameters that are typically based on the 95th percentile of the range of possible values. The Central Tendency (CT) exposure scenario evaluates the same receptor subpopulations, but uses parameter values that are typically based on the 50th percentile, mean, or median of the range of possible values.

The future commercial/industrial worker scenario is evaluated for the RME conditions only because all exposure parameters are based on published values that standardize this exposure scenario, and remedial decisions are typically made using the RME risks. The future land use scenarios that address potential exposures at the Marine Basin, Intertidal Flats, and Outfall 008 areas are evaluated for the RME and CT conditions. This approach is used because many of the exposure parameters that are used for these scenarios are not based on standardized default values (because standardized values are not published for these land uses). Consequently, it becomes more relevant for risk management decision-making to characterize the range of potential exposures and risks that would be associated with very conservative assumptions regarding potential exposures (RME case) and more typical assumptions regarding potential exposures (CT case).

The sources of the parameters that are used to quantitatively evaluate COPC intake by ingestion, dermal contact, and inhalation exposure pathways are identified below. The specific combination of variables that are used for each receptor scenario are subsequently discussed and are provided in Tables 11-13 through 11-16.

Ingestion Rate

The ingestion rate describes the amount of exposure medium that is consumed via the oral exposure route. Ingestion rates for soil and water (groundwater used as tap water) are specified by USEPA Region I (USEPA, 1994b), but ingestion rates for surface water and sediment are not specified. The ingestion rates for soil vary depending on the age of the receptor and the activities the receptor is engaged in. For young children (ages 1 through 6), the soil ingestion rate is 200 mg/day; for older children and adults the soil ingestion rate is 100 mg/day, and for contact intensive exposures to adults, the soil ingestion rate is 200 mg/day (USEPA, 2001d). The mechanism of ingestion exposure to soil is by adherence of soil particles to the hand (fingers), and subsequent transfer to the mouth during incidental hand-mouth contact. Based on this mechanism of exposure, ingestion exposure to sediment is unlikely to be substantial because sediment that is submerged would not adhere to skin, as the surrounding surface water would prevent binding of the sediment to the skin. In addition, when a body part that contacts sediment is removed from the surface water body, the sediment would wash off, thereby preventing adherence of the material. Hence, it is unlikely that any exposure to COPCs in submerged sediment would occur. Nonetheless, incidental ingestion of sediment is evaluated in the risk assessment to provide a conservative assessment of potential exposures to COPCs. The incidental ingestion rates for sediment are based on the soil ingestion rates recommended by USEPA Region I. Incidental ingestion of surface water is quantified using an ingestion rate for swimming published in USEPA guidance (USEPA, 1988a), decreased by a factor of two to account for lower exposures that would occur during wading activities.

Fraction Ingested

This variable describes the amount of exposure medium contacted daily that is derived from the site. A fraction ingested term of 50 percent, for example, indicates that one-half the daily contact with the exposure medium occurs at the site of interest. The soil ingestion rates used in this HHBRA represent the amount of soil (i.e., inorganic or soil-like material, including soil, sediment, and dust) that is ingested from all sources (e.g., residential yard, recreational facility, and work place) daily during waking hours. The fraction-ingested values used to evaluate potential sediment exposures are less than 100% because soil ingestion rates are used as sediment ingestion rates, and it is not reasonable to assume that the entire daily mass of ingested soil-like material would be associated with short-term exposures to sediment at the Site. The specific fraction-ingested values used were based on the receptor- and activity-specific conditions and are identified in the exposure scenarios discussed below.

Inhalation Rate

The inhalation rate describes the amount of air that a receptor breathes over a specific time period. Inhalation rates for various age groups and activity levels are published by USEPA (USEPA, 1997a). As described in Subsection 11.2.3.3 and 11.4, COPC exposures via the inhalation pathway are evaluated using reference concentrations and unit risk values in combination with daily exposure concentrations. This approach evaluates risk based on a direct evaluation of the COPC concentration in air to which a receptor is exposed, and not the dose of COPC that a receptor receives. Consequently, the inhalation rate parameter is not directly used to quantify exposure.

However, at the request of the CTDPH, the inhalation rate parameter is used in this HHBRA to incorporate the assumption that receptors with high respiration rates, by analogy, may have a greater exposure to chemicals in air (CTDPH, 2002). The inhalation rate is used in this HHBRA to establish an inhalation adjustment factor (IAF). The IAF accounts for receptor-specific inhalation rates that are greater than the inhalation rate that is used by USEPA for deriving inhalation dose-response values (20 m^3/day), which describes the volume of air inhaled by a typical adult engaged in rest, light, and moderate activities over a 24-hour period. The IAF is calculated by dividing the receptor-specific inhalation rate by the baseline inhalation rate of 20 m^3/day. An IAF value of 1.0 indicates that receptor-specific inhalation rate is not greater than the baseline inhalation rate (i.e., an IAF is essentially unnecessary), and an IAF value greater than 1.0 indicates that the receptor-specific inhalation rate is greater than the baseline inhalation rate. The IAF is applied to the algorithm used to estimate average daily exposure concentrations, as described in Subsection 11.2.3.3.

Adherence Factor

The adherence factor describes the amount of soil that adheres to the skin following dermal contact with soil. This variable is incorporated in the dermal intake calculation and, together with body surface area, describes the amount of sediment that is contacted by the skin during a given exposure event. Adherence factor values are area-weighted values published by USEPA (USEPA, 2001c). For commercial/industrial workers, the USEPA-recommended soil adherence factors for industrial receptors are used. Adherence factors for exposures to sediment are based on USEPA-recommended soil adherence factors, and are discussed in the exposure scenario descriptions below.

Surface Area

Surface area (SA) describes the area of body parts that come into contact with an exposure medium. Therefore, the dermal surface area values for each receptor are dependant on the exposure media evaluated (e.g., surface water vs. surface soil), the activities assumed to occur at those media (e.g., wading in surface water vs. swimming in surface water), and the ages of the receptors evaluated (because body surface area changes with age). Surface area values were calculated using 50th percentile values specific for each body part and receptor age, as published in USEPA guidance (USEPA, 1997a). Because body surface areas change with age while people are growing, the body surface areas for children and adolescents are calculated as age-adjusted surface areas. These values are calculated as the average surface areas for the age-groups evaluated (i.e., by summing the surface areas at each age, then dividing by the number of ages evaluated). Skin surface area values were selected from USEPA guidance, or calculated in accordance with USEPA guidance (USEPA, 1997a; 2001c). The dermal surface area value for commercial/industrial worker contact with soil is the USEPA default value for the industrial worker scenario (USEPA, 2001c). The basis of the dermal surface area values for receptors associated with possible exposures to surface water and sediment exposure areas are summarized as follows and additional rationale is provided in the exposure scenario descriptions below.

Receptor	Groundwater	Sediment	Surface Water
Recreational Angler – Intertidal Flats			
Adult	Lower legs, feet	Lower legs, feet	Legs, feet
Recreational Visitor - Marine Basin and Outfall 008 Drainage			
Adult	NA	Feet	Legs, feet
Adolescent	NA	Feet	Legs, feet
Child	NA	Feet	Abdomen, legs, feet

Commercial Fisherman			
Fin Fishing	NA	NA	Hands, forearms, face
Shell Fishing	NA	Hands, forearms	Hands, forearms, face

Exposure Frequency

The exposure frequency describes the number of times exposure occurs over a given exposure period, typically in units of days per year. Exposure frequency is highly dependent on the nature of the exposure setting and land use. Exposure frequency terms are described in the exposure scenario descriptions below.

Exposure Duration

Exposure duration represents the amount of time (typically in years) that a receptor is assumed to occupy, access, or use the site, and therefore contact the exposure media. Exposure duration values for some exposure scenarios (e.g., occupational exposures) are established by USEPA (USEPA, 1994b), whereas others (e.g., recreational visitor) are based on site-specific exposure conditions.

Exposure Time

The exposure time describes the amount of time over which contact with the exposure medium occurs during each exposure event. The exposure time variable is used in the dermal exposure calculations for water and the inhalation exposure calculations for air. Exposure time values are based on USEPA guidance and site-specific exposure conditions, and are described in the exposure scenario descriptions below.

Body Weight

Body weight values are used to normalize intake estimates, in units of chemical mass per kg body weight, so that intake estimates for various receptor scenarios are all comparable to the same dose-response values (which are also in units of chemical mass per kg body weight). USEPA (1994b; 1997a) publishes body weights for adults and children ages 1 through 6. Body weights for interim age groups (e.g., adolescents) are calculated as the average body weight for the age group evaluated from weight data published by USEPA (1997a).

Averaging Time

COPC exposures are quantified as an estimate of daily intake. Deriving the estimate of daily intake requires averaging an exposure over a time period; the time period used in this derivation is the averaging time. According to USEPA guidance, the averaging time for carcinogenic effects is assumed to be a 70-year lifetime (USEPA, 1989). This averaging time is used to evaluate carcinogenic effects for all receptors, regardless of the length of the receptor-specific exposure period. The averaging times for non-carcinogenic effects are equivalent to the duration of exposure and may vary depending on the nature of exposure. There is a wide range of possible estimates, from a day to a lifetime. However, based on USEPA guidance, exposure durations for non-carcinogenic effects can roughly be categorized into one of three periods: (1) chronic exposures of 7 years to a lifetime; (2) subchronic exposures of 2 weeks to 7 years; and (3) acute exposures of less than 2 weeks (USEPA, 1989). The length of the exposure period depends on the potentially exposed population and the characteristics of exposure. The averaging time for non-carcinogenic effects is always set equal to the exposure duration (USEPA, 1989).

Future Commercial/Industrial Worker. The foreseeable use of the facility is for commercial/industrial use. Commercial/industrial workers are assumed to be full-time workers (at facility 8 hours per day, 250 days per year, over a 25-year duration) who are either office workers or industrial workers. Office workers are assumed to be exposed to indoor air only, and industrial workers are assumed to be exposed to indoor air (each work-day), and to soil on days when the weather is not inclement and during months when the ground is not frozen or snow covered (150 days/year; USEPA, 1994b). The future commercial/industrial worker scenario is evaluated as an industrial worker who is exposed to indoor air 250 days per year and soil 150 days per year. This represents a very conservative scenario because it is unlikely that substantial contact with both soil and indoor air would occur on the same day. The nature of the work is assumed to involve moderate activities, so the IAF is greater than 1.0, as documented in Table 11-13. All other RME exposure parameters are based on values for commercial/industrial workers recommended in USEPA guidance, as shown in Table 11-13.

Although each building is a separate potential future indoor air exposure point, the EPCs for Building B-2 and Building B-6 are greater than the other buildings (Appendix S-3). Therefore, characterization of potential exposures and risks for these two buildings provides an assessment that is conservative for the other buildings at the Site. Soil exposure points include surface soil and subsurface soil at the Main Site, Former Chromium Plating Area, and South Lot.

The future commercial/industrial worker scenario is used to characterize risks for all potential future exposures to soil and indoor air that could be associated with commercial/industrial uses of the property, including potential exposures to construction or utility workers that could be associated with demolition and construction activities during re-development of the Site. The exposure assumptions that would be applicable to a construction worker scenario are consistent with those used for the commercial/industrial worker scenario, with the following exceptions: a) the soil ingestion rate for a construction worker would be two-times greater than the ingestion rate used for the commercial/industrial worker scenario; b) the dermal surface area exposed to soil would be approximately 60% greater for a construction worker; and c) the exposure duration for a construction worker would be 50-times less than the exposure duration for a commercial/industrial worker and the exposure frequency would be up to two-times lower for a construction worker. Finally, non-cancer risks for a construction worker scenario would be evaluated using sub-chronic reference dose (RfD) and reference concentration (RfC) values when available. Potential risks to a construction worker are discussed qualitatively in the uncertainty analysis.

Recreational Visitor. Under the current land use, it is possible that unauthorized wading occurs at the Marine Basin (the basin is posted with "No Swimming" signage). However, the occurrence of this activity is infrequent. There is no use of the Intertidal Flats and Outfall 008 Area under the existing land use conditions, and people are not known to access those areas.

Because the baseline risk assessment cannot use non-physical barriers, such as signage, to exclude exposure evaluations that would otherwise be considered relevant in the absence of such non-physical barriers, future exposures to media associated with the Marine Basin and Outfall 008 Area are assumed to occur in association with open space/recreational land uses of the surrounding area. Potential activities that may occur at these areas that could result in exposures to surface water and sediment include playing, exploring, and wading. At the Marine Basin, recreational angling and shellfish harvesting could also occur, resulting in exposures to sediment, surface water, and fish tissue.

The physical characteristics of the Intertidal Flats, notably the deep, mucky sediment, makes recreational wading or exploring difficult for adults and dangerous, if not impossible, for children. The land adjacent to the Intertidal Flats will be transferred as commercial/industrial property, indicating that recreational uses of the Intertidal Flats are neither likely nor commensurate with the area land uses. Therefore, the only conceivable uses of the Intertidal Flats that could potentially result in human exposures to surface water and sediment at the Intertidal Flats are angling or shell fish harvesting.

The recreational visitor is considered to be an area resident that is raised at and remains in the vicinity of the Site for a 30-year period (9 year period for CT). The age groups considered in the development of the scenario are young children (ages 1 through 6), older children/adolescents (ages 7 through 18), and adults (ages 19 through 30). The scenario is segregated into these age groups because adolescents and adults are likely to access the exposure areas with more frequency than young children who would require supervision and would not access the exposure areas independently. Exposure parameters are provided in Table 11-14 for surface water and sediment, and in Table 11-15 for fish consumption.

Potential exposures to surface water and sediment were assumed to occur during the warmest months of the year (June–September). It is not reasonable to assume that people engaged in recreational activities would contact surface water and sediment during fall, winter, and spring months of the year. For young children and older children/adolescents, it is assumed that exposures to surface water and sediment at the Marine Basin and Outfall 008 Drainage occur during wading. Older children/adolescents are assumed to contact surface water and sediment 4 times per week during July and August (36 days per year), and 2 times per week during the second half of June and first half of September. Each exposure event is assumed to occur over a two-hour period. Young children are assumed to contact surface water and sediment 2 times per week July and August (18 days per year), for one hour per event. The dermal surface area values for contact with sediment were based on the surface area of the feet, as these are the only body parts that would potentially be in constant contact with submerged sediment. Although hands may contact submerged sediment, the exposure would be transient because the sediment would wash off when removing the hand from surface water or re-submerging the hand in surface water. The dermal surface areas for contact with surface water while wading were selected as the feet and entire legs for adolescents, and the feet, legs, and abdomen for young children who, due to a shorter body height, would have more of their body submerged in the water while wading.

For adults it is assumed that exposures to surface water, sediment, and shallow groundwater at the Intertidal Flats occur during shell fishing activities, exposures to surface water and sediment at the Marine Basin occur during wading or shellfishing activities, and exposures to surface water and sediment at Outfall 008 Drainage Area occur during wading activities. To provide a conservative assessment that captures potential exposures associated with both wading and shell fishing, the exposure frequency is assumed to be 3 times per week June through September (52 days per year) at areas where shell fishing could occur (Marine Basin and Intertidal Flats). If shellfish harvesting occurs in other months, protective gear would be worn due to the cold temperatures of the water. Potential contact with surface water and sediment at the Outfall 008

Area would be during occasional visitation of the area, assumed to occur 2 times were week July and August (18 days per year). The dermal surface area values for contact with sediment and shallow groundwater at the Intertidal flats were based on the surface area of the feet and lower legs, as these are the body parts that would potentially be in constant contact with submerged sediment (lower legs are included because people sink into the mucky sediments at the Intertidal Flats). Although hands may contact submerged sediment, the exposure would be transient because the sediment would wash off when removing the hand from surface water or re-submerging the hand in surface water. At the Marine Basin and Outfall 008 Drainage Area, the dermal surface area for sediment was based on the area of the feet. The dermal surface area for contact with surface water while wading was selected as the area of the feet and entire legs.

The sediment ingestion rates are based on the soil ingestion rates, even though submerged sediment does not readily adhere to hands and thus is less likely to be transferred to the mouth during hand-mouth contact. However, in consideration of this as well as the assumption that these receptors spend only a portion of their daily outdoor time contacting sediment, the fraction-ingested values for sediment are established at 50%. Surface water ingestion rates are based on values for swimming as recommended in USEPA (1988) guidance. For wading in surface water, dermal exposures are based on the age-adjusted body surface areas for feet and legs. For sediment, it is assumed that dermal contact may occur at the feet and lower legs. Because the sediment is submerged, it is not likely that adherence to the skin would occur and, therefore, dermal absorption of chemicals in sediments is less likely. Dermal adherence factors have not been developed for sediments. Therefore, the RME dermal adherence factors for residential soils recommended in USEPA guidance (USEPA, 2001c) are used for younger children and older children/adolescents. The geometric mean value for “reed gatherers” is used for adults to represent possible enhanced contact with sediment during shellfishing.

Exposure parameters for fish ingestion are based on values published by USEPA for recreational marine angling, since the waters of the lower Housatonic River are tidally-influenced (USEPA, 1997a). The biota consumption scenarios assume that adults catch fish and shellfish, but that adults, adolescents, and children consume the fish and shellfish. The ingestion rates for adults are based on recreationally-caught marine fish (Atlantic Ocean), and the ingestion rates for children are based on age-specific consumption rates for recreationally-caught fish (USEPA, 1997a; Table 11-61). The scenario considers 100% of recreationally-caught fish to have come from areas affected by the Site (i.e., fraction ingested = 100%). Because the ingestion rates and fraction-ingested terms are for total recreationally-caught fish, it is inherently assumed that a recreational angler consumes only shellfish or only finfish (i.e., two separate exposure scenarios). This approach will define the maximum risk posed by either, and the risk posed by ingesting a combination will be less than the maximum.

Commercial Fisherman. As discussed previously, at present there is no commercial angling or shellfish harvesting occurring at the intertidal flats of Housatonic River adjacent to the site.. This scenario has been developed to evaluate potential exposures to COPCs in the event that commercial angling or shellfish harvesting occurs at the Intertidal Flats or Marine Basin at some point the future.

The commercial fisherman is evaluated using two scenarios: commercial finfishing, and commercial shellfishing. The scenarios are evaluated separately because they involve different activities and, thus, different exposure media, and because fish ingestion rates are established as total ingestion rates among fish and shellfish. Exposure parameters are provided in Table 11-16.

A commercial finfish angler is assumed to perform angling from a boat. Therefore, contact with sediment is very unlikely. The scenario considers angling 5 days per week, year round (250 days per year) over a 30-year period (9 years for CT). Commercial anglers use protective gear (e.g., boots and waders) to avoid getting wet. Therefore, exposures to surface water are assumed to occur at the hands and forearms (when hauling in nets), and to the face (due to wind spray). Incidental ingestion of surface water is unlikely, but is assumed to occur at a rate of 10 ml per day. Because the Intertidal Flats and Marine Basin areas each represent a very small portion of the overall range that a commercial angler would frequent (i.e., less than 1%), the fraction-ingested value for surface water is established at 25%. This reflects a conservative estimate of potential exposure to site-related surface water. Similarly, the exposure time to which a commercial finfish angler is assumed to be exposed to site-related surface water is conservatively assumed to be two hours per day, and not the entire day.

A commercial shellfish harvester is assumed to perform activities from a boat during cold months, and by wading and manually digging into sediments during warmer months of the year. Therefore, contact with surface water may occur all year, and contact with sediment may occur during a portion of the year. The scenario considers shellfish harvesting 5 days per week, year round (250 days per year) over a 30-year period (9 years for CT). However, it is unrealistic to assume that commercial shellfish harvesting could occur at the same shellfish beds each day without exhausting the supply of shell fish at each bed. Therefore, the scenario assumes that an individual shellfish harvester would visit each shellfish bed (i.e., Marine Basin or Intertidal Flats) once per week, and spend the entire day at each of those beds. Commercial shellfish harvesters use protective gear (e.g., boots and waders) to avoid getting wet. Therefore, exposures to surface water are assumed to occur at the hands and forearms and to the face (due to splashing and wind spray). Incidental ingestion of surface water is unlikely, but is assumed to occur at a rate of 10 ml per day. Dermal contact with sediments is assumed to occur at the hands and forearms. Dermal contact with sediment and shallow groundwater (Intertidal Flats only) at the feet and lower legs does not occur due to the use of boots and waders. Although the Intertidal Flats and Marine Basin areas each represent a very small portion of the overall range that a commercial angler would frequent (i.e., less than 1%), the fraction-ingested values for surface water and sediment are established at 100% to account for the fact that the shell fish harvester is assumed to spend the entire day at each shellfish bed on the days-exposed. This reflects a conservative estimate of potential exposure to site-related surface water and sediment, as it assumes that 20% of all commercial shellfishing occurs at the Intertidal Flats or Marine Basin (all day, one day per week). The exposure time to which a commercial shellfish angler is assumed to be exposed to site-related surface water is conservatively assumed to be eight hours to maintain consistency with the assumption of spending the entire day at each shellfish bed.

Fish ingestion rates are based on values for total fish intake among the general population (USEPA, 1997a). This scenario assumes that the commercial fisherman is not a subsistence angler, but that all fish consumed are self-caught (unlike recreational anglers or the general public who's annual dietary intake of fish is represented by only a small percentage, if any, of self-caught fish). However, it is unreasonable to assume that all fish consumed are from areas that have been affected by the Site, as those areas represent extremely small portions of the overall range of commercial fishing in the area. Therefore, the fraction-ingested parameters for fish consumption have been established at 25% for RME, and 10% for the CT. These values still represent a conservative estimate of the actual intake of fish that have been potentially affected by releases from the Site.

11.2.3.2 Exposure Point Concentrations. In accordance with USEPA Region I guidance, RME EPCs for all media except groundwater that is used as potable water (which is not an applicable exposure pathway at this Site) are based on the lesser of the 95 percent UCL on the arithmetic mean of the data set, or the maximum detected concentration in the data set, for each exposure point (USEPA, 1994b). The CT EPCs are based on the RME EPCs unless the RME EPCs are based on the maximum detected concentration, in which case the arithmetic mean concentrations are used as the EPCs (USEPA, 1995). In infrequent cases where an arithmetic mean concentration is greater than the maximum detected concentration (generally a result of infrequent detection at very low concentrations), the CT EPC is based on the maximum detected concentration.

USEPA cautions that the 95% UCL is unreliable when there are fewer than 10 samples in the data set (USEPA, 1992b). Therefore, for exposure points with fewer than 10 samples, a 95% UCL value was not calculated; the maximum detected concentration was used as the RME EPC and the arithmetic mean was used as the CT EPC.

For data sets with greater than 10 samples, the 95% UCL values were determined in accordance with the recent USEPA guidance "*Calculating Upper Confidence Limits for Exposure Point Concentrations at Hazardous Waste Sites"* (USEPA, 2002a) and the ProUCL Software (v 3.0; USEPA, 2004c). This guidance supercedes previous guidance concerning UCL calculations (USEPA, 1992b) and clarifies that the statistical basis of UCL values should be identified in consideration of the distribution of the data. Specifically, the guidance identifies methods that should be used to determine the data distribution and provides a decision-tree for identifying the statistically-appropriate method for calculating UCL values based on statistical metrics of the data set. The ProUCL software automatically tests the distribution of the data and implements the decision-tree to calculate UCL values. The ProUCL output files are provided in Appendix S-3. EPCs are shown in Tables 11-17 through 11-29.

Exposure Point Concentrations for Chromium and Hexavalent Chromium

The majority of media samples collected at the Site were analyzed for total chromium, which represents the total concentration of all oxidation states of chromium (primarily Cr[III] and Cr[VI]). However, a subset of soil samples collected at the former Chromium Plating Facility were analyzed for Cr[VI] in addition to total chromium. As discussed in Subsection 11.3, hexavalent chromium exhibits toxicological characteristics that are different from other forms of chromium. Therefore, it is necessary to quantify potential exposures separately for hexavalent chromium and other forms of chromium. This has been accomplished as follows:

- For soil samples collected at the former Chromium Plating Facility that were analyzed for both total chromium and hexavalent chromium, the analytical results were compared and a ratio of hexavalent chromium concentration to total chromium concentration was calculated.

- The 95th percentile of the hexavalent chromium/total chromium ratio was then identified; the ratio is 0.88.
- The hexavalent chromium/total chromium ratio (0.88) was then applied to the total chromium EPCs for surface soil and subsurface soil at the former Chromium Plating Facility. The resulting value was used as the EPC for hexavalent chromium.
- The portion of total chromium that is not hexavalent chromium was assumed to be trivalent chromium (Cr[III]). The EPC for trivalent chromium was derived by subtracting the hexavalent chromium EPC from the total chromium EPC.
- This procedure was applied to soils at the former Chromium Plating Facility only. For all other media and exposure areas, total chromium was conservatively evaluated as hexavalent chromium.

Exposure Point Concentrations for Indoor Air

The EPCs for indoor air are based on the 27 rounds of indoor air data have been collected between 1999 and 2002 at sampling locations in the ten buildings included in the indoor air monitoring program. The indoor air data do not suggest increasing or decreasing trends in concentrations, but do show seasonal variation. Therefore, the value selected to represent the concentration at each sample location is the temporal average concentration. EPCs for each exposure point (i.e., each building) were developed using the temporal average concentrations for the air sample locations within each exposure point. The air sample locations included in each exposure point are identified in the EPC calculations provided in Appendix S-3.

EPCs for indoor air were developed using the approach described above for the other media. However, the only exposure point with at least 10 samples is the Future B-2. Therefore, RME EPCs for the Building B-2 exposure point are based on the 95% UCL values.

Exposure Point Concentrations for Dust in Ambient (Outdoor) Air

Whereas EPCs for evaluating exposures to soil, sediment, surface water, fish tissue, and groundwater are based on measured data, EPCs for evaluating exposures to dust that may be released from soil to ambient (outdoor) air are modeled. To evaluate dust inhalation exposures, the EPC of the COPC in air is calculated using a particulate emission factor (PEF) recommended by USEPA (USEPA, 1996b), which is provided in the exposure parameter tables.

11.2.3.3 Calculation of Intakes. COPC intakes via the ingestion and dermal contact exposure routes are calculated using the exposure parameters and EPCs identified previously. The quantified intakes for these exposure routes are combined with the appropriate dose-response data to quantify risks, as discussed in Subsection 11.4.

The equations used to calculate intake are those presented in USEPA guidance (USEPA, 1989; 2001c). The chemical-specific intake, or the average daily dose (ADD), is the amount of COPC absorbed into the body. When appropriate, it is the product of the average daily exposure and an absorption factor (ABS). Chemical-specific intakes were calculated in a manner consistent with USEPA guidance for risk assessment (USEPA, 1989; 2001c).

A Lifetime Average Daily Dose (LADD) is calculated in order to estimate carcinogenic risk. The Averaging Time (AT) over which the total intake of COPC is averaged is 70 years for carcinogenic effects (USEPA, 1989).

For noncarcinogenic effects, depending on the duration of the exposure period, an Average Daily Dose, Chronic (ADD_c) for long-term exposure (seven years or longer) or Average Daily Dose, Subchronic (ADD_s) for exposure periods from a month up to seven years may be calculated.

Soil and Sediment Direct Contact Exposures

The ADD received by a receptor via direct contact with soil or sediment (ADD_{soil}) is the sum of the ADDs for exposure via the routes of dermal contact with the contaminated soil or sediment and ingestion of the contaminated soil or sediment.

Ingestion. The ADD due to the incidental ingestion of COPC contaminated soil or sediment ($ADD_{ingestion}$) may be calculated:

$$ADD_{ingestion} = \frac{[COPC\,]_{soil} * IR * EF * FI * ED * C}{BW * AT}$$

Where:

$ADD_{ingestion}$ = Average daily dose of COPC received through the ingestion of soil or sediment during the period of exposure (dimensions: mass/mass×time, typical units: mg/kg×day).

$[COPC]_{soil}$ = EPC of the COPC in soil or sediment (dimensions: mass/mass, typical units: mg/kg).

IR = Daily soil or sediment ingestion rate on days exposed during the exposure period (dimensions: mass/time, typical units: mg/day)

EF = Number of exposure events during the exposure period divided by the number of days in the exposure period (dimensions: events/time, typical units: days/year).

ED = Duration of the exposure period (dimensions: time, typical units: years).

FI = Fraction ingested (unitless)

C = Appropriate units conversion factor(s)

BW = Body weight of the receptor of concern during the averaging period (dimensions: mass, typical units: kg).

AT = Averaging Time (dimension: time, typical units: days).

Dermal Contact. The ADD due to dermal contact with COPC-contaminated soil or sediment ($ADD_{dermal\ absorption}$) may be calculated:

$$ADD_{dermal\ absorption} = \frac{DAevent * SA * EF * ED}{BW * AT}$$

and:

$$DA_{event} = [COPC\,]soil * AF * ABS * C$$

Where:

$ADD_{dermal\ absorption}$ = Average daily dose of COPC received through dermal contact with soil or sediment during the period of exposure (dimensions: mass/mass×time, typical units: mg/kg×day).

DA_{event} = Dose of COPC absorbed during each exposure event (dimensions: mg/cm^2)

$[COPC]_{soil}$ = EPC of COPC in the soil or sediment at the exposure point during the period of exposure (dimensions: mg/kg)

SA = Skin surface area in contact with the soil or sediment on days exposed (dimensions: cm^2/day)

AF = Mass of soil or sediment adhered to the unit surface area of skin exposed (dimensions: mg/cm^2)

ABS = Absorption Factor; represents the fraction of COPC that may be absorbed through the skin from soil (unitless)

EF = Exposure Frequency: the number of exposure events during the exposure period divided by the number of days in the exposure period (dimensions: days/year)

ED = Exposure Duration: the period of time over which exposure may occur (dimension: years)

BW = Body Weight of the receptor of concern during the exposure duration dimension: kg)

AT = Averaging Time (dimension: days)

C = Appropriate units conversion factor(s)

Dermal Absorption Factor. The dermal absorption factor (unitless) describes the amount of COPC that may be absorbed through the skin and into the blood stream (i.e., amount that may become bioavailable) following dermal exposure to sediment. Among the COPCs evaluated in the HHBRA, ABS values are published by USEPA for arsenic (0.03), cadmium (0.001), benzo(a)pyrene (0.13; used for all PAHs), and PCBs (0.14) (USEPA, 2001c). USEPA Region I guidance (USEPA, 1999a) indicates that dermal absorption need not be quantitatively evaluated for COPCs for which a USEPA-published ABS value was not available. However, per request of CTDEP (see CTDEP Comment No. 4 on the Draft BRA), dermal exposures to COPCs without published dermal absorption values are calculated using ABS values of 10% for organics and 1% for inorganics.

Inhalation Exposures

Receptors at the Site, under certain conditions, have the potential to be exposed via inhalation to COPC adsorbed to wind-eroded particles or dust, or to vapors that may migrate from soil or groundwater to air. The methodology for evaluating inhalation exposures differs from that used for other exposure pathways in that the toxicity values used are reference concentrations (RfCs) and unit risks (URs) instead of reference doses (RfDs) and cancer slope factors (CSFs). Because concentration and not dose is the basis for these toxicity values, body weight (BW) is not used in calculating potential risk estimates for carcinogenic and noncarcinogenic chemicals. At the direction of CTDPH (CTDPH, 2002), respiration rate (IR) is used indirectly to derive an adjustment factor that accounts for higher inhalation rates that some receptors may have. The general equation for calculating chemical exposure via inhalation is as follows:

$$\text{Exposure Concentration} = \frac{CA \times ET \times EF \times ED \times IAF}{CF \times AT}$$

Where:

Exposure Concentration = Representative concentration of COPC in the air at the exposure point during the period of exposure (mg/m^3)

CA = concentration of the COPC in air (mg/m^3),
EF = exposure frequency (days/year),
ED = exposure duration (years),
ET = exposure time (hours/day)
IAF = Air Inhalation Adjustment Factor (derived as receptor-specific inhalation rate / 20 m^3/day)
CF = conversion factor (24 hours/day)
AT = averaging time (for carcinogens, AT = 70 years times 365 days per year; for noncarcinogens, AT = ED times 365 days per year).

Water Direct Contact Exposures

Potable contact with surface water involves ingestion and dermal contact exposures, and possible contact with groundwater involves dermal contact exposures. Exposures are quantified in accordance with USEPA guidance (USEPA, 1989; 1995; 2001c).

Ingestion. The ADD due to the incidental ingestion of COPC contaminated surface water ($ADD_{ingestion}$) may be calculated:

$$ADD_{ingestion} = \frac{[COPC\,]_{water} * IR * EF * ED * FI * C}{BW * AT}$$

Where:

$ADD_{ingestion}$ = Average daily dose of COPC received through the ingestion of water during the period of exposure (dimensions: mass/mass×time, typical units: mg/kg×day).
$[COPC]_{water}$ = EPC of the COPC in water (dimensions: mass/mass, typical units: mg/L).
IR = Daily water ingestion rate on days exposed during the exposure period (dimensions: mass/time, typical units: L/day)
EF = Number of exposure events during the exposure period divided by the number of days in the exposure period (dimensions: events/time, typical units: days/year).
ED = Duration of the exposure period (dimensions: time, typical units: years).
FI = Fraction Ingested (unitless)
C = Appropriate units conversion factor(s)
BW = Body weight of the receptor of concern during the averaging period (dimensions: mass, typical units: kg).
AT = Averaging Time (dimension: time, typical units: days).

Dermal. The ADD due to dermal contact with COPC-contaminated water ($ADD_{dermal\ absorption}$) may be calculated:

$$ADD_{dermalabsorption} = \frac{DAevent * EV * ED * EF * SA}{BW * AT}$$

and:

<u>For Inorganics</u>: $DA_{event} = Kp * [COPC]_{water} * t_{event}$

<u>For Organics when $t_{event} \leq t^*$</u>: $DA_{event} = 2 * FA * Kp * [COPC]_{water} *[6 * T_{event} * t_{event} / 3.14]$

<u>For Organics when $t_{event} > t^*$</u>: $DA_{event} = FA * Kp * [COPC]_{water} *[(t_{event} /1 + B) + 2 * T_{event} (1+ 3 * B + 3 * B^2 (1 + B)^2)]$

Where:

$ADD_{dermal\ absorption}$ = Average daily dose of COPC received through dermal contact with water during the period of exposure (dimensions: mass/mass×time, typical units: mg/kg×day).

$[COPC]_{water}$ = EPC of COPC in the water at the exposure point during the period of exposure (dimensions: mg/L)

DA_{event} = Dose absorbed per event (mg/cm^2-event)

SA = Skin surface area in contact with the water on days exposed (dimensions: cm^2/day)

Kp = Permeability Constant; represents the amount of COPC that may be absorbed through the skin from water (units: cm/hr)

t_{event} = Number of hours per day the exposure occurs (dimensions: time, typical units: hours/event)

T_{event} = Lag time per event (hr/event)

FA = Fraction absorbed (unitless)

t* = Time to reach steady state (hr)

B = Ratio of the permeability coefficient of a compound through the stratum corneum relative to its permeability coefficient across the epidermis (dimensionless)

EF = Exposure Frequency: the number of exposure events during the exposure period divided by the number of days in the exposure period (dimensions: days/year)

EV = Event Frequency: the number of exposure events per day (dimensions: events/day)

ED = Exposure Duration: the period of time over which exposure may occur (dimension: years)

BW = Body Weight of the receptor of concern during the exposure duration dimension: kg)

AT = Averaging Time (dimension: days)

Values for FA, Kp, T_{event}, t_{event}, t*, and B were obtained from RAGS Part E (USEPA, 2001c).

<u>Fish Ingestion</u>

The ADD due to ingestion of COPC contaminated fish or shellfish tissue ($ADD_{ingestion}$) may be calculated:

$$ADD_{ingestion} = \frac{[COPC\,]_{fish} * IR * EF * ED * FI * C}{BW * AT}$$

Where:

$ADD_{ingestion}$ = Average daily dose of COPC received through the ingestion of fish during the period of exposure (dimensions: mass/mass×time, typical units: mg/kg×day).
$[COPC]_{fish}$ = EPC of the COPC in fish (dimensions: mass/mass, typical units: mg/kg).
IR = Daily fish ingestion rate on days exposed during the exposure period (dimensions: mass/time, typical units: mg/day)
EF = Number of exposure events during the exposure period divided by the number of days in the exposure period (dimensions: events/time, typical units: days/year).
ED = Duration of the exposure period (dimensions: time, typical units: years).
FI = Fraction ingested (unitless)
C = Appropriate units conversion factor(s)
BW = Body weight of the receptor of concern during the averaging period (dimensions: mass, typical units: kg).
AT = Averaging Time (dimension: time, typical units: days).

11.3 TOXICITY ASSESSMENT

The objective of the dose-response assessment is to quantify the relationship between the intake, or dose, of COPCs and the likelihood that an adverse health effect may result from exposure to the COPCs. There are two major types of adverse health effects evaluated in the HHBRA: non-carcinogenic, and carcinogenic. Following USEPA guidance (USEPA, 1989), these two effects (non-carcinogenic and carcinogenic) are evaluated separately.

There are two types of dose-response values: cancer slope factors (CSFs) and unit risk (UR) values for carcinogens, and RfDs and RfCs for non-carcinogens. For potentially carcinogenic COPCs, both types of values (CSF/UR and RfD/RfC values) have been developed by USEPA because these COPCs may elicit both carcinogenic and noncarcinogenic (systemic) effects. In addition, because toxicity and/or carcinogenicity can depend on the route of exposure (i.e., oral or inhalation), unique dose-response values have been developed for the oral, dermal, and inhalation exposure routes.

11.3.1 Dose-Response Values for Carcinogenic Effects

It has been generally assumed historically that carcinogenic effects are non-threshold effects. This means that any dose, no matter how small, is assumed to pose a finite probability of generating a response. Thus, no dose of a carcinogen is thought to be risk-free. For carcinogenic effects, USEPA uses a two-part evaluation in which the substance is first assigned a weight-of-evidence classification, and then a slope factor (SF) or UR is calculated to reflect the carcinogenic potency.

The weight-of-evidence evaluation involves determining the likelihood that the agent is a human carcinogen. USEPA has developed a system for characterizing the overall weight of evidence for a chemical's carcinogenicity based on the availability of animal, human, and other supportive data (USEPA, 1989). The weight-of-evidence classification rates the likelihood that an agent is a human carcinogen. It qualitatively affects the interpretation of potential health risks. Three major factors are considered in characterizing the overall weight-of-evidence for carcinogenicity: (1) the quality of evidence from human studies, (2) the quality of evidence from animal studies, and (3) other supportive information, such as mutagenicity data and structure-activity data.

USEPA's final classification of the overall weight-of-evidence has the following five categories:

Group A - Human Carcinogen. This category indicates there is sufficient evidence from epidemiological studies to support a causal association between an agent and human cancer.

Group B - Probable Human Carcinogen. This category generally indicates there is at least limited evidence from epidemiologic studies of carcinogenicity to humans (Group B1) or that, in the absence of data on humans, there is sufficient evidence of carcinogenicity in animals (Group B2).

Group C - Possible Human Carcinogen. This category indicates that there is limited evidence of carcinogenicity in animals in the absence of data on humans.

Group D - Not Classified. This category indicates that the evidence for carcinogenicity in animals is inadequate.

Group E - No Evidence of Carcinogenicity to Humans. This category indicates that there is evidence of noncarcinogenicity in at least two adequate animal tests in different species or in both epidemiologic and animal studies.

USEPA's draft revised guidelines for cancer risk assessment (USEPA, 1999c) have been adopted as agency policy for cancer risk assessment. These guidelines contain a revised classification system for carcinogenic effects with the following classifications:

- Carcinogenic to humans
- Likely to be carcinogenic in humans
- Suggestive evidence of carcinogenicity, but not sufficient to assess human carcinogenic potential
- Data inadequate for an assessment of human carcinogenic potential
- Not likely to be carcinogenic in humans

In IRIS, the weight of evidence classification for a given chemical may reflect either of the two classification schemes identified above.

CSF and UR values are typically calculated for chemicals in Groups A, B1, and B2. Cancer dose-response values for chemicals in Group C are calculated on a case-by-case basis. The CSF is an estimate of the upper 95% Confidence Limit of the slope of the dose-response curve extrapolated to low doses.

For some chemicals, human epidemiologic data are the basis of an estimate of the carcinogenic potency, although the most common basis of these values is an animal study. The CSF is given in units of $(mg/kg/day)^{-1}$ and is based upon the concept of a lifetime average daily dose. Oral CSFs are used to estimate the risks associated with exposure to carcinogens via ingestion. No CSFs are available for the dermal route of exposure, but are instead calculated from oral CSFs using the methodology described below.

The UR is the upper 95% Confidence Limit of the mean incremental lifetime cancer risk estimated to result from lifetime exposure to an agent if it is in the air at a concentration of 1 ug/m^3. In this HHBRA, inhalation URs are used to estimate the incremental risks associated with inhalation of carcinogenic COPCs in indoor air and dust emissions...

The CSF and UR values and supporting documentation for the chemicals selected as COPCs in the risk assessment are provided in Tables 11-30 and 11-32.

11.3.2 Dose-Response Values for Non-Carcinogenic Effects

In contrast to carcinogens, noncarcinogens are believed to have threshold exposure levels below which adverse effects are not expected. USEPA has derived standards and guidelines based on acceptable levels of exposure for such compounds. Noncarcinogenic effects of concern on which many of the standards and guidelines are based include liver toxicity, reproductive effects, neurotoxicity, teratogenicity, and other chronic toxicities. Various criteria have been developed from experiments that can be used to estimate the dose-response relationship of noncarcinogens. Some of the same uncertainties involved in deriving cancer risk estimates (namely, selection of an appropriate data set and extrapolation of high-dose animal data to low-dose human exposure) are also involved in deriving noncarcinogenic dose-response criteria. Dose-response values used most often to evaluate noncarcinogenic effects are RfDs.

The RfD, expressed in units of mg/kg/day, is defined as an estimate (with uncertainty spanning perhaps an order of magnitude or greater) of a daily exposure level for the human population, including sensitive subpopulations, that is likely to be without an appreciable risk of deleterious effects during a lifetime (USEPA, 1989). When available, the RfD is the dose-response criterion most appropriate for quantitatively estimating noncarcinogenic effects. The RfD is derived from the following equation:

$$RfD\ (mg/kg/day) = \frac{NOAEL\ or\ LOAEL}{UF\ and/or\ MF}$$

The No Observable Adverse Effect Level (NOAEL) represents the dose of a chemical at which there are no statistically or biologically significant differences in the frequency of an adverse effect between the exposed population and its appropriate control. The Lowest Observable Adverse Effect Level (LOAEL) represents the lowest dose at which a statistically significant difference in the frequency of an effect is noted. Both the NOAEL and the LOAEL are reported in terms of mg/kg/day. An uncertainty factor (UF) of ten per type of uncertainty is used to account for interspecies and intraspecies differences, severity of the adverse effect, whether the dose was an NOAEL or an LOAEL, and the adequacy of the data. The magnitude of the UF will therefore vary from chemical to chemical, ranging from 10 to 10,000. A modifying factor (MF), ranging from less than 0 to 10 may also be added to reflect qualitative uncertainties not explicitly addressed in the UFs. The toxicity endpoint upon which the RfD is derived and the UF and/or MF used in the calculation are presented in the dose-response tables. No RfDs are available for the dermal route of exposure but are instead calculated from oral RfDs using the methodology described below.

The Reference Concentration (RfC, in units of mg/m^3) is analogous to the RfD and is developed through a similar process. However, unlike RfDs, which represent a dose (in mg/kg/day) at which adverse or deleterious effects are unlikely, RfCs represent air concentrations (in mg/m^3) at which adverse or deleterious effects are unlikely (i.e., an air concentration corresponding to a HI = 1.0). In this HHBRA, inhalation RfCs are used to estimate the hazards associated with inhalation of COPCs in indoor air and dust emissions .

The use of chronic RfDs to evaluate the potential for adverse health effects resulting from substantially less-than-lifetime exposures may be overly protective. Subchronic Reference Doses and Subchronic Reference Concentrations (RfD_ss/RfC_ss) have been developed for many chemicals to evaluate the potential noncarcinogenic effects of limited duration exposures. Subchronic RfD_s are similar to chronic RfDs; the distinction is the length of exposure duration. In this HHBRA, all risks are calculated using chronic RfDs because the exposure duration for each of the receptors evaluated is of a chronic nature (i.e., more than a few years).

The RfD and RfC values and supporting documentation for the chemicals selected as COPCs are provided in Tables 11-31 and 11-33.

11.3.3 Adjustment for Dermal Exposure

Cancer CSFs and non-cancer RfDs were developed to evaluate risk associated with the dermal contact exposure route. In accordance with USEPA guidance (USEPA, 2001c), dermal dose-response values are calculated from oral dose-response values using an oral absorption factor. The oral absorption factor represents the amount of substance that is absorbed from the gastrointestinal tract following oral administration of a substance. The absorbed dose represents the amount of substance that is potentially available for biological interaction; it is by this dose-response relationship that the toxicity of a dermally absorbed substance must be evaluated.

Thus, for potentially carcinogenic substances, the dermal dose-response value is calculated as follows:

$$SF_d = SF_o \; / \; Oral\ ABS$$

The dermal dose-response value for evaluating non-carcinogenic effects is calculated as follows:

$$RfD_d = RfD_o \; X \; Oral\ ABS$$

Chemical-specific oral ABS values for are published by USEPA (USEPA, 2001c). In accordance with USEPA guidance (USEPA, 2001c), oral dose-response values are only adjusted using an oral ABS value if the COPC has an oral ABS value less than 50%. Otherwise, the oral dose-response value is used as the dermal dose-response value. Dermal SFs and RfDs are presented in Tables 11-30 (CSFs) and 11-31 (RfDs).

11.3.4 Sources of Dose-Response Values

The following hierarchy of sources for dose-response values has been utilized in identifying dose-response values for this HHBRA (USEPA, 2003a).

Tier 1- Integrated Risk Information System (IRIS) (http://www.epa.gov/iris/). In accordance with USEPA guidance, the main source of dose-response values is the USEPA Integrated Risk Information System (IRIS), which is a database established by USEPA containing all validated data on many toxic substances found at hazardous waste Sites. This database, current as of August 2004 (USEPA, 2004b), was used to identify the SFs, URs, RfDs, and RfCs applied in this risk assessment (USEPA, 2003a).

Tier 2- National Center for Environmental Assessement (NCEA)'s peer reviewed toxicity values (PRTVs). NCEA's PRTVs are developed by the Superfund Technical Support Center (STSC) for the EPA Superfund program. STSC's reassessment of (Health Effects Assessment Summary Table (HEAST) toxicity values, as well as development of PRTVs in response to Regional or Headquarters Superfund program requests, are consistent with Agency practices on toxicity value development, use the most recent scientific literature, and are supported by both internal and external peer review, providing a high level of confidence in the use of these values in the Superfund Program. Tier 2 values were obtained from the USEPA Region III Risk-Based Concentration Table, current as of April, 2004 (USEPA, 2004a).

Tier 3 - Other toxicity values

- California (Cal) EPA's toxicity values. Cal EPA develops toxicity values for both cancer and noncancer effects. Cal EPA toxicity values are obtained on the Cal EPA website at http://www.oehha.ca.gov/risk/chemicalDB//index.asp.

- ATSDR's MRLs address noncancer effects only, and are available on the ATSDR website at http://www.atsdr.cdc.gov/mrls.html.

- Toxicity values remaining in current versions of HEAST (1997a).

In this HHBRA, the majority of dose-response values used are published in IRIS. For some Site-related COPCs required dose-response data are only available as NCEA provisional values or from CAL-EPA. These dose-response values were used in this HHBRA in order to provide a more complete evaluation of potential risks.

For some COPCs, chronic inhalation RfCs are not available in the USEPA-recommended sources. USEPA Region I does not recommend route-to-route extrapolation of dose-response data (i.e., using an oral dose-response value to evaluate inhalation risks). Uncertainties related to the absence of dose-response data, particularly for COPCs for which the exposure pathway that represents the only pathway or most significant exposure pathway has no toxicity criterion, are discussed in the risk assessment uncertainty analysis.

Evaluation of Chromium in Site Media

The most common forms of chromium in environmental media are chromium III (trivalent chromium) and chromium VI (hexavalent chromium). Exposures to hexavalent chromium have been associated with carcinogenic effects via inhalation, and chronic non-cancer health effects via ingestion, dermal contact, and inhalation. The inhalation unit risk of 0.012 per ug/m^3 (IRIS) has been applied to estimated average lifetime air concentrations to characterize potential carcinogenic effects for inhalation of soil particulates. The chronic oral RfD of 0.003 mg/kg/day (IRIS) and the subchronic oral RfD of 0.02 mg/kg/day (HEAST) has been applied to the estimated daily doses in order to characterize potential non-cancer risks for ingestion and dermal contact with chromium in sediment, surface water, and biota. This approach provides a conservative estimate of potential health risks because it quantifies risks associated with chromium under the assumption that all chromium is present in the more toxic hexavalent form. Hexavalent chromium RfCs are published for chromium particulates and chromic acid mists. The chronic RfC for chromium particulates of 0.0001 mg/m^3 (IRIS) has been applied to the estimated daily air concentrations to characterize potential non-carcinogenic risks for inhalation of soil particulates. Exposures to trivalent chromium are not known to be associated with carcinogenic effects. The chronic oral RfD of 1.5 mg/kg/day (IRIS) has been used to evaluate non-carcinogenic effects. No inhalation RfC values have been published for trivalent chromium.

Evaluation of Cyanide in Site Media

In this risk characterization, the oral RfD for free cyanide of 0.02 mg/kg/day (published in IRIS) has been applied to characterize potential non-cancer risks. This approach assumes that all cyanide measured as total cyanide is either free cyanide or is capable of releasing free cyanide in the gastrointestinal tract.

Evaluation of 1,2-DCE in Site Media

1,2-DCE analyses were reported as cis- and trans-1,2-DCE and/or total 1,2-DCE (which is the total of cis- and trans-isomers). Separate RfD values are published for cis-, trans- and total 1,2-DCE isomers. For exposure points where cis- and trans- isomer data are available and sufficient (i.e., an equal or larger number of samples were analyzed for cis- and trans- isomers than for total 1,2-DCE), only the cis- and trans-1,2-DCE isomers are evaluated in the risk characterization using the cis- and trans-1,2-DCE specific RfD values. For exposure points where 1,2-DCE is reported as total 1,2-DCE and cis- and trans- isomer data are not available or are not sufficient, only the total 1,2-DCE EPCs are used in the risk characterization using the RfD for total 1,2-DCE (HEAST, 1997). EPCs for cis/trans and total 1,2-DCE were not used for the same exposure point to avoid double-counting risks associated with 1,2-DCE.

Evaluation of Lead in Site Media

No dose-response values are published for potential exposures to lead. In the absence of dose-response data, USEPA recommends use of lead biokinetic uptake models to evaluate potential lead exposures, and comparison of the lead intake estimated using the models to threshold blood lead levels for children and adults. Lead is screened as a potential COPC by comparing the maximum detected concentration to USEPA's Interim Soil Lead Screening Value of 400 mg/kg (USEPA, 1994d), which is considered by USEPA to be protective for residential exposures to lead in soil. The Office of Solid Waste and Emergency Response (OSWER) screening values are used to evaluate potential risks associated with lead exposure at these areas; comparisons of lead EPCs to the OSWER screening values are provided in the risk characterization (Subsection 11.4). If EPCs exceed the OSWER screening value, lead exposures are evaluated using the USEPA's biokinetic uptake model for lead. Specifically, the adult lead model (ALM; USEPA, 2003b) is used to evaluate potential exposures and risks associated with lead in soil. Documentation of the ALM is provided in Appendix S-4.

Evaluation of Mercury in Site Media
Mercury may exist as elemental mercury, inorganic mercury salts, and organic mercury. Typically, mercury is present in environmental media as inorganic mercury salts or organic mercury (methyl mercury) that may be produced by bacterial methylation of inorganic mercury. Methyl mercury is known to bioaccumulate. In this HHBRA, mercury detected in fish is evaluated using oral dose-response values for organic mercury (methyl mercury), whereas mercury detected in other media (e.g. sediment) is evaluated using oral dose-response values for inorganic mercury. The only inhalation RfC that is published for mercury is based on dose-response data for elemental mercury

11.4 RISK CHARACTERIZATION

Risk characterization, including uncertainty analysis, is the final step in the risk assessment process. The risk characterization integrates the exposure and toxicity information generated in previous sections to qualitatively or quantitatively evaluate the potential health risks associated with exposure to chemicals at the site. Risk estimates are then evaluated through a comparison to USEPA risk threshold criteria. The relative confidence in risk estimates may be further evaluated in the uncertainty analysis (Subsection 11.4.3).

11.4.1 Risk Characterization Methods

Quantitative estimates of both carcinogenic and noncarcinogenic risks are calculated for each complete exposure scenario selected for evaluation in the exposure assessment, in accordance with USEPA (1989) guidance. Methods of quantifying cancer and non-cancer risks, and summing total pathway risks, are discussed below.

Carcinogenic Risks. Cancer risks associated with exposure to each COPC are estimated by multiplying the exposure route-pathway specific intake (e.g., oral exposure to soil) by its exposure route-specific CSF (e.g., oral CSF). The calculated value is an ELCR and represents an upper bound of the probability of an individual developing cancer over a lifetime as the result of exposure to a COPC.

This process is repeated for all exposure pathways for a given exposure point (e.g., oral and inhalation exposures to groundwater).

Noncarcinogenic Risks. Non-cancer risk estimates are calculated by dividing the COPC intake for each exposure pathway by the appropriate RfD. The result is called the hazard quotient (HQ). The hazard index (HI) is the sum of the chemical-specific HQs for each exposure pathway.

An HI less than 1 indicates that noncarcinogenic toxic effects are unlikely to occur as a result of COPC exposure. HIs greater than 1 may be indicative of a possible noncarcinogenic toxic effect. As the HI increases, so does the likelihood that adverse effects might be associated with exposure. This determination is necessarily imprecise because the RfD is developed using uncertainty factors (uncertainty factors of 10 or greater are not uncommon) to be protective of human health. It is not at all certain, therefore, that an intake that exceeds the RfD would mean that adverse effects would be expected.

Summary. The risk estimates calculated for each receptor scenario are summarized in risk summary tables according to receptor exposure scenario. Risks are summed across all COPCs for each exposure route and each exposure point. Risks across multiple exposure points and multiple exposure media are then summed to yield cumulative cancer and non-cancer risk estimates for the receptor.

Within the risk characterization for each receptor scenario, the relative significance of the risk for each pathway, exposure point, and receptor scenario was evaluated in terms of a comparison with acceptable risk levels established by USEPA and CTDEP. The USEPA guidelines, established in the NCP, indicate that the total excess lifetime cancer risk due to exposure to the chemicals at a site, by each complete exposure pathway, should not exceed a range of 1 in 1,000,000 (1E-06) to 1 in 10,000 (1E-04) (USEPA, 1990b). Risks between 1E-06 and 1E-04 should be considered on a case-by-case basis during the risk management process. The CTDEP regulations state that for a release area polluted with multiple substances, the cumulative excess lifetime cancer risk for all carcinogenic substances should not exceed 1E-05 (22a-133k-2(d)(2)). For reference, the average cancer burden in the United States in 1993 was 1 in 3 for women and 1 in 2 for men (American Cancer Society, 1994).

According to the NCP, for non-cancer effects, the acceptable risk is associated with chemical concentrations that people (including sensitive individuals such as children) can be exposed to with an adequate margin of safety without adverse effects occurring. This level is generally interpreted by USEPA to be an HI of 1 or less, although values greater than 1 may also be acceptable. The CTDEP regulations state that for a release area polluted with multiple substances, the cumulative hazard index should not exceed 1 for non-carcinogenic substances with the same target organ (22a-133k-2(d)(2)).

The HI that is calculated by summing the HQs for all COPCs across all media is called the screening HI. The screening HI generally provides an overestimation of potential non-cancer risks. This is because the HQ for each COPC represents the ratio of the estimated COPC intake to the threshold dose for a *specific* adverse health effect, where the adverse health effect is determined by the basis of the RfD. Summing HQs that are based on risks for different adverse health effects does not provide an estimate of total risk for a specific adverse health effect. Therefore, according to USEPA guidance (USEPA, 1989), a screening hazard index that is above one and is based on exposures to multiple COPCs does not necessarily indicate that the potential for adverse health effects is unacceptable if the risks for the COPCs are not additive. A determination of whether HQs for multiple COPCs are additive can be made by examining the target organs that each COPC potentially exerts adverse effects on. The HQs for COPCs that affect the same target organ(s) are considered to be additive.

Consistent with USEPA guidance (USEPA, 2001b) and CTDEP regulations (22a-133k-2(d)(2)), a determination of whether the HQs for exposures to COPCs are additive was performed by segregating the HI according to the target organs or organ systems that each COPC effects. HQs are considered additive for those COPCs which have RfDs that are based on effects to the same target organ or organ system. The risk summary tables that present the HI values for each receptor scenario also provide a summary of the target organ or organ system associated with the dose-response relationship that the RfD for each COPC is based on. Using those data, the HQs for COPCs that have RfDs based on effects to the same target organ or organ systems were added together to obtain the target organ-specific HIs. Target organ-specific HIs are presented in the risk summary tables in Appendix S-4.

11.4.2 Risk Characterization Results

This section provides a discussion of risk characterization results for each receptor scenario. Cumulative receptor risks associated with possible exposures to multiple exposure media (e.g., groundwater, surface water, and sediment) are also discussed in this section. The risk calculation spreadsheets and risk summary tables for the exposure scenarios evaluated in this HHBRA are discussed below and are provided in USEPA Risk Assessment Guidance for Superfund (RAGS) Part D reporting format in Appendix S-4.

For all exposure scenarios, the risk characterization was performed for the RME scenarios. Risk characterizations for the CT scenarios were only performed for RME scenarios with cancer risks in excess of the CTDEP risk limit of 1E-05 or HI values in excess of 1.

Table 11-12 provides a summary of the exposure points that were evaluated for each receptor scenario. Separate exposure scenarios were evaluated for surface soil exposures and subsurface exposures, and risks for surface soil and subsurface soil were not added together. This is because soil contact rates (e.g., ingestion rates) are based on total soil ingestion per day (24-hour period). Therefore, adding risks for exposures to surface soil and subsurface soil would result in double-counting daily exposures to soil.

As indicated in Subsection 11.2, there are ten buildings at the Site in which indoor air data have been collected. The highest EPCs of each of the COPCs are associated with Building B-2 and Building B-6. Characterization of potential risks at these buildings provides an assessment that is conservative for potential exposures at the other facility buildings.

Cancer risks and non-cancer hazards for each scenario are summarized in Tables 11-34 through 11-39. COPCs that individually contribute cancer risk greater than 1E-06 or an HQ greater than 1 at an exposure point are identified in the risk characterization discussion and are summarized in Table 11-40.

11.4.2.1 Commercial/Industrial Worker - Future Land Use. Future commercial/industrial worker scenarios were evaluated for the Main Site, Chromium Plating Facility, and South Lot exposure areas. Within each exposure area, risks were evaluated for potential incidental ingestion, dermal contact, and dust inhalation exposure to surface soil or subsurface soil. Risks were also calculated for exposures to indoor air (based on an industrial worker indoors 8 hours per day, 250 days per year). To streamline the risk assessment, indoor air risks were calculated only for Buildings B-2 and B-6, because these buildings had the highest EPCs among buildings at the Site. Cumulative risks for each exposure area were estimated by combining risks for surface soil or subsurface soil with risks for indoor air. Since Building B-2 is located at the Main Site, cumulative risks for the Main Site and Chromium Plating Areas were calculated by summing the indoor air risks for Building B-2 with soil risks for the Main Site and Chromium Plating Areas. Since Building B-6 is located at the South Lot area, cumulative risks for the South Lot were calculated by summing the indoor air risks associated with Building B-6 with surface soil or subsurface soil risks for the South Lot area. Risk estimates are summarized in Table 11-34.

Main Site

- The cancer risk estimates for direct contact with surface soil (5E-05) or subsurface soil (7E-05) are within the USEPA cancer risk range of 1E-06 to 1E-04, but exceed the CTDEP cumulative risk limit of 1E-05. Cancer risks are associated with arsenic and PAHs.
- The non-cancer screening HI values for direct contact with surface soil (0.6) or subsurface soil (0.6) are below the USEPA and CTDEP risk threshold values of 1. Non-cancer HI values are associated with Aroclors, arsenic, and chromium.
- The cancer risk estimate for inhalation of indoor air at Building B-2 is 5E-04, which exceeds the USEPA cancer risk range of 1E-06 to 1E-04 and CTDEP cumulative risk limit of 1E-05. Cancer risks are primarily associated with TCE.
- The non-cancer screening HI value for inhalation of indoor air at Building B-2 is 0.4, which is below the USEPA and CTDEP risk threshold values of an HI of 1.
- The inhalation risks associated with background conditions are an ELCR of 1E-05 and a HI of 0.008. The inhalation risks associated with air at Building B-2 exceed the inhalation risks associated with background conditions.
- Cumulative HI values for exposure to surface soil and indoor air (1), or subsurface soil and indoor air (1), do not exceed the threshold limit (HI of 1).
- Cumulative risk estimates for exposure to surface soil and indoor air (6E-04), or subsurface soil and indoor air (6E-04), exceed the USEPA cancer risk range and CTDEP cancer risk limit.
- Lead concentrations in surface soil and subsurface soil at the Main Site do not pose health risks of concern as indicated by estimated blood-lead levels in adult workers that do not exceed the threshold blood lead levels. Specifically, as documented in Appendix S-4, the 95th percentile blood-lead levels that would potentially result in a full-time worker exposed to lead at the EPCs in surface soil (459 mg/kg) and subsurface soil (752 mg/kg) are 8.4 micrograms per deciliter (ug/dl), and 9.9 ug/dl, respectively, which are below the threshold blood lead level of 10 ug/dl established by the USEPA.

Chromium Plating Facility

- The non-cancer screening HI values for direct contact with surface soil (0.4) or subsurface soil (0.3) are below the USEPA and CTDEP risk threshold values of an HI of 1. The HI is primarily associated with hexavalent chromium.
- The cancer risk estimates for direct contact with surface soil (1E-06) or subsurface soil (7E-07) are within the USEPA cancer risk range of 1E-06 to 1E-04 and are below the CTDEP cumulative risk limit of 1E-05.

- Cumulative HI values for exposure to indoor air at Building B-2 and surface soil (0.8), or subsurface soil (0.7), are below 1.
- Cumulative risk estimates for exposure to indoor air at Building B-2 and surface soil (5E-04), or subsurface soil (5E-04), exceed the USEPA cancer risk range and CTDEP cumulative cancer risk limit due to the risks associated with indoor air. As discussed previously, cancer risks associated with indoor air are attributable to TCE.

South Lot

- The cancer risk estimates for direct contact with surface soil (7E-05) or subsurface soil (5E-06) are within the USEPA cancer risk range of 1E-06 to 1E-04, but the surface soil risks exceed the CTDEP cumulative risk limit of 1E-05. Cancer risks are primarily associated with PAHs.
- The non-cancer screening HI values for direct contact with surface soil (0.06) or subsurface soil (0.04) are below the USEPA and CTDEP risk threshold values of 1.
- The cancer risk estimate for inhalation of indoor air at Building B-6 is 3E-05, which does not exceed the USEPA cancer risk range of 1E-06 to 1E-04, but does exceed the CTDEP cumulative risk limit of 1E-05. Cancer risks are primarily associated with TCE.
- The non-cancer screening HI value for inhalation of indoor air at Building B-6 is 0.02, which is below the USEPA and CTDEP risk threshold values of an HI of 1.
- Cumulative HI values for exposure to surface soil and indoor air (0.08), or subsurface soil and indoor air (0.06), do not exceed the threshold limit (HI of 1).
- Cumulative risk estimates for exposure to surface soil and indoor air (1E-04), or subsurface soil and indoor air (4E-05), do not exceed the USEPA cancer risk range, but do exceed the CTDEP cancer risk limit.

Summary

The results of the HHBRA indicate that health risks associated with potential exposures to COPCs in indoor air and soil under possible future commercial/industrial land use conditions do not exceed an HI of 1 (highest HI = 1), but exceed the CTDEP cancer risk limit of 1E-05 and, with the exception of the South Lot Area, exceed the USEPA cancer risk range of 1E-06 to 1E-04. Cancer risks in excess of USEPA limits are due to TCE in indoor air, where as cancer risks in excess of CTDEP limits are due to PAHs and arsenic in soil, as well as TCE in indoor air. Non-cancer risks are attributable to Arolors, arsenic, and hexavalent chromium.

11.4.2.2 Recreational Visitor - Future Land Use. Future recreational visitor scenarios were evaluated for the Intertidal Flats, Marine Basin, and Outfall 008 Drainage area. Within each exposure area, risks were evaluated for potential incidental ingestion and dermal contact exposures to surface water and sediment during wading. Dermal contact with shallow groundwater in the interstitial sediment pore space was also evaluated at the Intertidal Flats. Ingestion of finfish and shellfish associated with recreational angling were evaluated at the Intertidal Flats and Marine Basin.

For each exposure point, cancer risks were summed among the recreational visitor subpopulations to derive a cumulative receptor cancer risk estimate. Because non-cancer HI values are not additive among receptor subpopulations, the highest HI value among the child, adolescent, and adult receptors was reported as the total receptor HI (this was typically the HI for the young child receptor).

Risk estimates are summarized in Table 11-35.

Intertidal Flats – Recreational Angling/Shellfishing

- The total receptor cancer risk estimate for direct contact with surface water (9E-07), sediment (1E-05), and groundwater (3E-05) of 4E-05 exceeds the CTDEP risk limit of 1E-05, but is within the USEPA cancer risk range of 1E-06 to 1E-04. However, the only exposure medium with a cancer risk that exceeds the risk limit is groundwater. Cancer risks in groundwater are attributable to vinyl chloride, and cancer risks in sediment are attributable to Aroclor-1248 and Aroclor-1254.
- The total receptor non-cancer screening HI value for direct contact with surface water (0.2), sediment (1), and groundwater (0.05) does not exceed the USEPA or CTDEP risk threshold value of 1.

Intertidal Flats – Recreational Finfish or Shellfish Consumption

- The total receptor cancer risk estimate for ingestion of finfish (6E-05) exceeds the CTDEP risk limit of 1E-05, but is within the USEPA cancer risk range of 1E-06 to 1E-04. The non-cancer HI values are greater than one for the young child, older child, and adult (HI value of 5 for the young child). Cancer and non-cancer risks are attributable to PCBs, primarily Aroclor-1254 and Aroclor-1260.
- The total receptor cancer risk estimate for ingestion of ribbed mussels (1E-04) exceeds the CTDEP risk limit of 1E-05, but is within the USEPA cancer risk range of 1E-06 to 1E-04. The non-cancer HI values are greater than one (HI value of 3 for the young child). Cancer risks are primarily associated with arsenic and to a lesser extent PCBs (Aroclor-1254 and Aroclor-1260). Non-cancer HI values are attributable to PCBs.
- The total receptor cancer risk estimate for ingestion of oysters (2E-04) is greater than the CTDEP risk limit of 1E-05, and exceeds the USEPA cancer risk range of 1E-06 to 1E-04. The non-cancer HI values are greater than one for each receptor (HI value of 22 for the young child). Cancer risks are entirely associated with PCBs (Aroclor-1254). Non-cancer HI values are attributable to Aroclor-1254, and to a lesser extent cadmium.
- The CT risks for recreational consumption of finfish and shellfish are shown in Table 11-37. Total receptor cancer risks for the CT scenario do not exceed the CTDEP risk limit of 1E-05 for finfish and ribbed mussles. However, the total receptor CT cancer risk estimate for ingestion of oysters (5E-05) exceeds the CTDEP risk limit, but is within the USEPA cancer risk range of 1E-06 to 1E-04. Non-cancer HI values are slightly lower than for the RME scenario, but still remain greater than 1.
- As shown in Table 11-39, risks associated with recreational consumption of finfish and ribbed mussels at the Intertidal Flats are less than or equal to risks associated with recreational consumption of finfish and ribbed mussels at background reference locations. This indicates that risks associated with recreational consumption of finfish and shellfish taken from the Intertidal Flats are no different than risks associated with recreational consumption of finfish and shellfish taken from locations that have not been affected by releases from the Site. It is important to note that recreational consumption of finfish and ribbed mussels taken from the Intertidal Flats in the area sampled is not known to occur; risks represent a hypothetical future land use activity.

- Risks associated with consumption of oysters taken from the Intertidal Flats are approximately two-times greater than risks associated with consumption of oysters taken at background reference locations (Table 11-39). This is due to differences in Aroclor-1254 concentration in oysters collected at the Intertidal Flats and oysters collected at background reference locations. It is important to note, however, that the EPCs for oysters are based on analyses of seed oysters that would not be consumed by humans; humans would only be potentially exposed to oysters from this area after they have been placed in Long Island Sound for a few years to grow and mature.

Marine Basin – Recreational Wading

- The total receptor cancer risk estimate for direct contact with surface water (5E-07) and sediment (2E-06) of 2E-06 is below the CTDEP risk cancer risk limit of 1E-05, and is within the USEPA cancer risk range of 1E-06 to 1E-04.
- The non-cancer screening HI values for direct contact with surface water (0.02) and sediment (0.03) do not exceed the CTDEP or USEPA risk threshold value of 1.
- The cumulative HI value for exposure to surface water and sediment is 0.04, which does not exceed 1.
- The cumulative risk estimate for exposure to surface water and sediment is 2E-06, which is below the CTDEP risk limit, and is within the USEPA cancer risk range of 1E-06 to 1E-04.

Marine Basin – Recreational Finfish or Shellfish Consumption

- The total receptor cancer risk estimate for ingestion of finfish (3E-05) exceeds the CTDEP risk limit of 1E-05, but is within the USEPA cancer risk range of 1E-06 to 1E-04. The non-cancer HI values are greater than one for the young child and adult (HI value of 2 for each receptor). Cancer and non-cancer risks are attributable to PCBs (Aroclor-1254 and Aroclor-1260).
- The total receptor cancer risk estimate for ingestion of ribbed mussels (1E-04) is greater than the CTDEP risk limit of 1E-05, but is within the USEPA cancer risk range of 1E-06 to 1E-04. Cancer risks are primarily associated with arsenic and to a lesser extent PCBs (Aroclor-1254). The non-cancer screening HI values are greater than 1 (HI value of 2 for the young child). However, the HQs for individual COPCs do not exceed 1; the largest contributors to non-cancer risk are Aroclor-1254 (HQ=0.8) and arsenic (HQ=0.7) are associated with target organ effects that are not additive (Aroclor-1254: immune system; arsenic: skin). Therefore, the target organ-specific HI values do not exceed 1.
- The cumulative cancer and non-cancer risk estimates for exposure to surface water and sediment (while wading or collecting shellfish), do not add appreciably to the risk estimates for consumption of fish and shellfish, as indicated in Table 11-35.
- The CT risks for recreational consumption of finfish and shellfish are shown in Table 11-37. Total receptor cancer risks for the CT scenario are below the CTDEP cancer risk limit and are within the USEPA cancer risk range. Non-cancer HI values are 2 for finfish and 1 for ribbed mussels.
- As shown in Table 11-39, risks associated with recreational consumption of finfish and shellfish are lower than or equal to risks associated with recreational consumption of fish and shellfish at background reference locations. This indicates that recreational consumption of finfish and shellfish taken from the Marine Basin are actually associated with a lower order of health risk than recreational consumption of finfish and shellfish

taken from locations that have not been affected by releases from the Site. It is important to note that recreational consumption of finfish and ribbed mussels taken from the Marine Basin is not known to occur; risks represent a hypothetical future land use activity.

Outfall 008 Drainage Area – Recreational Wading

- The total receptor cancer risk estimate for direct contact with surface water (7E-06) and sediment (8E-06) of 1E-05 does not exceed the CTDEP risk limit of 1E-05, and is within the USEPA cancer risk range of 1E-06 to 1E-04. Cancer risks in sediment are primarily associated with PAHs (e.g., benzo(a)pyrene).
- The total receptor non-cancer screening HI value for direct contact with surface water (0.1) does not exceed the CTDEP or USEPA risk threshold value of an HI of 1.
- The non-cancer screening HI value of 3 for direct contact with sediment exceeds the CTDEP and USEPA risk threshold values of an HI of 1; non-cancer risks are attributable to chromium. The HI for chromium is based on the assumption that chromium is present in the hexavalent state, which is associated with a higher order of toxicity than chromium that is in the trivalent state. Given the anaerobic environment of the sediment, the chromium is likely in the trivalent form. The HI associated with chromium in the trivalent form would be less than 0.1.

Summary

The results of the HHBRA indicate that health risks associated with potential exposures to COPCs in surface water and sediment under future use conditions at the Marine Basin do not exceed CTDEP risk limits and are within the USEPA cancer risk range and below an HI of 1. However, exposures to one or more of these media at the Intertidal Flats and Outfall 008 Drainage Area exceed the CTDEP cumulative cancer risk limit of 1E-05 and/or a HI of 1. Specifically:

- The cancer risk associated with dermal contact with shallow groundwater at the Intertidal Flats exceeds the CTDEP cancer risk limit of 1E-05 due to vinyl chloride;
- The hazard index associated with incidental ingestion and dermal contact with sediment at The Outfall 008 Drainage area exceeds a HI of 1 due to chromium, under the assumption that chromium is present as Cr(VI); if chromium was present as trivalent chromium, the HI would be below 1;

Surface water and sediment at the Intertidal Flats, and surface water at the Outfall 008 Drainage Area, are not associated with risks that exceed CTDEP risk limits or the USEPA cancer risk range or HI of 1.

The results of the HHBRA indicate that risks associated with ingestion of finfish, ribbed mussels, and oysters associated with recreational angling under hypothetical future conditions exceed an HI of 1 and/or the CTDEP cancer risk limit. In addition, ingestion of oysters associated with recreational angling at the Intertidal Flats exceeds the USEPA cancer risk range. Health risks associated with consumption of biota taken from the Intertidal Flats are higher than risks associated with consumption of biota taken from the Marine Basin. However, risks associated with recreational consumption of finfish and ribbed mussels taken from the either location are less than or equal to risks associated with recreational consumption of finfish and ribbed mussels at background locations. Based on available information, it does not appear that recreational finfishing or shellfish harvesting has occurred at the Intertidal Flats or Marine Basin. Moreover, in order for sport fishing to occur at these areas to an extent that could permit a person to be exposed to biota in a manner consistent with the exposure assumptions used to estimate risks, the shellfish and finfish populations in these areas would need to improve (in density) considerably.

11.4.2.3 Commercial Fisherman - Future Land Use. Future commercial fishing scenarios were evaluated for the finfishing at the Intertidal Flats, shellfishing at the Intertidal Flats, finfishing at the Marine Basin, and shellfishing at the Marine Basin. The commercial fisherman scenario has two distinct activities: collection of finfish and shellfish, and consumption of finfish and shellfish. As discussed previously, if finfishing was to occur in the future at the study areas, it is assumed that fishing would take place from a boat and, therefore, the only exposure medium other than biota would be surface water. It is assumed that if shellfish harvesting occurred in the future, it would take place from a boat during cold seasons, and via manually digging in sediments during warm seasons. Therefore, the exposure media include surface water and sediment. It is assumed that commercial fishermen would retain a portion of their catch for personal consumption.

Risk estimates are summarized in Table 11-36.

<u>Intertidal Flats – Finfish Angling</u>

- The cancer risk estimate for direct contact with surface water (4E-06) is less than the CTDEP cancer risk limit of 1E-05, and is within the USEPA cancer risk range of 1E-06 to 1E-04. The non-cancer screening HI value for direct contact with surface water (0.5) is below the CTDEP and USEPA threshold value of 1.
- The cancer risk estimate for ingestion of finfish (4E-05) is greater than the CTDEP cancer risk limit of 1E-05, but is within the USEPA cancer risk range of 1E-06 to 1E-04. The non-cancer HI value of 3 for ingestion of finfish is greater the CTDEP and USEPA risk threshold value of an HI of 1. Cancer and non-cancer risks are attributable to PCBs, primarily Aroclor-1254 and Aroclor-1260.
- The cumulative cancer and non-cancer risk estimates for exposure to surface water during angling do not add appreciably to the risk estimates for consumption of finfish, as indicated in Table 11-36.
- The CT risks for consumption of finfish by commercial fishermen are shown in Table 11-38. Cancer risks for the CT scenario are below the CTDEP cancer risk limit. Non-cancer HI values are below 1 for finfish.
- As shown in Table 11-39, risks associated with consumption of finfish at the Intertidal Flats are less than risks associated with consumption of finfish at background reference locations. This indicates that risks associated with consumption of finfish taken from the Intertidal Flats are actually lower than risks associated with consumption of finfish taken from locations that have not been affected by releases from the Site.

<u>Intertidal Flats –Shellfish Harvesting</u>

- The total receptor cancer risk estimate for direct contact with sediment (1E-05) and direct contact with surface water (2E-06) of 1E-05 does not exceed the CTDEP cancer risk limit of 1E-05, and is within the USEPA cancer risk range of 1E-06 to 1E-04. Risks are primarily associated with Aroclors. The non-cancer screening HI value for direct contact with sediment (0.7) and direct contact with surface water (0.4) of 1 does not exceed the CTDEP and USEPA threshold hazard index value of 1.

- The cancer risk estimate for ingestion of ribbed mussels (8E-05) exceeds the CTDEP risk limit of 1E-05, but is within the USEPA cancer risk range of 1E-06 to 1E-04. The non-cancer HI value is equal to one, which does not exceed the CTDEP or USEPA threshold of 1. Cancer risks are primarily associated with arsenic and to a lesser extent PCBs (Aroclor-1254 and Aroclor-1260).
- The cancer risk estimate for ingestion of oysters (2E-04) is greater than the CTDEP cancer risk limit of 1E-05, and exceeds the USEPA cancer risk range of 1E-06 to 1E-04. The non-cancer HI values are greater than 1 (HI value of 11). Cancer risks are entirely associated with PCBs (Aroclor-1254). Non-cancer HI values are attributable to Aroclor-1254.
- The cumulative cancer and non-cancer risk estimates for exposure to surface water and sediment associated with collection of shellfish, do not add appreciably to the risk estimates for consumption of fish and shellfish, as indicated in Table 11-36.
- The CT risks for consumption of shellfish by commercial fishermen are shown in Table 11-38. Cancer risks for the CT scenario are below the CTDEP cancer risk limit and are within the USEPA cancer risk range, ranging from 2E-06 for ribbed mussels to 7E-06 for oysters. Non-cancer HI values are below one for ribbed mussels and greater than 1 for oysters (HI = 2).
- As shown in Table 11-39, risks associated with consumption of ribbed mussels at the Intertidal Flats are equal to the risks associated with consumption of ribbed mussels at background reference locations. This indicates that risks associated with consumption of ribbed mussels taken from the Intertidal Flats are no different than risks associated with consumption of ribbed mussels taken from locations that have not been affected by releases from the Site. It is important to note that shellfish in the lower Housatonic River are not considered to be of a quality suitable for commercial markets.
- Risks associated with consumption of oysters taken from the Intertidal Flats are approximately two-times greater than risks associated with consumption of oysters taken at background reference locations (Table 11-39). This is due to differences in Aroclor-1254 concentration in oysters collected at the Intertidal Flats and oysters collected at background reference locations. It is important to note, however, that the EPCs for oysters are based on analyses of seed oysters that would not be consumed by humans; humans would only be potentially exposed to oysters from this area after they have been placed in Long Island Sound for a few years to grow and mature.

<u>Marine Basin – Finfish Angling</u>

- The cancer risk estimate for direct contact with surface water (5E-07) is below the CTDEP cancer risk limit, and is below the USEPA cancer risk range of 1E-06 to 1E-04. The non-cancer screening HI value for direct contact with surface water (0.03) is below the CTDEP and USEPA risk threshold value of 1.
- The cancer risk estimate for ingestion of finfish (2E-05) is greater than the CTDEP cancer risk limit of 1E-05, but is within the USEPA cancer risk range of 1E-06 to 1E-04. The non-cancer HI value of 1 for ingestion of finfish does not exceed the CTDEP or USEPA risk threshold value of 1. Cancer risks are attributable to PCBs, primarily Aroclor-1254 and Aroclor-1260.
- The cumulative cancer and non-cancer risk estimates for exposure to surface water during angling do not add appreciably to the risk estimates for consumption of finfish, as indicated in Table 11-36.

- The CT risks for consumption of finfish by commercial fishermen are shown in Table 11-38. Cancer risks for the CT scenario are below the CTDEP cancer risk range and are within the USEPA cancer risk range. Non-cancer HI values are below 1 for finfish.
- As shown in Table 11-39, risks associated with consumption of finfish at the Marine Basin are less than risks associated with consumption of finfish at background reference locations. This indicates that risks associated with consumption of finfish taken from the Marine Basin are actually lower than risks associated with consumption of finfish taken from locations that have not been affected by releases from the Site.

Marine Basin –Shellfish Harvesting

- The total receptor cancer risk estimate for direct contact with sediment (1E-06) and surface water (5E-07) of 2E-06 is below the CTDEP cancer risk limit of 1E-05, and is within or below the USEPA cancer risk range of 1E-06 to 1E-04. The non-cancer screening HI value for direct contact with sediment (0.02) and surface water (0.02) of 0.04 is below the CTDEP and USEPA threshold hazard index of 1.
- The cancer risk estimate for ingestion of ribbed mussels (7E-05) is greater than the CTDEP cancer risk limit of 1E-05, but is within the USEPA cancer risk range of 1E-06 to 1E-04. The non-cancer HI value (0.8) is below the CTDEP and USEPA threshold HI of 1. Cancer risks are primarily associated with arsenic and to a lesser extent PCBs (Aroclor-1254).
- The cumulative cancer and non-cancer risk estimates for exposure to surface water and sediment associated with collection of shellfish do not add appreciably to the risk estimates for consumption of fish and shellfish, as indicated in Table 11-36.
- The CT risks for consumption of shellfish by commercial fishermen are shown in Table 11-38. The cancer risk for the CT scenario is below the CTDEP cancer risk limit at 5E-07. The non-cancer HI value is below 1 (0.07).
- As shown in Table 11-39, risks associated with consumption of ribbed mussels at the Marine Basin are less than the risks associated with consumption of ribbed mussels at background reference locations. This indicates that risks associated with consumption of ribbed mussels taken from the Marine Basin are actually lower than risks associated with consumption of ribbed mussels taken from locations that have not been affected by releases from the Site.

Summary

The results of the HHBRA indicate that risks associated with hypothetical future commercial angling for finfish and commercial shellfish harvesting do not exceed the CTDEP or USEPA cancer risk limit and hazard index. The results of the HHBRA indicate that ingestion of finfish, mussels, and oysters at the Intertidal Flats and finfish and mussels at the Marine Basin exceed a HI of 1 and/or the CTDEP cancer risk limit of 1E-05. Cancer risk associated with ingestion of oysters at the Intertidal Flats exceeds the USEPA cancer risk range. Risks associated with consumption of finfish and ribbed mussels taken from the Site are less than or equal to risks associated with consumption of finfish and ribbed mussels at background locations. Based on current trends in aquaculture, commercial fishing is unlikely to occur at the Intertidal Flats or Marine Basin in the future. Moreover, in order for commercial fishing to occur at these areas to an extent that could permit a person to be exposed to biota in a manner consistent with the exposure assumptions used to estimate risks, the shellfish and finfish populations in these areas would need to improve (in density) considerably.

11.4.3 Uncertainty Analysis.

This subsection identifies and discusses uncertainties in the risk assessment. These uncertainties are identified in order to place the results in a context or perspective. Unlike some other assessments, risk assessments rely not just on measured or certain facts, but also on assumptions and estimates, and also policy decisions, in the face of limited or non-existent data. Historically, risk assessments have used highly conservative assumptions in the place of unavailable data, with the net result often being a substantial overestimation of potential risks. This approach was considered the "protective" approach, in that it would overestimate rather than underestimate potential risks. Consequently, the interpretation of risk estimates should be performed with the understanding that risk estimates are conservative values resulting from multiple layers of assumptions inherent in the risk assessment process.

This uncertainty discussion is not intended to identify "problems" with the risk assessment, only to point out how decisions made in the face of uncertainty may have affected the results and conclusions of the assessment. It should be emphasized that the potential risks estimated here are based on numerous assumptions. Each of these assumptions is associated with some uncertainty. The following types of uncertainties should be considered in any human health risk evaluation:

- uncertainties in the nature and extent of release of COPC;
- uncertainties associated with estimating the frequency, duration, and magnitude of possible exposure;
- uncertainties associated with assigning exposure parameters to a heterogeneous population that includes both men and women and young and old (e.g., body weight and ventilation rates);
- uncertainties in estimating carcinogenic slope factors and/or noncarcinogenic measures of toxicity (e.g., RfDs or RfCs); and
- uncertainties about possible synergistic or antagonistic chemical interactions of a chemical mixture.

The uncertainties associated with estimating possible exposure result from the variance in sampling and analytical techniques, and from quantifying parameters that are not directly observed (e.g., frequency and duration of exposure). Because some of these parameters are functions of the behavior patterns and personal habits of the exposed populations, no single value can be assumed to be representative of all possible exposure conditions. The standard of care for environmental risk assessments for addressing many of these uncertainties is to use upper-bound (90th or 95th percentile) estimates of input values, such as exposure parameters and toxicity values. When upper-bound estimates are compiled together, the total receptor risk reflects an estimate that is greater than the 99th percentile.

There are uncertainties in the four areas of risk assessment: data quality/COPC selection, exposure assessment, toxicity assessment, and risk characterization. Site-specific uncertainties that have a potentially significant bearing on the interpretation of this risk assessment are discussed below.

Analytes with SQLs in Excess of COPC Screening Values

Some analytes in the environmental media evaluated in the HHBRA that were not selected as COPCs due to maximum detected concentrations below COPC screening values, were reported at non-detect values (i.e., SQLs) greater than COPC screening values. For these analytes, it is hypothetically possible that they could be present in environmental media at concentrations that would include them as COPCs. This uncertainty has the greatest potential bearing on the risk assessment results for soil. For other media, COPC screening values were based on exposure pathways that are extremely conservative for this Site (e.g., drinking water screening values used to select COPCs in groundwater and surface water) and, therefore, even analytes with SQLs that exceed COPC screening values are unlikely to pose a health risk of concern. In addition, a review of the data summaries (Tables 11-2 through 11-9) indicates that very few analytes that were not selected as COPCs had SQLs in excess of risk-based COPC screening values.

For soil, several VOCs and SVOCs that were not selected as COPCs were associated with elevated SQLs in excess of the USEPA Region IX Residential Soil PRGs. However, with the exception of pentachlorophenol and N-nitroso-di-N-propylamine, all SQLs were below the CTDEP RSR values (Appendix S-1). The residential soil PRGs are extremely conservative when applied to the SAEP Site because current and future land uses are associated with potential exposures that are much lower than those assumed in the development of the PRGs. In addition, the detected concentrations of the chemicals identified in this uncertainty were generally orders of magnitude lower than the elevated SQL values, which suggest that these chemicals would not be present in Site media at concentrations in the range of the elevated SQL values. For these reasons, it is unlikely that exclusion of these chemicals as COPCs in the risk assessment would result in an underestimation of risks that would be substantial enough change the results or conclusions of the HHBRA.

COPCs without Dose-Response Values

Non-cancer inhalation dose-response values (RfCs) are unavailable for the majority of COPCs. However, inhalation exposures at the Site are only a potentially significant exposure pathway for indoor air. Although dust inhalation may be a complete pathway, comparison of cancer risk estimates for dust inhalation to cancer risk estimates for oral and dermal exposures indicates that risks associated with the inhalation pathways are orders of magnitude lower than risks associated with the direct contact pathways. By analogy, non-cancer risks for the dust inhalation pathway would also be insignificant relative to the oral and dermal pathways. Therefore, lack of inhalation RfC values does not represent an uncertainty that would affect the results or conclusions of the risk assessment for soil.

Among the six VOCs evaluated in indoor air samples (1,1,1-TCA, 1,1-DCE, cis-1,2-DCE, PCE, TCE, and VC), only one of the chemicals (cis-1,2-DCE) does not have a published inhalation RfC value. Although not recommended by USEPA Region I, the oral RfD values could be used as surrogate dose-response values (i.e., through route-to-route extrapolation) for evaluating the significance of possible non-cancer inhalation risks. Using the future industrial workers scenario to evaluate potential non-cancer risks for cis-1,2-DCE at the exposure point with the highest EPC (Building B-6), the HQ for this compound may be estimated, as shown in the following table.

COPC	Indoor Air EPC	Estimated Dose (mg/kg/day)	Oral RfD (mg/kg/day)	HQ
Cis-1,2-DCE	19 ug/m^3 (B-6)	0.0019	0.01	0.1

This HQ, when added to the HI for future industrial worker exposure to indoor air at B-6 (0.02), change the total HI for indoor air from 0.02 to 0.1, and the total receptor HI from 0.08 to 0.2; an increase that does not change the conclusions of the risk assessment. At Building B-2, the EPC for cis-1,2-DCE is only 1.1 ug/m^3; potential risks for cis-1,2-DCE at Building 2 would not change the conclusions of the risk characterization for the Main Site or Chromium Plating Area.

Lead in Fish

Lead was retained as a COPC in biota because no screening values have been published for consumption of biota that may contain lead. A review of the lead EPCs in finfish, ribbed mussels, oysters shows that lead EPCs in all biota except ribbed mussels collected at the Intertidal Flats are less than or consistent with (i.e., within 5% of) the lead concentrations detected in biota collected at background reference locations. Therefore, with the exception of ribbed mussels at the Intertidal Flats, potential risks associated with consumption of lead in biota collected at the Site are no different than potential risks associated with consumption of lead in biota collected at background reference locations.

Chromium Evaluated as Hexavalent Chromium

With the exception of soils at the former Chromium Plating Area, at which total chromium was evaluated as hexavalent chromium and trivalent chromium based on site-specific chromium speciation analyses, chromium was assumed to be present entirely as Cr(VI) in exposure media at the Site. Evaluation of chromium speciation data collected at the former Chromium Plating Area suggest that approximately 88% of the chromium may be present as Cr(VI) (based on the 95th percentile of the Cr(VI): total chromium ratios). Therefore, under the assumption that all chromium detected in other media and at other exposure areas is in the hexavalent form represents a conservative assumption that overestimates risk. However, even under that conservative assumption, chromium was not associated with an HQ greater than 0.1 in any of the media or exposure areas except shellfish at the Intertidal Flats, and sediment at the Outfall 008 Drainage area. If chromium at these exposure points was present entirely in the trivalent form, then HQ values would be well below a value of 0.1. However, if the compositional assumptions for chromium that were used at the former Chromium Plating Facility were applied to sediment at the Outfall 008 Drainage area, the HQ values for chromium would be reduced, but would still remain at values greater than 1.

In the absence of chromium speciation data for sediments in the Outfall 008 Drainage area, the toxicity of chromium in that medium was evaluated as Cr(VI), which exhibits a higher order of toxicity than trivalent chromium. Although SAEP used Cr(VI) solutions, it is likely that if Cr(VI) was released to sediments, the anaerobic conditions in the sediments would favor a chemical reduction of chromium from the hexavalent state to the trivalent state. Therefore, the chromium detected in sediments at the Outfall 008 Drainage area is likely present in part, if not all, as trivalent chromium. Under the assumption that chromium in the Outfall 008 Drainage area sediments is in the trivalent form, the HI for a child recreational visitor potentially exposed to sediments in the ditch would be 0.2. The HQ fro chromium evaluated as trivalent chromium would be. 0.01, reduced from an HQ of 4 under the assumptions that chromium is present as Cr(VI). Therefore, if chromium in the Outfall 008 Drainage area sediments is present as trivalent chromium, which it likely is, the HI for all receptors would be below the value of 1.

Vapor Migration from Surface Water

VOCs in surface water could, hypothetically, migrate to ambient air. This potential exposure pathway would be insignificant and, therefore, was not evaluated because VOCs were detected at very low concentrations in surface water, and dilution into ambient air would be very substantial.

Potential Risks to Construction Workers

The commercial/industrial worker scenario was used to provide an estimate of potential risks for exposures to soil and indoor air that would be protective for other receptors that could be exposed under commercial/industrial uses of the Site, including construction workers. The exposure parameters that differ between a full-time commercial/industrial worker and a construction worker are: a) soil ingestion rate (100 mg/day for commercial/industrial worker versus 200 mg/day for construction worker); b) dermal surface area exposed to soil (3300 square cm for commercial/industrial worker versus 5700 square cm for construction worker; and c) exposure duration (25 years for commercial/industrial worker versus 1 year or less for a construction worker) and exposure frequency (150 – 250 days per year for commercial/industrial worker versus 125 days for construction worker). In addition, non-cancer risks for a construction worker are evaluated using subchronic RfDs and RfCs when available. Due to the differences in exposure duration, the commercial/industrial worker scenario provides a very conservative assessment of potential cancer risks to a construction worker.

With respect to non-cancer risks for soil, construction worker HI values could be as much as 1.6 times greater than commercial/industrial worker HI values for incidental ingestion, and 1.4 times greater than commercial/industrial worker HI values for dermal contact. The highest non-cancer HI value for the commercial/industrial scenario is 0.6 for soil at the Main Site. Applying the differences in ingestion and dermal exposure parameters for a construction worker indicates that the soil HI could be as high as 0.9. However, that HI would be an overestimate because subchronic RfD values are available for the principal non-cancer risk contributors (Aroclors and chromium). Construction worker non-cancer HI values for indoor air would be one-half the HI values for a commercial/industrial worker, due to the lower exposure frequency for construction workers. Therefore, quantitative characterization of potential construction worker risks would not change the conclusions of the risk assessment.

Recreational and Commercial Angling and Shellfishing

With the exception of shallow groundwater that may be contacted if humans wade in Intertidal Flats without the use of waders, and sediment at the Outfall 008 Drainage Area, all of the health risks estimated in this HHBRA that are in excess of regulatory risk limits are associated with consumption of fin fish and shell fish taken from the Intertidal Flats or Marine Basin. Interviews were conducted with local commercial fishermen regarding commercial and sport (recreational) angling and shellfish harvesting activities at the Intertidal Flats and Marine Basin. Based on the interviews, it appears that no commercial or sport fishing is occurring in the lower Housatonic River and, more specifically, at the areas of the Intertidal Flats and Marine basin that have been sampled during the RI activities. Specifically, the interviews indicated that:

- seed oysters are no longer collected from the lower Housatonic River;
- commercial fishing no longer occurs in the lower Housatonic River, and has never been observed at the Marine Basin;
- some commercial fishing for bluefish occurs in Long Island Sound, and the majority of sport fishing for blue fish and striped bass occurs in Long Island Sound;

- no sport fishing has been observed at the Intertidal Flats or Marine Basin;
- shellfish (e.g., blue or ribbed mussels) in the lower Housatonic River are not of suitable quality for commercial harvesting.

Therefore, when interpreting the significance of these risks, it is important to consider the fact that recreational sport fishing is not known to occur at the Intertidal Flats and Marine Basin under the existing land uses, and commercial angling and shell fishing does not occur in the intertidal mudflats adjacent to the site. Moreover, risks associated with consumption of fin fish and ribbed mussels collected from the Marine Basin and Intertidal Flats are equal to or less than risks associated with consumption of those biota collected at background locations that are unaffected by releases from the Site.

12.0 BASELINE ECOLOGICAL RISK ASSESSMENT

As part of the RI, MACTEC has been tasked to prepare both a HHBRA to evaluate potential human health risks at the Site and a BERA to evaluate potential ecological risks at the Site. This section presents the approach and findings of the BERA.

The ecological risk assessment process conducted for the SAEP follows that prescribed in the USEPA Ecological Risk Assessment Guidance For Superfund, Process for Designing and Conducting Ecological Risk Assessments, 1997 (the Process Document, USEPA, 1997c). Steps 1 and 2 of the Process Document (Screening-level Ecological Risk Assessment, [SERA]) were completed as part of the initial steps of the Problem Formulation, mentioned below. However, the primary steps of the SERA included selection of ECOCs; this process has been revised in this BERA to be consistent with USEPA guidance. Step 3 of the Process Document (Problem Formulation) was followed in preparing the Draft Remedial Investigation Ecological Problem Formulation Technical Memorandum submitted in 1999 (URS-GWC, 1999). Step 4 of the Process Document (Study Design and DQO Process) was addressed in RI sampling plans associated with ecological risk assessment data collection activities. Steps 5 and 6 are verification of field sampling design, and Site investigation and data analysis, respectively; these steps were implemented prior to beginning the BERA. Step 7 of the Process Document is the Risk Characterization; the requirements of which are fulfilled by this BERA. The final step of ecological risk characterization as described in the Process Document is Risk Management. This BERA will be used to help make risk management decisions at the SAEP Site.

The BERA for the SAEP Site is also consistent with other current risk assessment guidance documents, including Guidelines for Ecological Risk Assessment (USEPA, 1998b), as well as various EcoUpdates (published since 1991). Other literature sources (Suter, 1993; Maughan, 1993) were also consulted.

The specific objective of the BERA is to conduct a more detailed evaluation of the risks associated with ecological COPCs (ECOPCs) and exposure pathways that were identified during previous ecological evaluations. This BERA includes a detailed evaluation of potential ecological risks at the site, and considers all of the available analytical data collected to date. It builds upon information presented in previous ecological investigations.

The remainder of the BERA is organized as follows:

- Section 12.1 – Problem Formulation
- Section 12.2 – Analysis (includes exposure assessment and effects assessment)
- Section 12.3 – Risk Characterization
- Section 12.4 – Summary and Conclusions

12.1 PROBLEM FORMULATION

Problem formulation is the initial step of the BERA process where the purpose and scope of the assessment are defined. A Draft Problem Formulation Technical Memorandum was submitted by

URSGWC for regulatory review in September 1999. USEPA comments were received in December 1999. The problem formulation presented below has been revised to reflect USEPA comments.

The Problem Formulation contains a brief site history and characterization (Subsection 12.1.1), conceptual site model development (Subsection 12.1.2), data summarization and identification of COPCs; previously referred to as ecological contaminants of concern, or ECOCs (Subsection 12.1.3), identification of ecological receptors and exposure pathways (Subsection 12.1.4), and the selection of assessment and measurement endpoints (Subsection 12.1.5).

12.1.1 Site History and Characterization

The SAEP is an industrial manufacturing facility that was used between 1929 and 1998 to manufacture aircraft, tank engines, and related components for the U.S. Government. The facility consists of 49 buildings and is located on 124 acres adjacent to the tidally-influenced portion of the Housatonic River approximately 1 mile upstream of Long Island Sound (Figure 1-1).

The portion of the facility containing all of the buildings (and historically all of the manufacturing operations) is referred to as the main site (Figure 1-2). The main site is located between Main Street, Sniffens Lane, and the Housatonic River. All portions of the main site (with the exception of a small landscaped area between Building B-2 and Main Street) are paved and enclosed with security fencing. The portion of the facility on the south side of Sniffens Lane, containing a large parking lot (South Lot), Building B-6, and the former wastewater treatment building, is referred to as the south lot area. A man-made causeway and dike are located at the portion of the main site that borders the Housatonic River. The Dike was constructed to control flooding caused by the Housatonic River. The Dike is approximately 15 feet high, contains a paved one-lane access road at the top, and is covered with rip-rap and stone on both sides. The Causeway was constructed to provide barge/boat access to the facility from the Housatonic River. The Causeway extends approximately 800 feet from the Dike into the Housatonic River. The Causeway is presently undergoing remediation under a NCRA; a permeable erosion control cover is being placed over the entire surface (extending to the water line) of the Causeway. The Site contains no significant areas of terrestrial habitat, and therefore the focus of this BERA is on risks associated with aquatic habitats.

As described below, there are three areas where surface water and sediment may have been affected by releases from the facility. These include the Tidal Flats (also referred to as the Tidal Flats), Outfall 008, and the Marine Basin. The Intertidal Flats are located in the Housatonic River adjacent to the Dike; the Causeway extends into the Intertidal Flats, essentially dividing it into two areas. Outfall 008 is the former outfall from the chemical treatment building. Outfall 008 discharged to a ditch that flowed to the Marine Basin. The Marine Basin is a tidally-influenced basin located on the Housatonic River approximately 1000 feet southeast of the Main Site. Additional discussion of each of these areas is provided in Subsection 12.1.4, Identification of Ecological Habitats, Receptors, and Exposure Pathways.

12.1.2 Conceptual Site Model

The conceptual site model (CSM) for the Site includes a description of:

- sources of contamination,
- release mechanisms to environmental media (e.g., overland transport, deposition, discharges),
- contaminated media (e.g., surface water, sediment, or groundwater),
- exposure routes (e.g., ingestion, direct contact, or root uptake), and
- aquatic and semi-aquatic receptors (e.g., invertebrates, fish, and wildlife).

Potential contaminant sources and release mechanisms are discussed below. Potential exposure routes and receptors are discussed in Subsection 12.1.4. Figure 11-1, presented as part of the HHBRA in Section 11, contains a graphical representation of the CSM for the SAEP Site.

During manufacturing processes, constituents were released to the environment through several mechanisms:

- Chlorinated solvents (e.g., PCE, TCE, and 1,1,1-TCA) used for degreasing and cleaning metal components were released (i.e., spilled or otherwise disposed of) from operations, primarily in Building B-2. Solvents migrated to soil beneath the building floor (e.g., via leaks in floor drain systems) to shallow groundwater. Groundwater flow is toward the Housatonic River; however, the rate of flow is extremely slow. DNAPL and elevated concentrations of chlorinated solvents have also been identified in deep groundwater, indicating that vertical migration of the plumes has occurred.
- Metal plating solutions containing chromium, including hexavalent chromium, were released (i.e., spilled or otherwise disposed of) from the former B-2 Chromium Plating Facility (attached to Building B-2). Chromium migrated to soil beneath the building floor (e.g., via leaks in floor drain systems) and into shallow groundwater. The RI has identified elevated concentrations of chromium in soil and a hexavalent chromium groundwater plume extending approximately 150 feet downgradient from the former B-2 Chromium Plating Facility.
- Potentially-contaminated soil from the Main Site area was placed in the vicinity of the South Parking Lot prior to paving the lot.
- Liquid wastes from various industrial operations were transported to the chemical treatment plant; treated wastes were then released to a drainage ditch that discharged through Outfall 008 to a ditch that emptied into the Marine Basin. Prior to construction of the CWTP, wastes were historically discharged directly to the Intertidal Flats through various outfalls. Some of the liquid wastes may have contained constituents (e.g., some inorganics) that could bioaccumulate in biota (fish and shellfish) in the Tidal Flats and Marine Basin areas.
- Stormwater that is collected from the SAEP is discharged to the Tidal Flats.

In addition, chlorinated solvents have been detected in shallow groundwater beneath and within the sediment pore space of the Tidal Flats. Since the groundwater plumes originating at Building B-2 are not discharging to surface water, the source of chlorinated solvent contamination at the Tidal Flats is suspected of being related to discrete releases (historic disposal) of solvent waste in the vicinity of the Dike area.

Site operations also resulted in waste streams potentially containing nickel, copper, cadmium, aluminum, magnesium, zinc, and cyanide. Additionally, manometers containing mercury were used at the engine testing facilities. These sources could potentially have resulted in releases to environmental media associated with the Site.

Erosion of soil and transport to surface water is not a complete pathway at SAEP because the site is covered by buildings, concrete and asphalt.

Based on this summary of the site conceptual model and the results of the RI sampling, the following media are considered to be affected by releases at the SAEP:

- Soil at the Main Site
- Soil at the Causeway/Dike area
- Soil at the South Lot area
- Groundwater at the Main Site
- Groundwater at the Causeway/Dike area
- Groundwater at the South Lot area
- Shallow groundwater beneath the Tidal Flats
- Indoor air in buildings overlying the chlorinated solvent groundwater plumes at the Main Site
- Surface water, sediment, and biota at the Tidal Flats
- Surface water, sediment, and biota at the Outfall 008 Area
- Surface water, sediment, and biota at the Marine Basin

Figure 11-1 in the HHBRA provides a graphical representation of the site conceptual model. The ecological media of concern are limited to:

- Groundwater beneath the Intertidal Flats
- Surface water, sediment, and biota at the Tidal Flats
- Surface water, sediment, and biota at the Outfall 008 Area
- Surface water, sediment, and biota at the Marine Basin

The potential ecological exposure pathways to each of the media identified in Figure 11-1 are discussed in Section 12.1.4, Identification of Ecological Habitats, Receptors, and Exposure Pathways. Rationale for exclusion of other site media is also discussed in that section.

12.1.3 Data Summarization and Selection of Chemicals of Potential Concern (COPCs)

This section presents the data summarization and selection process, and identifies the criteria used to select COPCs for each of the media evaluated in the BERA (surface water, sediment, shallow groundwater, and biological tissue). It is divided into three steps: available data; data summarization; and selection of COPCs.

12.1.3.1 Available Data. Analytical data are available for surface soil, subsurface soil, indoor air, groundwater, surface water, sediment, and biota. Surface and subsurface soil data are not included in the BERA because there is no direct ecological exposure pathway to onsite soils. The only area containing potential terrestrial habitat was the Causeway. Contaminated soils on the Causeway have been contained under a NTCRA, in which a permeable erosion control cover is being placed over the entire surface of the Causeway. Indoor air data are not included, because ecological receptors are not exposed to indoor air. With the exception of shallow groundwater data collected in the Tidal Flats, groundwater data were not included in the BERA, because surface water and sediment data are assumed to adequately represent the concentrations to which aquatic life are likely to be exposed.

Sampled abiotic media included in the BERA include surface water, sediment, and shallow groundwater in the Tidal Flats. In addition to these media, biological tissue samples were also collected and analyzed. As shown in Table 12-1, the analyses varied among the various RI field investigations (1992, 1994, 1999, 2001, and 2002); target analytes were selected according to specific objectives for each investigation.

Due to administrative issues, the amount (level of detail) of data varied among each of the RI investigations. The data limitations for each of the investigations are discussed below:

- 1992 Investigation: Data were available for analytical results that represent a positive detection (i.e., unqualified and "J" qualified results) for chemicals previously selected as COPCs in Phase I RI risk assessment (W-C, 1993). Data were not available for non-detected results (i.e., "U" qualified data).
- 1994 Investigation: Data were available for analytical results that represent a positive detection (i.e., unqualified and "J" qualified results). Data were not available for non-detected results (i.e., "U" qualified data). As part of this investigation, samples were collected from linear transects radiating outward from the nearshore areas near the outfalls in the Tidal Flats. For these transect samples, data for positive detections and non-detects were available if an analyte was detected in at least one sample in a given transect. If an analyte was not detected from any samples within a given transect, data for that analyte were not available for any of the samples within the transect (i.e., the analyte was not included in any of the available data tables or database files).
- 1999-2001 Investigations: Data were available for all analytical results (i.e., positive detections and non-detects).

The absence of data for non-detects from the 1992 and 1994 RI sampling introduces a high bias to statistics that are calculated from the data set and, therefore, contributes an uncertainty that is likely to overestimate rather than underestimate exposure point concentrations.

12.1.3.2 Data Summarization Methods. Data summaries were generated for each medium and data grouping included in the BERA. Summaries include the following parameters for each analyte: frequency of detection (number of samples in which a given chemical was detected divided by the number of samples for which that chemical was analyzed), range of analytical quantitation limits, maximum detected concentration, location of maximum detected concentration, arithmetic mean of all samples, 95% UCL on the mean, and background concentration (where available).

Additional factors in the data summarization process include the following:

- Field duplicate samples were collected throughout the RI sampling program to evaluate quality control. The duplicate samples were used to evaluate analytical variability, but were not included in the data set used in the risk assessment (i.e., only the analytical results for the "original" sample in the duplicate pair were included in the data sets).

- The arithmetic mean concentration was calculated using one-half the sample quantitation limit (SQL) for samples in which an analyte was not detected.
- Analytical results for samples that were collected from the same locations, but during different sampling events, were considered unique samples for the purpose of calculating summary statistics (i.e., they were not averaged together or area weighted). This likely resulted in overestimated exposures in some areas, since repeat sampling was generally focused on areas of higher chemical concentrations.

Data summaries were generated for the following data groupings:

<u>Surface water:</u>

- Tidal Flats
- Outfall 008 (Low Tide and High Tide, summarized separately)
- Marine Basin (Low Tide and High Tide, summarized separately)
- Reference (Background) Area (Low Tide and High Tide, summarized separately)

<u>Sediment:</u>

- Tidal Flats
- Outfall 008
- Marine Basin
- Reference Area

<u>Shallow Groundwater:</u>

- Tidal Flats

<u>Biota:</u>

- Oysters – Tidal Flats and Reference Area
- Mussels – Tidal Flats, Marine Basin, and Reference Area
- Aquatic Worm – Tidal Flats, Outfall 008, Marine Basin, and Reference Area
- Finfish – Tidal Flats, Marine Basin, and Reference Area

12.1.3.3 Selection Criteria Used to Identify COPCs. Several selection criteria were used to identify chemicals of concern for evaluation in the BERA, including comparison to conservative screening benchmarks, frequency of detection, and whether or not an analyte is considered to be an essential nutrient. Each of the COPC selection criteria is discussed in greater detail below.

Screening Benchmarks. The screening benchmarks for surface water include federal Ambient Water Quality Criteria, Connecticut Surface Water Quality Standards, Ecotox Threshold values, and Tier II screening benchmarks developed by Oak Ridge National Laboratories (Suter and Tsao, 1996). Screening benchmarks for sediment include USEPA Sediment Quality Criteria (SQC), Ontario Ministry of the Environment Lowest Effect Levels (LELs), NOAA Effects Range-Low (ER-L) values, Consensus-Based Threshold Effect Concentrations (TECs) and Probable Effect Concentrations (PECs), as well as sediment screening values developed by Oak Ridge National Laboratories (Secondary Chronic Values, and Lowest Chronic Values for Fish and Daphnids (Jones et al., 1997). The lowest of the available SQC, LEL, ER-L, and TECs were selected preferentially over the other values that were derived using limited data (e.g., Secondary

Chronic Values). If none of these screening values was available, additional values were obtained from the literature where available.

The lowest of the benchmarks for each medium were used in the COPC selection tables. If an analyte concentration is less than the selected conservative screening benchmark, it is unlikely to pose a risk to ecological receptors and was eliminated from further consideration in the BERA. If no screening benchmark was available, the analyte was retained as a COPC.

Frequency of Detection. Analytes that were infrequently detected are less likely to pose a risk to ecological receptors. For this BERA, analytes detected in 5% or less of the samples analyzed were eliminated from further consideration in the BERA. This criterion was only applied to data sets having 20 or more samples.

Essential Nutrients. There is little potential for toxic effects resulting from exposure to essential nutrients such as calcium, magnesium, potassium, and sodium. The highly controlled physiological and regulatory mechanisms of these inorganics suggest that there is little, if any, potential for bioaccumulation, and available toxicity data demonstrate that high dietary intakes of these nutrients are well-tolerated (National Academy of Sciences [NAS], 1977; NRC, 1982; 1984). As essential nutrients, these analytes are unlikely to pose a risk to ecological receptors, and they were eliminated from further consideration in the BERA. Iron is also considered an essential nutrient, but only in terrestrial systems; in aquatic systems iron and iron flocculates have been associated with toxicity to aquatic organisms. Therefore, iron was retained as a COPC in aquatic media if the screening benchmark was exceeded.

Background data are also presented in the data summary tables; however, per USEPA Region I guidance, background data were not used in the COPC selection process. Background concentrations were used in the risk characterization portion of the BERA to evaluate the magnitude of potential risks associated with historical practices at the SAEP relative to those that are associated with local background conditions. The incremental contribution of background risk relative to site risks is discussed in the risk characterization.

The remaining subsections discuss the COPCs selected for surface water, sediment, shallow groundwater, and biological tissue.

12.1.3.4 COPCs in Surface Water. Surface water samples used in the BERA were collected between 1992 and 1999. Surface water samples were collected from the Tidal Flats, Outfall 008 Drainage, Marine Basin, and background (reference) locations.

Surface water samples are summarized separately for each of these areas. A list of surface water sample locations evaluated in the BERA is presented in Appendix T, Table T-1.1. The analytical results for surface water are included in Appendix Q. Surface water sample locations are identified in Figure 5-2. Table 12-2 lists COPCs selected for surface water.

Tidal Flats

A total of 10 surface water samples were collected from Tidal Flats in 1992, primarily from locations near outfalls. Table 12-3 presents the analytical data summary for Tidal Flats surface water. COPCs include two VOCs (1,1,1-TCA and carbon disulfide), one SVOC (4-methylphenol), and a number of inorganic analytes. All inorganic results for this area are reported as total inorganics because no filtered samples were collected.

Outfall 008 Drainage
A total of 6 surface water samples were collected from the drainage ditch associated with Outfall 008 in 1999. Three samples were collected at low tide, and three samples were collected from the same locations at high tide. Tables 12-4 and 12-5 present the analytical data summaries for Outfall 008 surface water at low tide and high tide, respectively. COPCs for Outfall 008 Low Tide include one VOC (vinyl chloride), and the majority of inorganics that were detected in both filtered and unfiltered surface water. COPCs for Outfall 008 High Tide include one VOC (vinyl chloride), and the majority of inorganics detected in both filtered and unfiltered surface water.

Marine Basin
A total of 8 surface water samples were collected from the Marine Basin in 1999. Four samples were collected at low tide, and four samples were collected from the same locations at high tide. Tables 12-6 and 12-7 present the analytical data summaries for Marine Basin surface water at low tide and high tide, respectively. COPCs for Marine Basin Low Tide include one VOC (methylene chloride), and a number of inorganics detected in both filtered and unfiltered surface water. Barium, lead, and silver were detected in unfiltered samples, but were not detected in filtered samples, suggesting that their presence in unfiltered samples may be associated with suspended particulates in the water column. COPCs for Marine Basin High Tide include one VOC (methylene chloride), and a number of inorganics in both filtered and unfiltered surface water. Chromium was detected in unfiltered samples, but was not detected in filtered samples, suggesting that the presence of this analyte may be associated with suspended particulates.

Reference (Background)
A total of seven surface water reference (background) samples were collected from areas upstream or across the Housatonic River in the Nells Island reference area, at locations not believed to be influenced by the SAEP Site. An additional discussion of the Nells Island reference area is provided in Section 12.1.4.1. The reference surface water sampling locations are included in Figure 5-2.

One sample was collected in 1992; six additional samples were collected in 1999 (three at low tide, and three from the same locations at high tide). COPCs are not selected for reference locations, but the data summary is presented in Tables 12-8 and 12-9 for low tide and high tide, respectively. SVOCs and inorganic analytes were detected in surface water from reference locations. These data were used in the risk characterization portion of the BERA to help discriminate between site-related and non-site-related risks.

12.1.3.5 COPCs in Sediment. Only surficial sediment data (0-6 inches or 0-1 foot) were included in the BERA; samples from subsurface sediment (6-24 inches) were collected to provide an indication of vertical extent of contamination, but are not included in the BERA since they are below the depth at which ecological exposure is likely to occur. Sediment samples used in the BERA were collected between 1992 and 1999. Sediment samples were collected from the same locations as surface water samples described above: Tidal Flats, Outfall 008, Marine Basin, and several background locations. A list of sediment sample locations evaluated in the BERA is presented in Appendix T, Table T-1.2. Sediment sampling locations are presented in Figure 5-1. As with surface water samples discussed above, sediment samples were summarized by area. The analytical results for sediment are included in Appendix Q. Table 12-10 lists COPCs selected for sediment.

Tidal Flats
Table 12-11 presents the analytical data summary for Intertidal Mudflat sediment. A total of 50 sediment samples were collected from this area between 1992 and 1999. COPCs identified for this area include one VOC (acetone), SVOCs (mostly PAHs, as well as dibenzofuran and N-nitrosodiphenylamine), the majority of detected inorganics, three PCB congeners (Aroclors 1248, 1254, and 1260), and total PCBs.

Outfall 008 Drainage
Table 12-12 presents the analytical data summary for Outfall 008 sediment. A total of four sediment samples were collected from the ditch associated with Outfall 008 between 1992 and 1999. COPCs identified for this area include one VOC (acetone), SVOCs (primarily PAHs, as well as N-nitrosodiphenylamine), the majority of detected inorganics, two PCB congeners (Aroclors 1254 and 1260), and total PCBs.

Marine Basin
Table 12-13 presents the analytical data summary for Marine Basin sediment. A total of 5 sediment samples were collected from the Marine Basin between 1994 and 1999. COPCs identified for this area include one VOC (acetone), PAHs, the majority of detected inorganics, three PCB congeners (Aroclors 1248, 1254, and 1260), and total PCBs.

Reference (Background)
Four sediment reference (background) samples were collected between 1994 and 1999. The reference sediment sampling locations are the same as those for reference surface water sampling locations, and are included in Figure 5-1. COPCs are not selected for reference locations, but the analytical data summary is presented in Table 12-14. PAHs, inorganic analytes, and two PCBs (Aroclors 1248 and 1260) were detected in sediment from reference locations. These data were used in the risk characterization portion of the BERA to help discriminate between site-related and non-site-related risks.

12.1.3.6 COPCs in Shallow Groundwater. Table 12-15 presents the data summary for shallow groundwater data collected from areas within the Tidal Flats. These data were collected to better characterize VOC concentrations in sediment porewater in the Tidal Flats. Twenty-nine shallow groundwater samples were collected from piezometers located within the Tidal Flats in 2001 and 2002; these samples were analyzed for VOCs. Sample locations are shown in Figure 3-2. A list of shallow groundwater locations evaluated in the BERA is included in Appendix T, Table T-1.3. The analytical results for shallow groundwater samples are included in Appendix P. Table 12-16 lists COPCs selected for groundwater.

Six VOCs were identified as COPCs in shallow groundwater, including 1,1-dichloroethane, carbon disulfide, cis-1,2-DCE, ethylbenzene, toluene, and xylenes.

12.1.3.7 COPCs in Biological Tissue. Biological tissue data available at the SAEP site were obtained from oysters, ribbed mussels, and finfish (Atlantic silverside and mummichog). Table 12-17 presents a summary of the numbers and types of biological tissue data available for each area. A complete list of sample locations evaluated in the BERA is included in Appendix T, Table T-1.4. Analytical results for biota are included in Appendix W. Sample locations are presented in Figure 5-3.

Additional data from striped bass filleted samples were not included in the BERA, because there are no ecological receptors that forage on striped bass, and the filleted samples do not represent a portion to which ecological receptors would likely be exposed. These samples were collected to evaluate potential risks to human health in the HHBRA (Section 12).

Oysters were collected from the Intertidal Flats during the 1994 RI sampling program, and ribbed mussels and finfish were collected from the Intertidal Flats and Marine Basin during the 1999 RI sampling program. The laboratory sediment bioassay study (described in Section 12.2.2, Effects Assessment) was designed to provide invertebrate tissue for chemical analysis as well. At the conclusion of the bioassay, tissue samples were collected and submitted for chemical analysis. Shellfish were collected from the same locations as sediment samples. Whole body samples of shellfish (ribbed mussels and oysters, excluding shells) and finfish were analyzed for PCBs and inorganics.

Five finfish samples, three oyster samples, and five ribbed mussel samples were collected from background reference locations in the lower Housatonic River, upgradient of the Site. Background samples were analyzed for the same parameters as samples collected at the Site.

The primary purpose of these sampling programs was to obtain site-specific information regarding tissue levels in probable prey items, rather than estimating the levels using default bioaccumulation factors which are not site-specific. The tissue data were used to calculate site-specific biota-to-sediment accumulation factors (BSAFs). These were used in the food chain modeling to assess exposures to higher trophic level organisms in the BERA. COPCs were not identified specifically for biota; the COPCs identified for sediment were considered to be COPCs in the semi-aquatic food chain models, respectively. However, analytical data were also compared to Critical Body Residue values to provide an indication whether the COPC concentrations present in tissue might pose a risk to the prey items themselves. Tables 12-18 through 12-29 present data summaries for each type of tissue (shellfish [ribbed mussel], oysters, invertebrates, and finfish) in each area.

12.1.4 Identification of Ecological Habitats, Receptors, and Exposure Pathways

This subsection contains a description of the ecological habitats present at the site, the receptors that occur or could occur in those habitats, and pathways by which these receptors may be exposed to contaminants from the SAEP Site.

12.1.4.1 Ecological Characterization. The ecological characterization is based on the description of the Site setting compiled during the Phase I and II investigations including Section 2.0 of the Phase II investigation Work Plan (W-C, 1994) and a previous Draft Baseline Ecological Risk Assessment for the SAEP (W-C, 1996). These data are supplemented with more current information as appropriate. The purpose of this description is to characterize potentially exposed ecosystems which may be at risk, and identify biological populations that represent potential receptors therein. Brief descriptions of significant natural features such as physiography, regional hydrogeology and surface water hydrology, that may directly or indirectly influence Site ecology, are also presented. Ecosystems evaluated in these studies include aquatic resources and, to a limited degree, terrestrial resources in the following five areas:

- Tidal Flats
- Marine Basin
- Outfall 008 Drainage
- Causeway
- Nell's Island Reference Area

Information sources include the New York State Department of Environmental Conservation (NYSDEC), USGS, U.S. Fish and Wildlife Service (USFWS), NOAA, CTDEP Marine Fisheries Division, CTDEP Western District Wildlife, and the Connecticut Department of Agriculture - Aquaculture Division.

The SAEP Site consists of approximately 124 acres. Site photographs showing the SAEP facility and areas evaluated in the BERA are included as Appendix X. About 76 acres are improved land and 48 acres are riparian rights. (A riparian right is a right of access to, or use of, the shoreline or waterline land on the bank of a natural watercourse.) The 76 acres of improved land consist of 49 buildings, paved roadway and grounds, and five paved parking lots. Included in the improved land are an estimated 10 acres along the Housatonic River where fill was placed over tidal flats during the early development of SAEP. The 48 acres of riparian rights property consist of Tidal Flats of the Housatonic River. An estimated 2 acres of this property comprise a Causeway constructed in the 1930s to provide access to the river channel.

The SAEP Site is located on, and drains to the tidal portion of, the Housatonic River less than one mile upstream from Long Island Sound (Figure 1-1). The Housatonic River is approximately one-quarter mile wide in the Site vicinity, and conveys most of the region’s drainage to the Long Island Sound. Currents in the river channel are variable in this area, reversing in response to tidal cycles.

These coastal and marine surface waters including Marine Basin, just east of the Site, have been classified by the CTDEP, Water Quality Standard regulations as SC/SB (CTDEP, 1990) indicating that the CTDEP recognizes existing water quality problems in the coastal waters. The SB classification denotes CTDEP’s goal of improving the water quality conditions.

Frash Pond is a freshwater body subjected to tidal fluctuation located west of the Site, but not exposed to Site drainage. Frash Pond is not currently classified. According to CDEP, unclassified surface waters default to an A classification, which designates the following water uses: potential drinking, agricultural, or industrial water supply; fish and wildlife habitat; and recreational.

All of the region's drainage flows to Long Island Sound, largely via the Housatonic River. Discharge from the Housatonic River ranges from 40 to over 100,000 cubic feet per second (cfs) and averages 3,000 cfs (USGS, 1989). Reported tidal levels for the Housatonic River at Stratford are:

Low tide level	0.8 feet MSL
Mean tide level	2.9 feet MSL

High tide level	5.5 feet MSL

Tidal marsh habitat surrounds much of the SAEP Site. Sediments that form the basis for these marshes are deposited at, and upstream, from the mouths of tidal inlets (due to rise in sea level since the last glaciation and daily tides) which discharge to the ocean (Flint, 1968). These sediments consist of decayed plant matter, peat, and mixtures of silt and clay with high amounts of organic matter. These deposits may be as thick as 15 feet and form in wedge shapes that become thicker towards the ocean or mouth of the streams.

Water quality information was collected during the spring 1999 sampling effort as well as previous investigations in late spring and early summer (W-C, 1993). Surface water pH ranged from 7.4 to 8.5, and was typically around 8.1. Salinity ranged from 11.8 to 27 parts per thousand, typical of estuarine waters. Specific conductance ranged from 14,500 to 40,300 µmhos/cm. Surface water temperature at the time of sampling in 1999 was about 15°C. Prior sampling events recorded temperature ranges from 15.2 to 25.0°C, with most values in the low 20s°C. Dissolved oxygen readings taken at the surface in the Marine Basin and Nell's Island reference area, located across the Housatonic River, indicated saturation levels.

The habitats and biota that occur in the Site vicinity are largely dictated by tidal fluctuations in water level, salinity and surface water chemistry, and the proximity to Long Island Sound. Site habitats are described in the following section.

Tidal Flats

The Tidal Flats adjacent to the SAEP are generally level to gently sloping. Local currents from the Housatonic River flow toward land into the sheltered cove on a flooding tide. In the absence of current on a slack tide, suspended fine sediments settle out of the water column and contribute to the sediment accumulation in the mudflat areas. The mudflats are exposed twice daily at low tide when all water recedes except for flow in several rivulets near stormwater outfalls from the Site. The area is largely unvegetated with the northwest portion supporting limited emergent vegetation.

Tidal Flats are important resources within the estuarine ecosystem. Continuous deposition of fine sediments high in organic material creates an enriched environment that supports populations of macroinvertebrates. In turn, these benthic species provide food sources for fish and shore birds, playing a vital role in the transfer of energy through the food chain.

Qualitative descriptions of the Intertidal Mudflat sediments were recorded during the May 1999 field investigations. Surface sediments were comprised mostly of unconsolidated organic silts and sand, with grain-size distributions appearing to be influenced by degree of exposure to the tidal current. Both debris (in the form of shells and sticks) and epibenthic biota were common and collected at almost every sampling location. The sediment was consistently soft and generally gray-brown in color. Core sediment samples taken at some of the same locations were more compacted, but resembled surface grabs in color and composition. An organic flocculent layer of fine material from vegetative breakdown was always present on surface sediments. With the exception of stations TD006 and TE001, where petroleum odor and sheen were noted, there was no visual or olfactory evidence of contamination (Figure 5-1).

Marine Basin

The Marine Basin is located about 1,000 feet southeast of SAEP on the Housatonic River (Figure 5-1). It is surrounded primarily by stands of common reed (Phragmites communis). The basin is a shallow (maximum depths at high and low tides are 8 and 4 feet, respectively), permanently inundated, tidal embayment that receives discharge from a drainageway for the Sikorsky National Airport and Site Outfall 008. Due to the bathymetry of the basin, tidal flushing is incomplete and surface waters drain and fill continuously throughout the tidal cycle. A tidal gate at the western end of the basin was inspected during the 1999 Field Investigation and appeared to be slightly open (debris in the gate restricted operation and closure during high tide cycles was not complete, allowing water to seep behind the structure). This allows the drainageway to fluctuate with the tidal cycles, creating a constant current in the Marine Basin.

The Marine Basin substrate was characterized during the May 1999 RI field sampling program. Surface sediments consist of fine sands and organic silts with an organic flocculent surface layer. Particle size varies with current velocities in the basin. The color tends to be gray-black to brown depending on the level of organic silts present. Strong hydrogen sulfide odors have been noted at several locations, indicating anaerobic conditions in the sediments. Sediment grabs have a soft consistency and contain debris and benthic macroinvertebrates. The only evidence of contamination is at MB001, where faint petroleum odor has been noted and a slight oily sheen observed (Figure 5-1).

Outfall 008 Drainage

Outfall 008 discharges into a drainageway that travels several hundred feet to its confluence with the Marine Basin. The channel of the Outfall 008 drainageway is approximately 10 to 12 feet wide and generally less than 2 feet deep. Upgradient portions of the Outfall 008 drainageway are exposed during low tide. Outfall 008 merges with another channel conveying drainage for the Sikorsky National Airport. The entire system is tidally-influenced. The adjacent land area is generally disturbed and vegetated primarily with common reed.

Three surface sediment samples were collected along the Outfall 008 Drainage in May 1999, and site observations were consistent among sampling locations. Sediments consisted of soft highly organic sandy silt and were black in color. They contained an organic flocculent layer, plant debris and limited macroinvertebrate fauna. All sediment samples displayed an oily sheen and persistent petroleum odor.

The Causeway

Due to the industrial nature of the SAEP Site, there is no terrestrial habitat on the Site with the exception of approximately 2 acres on top of the Causeway extending into the Tidal Flats along the Housatonic River (Figure 1-2). The Causeway was constructed to provide barge/boat access to the facility from the Housatonic River. The Causeway extends approximately 800 feet from the dike into the channel of the Housatonic River. The Causeway is presently undergoing remediation under a NCRA; a permeable erosion control cover is being placed over the entire surface (extending to the water line) of the Causeway.

Regional Background (Reference) Areas

The reference area is located within Nell's Island salt marsh about 2,000-ft north-northeast, upstream and across the Housatonic River from the SAEP study area. Salt marshes in this area are influenced by tidal water, and generally have high nutrient levels and biological productivity. They are important feeding, breeding, and rearing grounds for shore birds, waterfowl, fish, and shellfish. Within the reference area, plant life consists primarily of soft-stemmed plants such as sedges, rushes, and grasses. Cordgrass (Spartina alterniflora) and common reed are the dominant species. Nell's Island Salt Marsh has recognized value to wildlife, and is a protected refuge.

Sediment for toxicity and bioaccumulation testing, surface water, and biota samples were collected at three reference stations within Nell's Island Salt Marsh in May 1999 (Figure 5-1). Areas within Nell’s Island containing similar habitat to the SAEP Site could not be located. Sediment grain-size distribution tended to be coarser, and organic content lower, than sediments collected at the SAEP study areas. Sediments consisted mostly of fine grain sands, with little organic matter and silt. Sample color ranged from gray-brown to gray-black, and was consistently soft with an organic flocculent surface layer. Plant debris, shells and macroinvertebrates were observed in most sediment grabs. No visual or olfactory indicators of contamination were noted.

At the request of EPA, background sediment samples collected for investigations at the Raymark Site for EPA Contract No. 68-W6-0045 were used to represent regional sediment chemistry conditions (Tetra Tech (NUS), 1999). The sediment samples were collected by Tetra Tech NUS, Inc. and determined by the EPA to represent background data within the Housatonic River Estuary. Sediment samples were taken from areas along Great Meadow (located in the southern portion of Stratford, along Long Island Sound adjacent to the Igor Sikorsky Memorial Airport) and Nell’s Island (Tetra Tech (NUS), 1999). These data are not available electronically, but are included in the risk characterization discussions in Section 12.3.

12.1.4.2 Ecological Receptors. This section describes biota that may utilize the habitat available at the Site. These data form the basis for the selection of receptor species in the BERA. General categories of ecological receptors include benthos, fish, birds, and mammals, each discussed in greater detail below.

Benthos

Information on indigenous benthic fauna in the vicinity of the SAEP Site is based on extensive data collected in December 1994 as part of the Phase II effort and a field reconnaissance performed in May 1995 (W-C, 1996). Additional collections were taken in May 1999 from the Tidal Flats, Marine Basin, Outfall 008 Drainage, and upstream reference area to further supplement the data and provide Site comparisons (Figure 12-1). All samples were collected using a petite Ponar grab apparatus, and sieved before being stained and preserved in formalin.

At least 46 different macroinvertebrate taxa are known to occur in the Site vicinity (see Tables 12-30 and 12-31 for 1994 and 1999 data, respectively). Oligochaetes and polychaetes were present at all stations and dominated every sample. The percent contribution of oligochaetes ranged from 1 % (Station US-2) to 95 % (Station 08-1), while percent contribution of polychaetes ranged from 3 % (Station 08-2) to 85 % (Station US-2) in the 1999 sampling event. Total number of taxa observed, and total number of individuals (abundance) was comparable at all locations with the exception of one location in the Marine Basin, where total number of individuals was lower than at other locations.

At most stations, the polychaete Streblospio benedicti were the dominant taxon. Its presence alone accounted for 43 % of all individuals at Station US-3. At one location in the Tidal Flats Mudflats, the density of the polychaete Mediomastus ambiseta, exceeded the density of S. bendicti. Amphipods were recovered from all stations, with the exception of Station 08-1. The most common amphipod species were Leptocheirus plumulosus and Monoculodes edwardsi. Molluscs were collected from ten of the twelve stations. Hydrobia minuta (a gastropod) and Gemma gemma (a bivalve) were the predominate mollusc species in most samples. In addition to the infaunal invertebrates described above, Palaemonetes sp. (shore shrimp), Carcinus maenas (green crab) and Nassarius obsoletus (mud dog whelk) were collected in high numbers in the Marine Basin and Tidal Flats.

During the field investigation, benthic macroinvertebrates were observed inhabiting the rocky intertidal zone around the Tidal Flats, which has formed as a result of rip-rap placement. The following species were recorded in the rocky intertidal zone Mytilus edulis (blue mussel), Modiolus demissus (ribbed mussel), and green crab. In addition, the river channel downstream of the Site supports beds of immature Crassostrea virginica (eastern oyster), the primary species of shellfish harvested in Long Island Sound.

Fish

Fish collections were made during May 1999 in the Marine Basin, Tidal Flats and the upstream reference area to qualitatively describe the species using these habitats. Fish were collected over a two-day period using a variety of gear depending on habitat, including seines, minnow traps, trap nets, and rod and reel. Effort was similar between the investigation areas and all species collected were identified and most were released.

The fish community assemblages collected are typical of estuarine habitats within the Long Island Sound. A total of 12 species were identified during field sampling, as shown below:

Scientific Name	Common Name	Collection Location		
		Tidal Flat	Outfall 008/Marine Basin	Upstream Reference
Ammodytes hexapterus	American sand lance			X
Anguilla rostrata	American eel	X		X
Cyprinodon variegatus	Sheepshead minnow			X
Fundulus heteroclitus	Mummichog	X	X	X
Fundulus majalis	Striped killifish		X	X
Gasterostues aculeatus	Three-spined stickleback	X	X	X
Menidia menidia	Atlantic silverside	X	X	X
Microgadus tomcod	Atlantic tomcod	X		
Morone americanus	White Perch	X	X	
Morone saxitalis	Striped bass	X	X	X
Pseudopleuronectes americanus	Winter flounder			X
Syngnathus fuscus	Northern pipefish			X

Taxonomic diversity by area included 6 species in the Marine Basin, 7 species in the Tidal Flats and 10 species in the reference area upstream. The principal species recorded were Atlantic silverside (Menidia menidia), mummichog (Fundulus heteroclitus), and striped bass (Morone saxatilis). Numerous species were observed either gravid, flowing, or in breeding color; these include mummichog, white perch (Morone americanus), three spined stickelback (Gasterosteus aculeatus), and sheepshead minnow (Cyprinodon variegatus). In addition, juvenile Atlantic tomcod (Microgadus tomcod) and winter flounder (Pseudopleuronectes americanus) were identified. The presence of juveniles and mature fish in breeding condition suggests that these locations are functioning as both spawning and nursery areas.

Avifauna

Avifauna is an important component of the ecosystem within the SAEP Site area. There have been approximately 220 bird species observed in the Site vicinity comprising four general categories; these include shore birds, waterfowl, song birds, and piscivorous birds (W-C, 1991). The wetlands near SAEP are known breeding grounds for summer resident species such as the great egret (Casmerodius albus), snowy egret (Egretta thula), black-crowned night heron (Nycticorax nycticorax), glossy ibis (Plegadis falcinellus), and fish crow (Corvus ossifragus). Three bird species that nest in the general vicinity of the Site are the least tern (Sterna antillarum), mute swans (Cygnus olor), and piping plover (Charadrius melodus). These species utilize tidal marshes as feeding areas during the breeding season.

During field reconnaissance of the Site performed in May 1995 (W-C, 1996) and May 1999, a variety of species was observed including: great blue heron (Ardea herodias), snowy egret, black-crowned night heron, American black duck (Anas rubripes), sandpiper (Calidris sp.), double-crested cormorant (Phalacrocorax auritus), mallard (Anas platyrhynchos), osprey (Pandion haliaetus), common tern (Sterna hirundo), lesser yellow legs (Tringa flavipes), piping plover, greater black-backed gull (Larus marinus), herring gull (Larus argentatus), and mute swans.

Small Mammals

Mammal use of the SAEP Site is restricted to small, highly modified riparian corridors. Areas along the Causeway and Tidal Flats are primarily rip-rap shoreline with little cover or access to quality habitats. The Marine Basin and Outfall 008 Drainage offer slightly larger areas of plant cover and wetland habitat. These areas are highly disturbed when compared to wildlife refuges in the vicinity. The primary mammal species that may use these areas near the Site include various small rodents, muskrat (Ondatra zibethica) and raccoon (Procyon lotor). While no mammals were observed during the 1999 Field Investigation, both raccoon and muskrats have adapted to use habitats in fringe areas near industrial and urban activity. Since no small semi-aquatic mammals were observed and these areas offer limited habitat and food supplies, it is doubtful that large populations exist at this site.

Critical Habitats and Species

Critical habitats in the vicinity of the Site include extensive tracts of salt marshes, saltwater Tidal Flats and shores, and coastal sand dunes that provide habitat for a variety of biota. The Tidal Flats in the vicinity of SAEP are located in a band along the shoreline of the Housatonic River. Plant life in the vicinity of SAEP is limited to the tidal marshes. The tidal marsh plant life consists primarily of soft-stemmed plants such as sedges, rushes, and grasses. Cordgrass and common reed are the dominant species in the marshes. A number of southeastern Piedmont and Coastal Plain plant species reach their northern native range limits in this region.

The USFWS, National Marine Fisheries, and the CTDEP were contacted to obtain information on federal- and/or state-listed threatened or endangered species that have been reported to occur near the SAEP. A detailed description of the federal- and state-listed endangered, threatened, or special concern species that have the potential to occur in the vicinity of SAEP and the status of each is provided in the Preliminary Assessment Screening (W-C, 1991). Appendix T-2 contains updated information regarding rare, threatened, or endangered species in Connecticut.

No federally-listed threatened and endangered plant species have been reported to occur in the vicinity of SAEP, although a total of 13 plant species were proposed to be listed by the State of Connecticut in 1999. Several of these species (saltpond grass – last seen in 1901, seabeach sandwort - 1907, mudwort - 1897, yellow-fringed orchid - 1909, salt marsh bulrush - 1943, and coast violet - 1905) have not been reported recently in the vicinity of SAEP and may be extirpated from the area. Since no native terrestrial vegetation exists at SAEP, no state-listed plant species are expected to be present.

Two federally-listed and 11 additional stated-listed threatened, endangered, or special concern birds have the potential to occur in the vicinity of SAEP. The two federally listed species include the piping plover and roseate tern. In 1997, Lee LeBlanc (1997) reported 2 roseate terns at one occasion on a sand bar 200 yards offshore from the Connecticut Audubon Coastal Center near the SAEP Site, the most recent report of the Roseate tern in the vicinity of SAEP prior to that was in 1969. No further sightings were recorded during the 1997 survey.

The piping plover frequently occurs in the vicinity of SAEP, and nesting habitat is located on Short Beach, Long Beach, and Milford Point. These areas are extremely important to the continued survival of the piping plover and select areas of these beaches are closed to public activity during the nesting and rearing period. Short Beach is of particular interest since it borders the Marine Basin. During brood raising, piping plovers usually feed in areas contiguous to nesting territories increasing the probability that these birds may forage in the Marine Basin, both chicks and adults (USFWS, 1995). During migrations, mating, and after fledging, both adults and yearling may gather at non-territorial feeding grounds such as the Tidal Flats.

Piping plover nesting patterns have been documented in the Site vicinity for over 15 years. During 1982-1990, between 8 and 19 pairs have nested annually on Short Beach, Long Beach, and Milford Point (CTDEP, 1990). The highest nesting activity occurred in 1989 and 1990 when 17 and 19 pairs nested, respectively. The previous high had been 12 pairs. During this period 11 to 29 young birds (fledges) were produced annually, with 1989 and 1990 also being the highest years (CTDEP, 1990). In 1997, 6 nests were observed in these areas producing 24 eggs and 4 young fledglings (LeBlanc, 1997). The mudflats surrounding the SAEP Site were monitored regularly between May and August, 1997 and no piping plovers were observed feeding in the Tidal Flats (LeBlanc, 1997).

The beach area utilized as nesting habitat by the piping plover is probably also used by the least tern, which is a state-listed threatened species. Again, during the shorebird monitoring program in 1997, approximately 30-40 nesting least terns were observed on a sand bar 200 yards offshore from the Connecticut Audubon Coastal Center in the vicinity of SAEP (LeBlanc, 1997). Since these birds are mainly fish eaters, birds were observed only twice for short periods during high tides in the Tidal Flats at the SAEP Site.

Nesting Success Data for Piping Plover and Least Tern in Connecticut

	1997	1998	1999	2000
Piping Plover				
pairs nesting	25	21	22	22
# young fledged	44	22	32	41
fledged/pair	1.8	1.0	1.4	1.9
Least Tern				
pairs nesting	~ 400	447	335	239
# young fledged	75	105	104	26
fledged/pair	0.20	0.23	0.31	0.11

Statewide monitoring for these two bird species reveals that during 1997 –2000, 21 to 25 pairs of piping plovers were recorded nesting in Connecticut (DEP 1998, 1999, 2000, and 2002). Nesting success during these years resulted in 22 to 44 young fledglings. Similar information exists for the least tern. Numbers of pairs nesting ranged from 239 to 447. Nesting success during these years resulted in 26 to 105 young fledglings (CTDEP 1998, 1999, 2000b, 2002). Appendix T-2 contains additional detail regarding the statewide monitoring results.

CTDEP records also list the Atlantic sturgeon (Acipenser oxyrhynchus) as a state-threatened species, which may use the Housatonic River. A 1996 report entitled Spatial and Temporal Composition of the Fisheries Resources of the Lower Housatonic River, Connecticut, records the capture of one adult sturgeon in 1993, over 8 river miles upstream from the SAEP Site (CTDEP 1996b). While this capture was the first recorded in "modern times", Atlantic sturgeons no longer spawn in Connecticut (CTDEP, 1996b). The report classifies the fish as an occasional saltwater visitor. Any use of the Site by this species would be transient at most. No specific exposures or risks are predicted for this species, and therefore no further assessment will be conducted in this BERA.

Although not identified by USFWS or DEP, the federally listed southern bald eagle (Haliaeetus 1. leucocephalus) has been reported to possibly pass through the SAEP area as migrating transients. Any use of the Site by this species is likely to be transient, and given the eagle's extremely large foraging range relative to the portion of river adjacent to the site, no significant exposures or risks are predicted for this species.

Since SAEP does not contain habitat utilized by most of the listed threatened, endangered, or special concern bird species, they would not be expected to occur on the site. However, the intertidal flats along the Housatonic River provide feeding areas for waterfowl and shore birds. Therefore, it would be expected that piping plover and least tern that nest in the general area would use this feeding area to some degree.

12.1.4.3 Complete Exposure Pathways. Exposure pathways are identified for two groups of ecological receptors (aquatic receptors, and semi-aquatic wildlife). The exposure pathway includes a source of contamination, potentially contaminated media, and an exposure route.

Aquatic Receptors
Exposure pathways for aquatic receptors (e.g., fish, invertebrates, juvenile amphibians, and plants) at the SAEP Site include direct contact with and ingestion of surface water and sediment in the Tidal Flats, Outfall 008, Marine Basin, and Reference areas.

Aquatic receptors may also be exposed to contaminants in groundwater as it discharges to and mixes with surface water. Primary contaminants in shallow groundwater are VOCs. In Section 8, the evaluation of potential sources and groundwater data shows that, with the exception of a metals plume located under Buildings B-2 and B-10 (400 feet upgradient from the Tidal Flats), there are no other release-related metals present in the aquifer. As discussed in Section 9 of the RI (Fate and Transport), the groundwater model simulations indicate that chromium transport toward the Housatonic River is very slow. The chromium has moved little after 50 years, and has not yet migrated to the Tidal Flats. Based on the hydrogeologic conceptual model, groundwater from SAEP does not discharge to the Marine Basin. Since there are no metals plumes originating from the site known to be discharging to the Tidal Flats or other areas, the transport pathway for metals in groundwater is incomplete, and the evaluation is limited to VOCs in groundwater.

Aquatic receptors evaluated in this BERA are the same as those identified in the original Draft Problem Formulation (URSGWC, 1999). These include:
- Benthic macroinvertebrates (collectively),
- Forage fish (e.g., Atlantic silverside and mummichog), and
- Piscivorous fish (e.g., striped bass).

Semi-Aquatic Wildlife
The wildlife exposure routes that are believed to contribute the highest potential contaminant exposures include incidental ingestion of site media, and ingestion of food items that have bioaccumulated and bioconcentrated contaminants from Site media. Wildlife may also be exposed to chemicals in surface water and sediment by foraging in aquatic environments and eating prey that has bioaccumulated chemicals in their tissue.

Dermal exposures to wildlife are not evaluated because there are few data relating dermal exposures to toxic responses in wildlife. An assumption was made that fur and feathers limit the transfer of contamination across the dermis.

Inhalation of VOCs is also not evaluated because this does not represent a complete exposure pathway for ecological receptors. VOCs in surface water or sediment are unlikely to be released to the air at concentrations that would result in significant exposures.

Semi-aquatic wildlife receptors evaluated in this BERA include the following species:
- Black duck
- Spotted sandpiper
- Great blue heron
- Raccoon

The semi-aquatic wildlife species evaluated in this BERA are the same as those identified in the original Draft Problem Formulation (URS-GWC, 1999), with one exception. The black duck was selected in place of the mute swan, because the latter is an introduced species that has become problematic in many areas.

12.1.5 Identification of Assessment and Measurement Endpoints

Based on USEPA guidance (USEPA, 1997c), an important step in the problem-formulation process is the identification of assessment and measurement endpoints, which must be determined before exposure, toxicity, or risk can be estimated. Endpoints are used in the BERA to define the ecological attributes to be protected or assessed (i.e., assessment endpoints) and to define measurable characteristics of those attributes that can be used to gauge the degree of impact that may occur (i.e., measurement endpoints).

An assessment endpoint is an ecological attribute that, if found to be significantly affected, would indicate a need for remediation. Assessment endpoints most often relate to attributes of biological populations or communities (e.g., abundance, richness, and productivity); individual-based assessment endpoints typically are relevant only if endangered species are present.

In general, assessment endpoints should have social relevance (i.e., be valued by society) and biological relevance, be measurable or predictable, and be susceptible to the hazard or stress being evaluated. Assessment endpoints generally are populations, communities, or trophic guilds. Measurement endpoints should be related to or predictive of the assessment endpoint, readily measurable, and appropriate to the exposure pathways, size, and temporal dynamics of the site.

The overall assessment endpoints for the BERA are the protection and maintenance of aquatic and semi-aquatic receptor populations at the Site, and protection of individuals of endangered or threatened species. The objective of the BERA is to determine if exposure to contaminant concentrations detected in site media is likely to cause a decline in receptor populations or to adversely affect the integrity of aquatic communities. The general types of effects of concern include:

- Mortality, growth, or reproductive effects resulting from direct exposure to contaminants that affect a significant proportion of a receptor population;
- Mortality, growth, or reproductive effects resulting from exposure to contaminants that have bioaccumulated in the ecological food chain that affect a significant proportion of a (higher trophic level) receptor population; and,
- Indirect effects in higher trophic level receptors associated with a substantial reduction in abundance of prey populations.

Assessment and measurement endpoints for the SAEP Site are the same as those identified in the draft BERA.

12.1.5.1 Assessment Endpoints. Assessment endpoints were developed for aquatic habitats. By evaluating and protecting these assessment endpoints, it is assumed that the ecosystem as a whole is also protected. For each assessment endpoint, a risk question is identified that clearly states the question to be answered. Each assessment endpoint and risk question is identified and discussed in detail below. These endpoints are presented in greater detail in Table 12-32.

Benthic Macroinvertebrate Communities
Benthic macroinvertebrate communities are keystone components of the aquatic ecosystem. Their most important function at the SAEP site is as a food resource for higher order consumers, including rare, threatened, and endangered shorebirds known to occur in the vicinity. Benthic organisms also play vital roles in the transfer of biomass through the food chain. Decreases in overall abundance would have negative effects on these higher order consumers and the estuarine ecosystem.

Assessment Endpoint:

- Protection of benthic community structure and function in the Tidal Flats, Marine Basin, and Outfall 008 Drainage. Are levels of site contaminants sufficient to adversely affect benthic communities in the vicinity of the site?

Fish Community
Estuarine ecosystems are recognized as important spawning and nursery habitats for marine fishes. These habitats are complex and provide a variety of niches for fish growth and community development. While the Tidal Flats and Marine Basin around the SAEP Site are important in providing these habitats, assessment endpoints that consider community effects and structure are not easily measurable. The Tidal Flats support food resources important for fish, however their use is temporally limited because they are exposed twice daily during low tides. The small size and tidal nature of the Marine Basin is also conducive to transient movement of fish within the general area. Further, more favorable, permanently inundated salt marsh habitat occurs upstream and across the Housatonic River in Nell's Island Wildlife Refuge. Therefore, no assessment endpoints were identified for evaluation of the overall fish community structure.

Direct contact and direct ingestion of sediments and biota by piscivorous and omnivorous fish are important exposure pathways at the SAEP Site. Forage fish are likely to represent the most critical group for several reasons. These forms are not migratory and have a small home range, hence, individual use rates within Site areas may be high. Bioaccumulation of contaminants in prey species and their impacts on growth and reproduction within the food chain is important. Transfer and magnification in predatory species can pose significant threats to other environmental consumers as well as humans.

Assessment Endpoints:

- Protection of forage fish in the Tidal Flats and Marine Basin against toxicity effects on survival, growth and reproduction from bioaccumulation of COPCs: Are levels of site contaminants sufficient to cause survival, growth, or reproductive impairment of forage fish populations?
- Protection of predatory fish using the Tidal Flats and Marine Basin against toxicity effects on survival, growth and reproduction from bioaccumulation of COPC: Are levels of site contaminants sufficient to cause survival, growth, or reproductive impairment of predatory fish populations?

Avian Community
The area around the SAEP Site is important for avifauna use. Four categories of avifauna have the potential for exposure including shore birds, song birds, waterfowl, and piscivorous birds. Site-related chemicals in sediments and biota have the potential to adversely affect these

populations. The small areas of marginal terrestrial and wetland habitats along the Causeway and Outfall 008 Drainage (especially when compared to the surrounding wildlife refuges) do not provide sustainable resources and preferred niches attractive to song birds. Therefore, species of song birds will not be included as valuable potential receptors with ecological assessment endpoints. Key receptor species representing the remaining three categories have the following assessment endpoints.

Assessment Endpoints:

- Protection of shore birds using the Tidal Flats, Marine Basin, and Outfall 008 Drainage against toxicity effects on survival, growth and reproduction from bioaccumulation of COPCs: Are levels of site contaminants sufficient to cause survival, growth , or reproductive impairment in shorebird populations?
- Protection of waterfowl using the Tidal Flats, Marine Basin, and Outfall 008 Drainage against toxicity effects on survival, growth and reproduction from bioaccumulation of COPCs: Are levels of site contaminants sufficient to cause survival, growth , or reproductive impairment in waterfowl populations?
- Protection of piscivorous birds using the Tidal Flats and Marine Basin against toxicity effects on survival, growth and reproduction from bioaccumulation of COPC: Are levels of site contaminants sufficient to cause survival, growth, or reproductive impairment in piscivorous bird populations?

Mammalian Community

Mammal use of the SAEP Site is restricted to small, poorly developed riparian corridors that include areas along the Tidal Flats, Marine Basin, and Outfall 008 Drainage. Assessment endpoints for mammals will be developed for the Tidal Flats, and Marine Basin because these areas are most likely to have complete exposure pathways to their habitat size, history and proximity to the SAEP facility. These areas are highly disturbed when compared to wildlife refuges in the Site vicinity, and offer little quality habitat for communities or large populations. Semi-aquatic mammals are most likely to use the Tidal Flats and Marine Basin because they provide the most cover and habitat of the areas near the Site. Although semi-aquatic mammals may use the Outfall 008 Drainage, due to the degraded habitat (channelization) and extreme fluctuations in water level and salinity, aquatic prey species are not abundant enough to sustain a foraging resource.

Assessment Endpoint:

- Protection of semi-aquatic mammals using the Tidal Flats and Marine Basin against toxicity effects on survival, growth and reproduction from exposure to or bioaccumulation of COPCs in their prey items: Are levels of site contaminants sufficient to cause survival, growth, or reproductive impairment in semi-aquatic mammal populations?

Rare, Threatened, Or Endangered Species

Rare, threatened, or endangered species are evaluated separately. Their protected status is due to the increased susceptibility of their populations as a result of numerous factors, including habitat loss, over-harvesting, and other anthropogenic effects. Because of their susceptibility, loss or adverse impacts to even one individual is considered a significant impact.

The two federally listed species known to occur in the vicinity of the SAEP Site include the piping plover and least tern. Exposures to these species are expected to be similar to those estimated for other semi-aquatic organisms at the site. However, given their protected status and vulnerable populations, emphasis is placed on protecting individual organisms (rather than populations), and emphasis must also be given to maximum concentrations (rather than average exposures).

Assessment Endpoint:

- Protection of individuals of these species against toxicity effects on survival, growth, and reproduction from exposure to or bioaccumulation of COPCs in their prey items. : Are levels of site contaminants sufficient to cause survival, growth, or reproductive impairment in individual piping plovers or least terns?

12.1.5.2 Measurement Endpoints. Two general types of measurement endpoints are used in this BERA: 1) comparison of estimated or measured exposure levels of COPCs to levels known to cause adverse effects, and 2) bioassay testing of site and reference media. Specific measurement endpoints selected to address each assessment endpoint are discussed below, and summarized in Table 12-32.

<u>Benthic Macroinvertebrate Community (Assessment Endpoint #1)</u>

- Measured concentrations of COPCs in the tissue of field-caught aquatic macroinvertebrates to determine uptake and bioavailability. The tissue chemical results will also be used in food chain models to determine ingestion doses for wildlife receptors that feed on these invertebrates. Medium to High Inference Weight.
- Toxicological response of laboratory animals exposed to whole sediment collected from the Tidal Flats, Marine Basin, and Outfall 008. Assess degree of correlations between COPC concentration and biological response to identify potential stressors. Medium Inference Weight.
- Relationship between measures of aquatic macroinvertebrate community structure and function and sediment COPCs. Medium to High Inference Weight.
- Comparison of surface water and sediment COPC concentrations with aquatic effects criteria/guidelines to provide measure of effect. Low to Medium Inference Weight.

<u>Forage Fish Populations (Assessment Endpoint #2)</u>

- Measured concentrations of COPCs in the tissue of field-caught fish to determine uptake and bioavailability. The tissue chemical results will also be used in food chain models to determine ingestion doses for wildlife receptors that feed on these fish. Compare to Critical Body Residues (CBRs) to provide measure of effect. CBRs are whole body tissue threshold concentrations, above which adverse effects to survival, growth, or reproduction might be anticipated. High Inference Weight.
- Comparison of surface water and sediment COPC concentrations with aquatic effects criteria/guidelines to provide measure of effect. Low to Medium Inference Weight.

Predatory Fish Populations (Assessment Endpoint #3)

- Measured concentrations of COPCs in tissue of field-caught fish to determine uptake and bioavailability. The tissue chemical results will also be used in food chain models to determine ingestion doses for wildlife receptors that feed on these fish. Comparison to CBRs to provide a measure of effect. Medium Inference Weight (because the fish tissue data are for forage fish).
- Comparison of surface water and sediment COPC concentrations with aquatic effects criteria/guidelines to provide measure of effect. Low to Medium Inference Weight.

Shorebirds, Waterfowl, and Fish-Eating Bird Populations (Assessment Endpoint #4)

- Estimate dietary doses of sediment COPCs using food chain models. Input parameters will be based on site-specific BSAFs to approximate concentrations in prey items, and literature-derived exposure factors (e.g., receptor-specific ingestion rate, body weight, home range, and dietary composition). Comparison of estimated doses to literature-derived Reference Toxicity Values (RTVs) will provide a measure of effect. Medium Inference Weight.

Semi-Aquatic Mammal Populations (Assessment Endpoint #5)

- Estimate dietary doses of sediment COPCs using food chain models. Input parameters will be based on site-specific BSAFs to approximate concentrations in prey items, and literature-derived exposure factors. Comparison of estimated doses to literature-derived RTVs will provide a measure of effect. Medium Inference Weight.

Rare, Threatened, or Endangered Species (Assessment Endpoint #6)

- Extrapolation of food chain model results for other semi-aquatic organisms, with emphasis on NOAEL-based endpoints (as opposed to LOAEL-based endpoints). Medium Inference Weight.

12.2 ANALYSIS

This subsection of the BERA characterizes the potential ecological exposures and associated ecological effects. The exposure assessment identifies the receptors and exposure pathways that will be evaluated, identifies the exposure point concentrations that will be used to quantify exposures to these receptors, and describes the method that will be used to quantify exposure. The effects assessment includes a discussion of the available toxicity benchmarks and a discussion of the Site-specific toxicity tests used to quantify effects.

12.2.1 Exposure Assessment

Exposure assessment is the process of estimating or measuring the amount of an ecological COPC in environmental media (surface water, sediment, shallow groundwater, and biological tissue) to which an ecological receptor may be exposed via respective exposure routes (e.g., ingestion or direct contact). Indirect exposures associated with consumption of contaminated prey items are also quantitatively evaluated based on the concentrations of COPCs measured or estimated in prey items. The exposure assessment identifies receptors and exposure pathways that will be quantitatively evaluated; receptors and pathways are those identified previously in Subsection 12.1.4.3. The exposure assessment also identifies the exposure point concentrations that will be used and includes a discussion of the methods used to quantify exposure to semi-aquatic wildlife receptors via a food chain model.

The following table summarizes exposure areas at the SAEP Site and media evaluated for those areas:

Exposure Area	Sediment	Surface Water	Finfish	Shellfish (Ribbed mussel)	Shellfish (Oyster)	Invertebrates	Shallow Ground-water
Intertidal Flats	X	X	X	X	X	X	X
Marine Basin	X	X	X	X		X	
Outfall 008 Area	X	X				X	
Reference Area	X	X	X	X	X	X	

12.2.1.1 Calculation of Exposure Point Concentrations. EPCs were identified for surface water, sediment, and shallow groundwater at the SAEP. Surface water and sediment EPCs were identified for Tidal Flats, Outfall 008 Drainage, and Marine Basin. EPCs were also identified for the background locations; these concentrations are used in the risk characterization to evaluate the magnitude of risks associated with historical practices at SAEP relative to those that are associated with local background conditions. The incremental contribution of background risk relative to site risks is discussed further in the risk characterization.

USEPA Region I guidance requires consideration of RME; however, CT exposures are also presented because they represent the more likely exposures, and along with the RME provide a range of risk estimates for risk management decision-making.

CT and RME EPCs were evaluated in the BERA. Arithmetic mean concentrations are intended to represent the most likely concentration of a COPC to which an ecological receptor might be exposed; the arithmetic mean concentration is referred to as the CT EPC. If the average concentration exceeded the maximum detected concentration (due to elevated SQLs), the maximum concentration was used as the CT EPC. The 95% UCL on the arithmetic mean was used, where available, to represent the upper bound concentrations of a COPC to which an ecological receptor might be exposed; this concentration is referred to as the RME EPC. Methods used to calculate the RME EPC are discussed in Section 12.2.3.2 of the HHRA. For data sets that are too small to calculate a 95% UCL, the maximum detected concentration was used to estimate the RME EPC.

12.2.1.2 Quantification of Exposure for Wildlife - Food Chain Model. Appendix T-3 presents the food chain model used to estimate food chain exposures to COPCs in surface water and sediment for representative wildlife species. Ecological receptor species discussed above were identified for each exposure area. For each receptor, information that determines their likely exposure (e.g., body weight, food ingestion rate, dietary preferences, and foraging range) is summarized in Appendix T-3, Table T-3.1.

Dietary exposures to contaminated prey items were estimated using site-specific BSAFs that were calculated using tissue data from organisms collected from the site (mussels, shellfish, oysters, and fish) or exposed to site media (the aquatic worm Nereis virens). Biota were analyzed for analytes that were identified as potential risk drivers in previous investigations. These were limited to inorganic analytes (including mercury), and PCBs. They were also analyzed for percent lipid. These site-specific data were used to help reduce uncertainties associated with COPC bioavailability and uptake. Calculating a site-specific BSAF meant that it could be more easily used in the food chain model in combination with literature-based BSAFs for analytes lacking site-specific tissue data. Dietary exposures to contaminated prey items for other analytes (e.g., PAHs in invertebrates) were estimated using literature-derived bioaccumulation factors (BAFs) and BSAFs.

A daily dose of ingested food and abiotic media was calculated to estimate exposures to wildlife receptors. The daily dose of a COPC to a given receptor depends upon (1) the COPC concentration in plant and animal prey items, (2) the endpoint receptor's trophic level, (3) the trophic level of animal food items, and (4) the measurement endpoint receptor's food and media ingestion rates (USEPA, 1999c). The complexity of the daily dose equation depends upon the number of food items in an endpoint receptor's diet and the trophic level of each prey item and of the endpoint receptor.

The equations used to derive an Estimated Daily Intake (EDI) dose are discussed below.

<u>Surface Water Ingestion Pathway</u>. The receptor-specific average daily dose ($\text{ADD}_{\text{water}}$), expressed in mg/kg/day, will be estimated as the quotient of the mass of a given COPC ingested on a daily basis and the body mass of the receptor being evaluated:

$$\text{ADD}_{\text{water}} = \frac{(\text{EPC}_{\text{water}} * \text{IR}_{\text{water}})}{\text{BW}} * (\text{SFF})$$

where:

$\text{ADD}_{\text{water}}$ = Daily dose of a COPC from consuming surface water (mg/kg/day);
$\text{EPC}_{\text{water}}$ = Exposure point concentration (mg/L) in surface water;
IR_{water} = Water ingestion rate (L/day) for wildlife receptor;

SFF = Site Foraging Frequency (unitless) which represents the fraction of the exposure area relative to the home or forage range of the receptor; and,
BW = Body weight (kg) of receptor

Incidental Sediment Ingestion Pathway. The receptor-specific average daily dose ($ADD_{sediment}$), expressed in mg/kg/day on a dry weight basis, will be estimated for each receptor as follows:

$$ADD_{sediment} = \frac{(EPC_{sediment} * PD_{sediment} * IR_{total})}{BW} * (SFF)$$

where:

$ADD_{sediment}$ = Daily dose of a COPC from incidental ingestion of sediment (mg/kg/day);
$EPC_{sediment}$ = exposure point concentration (mg/kg) in sediment (dry-weight basis);
$PD_{sediment}$ = Fraction of sediment in overall diet (unitless);
IR_{total} = Total ingestion rate (kg/day) for wildlife receptor;
SFF = Site Foraging Frequency (unitless) which represents the fraction of the exposure area relative to the home or forage range of the receptor; and,
BW = Body weight (kg) of receptor

Contaminated Prey Consumption Pathway. The receptor-specific average daily dose (ADD_{diet}), expressed in mg/kg/day on a wet weight basis, will be estimated for each receptor as follows:

$$ADD_{diet} = ADD_{fish} + ADD_{inv}$$

where:

ADD_{diet} = Average or 95% UCL daily dietary dose of a given COPC in diet;
ADD_{fish} = Average or 95% UCL daily dietary dose of a given COPC from consumption of fish; and,
ADD_{inv} = Average or 95% UCL daily dietary dose of a given COPC from consumption of invertebrates.

The average daily dose of a COPC that a given receptor receives from the portion of its diet represented by fish, expressed on a wet weight basis, will be estimated as follows:

$$ADD_{fish} = \frac{(EPC_{fish} * PD_{fish} * IR_{total})}{BW} * (SFF)$$

where:

ADD_{fish} = Daily dose of a COPC from ingestion of fish (mg/kg/day);
EPC_{fish} = Average or 95% UCL exposure concentration (mg/kg) in fish;
PD_{fish} = Fraction of fish in overall diet (unitless);
IR_{total} = Total ingestion rate (kg/day) for wildlife receptor;
SFF = Site Foraging Frequency (unitless) which represents the fraction of the exposure area relative to the home or forage range of the receptor; and,
BW = Body weight (kg) of receptor

The average daily dose of a COPC that a given receptor receives from the portion of its diet represented by invertebrates, expressed on a wet weight basis, will be estimated as follows:

$$ADD_{inv} = \frac{(EPC_{inv} * PD_{diet} * IR_{total})}{BW} * (SFF)$$

where:

ADD_{inv}= Daily dose of a COPC from ingestion of invertebrates (mg/kg/day);
EPC_{inv} = Average or 95% UCL exposure concentration (mg/kg) in invertebrates;
PD_{inv} = Fraction of invertebrates in overall diet (unitless);
IR_{total} = Total ingestion rate (kg/day) for wildlife receptor;
SFF =Site Foraging Frequency (unitless) which represents the fraction of the exposure area relative to the home or forage range of the receptor; and,
BW = Body weight (kg) of receptor

The receptor-specific exposure parameters, including dietary composition, daily ingestion rates, body weight, and foraging range) are summarized in Appendix T, Table T-3.1. Food and water ingestion rates were estimated using allometric equations that relate food and water intake to body weight of the receptor species, as presented in the Wildlife Exposure Factors Handbook (USEPA, 1993a). Information on the dietary composition for specific receptor species was obtained from a variety of sources. The amount of each prey item ingested daily (kg/day) was calculated by multiplying the food ingestion rate (kg/day) by the fraction of the diet comprised by each prey item. The incidental sediment ingestion rate for some receptors was based on data presented in the Wildlife Exposure Factors Handbook (USEPA, 1993a), or on professional judgment.

Exposure concentrations in prey items were calculated using site-specific BSAFs. These BSAFs were calculated using measured concentrations in site biota, and the sediment concentrations to which those biota have been exposed. BSAF calculations are provided in Appendix T-3, Tables T-3.3 and T-3.4 for invertebrates and fish, respectively.

In identifying an invertebrate concentration, oyster, ribbed mussel, and aquatic worm tissue concentrations were all considered. For each COPC, the highest average tissue concentration from among these three groups was selected and used to represent the invertebrate tissue concentration for that area. The average sediment concentration from within that exposure area was used to calculate a BSAF. BSAFs are also adjusted to incorporate lipid content in tissue, and organic carbon content in sediment. However, lipid data for fish tissue were not used, because reported lipid values in the database were outside the expected range of lipid content (i.e., ranging upwards from 70 % to over 100 %) and suggestive of analytical or recording error. Therefore, an assumed percent lipid content of 5% was used for fish tissue. This was based on professional judgment given the relatively small size of the species of fish included in the risk assessment and lipid contents in forage fish from other New England sites.

Biota samples were not analyzed for PAHs, and therefore BSAFs for PAHs could not be calculated using site data. Therefore literature-based BSAFs were used. These BSAFs were derived using data presented in the U.S. Army Corps of Engineers Waterways Experiment Station BSAF database (USACE, 1999). Fluoranthene was used as a surrogate for other PAHs. These BSAFs are included in Table T-3.3. PAHs are not known to bioaccumulate in fish tissue, and therefore the fish BSAF were assumed to be zero, as shown on Table T-3.4.

A Site Foraging Frequency (SFF) was used to estimate the percentage of time the indicator species will forage in a particular area based on the animal's home range and the exposure area. The SFF was estimated by dividing the receptor's home range by the area of contamination; by definition, the SFF cannot exceed 1. Exposure areas for each of the exposure point areas shown in Figure 11-3 calculated using the SAEP GIS system are as follows:

Exposure Area	Area (hectares)
Tidal Flats	25
Outfall 008 Drainage	3.4
Marine Basin	8.5
Reference Area	24

12.2.2 Ecological Effects Assessment

The methods used for identifying and characterizing potential adverse ecological effects for aquatic and semi-aquatic receptors are described in the following subsections.

12.2.2.1 Aquatic Receptors. Risks to aquatic receptors are evaluated in this BERA based on site-specific toxicity test results, benthic community data, and/or published RTVs, each of which are discussed below.

Toxicity Tests

Solid phase toxicity testing was conducted in 1994 and 1999 using three test species; two amphipods, Ampelisca abdita and Leptocheirus plumulosa and a polychaete, Neanthes arenacedentata. Tests were performed on sediment samples collected at one reference station and 15 Site stations in 1994, and four reference stations and 17 Site stations in 1999 (see Figure 12-1). These results are a site-specific, direct measurement of sediment toxicity. Results of the testing are considered as part of the weight of evidence evaluation for Site areas. Data are included in Tables 12-33, 12-34, and 12-35. Testing was performed in accordance with the ASTM Standard E-1367-92. Exposure tests were conducted within two weeks of collection of the test sediments for 1999 samples. In 1994, an initial test resulted in unacceptable control mortality and the test was restarted at a later date. The length of the storage period for 1994 exceeded the ASTM recommendations, but is not believed to have affected the results. Storage of sediments for periods up to 6 months have shown little to no change in toxicity and/or bioaccumulation (ASTM, 1994; USEPA, 1994c). Toxicity test results are presented in the Risk Characterization section.

Benthic Community Analyses
Benthic macroinvertebrate community analyses were performed at reference locations and Site areas in 1994 and 1999. The communities observed at these locations have already been characterized in Subsection 12.1.4.2. Tables 12-30 and 12-31 present summaries of the numbers of organisms and taxa observed in each area in 1994 and 1999. Three different metrics were used to evaluate the benthic community data for the SAEP as follows:

- Taxa richness
- Taxa density (a measure of overall abundance)
- Percent contribution of the dominant taxon

Emphasis is placed on overall taxa richness and density, as an important function of the benthic community at the sits is as a food source for higher order consumers, including protected shorebirds.

Table 12-36 includes summaries of both the toxicity test results (survival only) and benthic community metrics by area. Figure 12-1 identifies toxicity test and benthic sampling locations. Benthic community results are presented in the Risk Characterization section.

Reference Toxicity Values
RTVs provide another useful measure of potential risks to aquatic life. RTVs for surface water (and groundwater), sediment, and biota are discussed below.

Surface Water
The surface water benchmarks used for the BERA are presented in Table 12-37. This table represents a compilation of criteria and guidelines from different regulatory agencies that were derived to be protective of aquatic receptors exposed to contaminants in surface water. The different sources of information are discussed briefly below. The lowest identified surface water benchmarks are used in the BERA as a measure of effect.

The primary criteria and guidelines used to select surface water benchmarks include: chronic Ambient Water Quality Criteria (AWQC) (USEPA, 2002c; 1991c), CTDEP chronic Surface Water Quality Standards (SWQS) for freshwater (CTDEP, 1997), chronic ecotoxicity thresholds (ETs) for freshwater (USEPA, 1996b), and Tier II screening benchmarks developed by Oak Ridge National Laboratories (Suter and Tsao, 1996). These criteria are used as NOAEL-equivalent toxicity effects endpoints at both the individual and community levels for pelagic (i.e., water column) species.

Chronic AWQC are equal to the 4-day average concentration not to be exceeded more than once every three years. AWQC are intended to be protective of aquatic life and its uses, and therefore, for chemicals known to bioaccumulate (e.g., mercury and pesticides), the AWQC are often protective of wildlife and/or humans that may ingest aquatic life. When chronic AWQC are not available, acute AWQC were used instead. The USEPA has also identified Lowest Observed

Effects Concentrations (LOECs) for many chemicals (including chlorinated solvents) for which there were not enough data to generate an AWQC; these values are generally accepted in place of AWQC. The CTDEP SWQSs are based on the 1991 AWQC, with the exception that the hardness adjusted standards are based on filtered metals.

The AWQC and CTDEP SWQS for cadmium, copper, lead, nickel, and zinc are normalized to a hardness concentration of 100 mg/L as $CaCO_3$; hardness measurements were not available; however, hardness values estimated based on calcium and magnesium indicate that 100 mg/L is a conservative estimate of hardness in these water bodies.

Sediment

The sediment benchmarks used for the BERA are presented in Table 12-38. This table represents a compilation of criteria and guidelines derived by several methods (described below) to be protective of aquatic receptors exposed to contaminants in sediment. The different sources of information are discussed briefly below. For each chemical, the lowest benchmark from different sources of information is identified in Table 12-38. The lowest identified sediment benchmarks are used in the BERA as a measure of effect.

Sediment benchmarks were derived from several sources of information, including: the USEPA SQCs and guidelines (USEPA, 1993 b, c, d, e, f; USEPA, 1988b), the Ontario Ministry of the Environment (OME) LELs (Persaud et al., 1996), and the NOAA ER-L values (Long et al., 1995; Long and Morgan, 1990). MacDonald et al, (2000) have identified consensus-based TEC and PEC. TECs are levels below which harmful effects are unlikely to be observed, whereas PECs are levels above which harmful effects are likely to be observed.

SQCs have been proposed or established by USEPA for a limited number of organic chemicals (USEPA, 1993b,c,d,e,f; USEPA, 1988b). These are the only published sediment criteria. The USEPA SQCs, which are adjusted by the TOC content in sediment, are bulk sediment concentrations that are equivalent to the chronic AWQC. Since the AWQC are designed to be protective of 95 % of aquatic species, these bulk sediment concentrations (generated using an equilibrium partitioning approach) are expected to be equally conservative in evaluating adverse effects to benthic organisms (USEPA, 1992c).

The OME LELs represent chemical concentrations below which biological effects are improbable (by the Apparent Effects Threshold method [USEPA, 1992c]). The OME values were derived largely using data from the Great Lakes and other freshwater bodies.

NOAA ER-Ls represent concentrations that may rarely (i.e., 10th percentile) be associated with toxicity to benthic organisms (by the National Status and Trends Program Approach [USEPA, 1992c]). These sediment guidelines are based on a database of primarily marine and estuarine studies.

If no USEPA SQCs, OME LELs, NOAA ER-Ls, or TECs were available for a particular analyte, other sources of information were consulted to derive an appropriate benchmark value. These included the ORNL secondary chronic values for screening COPCs for effects on sediment-associated biota (Jones et al., 1997), and 10th percentile predicted effects thresholds developed by Fuchsmans et al. (1999) for chlorinated benzenes.

Biological Tissue
RTVs were developed by querying available databases to develop Critical Body Residues (CBRs) for aquatic receptors. CBRs are defined as whole body tissue threshold concentrations above which adverse effects to survival, growth, or reproduction might be anticipated. CBRs are summarized in Table 12-39 for finfish, invertebrates (other than shellfish), and shellfish; the development of CBR tissue effect concentrations is discussed in Appendix T-4.

AVS:SEM
One additional measure of sediment effects is the ratio of molar concentrations of Acid Volatile Sulfide/Simultaneously Extracted Metals (AVS:SEM). AVS:SEM data were collected for 21 site locations and 3 reference locations. AVS plays a critical role in partitioning and bioavailability of some metals (primarily divalent cations) in sediments. SEM is a measure of the extractable metals concentration. If the molar SEM concentration is less than the molar AVS concentration, these metals are assumed to be bound to AVS and not bioavailable or toxic. Table 12-40 presents a summary of AVS: SEM data for sediments at SAEP.

12.2.2.2 Semi-Aquatic Wildlife. RTVs were identified from the literature for each selected wildlife receptor. The RTV relates the dose of a COPC in an oral exposure to the likelihood of an adverse effect. RTVs representing dietary ingestion thresholds for lethal and sublethal effects have been identified. Application factors were used to adjust toxicological data depending on the nature of the effects reported and how closely they relate to the assessment endpoints. The toxicity data and RTVs are included in Appendix T-3.

12.3 RISK CHARACTERIZATION

Risk characterization involves the integration of exposure and effects data to determine the likelihood of adverse effects. The HQ method was used primarily to characterize the magnitude of potential risks associated with exposure to the identified COPCs for each of the assessment endpoints. The following sections present the risk characterization results for assessment endpoints for the SAEP Site. Ecological risks are summarized at the end of the section. Both CT and RME exposures were considered in characterizing risk in the BERA. USEPA Region I guidance requires consideration of RME; however, average exposures are also presented because they represent the more likely exposures, and they provide a better perspective for risk management decision-making.

For aquatic receptors, the HQ is expressed as follows:

$$HQ = \frac{EPC_{medium}}{RTV_{medium}}$$

where:

EPC_{medium} = Exposure Point Concentration in a medium (mg/kg or mg/L)
RTV_{medium} = Toxicity benchmark for the COPC in the given medium (mg/kg or mg/L)

For wildlife receptors, the HQ was calculated as follows:

$$HQ = \frac{EDI_{total}}{RTV}$$

where:

EDI_{total} = Estimated Daily Intake for a COPC (total for all exposure pathways) (mg/kg-day)

RTV = Toxicity Reference Value for a measurement endpoint receptor (mg/kg-day).

A calculated HQ that exceeds one may indicate that the assessment endpoint receptor in question may be at risk of an adverse effect from a particular COPC, although the conservative assumptions made throughout the development of the exposure and effects assessments introduces uncertainty regarding the interpretation of the HQ. A cumulative HI consisting of the sum of all HQs for a given receptor was also calculated for each measurement endpoint receptor. It is important to note that the interpretation of ecological risks, as estimated using HQs and HIs, is different from the interpretation of human health risks. In human health risk assessment, any HQ or HI greater than 1 warrants close scrutiny and may be interpreted as posing a potential risk to human receptors. The focus of a human health risk assessment is protection of each individual that might be exposed, and therefore any HQ greater than one may be of concern. The focus of ecological risk assessment, on the other hand, is typically protection of populations of receptors. One exception to this focus is when there are rare, threatened, or endangered species known to occur at or use a site; in this case, the focus is on protection of individual receptors, because the survival of each individual is considered critical for the overall protection of the species. Risks to rare, threatened, or endangered species are discussed in Subsection 12.3.4.

The interpretation of ecological risks is generally categorized using the magnitude of the estimates, with the probability of an unacceptable risk increasing with the size of these ratios. HIs are derived as the sum of chemical-specific HQs, assuming an additive effect. However, HQs for different chemicals are based on different endpoints and different mechanisms, and the assumption of additivity introduces considerable uncertainty to the interpretation of potential risks. Although HIs are presented in food chain spreadsheets and summary tables, emphasis should be placed on HQs as an indication of ecological levels of concern. This emphasis is brought out in the discussion of risks to individual receptors.

HQs and HIs that fall between 1 and 5 are generally interpreted to represent a de minimus risk (Suter, 1993) to the particular receptor population or community. This interpretation is an acknowledgement of the conservative assumptions that are used in developing exposure and effects estimates. HQs and HIs that fall between 5 and 10 represent a “possible” risk, and those greater than 10 represent a “likely” risk to ecological receptors. This is just a generalization; HQs are examined on a receptor and area-specific basis, as needed to examine the basis for those HQs.

The risk characterization also includes an evaluation of incremental risks, which take into account the contribution of background concentrations to the overall site risks. The incremental HQ or HI identifies that portion of the HQ or HI that might be assumed to be site-related (i.e., cannot be attributed to background). This is primarily based upon concentrations in media at reference locations, relative to those at site locations, and is discussed as such in each section. For this BERA the following assumptions are used to interpret incremental risks:

- if the LOAEL-based incremental risk for the central tendency exposure scenario (Site HI – Reference HI) is <1, there is no significant risk attributable to the site,
- If the LOAEL-based incremental risk for the central tendency exposure scenario is between 1 and 5, there may be a significant risk attributable to the site that warrants further evaluation, and
- If the LOAEL-based incremental risk for the central tendency exposure scenario is >5, there is likely to be a significant risk attributable to the site.

The conclusions regarding overall risk to ecological receptors are made by considering various lines of evidence from the results of all components of the assessment (i.e., the approach integrates results of physical, biological, toxicological, and modeling studies to draw risk-based conclusions). The components provide measures of risks for different ecological receptors, exposure pathways, and potential adverse effects. A qualitative weight-of-evidence approach is employed to integrate multiple measurement endpoints in making conclusions about the risks to the selected indicator organisms.

12.3.1 Risks To Benthic Macroinvertebrates

The assessment endpoint is protection of benthic macroinvertebrate community structure in the Tidal Flats, Marine Basin, and Outfall 008 Drainage areas: Are levels of site contaminants in surface water and sediment sufficient to cause adverse effects to benthic communities?

This endpoint was assessed using a number of measurement endpoints:

- comparison of tissue EPCs to tissue CBRs;
- evaluation of sediment toxicity test results;
- evaluation of aquatic macroinvertebrate community structure and function; and
- comparison of surface water and sediment EPCs to aquatic benchmarks.

These endpoints were evaluated separately for each water body at the SAEP Site: Tidal Flats, Outfall 008 drainage area, and Marine Basin.

12.3.1.1 Tidal Flats

<u>Comparison Of Tissue EPCs To Tissue CBRs</u>
Tables 12-41, -42, and -43 present RME, CT and background exposure point concentrations in mussels, oysters, and aquatic worms collected from or exposed to Tidal Flats sediments. These tables also include screening toxicity values for tissue (CBRs) compiled as described in Subsection 12.2.2.1.

CT and RME mussel concentrations of aluminum, chromium, lead, selenium, and vanadium, and Aroclor 1254 in the Tidal Flats exceed CBRs, with HQs ranging from 1 to 158. The RME mussel concentrations of arsenic, copper, and manganese also exceed the CBR, but the CT concentration is below the CBR. CT concentrations of several inorganics (aluminum, chromium, lead, and vanadium) exceed background concentrations.

CT and RME oyster concentrations of cadmium, chromium, lead, mercury, and Aroclor 1254 in the Tidal Flats exceed CBRs, with HQs ranging from 1 to 65. HQs for these analytes are comparable to (i.e., within the same order of magnitude as) those for oysters from the background (reference) location, indicating that these analytes are not site-related.

CT and RME aquatic worm concentrations of aluminum, arsenic, cadmium, copper, mercury, nickel, and zinc exceed their respective CBRs, with HQs ranging from 1.7 to 8.2. These HQs exceed those for aquatic worms from the background location. Mercury was only detected in one of 50 tissue samples. Therefore, mercury tissue levels, even if present at levels below the detection limit, are unlikely to pose a risk to these organisms. Zinc was detected in worm tissue at concentrations comparable to background levels.

Evaluation Of Sediment Toxicity Test Results

Toxicity test results for Tidal Flats were included in Tables 12-33, -34, and –35, presented previously. Among the site areas, polychaete Neanthes survival was the lowest in the Tidal Flats station samples when compared to the reference area. Survival was significantly different from the control sample at two stations, SDTA007 and SDTC006, and growth was significantly reduced at station SDTH001 as compared to growth of the control. Survival at the remainder of sample locations was not significantly different from the laboratory control.

Results of amphipod (Leptocheirus and Ampelisca) toxicity testing in the Tidal Flats in 1994 and 1999 were similar. Based on the analysis, all stations exhibit significant toxicity. Both tests recorded 0% survival for amphipods at nearshore stations in the SDTC transect (SDTC006A in the 1999 test, and TC-1 in the 1994 test).

Table 12-44 presents a summary of results for the amphipod Leptocheirus from the 1999 tests. These results indicated significant toxicity at all stations within the Tidal Flats, and therefore a comparison of detected analytes and benchmarks was included in this table to determine if there were any noticeable trends or relationships between the observed toxicity and contaminant concentrations. The NOAA Effects Range-Median (ER-M) values were used for this comparison at the suggestions of CTDEP. Concentrations of acenapthene, Total PCBs, copper, mercury, methyl mercury, nickel, and silver exceed their respective ER-Ms. Concentrations of these and other analytes are fairly consistent among the stations, differing very little between the station with 0% survival (SDTC006A), and that with 64% survival (SDTA006A). Concentrations of Total PCBs increase with increasing toxicity (i.e., decreased survival), suggesting a possible causal relationship. PCBs are typically associated with bioaccumulation and food-chain related risks, however, and therefore are not usually considered to be highly toxic to lower trophic-level organisms such as invertebrates.

Table 12-45 presents a similar comparison for the Leptocheirus results from the reference area, in which significant toxicity was observed at two of the three stations, even though concentrations of all analytes were below benchmarks. This indicates that observed effects are likely due to non-chemical stressors. A review of Table 12-44 shows consistent TOC and percent fines (grain size) results across sample locations, and therefore these parameters do not appear to be correlated with any observed toxicity.

Evaluation Of Aquatic Macroinvertebrate Community Structure and Function

Table 12-36 contains a summary of benthic macroinvertebrate data from surveys performed at reference locations and Site areas in 1994 and 1999. The results of the benthic macroinvertebrate community analyses in the Tidal Flats serve as another line of evidence in evaluating impacts to the benthic community in this area.

Three different metrics were used to evaluate the benthic community data for the SAEP as follows:

- Taxa richness
- Organismal density
- Percent contribution of the dominant taxon

Additional metrics (Percent composition of various other taxa groups) were added at the request of CTDEP.

Taxa richness is the number of different taxa (or types of organisms) that are collected at a station. Taxa richness reflects the health of the community through a measurement of the variety of taxa present. Taxa richness generally increases with increasing water quality, habitat diversity and habitat suitability. With several exceptions, the taxa richness measured at the reference stations and stations in the vicinity of the Site are within the range of what would be expected for intertidal mudflat communities in an estuarine system within an urbanized watershed. There is little discernable pattern in the richness values for the stations sampled. Several stations (including reference station U-1 and U-2, 10 taxa; Tidal Flats TC01, 11 taxa) had a low richness when compared to other areas on-Site. Based on the data it is difficult to determine whether the differences in individual station richness values are attributable to physical habitat characteristics or some other factor(s). However, the richness observed at stations within the Tidal Flats is comparable to that observed at the reference stations.

Organismal density is the total number of individual organisms per unit area sampled and is a good measure of overall abundance. When evaluated in combination with other metrics, such as richness and percent dominance and community composition, density is a good indicator of environmental quality.

Overall, density in the Tidal Flat samples was high, ranging from 2,622 indiv/ft^2 at Station TB03 to 791 indiv/ft^2 at Station TF03. While densities in the Transect E through F area are lower, the greater taxa richness in this area likely indicates a better overall habitat condition.

Percent contribution of the dominant taxon to the total number of organisms uses abundance of the numerically dominant taxon relative to the rest of the population as an indication of community balance. A community dominated by relatively few taxa likely indicates some form of "natural" or "man-induced" environmental stress. Owing to the relative physical uniformity in the sediment habitat sampled for this investigation, two taxa (oligochaetes as a group, the polychaete Mediomastus ambiseta, and Streblospio benedicti) tended to dominate the benthic biota at all stations. These invertebrates are typical dominants of intertidal muds in estuaries. Overwhelming dominance in the absence of other fauna, however, is likely to represent a poor or stressed condition. This seems to be the case in two stations (TC01 and TB01) in the Tidal Flats where one species of polychaete comprises almost 90% of the total numbers and three stations recorded majorities (>50%) including TA01, TD01 and TB05. This indicates that several nearshore stations within the Tidal Flats have a high percentage dominance by one species (e.g., TB01 and TC01), and do appear to be stressed compared to surrounding stations within the mudflat. However, this effect may be habitat-related, as nearshore habitats will generally be more physically challenging due to the extreme fluctuations in conditions (e.g., longer periods of exposed sediments, etc.). Overall richness and abundance of benthic organisms in the Tidal Flats are comparable to those at reference locations and the function of the benthic community as a whole does not appear to be impacted.

Comparison Of Surface Water and Sediment EPCs To Aquatic Benchmarks
Table 12-46 contains a comparison of RME and CT surface water concentrations in the Tidal Flats to surface water screening toxicity values. Because these screening benchmarks were used to identify COPCs, benchmarks are invariably exceeded by the RME concentrations. CT concentrations, which are a more realistic approximation of exposure point concentrations, also exceed benchmarks for every analyte except vanadium. CT concentrations are generally an order of magnitude or more higher than background levels. Concentrations of the two VOCs that were identified as COPCs are only slightly above their respective benchmarks (CT HQs of 1.4 and 1.6).

Table 12-47 contains a comparison of RME and CT sediment concentrations in the Tidal Flats to sediment screening toxicity values. Background HQs are nearly all at or below one, whereas site HQs for analytes are nearly all above one. However, background concentrations from the Raymark site contained similar concentrations of several PAHs, including fluoranthene (average background concentration 3.77 mg/kg), indeno 1,2,3-cd pyrene (average background concentration 1.55 mg/kg), phenanthrene, average concentration 1.9 mg/kg) and pyrene (average concentration 2.49 mg/kg), suggesting that these analytes detected in the tidal flats may not be elevated above typical background levels. Background concentrations from the Raymark site for the majority of inorganic analytes are higher than background concentrations used in this BERA, and examination of the Tidal Flats sediment data indicates that average concentrations of the majority of inorganic analytes area consistent with the background concentrations from the Raymark site (Tetrated [NUS], 1999). However, Aroclors were non-detect in those background samples. In general, Aroclors are not highly toxic to aquatic invertebrates (which is why they biomagnify in the food chain); therefore it is unlikely that any observed effects are attributable to Aroclors.

AVS:SEM data, summarized in Table 12-40, indicate that SEM:AVS ratios at 9 of 14 stations sampled are less than or equal to one, indicating that divalent inorganic analytes present in sediment are bound with sulfides and not likely to be bioavailable or toxic. Ratios greater than one were reported at one location on Transect A (SDTA006A), one on Transect B (SDTB007A), and at both locations at Transects C and D. Ratios at Transects G and H were all less than one (sample locations are shown on Figure 5-1).

Table 12-48 contains a comparison of RME and CT shallow groundwater concentrations in the Tidal Flats to surface water screening toxicity values. RME and CT HQs for all analytes except carbon disulfide are below one. The HQ for carbon disulfide is only slightly greater than one (HQ=1.2). A benchmark was not available for vinyl chloride. However, the concentration of this VOC is comparable to those of other VOCs detected in shallow groundwater; based on the screening toxicity values identified for those compounds, it does not appear likely to pose a risk to benthic organisms at these levels.

12.3.1.2 Outfall 008 Drainage

Comparison Of Tissue EPCs To Tissue CBRs
Aquatic worms are the only organisms for which tissue data are available at Outfall 008 drainage.

Table 12-49 contains a comparison of RME and CT tissue concentrations to CBRs. CT and RME concentrations of arsenic, cadmium, copper, nickel, and zinc exceed CBRs, with HQs ranging from 1.1 to 2.5. These metals were all detected at concentrations consistent with background levels. Nickel was only detected in 1 of 15 samples from this area.

Evaluation Of Sediment Toxicity Test Results
Toxicity test results for the Outfall 008 Drainage were summarized previously in Tables 12-33 and -34. Survival and growth of the polychaete Neanthes arenacedentata were not significantly different from controls. Amphipod Ampelisca survival in the Outfall 008 Drainage in the 1994 test was 0% at the only station tested compared to 65 % in the reference area. Data from 1999 showed that Leptocheirus plumulosa survival was different from control at one station in the Outfall 008 Drainage and two stations in the reference area. These data suggest that some stations in the Outfall 008 Drainage may affect survival of sensitive species. Given the harsh habitat conditions in the Outfall 008 Drainage (drastic changes in salinity, temperature, and water level) sensitive and intolerant species are not likely to establish viable populations.

Table 12-50 presents a summary of results for the amphipod Leptocheirus from the 1999 tests. These results indicated significant toxicity at one of the three stations within the Outfall 008 Drainage (SD08002A). A comparison of detected analytes and ER-Ms was included in this table to determine if there were any noticeable trends or relationships between the observed toxicity and contaminants. Examination of the concentrations in this table shows that concentrations of chromium, copper, and Aroclor 1260 are notably higher at SD08002A. In general, Aroclors are not highly toxic to aquatic invertebrates (which is why they biomagnify in the food chain); therefore it is unlikely that any observed effects are attributable to Aroclor 1260. Concentrations of nickel and silver exceed their respective ER-Ms at two of 3 locations, but concentrations do not appear to be correlated with toxicity. A review of Table 12-50 shows no obvious relationship between TOC, grain size (percent fines) and observed toxicity.

Evaluation Of Aquatic Macroinvertebrate Community Structure and Function
Table 12-36 contains a summary of benthic macroinvertebrate data from surveys performed at reference locations and Site areas in 1994 and 1999. The results of the benthic macroinvertebrate community analyses in the Outfall 008 Drainage serve as another line of evidence in evaluating impacts to the benthic community in this area.

Taxa richness. The taxa richness at the Outfall 008 Drainage station 008-01 was notably low, with only 6 species recorded in 1999. Several stations (including reference station U-1 and U-2, 10 taxa) also had a low richness. Based on the data it is difficult to determine whether the differences in individual station richness values are attributable to physical habitat characteristics or some other factor(s).

Organismal density. At the Outfall 008 Drainage station 008-01, a low taxa richness coupled with a density of only 270 indiv/ft^2 in 1994 and 440 indiv/ft^2 in 1999 indicates a stressed condition in this general area.

Percent contribution of the dominant taxon. Two taxa (oligochaetes and polychaetes) tended to dominate the benthic biota at all stations. These invertebrates are typical dominants of intertidal muds in estuaries. Overwhelming dominance in the absence of other fauna, however, is likely to represent a poor or stressed condition. This seems to be the case in the Outfall 008 Drainage where one species of oligochaete comprises a majority at all stations, 95%, 57%, and 80% at locations 008-01, 008-02, and 008-03 respectively. The benthic community in the Outfall 008 Drainage tends to be dominated by one species and exhibits sign of stress; this condition is most evident at stations closer to the Site outfall.

Comparison Of Surface Water and Sediment EPCs To Aquatic Benchmarks

Tables 12-51 and -52 contain a comparison of low tide and high tide surface water concentrations in the Outfall 008 Drainage to surface water screening toxicity values. Because these screening benchmarks were used to identify COPCs, RME concentrations by definition exceed them. CT concentrations, which are a more realistic approximation of exposure point concentrations, also exceed screening benchmarks for the majority of COPCs.

CT concentrations from the Low Tide dataset are generally within the same order of magnitude as available background levels. Dissolved concentrations of many of the inorganics are comparable to total (unfiltered) concentrations, suggesting that these metals are not sorbed to suspended sediments. No screening values is available for vinyl chloride, and therefore risks associated with this analyte could not be evaluated.

CT concentrations from the High Tide dataset are comparable to background levels for analytes that were detected in background samples; however, a number of analytes were not detected in background samples. Low Tide and High Tide concentrations appear to be comparable.

Table 12-53 contains a comparison of RME and CT sediment concentrations in the Outfall 008 Drainage to sediment screening toxicity values. Background HQs are nearly all at or below one, whereas site HQs for analytes are nearly all above one. AVS:SEM data, summarized in Table 12-40, indicate that SEM:AVS ratios at all three stations sampled are less than one, indicating that divalent inorganic analytes present in sediment are bound with sulfides and not likely to be bioavailable or toxic. As discussed above, although site concentrations of PAHs and inorganics exceed background concentrations used in this BERA, they are consistent with background concentrations from the upstream Raymark site.

12.3.1.3 Marine Basin

Comparison Of Tissue EPCs To Tissue CBRs

Tables 12-54 and -55 present RME, CT and background exposure point concentrations in mussels and aquatic worms collected from or exposed to marine basin sediments. Oysters were not collected from the Marine Basin. These tables also include CBRs compiled as described in Subsection 12.2.2.1.

RME mussel concentrations of aluminum, arsenic and selenium in the Marine Basin exceed CBRs, with HQs ranging from 1 to 53. CT concentrations of aluminum, arsenic, and selenium also exceed CBRs. However, these concentrations are consistent with background concentrations, indicating that these analytes do not appear to be site related.

CT and RME aquatic worm concentrations of arsenic, cadmium, chromium, nickel, and zinc exceed their respective CBRs, with HQs ranging from 1.0 to 3.5. Concentrations are consistent with those in background samples, indicating that these analytes do not appear to be site-related.

Evaluation Of Sediment Toxicity Test Results

Toxicity test results in the Marine Basin and reference area were similar. Survival and growth of Neanthes arenacedentata were not significantly different from controls with the lowest percentage survival, 88% recorded at one sampling station in the reference area and Marine Basin. All organisms (100%) survived at two of the Marine Basin stations.

Toxicity test results for amphipods in 1994 were all significantly different from the control. Data from 1999 showed that Leptocheirus plumulosa survival was different from control at one station in the Marine Basin and two stations in the reference area.

Table 12-56 presents a summary of results for the amphipod Leptocheirus from the 1999 tests. These results indicated significant toxicity at one of the four stations within the Marine Basin (SDMB002A). A comparison of detected analytes and ERMs was included in this table to determine if there were any noticeable trends or relationships between the observed toxicity and contaminants. Examination of the concentrations in this table shows that none of the analyte concentrations in this sample appear to be elevated relative to the other samples, indicating that the toxicity observed at this station may be due to a non-chemical stressor. A review of Table 12-56 shows no obvious relationship between TOC, grain size (percent fines) and observed toxicity.

Although toxicity tests from these areas showed significantly reduced survival, both species used in the toxicity tests are recorded in benthic samples from this area. Leptocheirus plumulosa was recorded in benthic samples from all Site areas in 1994 and 1999. Ampelisca was collected in samples from the Marine Basin in 1999 and the reference area in 1994 and 1999. With this in mind, these data were evaluated considering each species' sensitivity and tolerance. The presence of test amphipods in benthic invertebrate collections in the Marine Basin, and the high survival of test organisms in most samples indicate that Site constituents probably do not affect survival of test organisms (i.e. results are similar to or better than the reference area).

Evaluation Of Aquatic Macroinvertebrate Community Structure and Function

Table 12-36 contains a summary of benthic macroinvertebrate data from surveys performed at reference locations and Site areas in 1994 and 1999. The results of the benthic macroinvertebrate community analyses in the Marine Basin serve as another line of evidence in evaluating impacts to the benthic community in this area.

Taxa Richness. With several exceptions, the taxa richness measured at the reference stations and stations in the vicinity of the Site are within the range of what would be expected for intertidal mudflat communities in an urban, estuarine system. Sample MB01had a relatively low richness (9 taxa). Based on the data, it is difficult to determine whether the differences in individual station richness values are attributable to physical habitat characteristics or some other factor(s). However, the richness observed at the remaining stations in the Marine Basin is comparable to that observed at the reference stations.

Organismal Density. Benthic invertebrate densities in the Marine Basin are lower than the Tidal Flats; however, the only area exposed at low tide is MB03, where high densities were observed. In general, densities were similar to or higher than the reference area with the exception of MB01 where both richness and density were low. Reasons for the variability are uncertain, but may be related to the station's proximity to the drainageway input, and the fact that it is permanently inundated and more accessible to aquatic predators.

Owing to the relative physical uniformity in the sediment habitat sampled for this investigation, two taxa (oligochaetes and polychaetes) tended to dominate the benthic assemblages at all stations. These invertebrates are typical dominants of intertidal muds in estuaries. Overwhelming dominance in the absence of other fauna, however, is likely to represent a poor or stressed condition. Two stations in the Marine Basin had percentage dominance by one oligochaete species of 51% at MB04 and 62% at MB03. Variable results in the Marine Basin and similarities with reference area collections indicate that benthic communities in both these areas are probably comparable. Overall richness and abundance of benthic organisms in the Marine Basin are comparable to those at reference locations, and the function of the benthic community as a whole does not appear to be impacted.

Comparison Of Surface Water and Sediment EPCs To Aquatic Benchmarks.
Tables 12-57 and 12-58 contain a comparison of low tide and high tide surface water concentrations in the Marine Basin to surface water screening toxicity values. Because these screening benchmarks were used to identify COPCs, RME concentrations by definition exceed them. CT concentrations, which are a more realistic approximation of exposure point concentrations, also exceed benchmarks for the majority of COPCs.

CT concentrations from the Low Tide dataset are generally consistent with (i.e., within the same order of magnitude as) background levels. There were only 3 inorganic analytes whose dissolved RME concentration exceed the benchmark (aluminum, copper, and manganese). Concentrations of these analytes are comparable to (i.e., within the same order of magnitude as) background levels and do not appear to be site related. No screening value is available for methylene chloride, and therefore risks associated with this analyte could not be evaluated. However, this analyte is a common laboratory contaminant and is not believed to be site related.

CT concentrations from the High Tide dataset are comparable to background levels for analytes that were detected in background samples; however, a number of analytes were not detected in background samples. Low Tide and High Tide concentrations appear to be comparable.

Table 12-59 contains a comparison of RME and CT sediment concentrations in the Marine Basin to sediment screening toxicity values. Background HQs are nearly all at or below one, whereas site HQs for analytes are nearly all above one, and in several cases exceed 10 (acenaphthene, copper, and Aroclor 1260). As discussed above, although site concentrations of PAHs and inorganics exceed background concentrations used in this BERA, they are generally consistent with (i.e., within the same order of magnitude as) background concentrations from the upstream Raymark site.

AVS:SEM data, summarized in Table 12-40, indicate that SEM:AVS ratios at all four stations sampled are less than one, indicating that divalent inorganic analytes present in sediment are bound with sulfides and not likely to be bioavailable or toxic.

12.3.1.4Weight of Evidence Evaluation for Benthic Invertebrates. Table 12-60 contains assessment and measurement endpoints, and a summarization of risk results for each measurement endpoint. A qualitative weight of evidence evaluation is then used to draw conclusions regarding potential risks for each assessment endpoint.

The weight of evidence suggests potential impacts to benthic invertebrates in the Outfall 008 Drainage near the outfall, potentially linked to chromium, Aroclor 1260, and possibly barium and copper. The weight of evidence also suggests potential impacts to benthic invertebrates in some nearshore stations in the Tidal Flats. However, the assessment endpoint is protection of the benthic community as a whole. The results of the sediment toxicity tests and benchmark comparisons did not indicated any particular analytes responsible for the observed effects, and the observed effects may be due to habitat differences or other non-chemical stressors. Direct measurements of the benthic community at 12 stations in the Tidal Flats in 1994 and 1999 indicate no correlation between amphipod toxicity observed in the toxicity test and benthic community metrics or chemical concentrations. Tissue concentrations observed in invertebrates at these locations are comparable to those measured in reference locations. The amphipod Leptocheirus was present in nearly all benthic samples in the Tidal Flats. It is therefore concluded that the overall weight of evidence indicates no significant risk to the benthic community in both the Tidal Flats and the Marine Basin.

12.3.2 Risks To Forage Fish

The assessment endpoint is protection of forage fish in the Tidal Flats and Marine Basin against adverse effects on survival, growth, and reproduction from exposure to COPCs: Are levels of site contaminants in surface water and sediment sufficient to cause adverse effects to forage fish populations?

This endpoint was assessed using two measurement endpoints:

- comparison of tissue EPCs to tissue CBRs; and
- comparison of surface water and sediment EPCs to aquatic benchmarks.

These endpoints were evaluated separately for the Tidal Flats and Marine Basin. They were not evaluated for the Outfall 008 Drainage because this area does not provide significant habitat for forage fish.

12.3.2.1Tidal Flats

Comparison Of Tissue EPCs To Tissue CBRs
Table 12-61 presents RME, CT and background exposure point concentrations in forage fish collected from the Tidal Flats. This table also includes CBRs compiled as described in Subsection 12.2.2.1. RME and CT concentrations of aluminum and zinc exceed their respective CBRs. However, concentrations of aluminum are below background levels, and those for zinc are comparable (RME and CT concentrations of 40 and 36 mg/kg, compared to a background concentration of 31 mg/kg).

Comparison of Surface Water and Sediment EPCs to Aquatic Benchmarks
Table 12-46, presented previously, contains a comparison of RME and CT surface water concentrations in the Tidal Flats to surface water screening toxicity values. Because these screening benchmarks were used to identify COPCs, RME concentrations by definition exceed them. CT concentrations, which are a more realistic approximation of exposure point concentrations, also exceed benchmarks for every analyte except vanadium. CT concentrations are generally an order of magnitude of more higher than background levels. Concentrations of 2 VOCs that were identified as COPCs are only slightly above their respective benchmarks (CT HQs of 1.4 and 1.6).

Table 12-47, presented previously, contains a comparison of RME and CT sediment concentrations in the Tidal Flats to sediment screening toxicity values. Background HQs are nearly all at or below one, whereas site HQs for analytes are nearly all above one. As discussed above, although site concentrations of PAHs and inorganics exceed background concentrations used in this BERA, they are consistent with background concentrations from the upstream Raymark site.

12.3.2.2 Marine Basin

Comparison Of Tissue EPCs To Tissue CBRs
Table 12-62 presents RME, CT and background exposure point concentrations in forage fish collected from the Marine Basin. This table also includes CBRs compiled as described in Subsection 12.2.2.1. RME and CT concentrations of aluminum and zinc exceed their respective CBRs. However, concentrations of aluminum are below background levels, and those for zinc are comparable (RME and CT concentrations of 42 and 35 mg/kg, compared to a background concentration of 31 mg/kg).

Comparison of Surface Water and Sediment EPCs to Aquatic Benchmarks
Tables 12-57 and 12-58, presented previously, contain a comparison of low tide and high tide surface water concentrations in the Marine Basin to surface water screening toxicity values. Because these screening benchmarks were used to identify COPCs, RME concentrations by definition exceed them. CT concentrations, which are a more realistic approximation of exposure point concentrations, also exceed benchmarks for the majority of COPCs.

CT concentrations from the Low Tide dataset are generally within the same order of magnitude as background levels. There were only 3 inorganic analytes whose dissolved RME concentration exceed the benchmark (aluminum, copper, and manganese). Concentrations of these analytes are comparable to background levels and do not appear to be site related. No screening value is available for methylene chloride, and therefore risks associated with this analyte could not be evaluated. However, this analyte is a common laboratory contaminant.

CT concentrations from the High Tide dataset are comparable to background levels for analytes that were detected in background samples; however, a number of analytes were not detected in background samples. Low Tide and High Tide concentrations appear to be comparable.

Table 12-59, presented previously, contains a comparison of RME and CT sediment concentrations in the Marine Basin to sediment screening toxicity values. Background HQs are nearly all at or below one, whereas site HQs for analytes are nearly all above one, and in several cases exceed 10 (acenaphthene, copper, and Aroclor 1260). As discussed above, although site concentrations of PAHs and inorganics exceed background concentrations used in this BERA, they are consistent with background concentrations from the upstream Raymark site. Aroclors were not detected in background samples from the Raymark site.

12.3.2.3 Weight of Evidence Evaluation for Forage Fish. Table 12-60 contains assessment and measurement endpoints, and a summarizationof risk results for each measurement endpoint. The weight of evidence for forage fish suggests no significant risk to forage fish inhabiting the Tidal Flats and Marine Basin. Although there are sediment benchmark exceedances, the tissue concentrations in fish collected from this area are similar to those from the reference areas. The comparison of fish tissue concentrations to CBRs indicates that tissue levels do not appear to pose a risk to forage fish. In addition, the fish collected as part of the ecological characterization, discussed in 12.1.4.2, are typical of those that would be expected to occur in these habitats. Numerous species were observed gravid, flowing or in breeding color, and the presences of both juveniles and mature fish in breeding condition suggests that these locations are functioning as both spawning and nursery areas. The results of this evaluation support a conclusion of no significant risk to forage fish.

12.3.3 Risks to Predatory Fish

The assessment endpoint is protection of predatory fish in the Tidal Flats and Marine Basin against adverse effects on survival, growth, and reproduction from exposure to COPCs: Are levels of site contaminants in surface water and sediment sufficient to cause adverse effects to predatory fish populations?

This endpoint was assessed using two measurement endpoints:

- comparison of tissue EPCs to tissue CBRs; and
- comparison of surface water and sediment EPCs to aquatic benchmarks.

These endpoints were evaluated separately for the Tidal Flats and Marine Basin, and due to a lack of detailed benchmarks, findings are the same as those discussed above for forage fish. No whole body tissue data are available for predatory fish species; striped bass samples collected for evaluation of human health risks were filleted and would not represent an actual whole body tissue concentration. Risks to predatory fish were not evaluated for the Outfall 008 Drainage because this area does not provide significant habitat for these larger organisms.

12.3.3.1 Weight of Evidence Evaluation for Predatory Fish. As summarized in Table 12-60, the weight of evidence for predatory fish suggests no significant risk to predatory fish inhabiting the Tidal Flats and Marine Basin. Although there are sediment benchmark exceedances, the tissue concentrations in fish collected from this area are similar to those from the reference areas. The comparison of fish tissue concentrations to CBRs indicates that tissue levels do not appear to pose a risk to predatory fish. The comparison of fish tissue concentrations relative to background levels, and the relatively low concentrations when compared to CBRs, support a conclusion of no significant risk to predatory fish. Although only forage fish tissue concentrations were evaluated in the BERA, predatory fish (i.e., striped bass) were evaluated in the HHBRA (Section 11). As can be seen in Table 11-7 of the HHBRA, tissue concentrations in predatory fish are lower than those detected in the forage fish analyzed in the BERA. In addition, the fish collected as part of the ecological characterization, discussed in 12.1.4.2, are typical of those that would be expected to occur in these habitats. Numerous species were observed gravid, flowing or in breeding color, and the presences of both juveniles and mature fish in breeding condition suggests that these locations are functioning as both spawning and nursery areas. Results of this weight of evidence evaluation support a conclusion of no significant risk to predatory fish inhabiting the Tidal Flats or Marine Basin.

12.3.4 Risks to Shorebirds, Waterfowl, and Fish-Eating Birds

The assessment endpoint is protection of shorebirds, waterfowl, and fish-eating birds using the Tidal Flats, Marine Basin, and Outfall 008 Drainage, against adverse effects on survival, growth, and reproduction from exposure to COPCs: Are levels of site contaminants in surface water and sediment sufficient to cause adverse effects to avian receptors?

This endpoint was assessed using one measurement endpoint: Comparison of modeled dietary doses of sediment COPCs to literature-derived RTVs. This endpoint was evaluated separately for each of the exposure areas.

Two exposure scenarios were evaluated for semi-aquatic wildlife: RME and CT. However, if HIs calculated based on RME exposures were less than one, the CT exposure scenario was not evaluated, since it would, by definition, be less than one. HIs were calculated using both LOAEL and NOAEL toxicity values, because these two values help bound the range of potential effects, but emphasis is placed on LOAEL-based results.

12.3.4.1 Tidal Flats. Tables 12-63 and 12-64 present summaries of RME and CT hazard indices (HIs; the sum of individual HQs), respectively. Values are presented for HIs calculated based on NOAELs and on LOAELs.

For the RME exposure scenario, NOAEL-based HIs for the three avian receptors were greater than or equal to one. The HI for the black duck was 1.3, although HQs for individual analytes were all below one. NOAEL-based HIs for the other two avian receptors are 11 and 23. LOAEL-based HIs for the three receptors ranged from less than one to 3.5.

CT exposures were evaluated for great blue heron and sandpiper; CT exposures were not evaluated for black duck because the RME exposures resulted in HQs less than one. NOAEL-based HIs for these two receptors were greater than or equal to one. LOAEL-based HIs for the heron and sandpiper were also greater than one (1.2 and 3.9, respectively). For the heron, however, HQs for individual analytes were all less than one, and for the sandpiper, the only analyte with an HQ>1 was chromium, with an HQ. of 1.1, due to sediment ingestion. The incremental risk for the heron and sandpiper under the CT scenario are 1.1 and 2.0. The incremental risk for the heron suggests any potential site-related risks are negligible (i.e., not discernable from reference conditions). The incremental risk for the sandpiper suggests a potential site-related risk, from chromium in sediment and mercury (presumed to be methylmercury) in tissue. The average chromium concentration out of 50 samples was 255 mg/kg. The food chain model assumes a foraging range of 0.25 hectares (0.64 acres). Since the area of the Tidal Flats is greater than that (25 hectares, or 64 acres), the food chain model assumes that the sandpiper feeds exclusively at the Tidal Flats. If a significant portion of a sandpiper or other shorebird population forages exclusively at the Tidal Flats, there is a potential risk to that population; however, the extent to which this occurs is uncertain.

The results of this evaluation indicate that there is no significant risk to waterfowl, or fish-eating birds foraging in the Tidal Flats, but a potential risk to shorebirds such as the sandpiper from chromium in Tidal Flats sediment if they feed exclusively in the Tidal Flats.

12.3.4.2 Outfall 008 Drainage. Tables 12-63 and 12-64, presented previously, include summaries of RME and CT HIs, respectively. Values are presented for HIs calculated based on NOAELs and on LOAELs. NOAEL-based HIs at Outfall 008 under the RME exposure scenario exceed one for all of the receptors. The HI for the black duck is 1.4. HIs for the sandpiper and great blue heron are 370 and 570, respectively. LOAEL-based HIs for the sandpiper and heron are 74 and 120. The HIs for the heron are somewhat misleading, however, because its diet is primarily comprised of fish, and fish tissue data are not available for Outfall 008. The Outfall 008 drainage does not represent significant fish habitat. Therefore, because site-specific data were not entered, the food chain model estimates fish tissue concentrations using literature-based bioaccumulation factors that likely overestimate exposure and risk.

CT exposures were evaluated for black duck, great blue heron and sandpiper. Both NOAEL- and LOAEL-based HIs for the black duck were less than one. However, both NOAEL- and LOAEL-based HIs for heron and sandpiper were greater than one.

Under the CT exposure scenario, the LOAEL-based HI of 25 for the sandpiper is almost entirely attributable to incidental ingestion of chromium in sediment (HQ= 22); HQs for all other analytes were less than one. The average chromium concentration in Outfall 008 Drainage sediment is substantially elevated, at 5,319 mg/kg.

Under the CT exposure scenario, the LOAEL-based HI of 39 for the great blue heron is due to ingestion of fish containing chromium (HQ=35); however, the fish ingestion pathway for Outfall 008 was based on estimated fish tissue concentrations calculated by applying literature-based bioaccumulation factors; this elevated HI is therefore also attributable to the elevated chromium in sediment.

The results of this evaluation indicate that there is no significant risk to waterfowl, but there could be a potential risk to shorebirds and fish-eating birds foraging at the Outfall 008 drainage from exposure to chromium in sediment. However, differences in habitat quality, or foraging preferences due to that quality, were not incorporated into the food chain model. The habitat quality of Outfall 008 is quite limited, relative to that in adjacent areas (Marine Basin and Tidal Flats), and this habitat quality limitation should be considered in determining the overall significance of this finding.

12.3.4.3 Marine Basin. Tables 12-63 and 12-64, presented previously, include summaries of RME and CT HIs, respectively. Values are presented for HIs calculated based on NOAELs and on LOAELs. NOAEL-based HIs at the Marine Basin under the RME exposure scenario exceed one for all of the receptors except the black duck. HIs for the sandpiper and great blue heron are 17 and 10, respectively. LOAEL-based HIs for these 2 receptors at the Marine Basin under the RME exposure scenario are only slightly greater than one (1.3 and 2.3 for heron and sandpiper, respectively), and individual HQs are all below one.

Under the CT exposure scenario, the NOAEL-based HIs for heron and sandpiper are 6.9 and 11, respectively. LOAEL-based HIs for heron and sandpiper are 0.84 and 1.5. For both receptors, HQs for individual analytes are all below one, and the incremental risk is below 1.

The results of this evaluation indicate that there is no significant risk to shorebirds, waterfowl, or fish-eating birds foraging in the Marine Basin.

12.3.4.4 Weight of Evidence Evaluation for Semi-Aquatic Birds. Table 12-60 contains assessment and measurement endpoints, and a summary of risk results. The results of the food chain model, which incorporates site-specific tissue data to estimate potential exposures and risks, suggest no significant risk to semi-aquatic birds foraging in the Marine Basin. Potential risks have been identified, however, for shorebirds foraging in the Tidal Flats and Outfall 008 drainage area due primarily to elevated chromium. At Outfall 008, the significance of this finding is uncertain, however, due to the limited habitat quality relative to habitat quality in adjacent areas (Marine Basin). It is unlikely that a significant proportion of a local population would feed in the Outfall 008 Drainage a significant portion of the time. The food chain model also assumes that the sandpiper feeds exclusively in the Tidal Flats. Individual sandpipers feeding exclusively in the Tidal Flats may be at risk, however, population-level risks would only exist if a significant portion of the population were to feed exclusively at the Tidal Flats.

12.3.5 Risks to Semi-Aquatic Mammals

The assessment endpoint is protection of semi-aquatic mammals using the Tidal Flats, Marine Basin, and Outfall 008 Drainage, against adverse effects on survival, growth, and reproduction from exposure to COPCs: Are levels of site contaminants in surface water and sediment sufficient to cause adverse effects to semi-aquatic mammal receptors?

This endpoint was assessed using one measurement endpoint: Comparison of modeled dietary doses of sediment COPCs to literature-derived RTVs. The representative semi-aquatic mammal selected for evaluation is the raccoon; this endpoint was evaluated separately for each of the exposure areas.

12.3.5.1 Tidal Flats. Under the RME exposure scenario, the NOAEL-based HI for the raccoon is 42; the LOAEL-based HI is 5.5. Under the more realistic CT exposure scenario, the NOAEL-based HI for the raccoon is 35; the LOAEL-based HI is 4.4. The LOAEL-based CT HI is primarily attributable to incidental ingestion of aluminum in sediment (HQ of 2.7); HQs for all other analytes were less than one. The incremental risk for the raccoon is <1, indicating no significant risk to semi-aquatic mammals foraging in the Tidal Flats.

12.3.5.2 Outfall 008 Drainage. Under the RME exposure scenario, the NOAEL-based HI for the raccoon is 9.9. The LOAEL-based HI is 1.4, but HQs for individual analytes are all less than one. Under the more realistic CT exposure scenario, the NOAEL-based HI for the raccoon is 7.0 and the LOAEL-based HI is less than one. The incremental risk for the raccoon is <1, indicating no significant risk to semi-aquatic mammals foraging in the Outfall 008 Drainage.

12.3.5.3 Marine Basin. Under the RME exposure scenario, the NOAEL-based HI for the raccoon is 16; the LOAEL-based HI is 2.0. Under the CT exposure scenario, the NOAEL-based HI for the raccoon is 10; the LOAEL-based HI is 1.3, but HQs for individual analytes were all less than one. The incremental risk for the raccoon is <1, indicating no significant risk to semi-aquatic mammals foraging in the Marine Basin.

12.3.5.4 Weight of Evidence Evaluation for Semi-Aquatic Mammals. As summarized in Table 12-60, the results of the food chain model, which incorporates site-specific tissue data to estimate potential exposures and risks, suggest no significant risk to semi-aquatic mammals such as the raccoon from foraging at any of the three areas (Tidal Flats, Marine Basin, and Outfall 008).

12.3.6 Risks to Rare, Threatened, or Endangered Species

The assessment endpoint is protection of individuals of rare, threatened, or endangered species using the Tidal Flats, Marine Basin, and Outfall 008 Drainage, against adverse effects on survival, growth, and reproduction from exposure to COPCs: Are levels of site contaminants in surface water and sediment sufficient to cause adverse effects in individuals of any rare, threatened, or endangered species (piping plover and least tern)?

This endpoint was assessed using one measurement endpoint: Extrapolation of food-chain results for other semi-aquatic wildlife of similar habits and feeding preferences. Results for sandpiper are considered for the piping plover, as they have similar dietary requirements. The results for the great blue heron are considered for the least tern, because this bird has dietary requirements more similar to those of the heron. Emphasis is given to the NOAEL-based HQs and HIs for these species of concern. This endpoint was evaluated separately for each of the exposure areas.

12.3.6.1 Tidal Flats

Piping Plover
Under the RME exposure scenario, the NOAEL-based HI for the sandpiper (surrogate for the piping plover) is 23, due primarily to methylmercury concentrations in aquatic invertebrates. Invertebrates were analyzed for mercury, but not methylmercury; however, for purposes of assessment it was assumed that all of the mercury detected in the invertebrates was methylmercury.

Under the CT exposure scenario, the NOAEL-based HI for the sandpiper is 20, primarily due to mercury (presumed to be methylmercury) in aquatic invertebrates (HQ=5.9) and chromium in sediment (HQ=5.4). If only a fraction of that mercury was methylmercury, the CT NOAEL-based HQ would be substantially lower.

Mercury was only detected in 5 of 15 oysters, 0 of 5 ribbed mussels, and 1 of 50 invertebrates. Since both plovers and sandpipers are more likely to ingest aquatic worms and other invertebrates lacking a protective shell, it is unlikely that these mercury levels pose a significant risk.

The NOAEL-based incremental risk for the sandpiper at the Tidal Flats is 11, indicating a potential site-related risk to any individual plovers foraging in the Tidal Flats due primarily to chromium in sediment. However, the foraging range for the sandpiper (and presumably plover) is 0.25 hectares (0.64 acres). Since the area of the Tidal Flats is greater than that (25 hectares, or 64 acres), the food chain model assumes that the sandpiper feeds exclusively at the Tidal Flats. As stated previously, the Tidal Flats were monitored regularly between May and August, 1997 and no piping plovers were observed feeding there (LeBlanc, 1997). Based on those observations, it is uncertain if piping plovers utilize the Tidal Flats, and if so, with what frequency. Risks to this species may well have been overestimated.

Least Tern

Under the RME exposure scenario, the NOAEL-based HI for the great blue heron is 11, due primarily to mercury (presumed to be methylmercury) concentrations in fish. Under the CT exposure scenario, the NOAEL-based HI for the heron is 8.9, again primarily due to mercury (presumed to be methylmercury) in fish.

The NOAEL-based incremental risk for the heron at the Tidal Flats is 8, due to estimated mercury concentrations in fish. Mercury was not detected in fish tissue; estimated fish tissue concentrations were based on one-half the quantitation limit of mercury. Fish were not analyzed for methylmercury (but it was conservatively assumed that all of the mercury potentially present in the fish was methylmercury). These assumptions may have overestimated risks to this receptor.

The results of this analysis suggest that individual terns feeding at the Tidal Flats are unlikely to be affected by mercury in prey tissue, but there remains some uncertainty due to the detection limits achieved in fish tissue and the lack of methylmercury data in fish tissue.

12.3.6.2 Outfall 008 Drainage

Piping Plover

Under the RME exposure scenario, the NOAEL-based HI for the sandpiper is 370. This is primarily due to incidental ingestion of chromium in sediment (HQ=340), but also contributed to by mercury concentrations (presumed to be methylmercury) in invertebrates(HQ=6.2), and antimony in sediment (HQ=12).

Under the CT exposure scenario, the NOAEL-based HI for the sandpiper is 130, primarily due to the incidental ingestion of chromium in sediment (HQ=110), but also contributed to by mercury concentrations (presumed to be methylmercury) in invertebrates (HQ=5.2) and antimony in sediment (HQ=4.6).

The only aquatic invertebrates analyzed in Outfall 008 drainage were aquatic worms. Mercury was not detected in any of the 15 samples analyzed from this location. Estimated tissue concentrations were based on one-half the quantitation limit of mercury. These organisms were not analyzed for methylmercury (but it was conservatively assumed that all of the mercury potentially present in the fish was methylmercury). These assumptions may have overestimated mercury-associated risks to this receptor. However, chromium is the primary risk driver at Outfall 008; based on the NOAEL-based risk estimates under both RME and CT scenarios, there is a potential risk to any plovers that spend a significant portion of their time foraging in the Outfall 008 drainage. Given the availability of other more desirable habitat in the vicinity, these exposures and risks may have been overestimated. No terns or plovers have been observed utilizing this area of the site.

Least Tern
Under the RME exposure scenario, the NOAEL-based HI for the great blue heron is 570, due almost entirely to ingestion of chromium in fish (HQ=440) and incidental ingestion of sediment (HQ=83). There are no fish tissue data for Outfall 008, and therefore the fish tissue concentrations in the food chain model are estimated using conservative, literature-based BSAFs. The HQs for several other inorganics (zinc, copper, antimony, mercury, cadmium, and aluminum) also exceed one, again primarily due to fish ingestion based on estimated fish tissue concentrations. Application of these BSAFs to the elevated chromium concentrations present in the sediment samples from the Outfall 008 drainage resulted in this high exposure estimate.

Under the CT exposure scenario the NOAEL-based HI for the heron is 190, again due to ingestion of chromium in fish (HQ=150) and incidental ingestion of chromium in sediment (HQ=27). The HQs for several other inorganics slightly exceed one, again primarily due to fish ingestion based on estimated fish tissue concentrations. The degree to which Outfall 008 Drainage ditch provides habitat for forage fish, and the frequency with which wading birds and shorebirds are likely to forage there, are uncertain. No terns or plovers have been observed utilizing this area of the site.

12.3.6.3 Marine Basin

Piping Plover
Under the RME exposure scenario, the NOAEL-based HI for the sandpiper is 17, due primarily to mercury concentrations (assumed to be methylmercury) in aquatic invertebrates. Invertebrates were analyzed for mercury, but not methylmercury; however, for purposes of assessment it was assumed that all of the mercury detected in the invertebrates was methylmercury.

Under the CT exposure scenario, the NOAEL-based HI for the sandpiper is 11, again primarily due to methylmercury concentrations (assumed to be methylmercury) in aquatic invertebrates (HQ=5.3).

Mercury was not detected in any of the ribbed mussel (N=5) or aquatic worm (N=20) samples analyzed (oysters were not analyzed at this location). Estimated invertebrate tissue concentrations were based on one-half the quantitation limit of mercury. These organisms were not analyzed for methylmercury (but it was conservatively assumed that all of the mercury potentially present in the invertebrates was methylmercury). These assumptions may have overestimated risks to this receptor.

The NOAEL-based incremental risk to sandpipers in the Marine Basin under the CT scenario is 1.8, although again, risks are primarily attributable to estimated mercury concentrations in invertebrates. Based on this information, there does not appear to be a significant risk to plovers at the Marine Basin.

Least Tern

Under the RME exposure scenario, the NOAEL-based HI for the great blue heron is 10, due primarily to assumed methylmercury concentrations in fish. Under the CT exposure scenario, the NOAEL-based HI for the heron is 6.9, again due primarily to assumed methylmercury in fish (HQ=4.6). HQs for all other analytes were less than one.

Mercury was not detected in fish tissue; estimated fish tissue concentrations were based on one-half the quantitation limit of mercury. Fish were not analyzed for methylmercury (but it was conservatively assumed that all of the mercury present in the fish was potentially methylmercury). These assumptions may have overestimated risks to this receptor.

The results of this analysis suggest that individual terns feeding at the Marine Basin are unlikely to be affected by mercury in prey tissue, but there remains some uncertainty due to the detection limits achieved in fish tissue and the lack of methylmercury data in fish tissue.

12.3.6.4 Weight of Evidence Evaluation for Rare, Threatened, and Endangered Species. Risks to the plover and tern are evaluated by considering NOAEL-based risk estimates under both RME and CT exposure scenarios for the sandpiper and heron, respectively. The overall weight of evidence indicates a potential risk to the plover from chromium in Tidal Flats sediment, and a potential risk to both receptors in the Tidal Flats and Marine Basin due to estimated mercury concentrations (assumed to be methylmercury) in prey items. Mercury was detected infrequently in prey items, and at relatively low concentrations. One-half the quantitation limit was used in evaluating risks for non-detected analytes such as mercury; this combined with the assumption that all of the mercury is methylmercury, may have overestimated risks to these receptors. If either the plover or tern makes use of the Outfall 008 Drainage, there is also a potential risk associated with exposure to chromium in sediment in this area; however, neither species has been observed in this area, and frequent foraging in this area is considered to be somewhat unlikely due to the availability of higher quality habitat nearby.

For the plover, there is uncertainty as to if and how often they utilize the Tidal Flats. The food chain model assumed that the sandpiper (surrogate for the plover) feeds exclusively at the Tidal Flats. As stated previously, the Tidal Flats were monitored regularly between May and August, 1997 and no piping plovers were observed feeding there (LeBlanc, 1997). Based on those observations, it is uncertain if piping plovers utilize the Tidal Flats, and if so, with what frequency. Risks to this species may well have been overestimated.

12.3.7 Uncertainty Analysis

Data gaps and assumptions were identified during each phase of the SAEP BERA. Each assumption contributes to a measure of uncertainty that affects the robustness of the conclusions of the risk characterization. Uncertainties related to exposure and modeling, effects assessment, and risk characterization are discussed below.

12.3.7.1 Exposure and Modeling Uncertainties. Uncertainties associated with the exposure assessment step of the BERA affect the interpretation of the significance of the HQs. The main uncertainties are associated with the exposure point concentrations, exposure parameters, estimation of bioaccumulation factors for various environmental media, and assumptions of bioavailability of COPCs.

Data Summary/Exposure Point Concentrations.

There were a number of administrative issues with the existing database that affect the overall data quality. For example, some of the older datasets are incomplete, missing non-detect data or data for specific analytes. A degree of uncertainty was associated with the percent lipid data, particularly in fish tissue, as several fish tissue lipid results were reported as > 100%. Therefore the decision was made to use a standard assumption of 5% lipid for small forage fish.

Trivalent arsenic data were suspect, as in many cases concentrations exceed those for total arsenic. This may be due to differences in analytical technique, but may also be due to improper documentation of units of measurement (i.e., database reports mg/kg, but may actually be micrograms per kilogram [µg/kg]). Therefore, trivalent arsenic data were not included in the BERA data sets.

Methylmercury was not analyzed for in biota; for the BERA, it was therefore assumed that all of the mercury detected in biota was methylmercury. This may have greatly over-estimated risks to semi-aquatic birds and mammals feeding on biota at the Site.

Data sets for several areas are limited to just a few samples – this introduces a great degree of uncertainty in the exposure and risk estimates for these areas.

Exposure Parameters. The relationship between receptor size and dietary intake is a critical factor in estimating exposure. In addition, dietary composition affects exposure because different food sources contain varying levels of COPCs. Although literature data exist for dose calculation inputs such as body weight, ingestion rate, and dietary composition for the measurement endpoint receptors evaluated at the SAEP site, there is a natural level of variability in these parameters within a population of organisms. Uncertainty is inherent in the use of these values because they were generated from literature sources rather than being empirically measured at the site or surrounding area. Therefore, use of literature-derived exposure parameters increases uncertainty that may have over- or underestimated actual exposures encountered by receptors at the Site.

The food chain model evaluated exposures within each of the exposure areas, but did not specifically evaluate combined exposures across areas (i.e., risks to organisms foraging among all of the areas). However, evaluating the exposure areas separately allows the BERA to better identify potential risks associated with a particular area or source.

Bioaccumulation Factors. Site-specific prey tissue data were available for fish and several different invertebrates (oysters, mussels, and aquatic worms). Although the risks identified for various receptors were calculated using site-specific BSAFs, these uptake factors were derived using relatively few samples, which may not be representative of the conditions throughout the Site. Further, for each analyte in each area, the highest average concentration from among the three invertebrate receptor groups was used. Thus, if oysters contained the highest concentrations, it may be overly conservative to have used oyster-BSAFs to estimate risks to sandpipers, which ingest primarily aquatic insects and worms. In general, receptors at the Site are likely to ingest a far greater variety of organisms than those assumed for purposes of the food chain evaluation.

Literature-based BAFs were routinely employed to estimate uptake into other food chain components (e.g., plants) and for COPCs that were not analyzed for in some biota types (e.g., PAHs); however, these analytes were not risk-drivers in any of the exposure areas. One exception was for fish in the Outfall 008 Drainage; fish samples were not collected from this ditch, and therefore the food chain model estimated fish tissue concentrations at this location by applying literature-derived BSAFs, which may significantly overestimate fish tissue concentrations at this location. This, combined with the fact that the ditch provides marginal habitat at best for small forage fish, primarily in the lower reaches, suggests that risks at the Outfall 008 drainage were overestimated for fish-eating wildlife.

Bioavailability of COPCs. Typically, data useful for determining COPC bioavailability in sediments include pH, SEM/AVS, exchangeable cations, sediment organic carbon content, and grain-size. Although some information was collected to assess bioavailability of COPCs, doses were calculated based on sediment COPC concentrations, which assumes that COPCs are completely bioavailable. But it is likely that various environmental factors, such as TOC, affect the bioavailability of these COPCs, and therefore this assumption may have overestimated risks.

Exposure Parameters. Exposure parameters are not available for the two listed bird species (piping plover and least tern), and therefore risks were evaluated based on results for the sandpiper and great blue heron, respectively. Exposure parameters, including foraging range and food ingestion rate, may vary considerably, and therefore risks to the listed species may have been over- or under-estimated.

12.3.7.2 Toxicological Uncertainties. Potential uncertainties are related to the appropriateness of literature-derived toxicity data. RTVs used in the BERA are based on an extensive search of both primary peer-reviewed literature and secondary literature, such as government reports and technical conference proceedings. The number and types of information sources reviewed is believed to be adequate to capture the majority of relevant source of ecotoxicological literature. Very limited data exist for specific wildlife receptor types (e.g., omnivorous birds, herbivorous mammals) in the toxicological literature. Therefore, RTVs were selected from toxicity data and applied to broad classes of animals, and used to quantify risk for specific measurement endpoint receptors.

Chronic data were selected preferentially in developing RTVs. However, available toxicological data are not always associated with chronic exposure duration. Therefore, there are uncertainties in extrapolating the results of shorter term exposures to the chronic exposures assumed for receptors at the SAEP Site. Chronic NOAELs were the preferred toxicity endpoint for selection of RTVs; however, ecological toxicity data were limited for some COPCs and some receptor groups. Therefore, other endpoints (for example, subchronic NOAELs, LC50 values) were selected for use as RTVs. When an endpoint other than a chronic NOAEL was selected as a RTV, an uncertainty factor was applied to the reported value to provide an additional level of conservatism in the risk estimation process. Application of conservative application factors may result in risks being overestimated.

Some COPCs had very little or no toxicological data available. When appropriate, toxicity data for similar chemicals were used as surrogates for these COPCs. For example, oral ingestion toxicity data for benzo(a)pyrene was used to derive RTVs for many of the PAH compounds lacking relevant information. Use of such surrogate toxicity data may underestimate or overestimate risks from the COPCs. In some cases, no appropriate toxicity or surrogate data were identified in the toxicological literature. In addition, there was no information available to establish CBRs for some of the analytes detected in biological tissue. As a result, risk from some COPCs could not be quantified. This may underestimate ecological risks at the site.

In general, uncertainty is also associated with the extrapolation of literature-derived toxicity endpoints (especially previously published laboratory-based studies) to equivalent endpoints for measurement endpoint receptors at the SAEP Site because of discrepancies in exposure conditions. The majority of the literature-based toxicity data evaluated and used in the BERA were derived from laboratory studies. Laboratory settings do not necessarily mimic field conditions and exposures, and typically are designed to control various factors in order to isolate one parameter in particular. Although controlled experiments result in a more valid interpretation of the isolated parameters, uncertainty is associated with the assumption that field exposures are equivalent to laboratory exposure conditions.

The sediment toxicity test results for the Site are extremely variable. Mortality data were not available in the Laboratory Control for the 1994 Ampelisca abdita tests, and therefore the statistical significance of the high mortality in the site samples (Table 12-35) is uncertain. Additionally, there was significant mortality in the Leptocheirus tests at the upstream reference area, but not in the Neanthes tests, conducted at the same upstream area, which makes interpretation of the toxicity observed at site locations more difficult to interpret.

12.3.7.3 Uncertainty in Risk Characterization. In addition to the uncertainties introduced as a result of the data availability and representativeness issues discussed above, the main sources of uncertainty associated with risk characterization involve the interpretation of HQ results. Emphasis was placed on CT HQs, which are lower than RME HQs, and therefore risks to organisms exposed to RME concentrations have been understated. However, it is unlikely that a significant proportion of any ecological population would be exposed to the RME concentrations identified for each exposure area.

Due to the conservative assumptions that were made in developing both exposure and effects assessments, HQs that are greater than one do not indicate that potential adverse effects necessarily exist. Attempts were made to bound these uncertainties in the BERA by considering risk associated with both chronic NOAELs and LOAELs. For listed species, emphasis was placed on NOAEL-based HQs. Since NOAELs represent a no-observed adverse effect level, exceedance of these benchmarks is not necessarily indicative of potential risks. For the majority of COPCs, no dose-response relationship has been established, and therefore as the HQ increases, it is an oversimplification to assume that risk increases in a linear fashion.

12.4 SUMMARY AND CONCLUSIONS

The BERA was conducted to characterize ecological risks at the site in accordance with USEPA performance standards for risk characterization. The BERA included a problem formulation, analysis (exposure assessment and effects assessment), and risk characterization. A summary of the ecological risk characterization is presented below.

A summary of the ecological risk characterization for receptors in each area of the site is presented in Table 12-60. The table provides the overall interpretation of risks to each receptor in each area based on a weight of evidence evaluation.

The following paragraphs present a summary of the BERA findings:

<u>Protection of benthic invertebrate community structure in the Tidal Flats, Outfall 008 Drainage, and Marine Basin</u>

The results of the BERA suggest possible impacts to benthic invertebrateds at some nearshore stations of the Tidal Flats; however, overall richness and abundance are not affected. The results of the sediment toxicity tests and benchmark comparisons did not indicate any particular analyte responsible for the observed effects, and observed effects may be due to habitat differences or other non-chemical stressors. The comparison of invertebrate tissue concentrations to Critical Body Residues (CBRs) suggest that levels of some inorganics in invertebrate tissue slightly exceed CBRs. However, the primary inorganic of potential concern, mercury, was only detected in 1/50 invertebrates analytes. Mercury was not detected in background tissue samples. All other inorganics were detected at tissue concentrations comparable to background levels. The overall weight of evidence indicates no significant risk to the benthic community as a whole in the Tidal Flats.

The results of the BERA indicate that there is a potential risk to macroinvertebrates in the Outfall 008 Drainage, due to inorganics (barium, chromium, and copper) and Aroclor-1260 in sediment. The Outfall 008 Drainage represents a physically disturbed habitat of lower value than the surrounding Marine Basin and Tidal Flats.

The results of the BERA, particularly the sediment toxicity test results, benthic community results, and the comparison of tissue concentrations to CBRs, indicate that there is no significant risk to macroinvertebrates in the Marine Basin.

<u>Protection of forage fish and predatory fish in the Tidal Flats and Marine Basin.</u>

The results of the BERA indicate that there is no significant risk to forage fish inhabiting the Tidal Flats or Marine Basin. Tissue concentrations are comparable to tissue concentrations from reference locations. Although there are some elevated concentrations in sediments, tissue concentrations are similar to those from reference areas. In addition, species observed at these locations are typical of those that would be expected; the presence of juveniles and fish in breeding condition are indicative of a healthy fish community.

Protection of semi-aquatic birds and mammals

The results of the BERA indicate that there is no significant risk to semiaquatic birds (represented by mallard, great blue heron, and sandpiper) that may forage at the Marine Basin. Potential risks have been identified, however, for shorebirds foraging in the Tidal Flats and Outfall 008 drainage area due primarily to elevated chromium. At Outfall 008, the significance of this finding is uncertain, however, due to the limited habitat quality relative to habitat quality in adjacent areas (Marine Basin). It is unlikely that a significant proportion of a local population would feed in the Outfall 008 Drainage a significant portion of the time. The food chain model also assumes that the sandpiper feeds exclusively in the Tidal Flats. Individual sandpipers feeding exclusively in the Tidal Flats may be at risk, however, population-level risks would only exist if a significant portion of the population were to feed exclusively at the Tidal Flats.

Protection of rare, threatened, or endangered species

Risks to protected bird species, the piping plover and least tern, were evaluated in the BERA. These two species are known to nest in the vicinity of the site, and they may forage at the site. Risks to these two birds were evaluated indirectly by evaluating results for the great blue heron and sandpiper. The feeding habits of the plover are similar to those of the sandpiper, as they both feed largely on aquatic invertebrates. The feeding habits of the tern are similar to those of the heron, as they both feed largely on small fish. In evaluating risks to the protected species, emphasis was placed on NOAEL-based endpoints, rather than LOAEL-based endpoints. The results of the BERA indicate that there is no significant risk to rare, threatened or endangered bird species that may forage at the Marine Basin, but a potential risk to plovers at the Tidal Flats, and a potential risk to both plovers and terns at Outfall 008, from chromium in sediment. Given the limited habitat quality at Outfall 008, plovers and terns are unlikely to spend a significant amount of time foraging there. There is uncertainty as to whether and how frequently plovers utilize the Tidal Flats. The food chain model assumed that the sandpiper (surrogate for the plover) feeds exclusively at the Tidal Flats. As stated previously, the Tidal Flats were monitored regularly between May and August, 1997 and no piping plovers were observed feeding there (LeBlanc, 1997). Based on those observations, it is uncertain if piping plovers utilize the Tidal Flats, and if so, with what frequency. Risks to this species may well have been overestimated.

ACRONYMS

1,1,1-TCA	1,1,1-trichloroethane
1,1,2-TCA	1,1,2-trichlorethane
1,1,2,2-TCA	1,1,2,2-tetrachloroethane
1,1-DCA	1,1-dichloroethane
1,1-DCE	1,1-dichloroethene
1,2-DCA	1,2-dichloroethane
1,2-DCE	1,2-dichloroethene
ABB-ES	ABB Environmental Services, Inc.
ABS	absorption factor
ACSIM	U.S. Army Assistant Chief of Staff for Installation Management
ADD	average daily dose
ADD$_c$	average daily dose, chronic
ADD$_s$	average daily dose, subchronicAOC area of concern
ARARs	Applicable or Relevant and Appropriate Requirements
AST	aboveground storage tank
ASTM	American Society for Testing and Materials
AT	averaging time
ATL	Air Toxics Limited
AVCO	Avco Corporation
AWQC	Ambient Water Quality Criteria
BAF	bioaccumulation factor
bgs	below ground surface
BERA	Baseline Ecological Risk Assessment
BRAC	Base Closure and Realignment Act
BSAF	biota-sediment accumulation factor
BTEX	benzene, toluene, ethylbenzene, and xylene
BW	body weight
CBR	critical body residue
CDF	Cyanide Destruction Facility
CDM FPC	Camp, Dresser and McKee Federal Programs Corporation
CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act
CERCLIS	Comprehensive Environmental Response, Compensation, and Liability Identification System
CERFA	Community Environmental Response Facilitation Act
cf	cubic feet
CF	conversion factor
CFR	Code of Federal Regulation
cfs	cubic feet per second
cis-1,2-DCE	cis-1,2-dichloroethene
cm	centimeter
cm/hr	centimeters per hour
CN	cyanide
COD	chemical oxygen demand

COPC	chemical of potential concern
CR(III)	trivalent chromium
Cr(VI)	hexavalent chromium
CRDL	contract required detection limit
CSF	cancer slope factor
CSM	conceptual site model
CT	central tendency
CTDEP	Connecticut Department of Environmental Protection
CTDPH	Connecticut Department of Public Health
cVOCs	chlorinated volatile organic compounds
CWTP	Chemical Waste Treatment Plant
DEC	Direct Exposure Criteria
DGPS	digital global positioning system
DNAPL	dense non-aqueous phase liquid
DO	dissolved oxygen
DQO	data quality objective
EBS	Environmental Baseline Survey
EC	electrical conductivity
ECAF	environmental condition assessment form
ECOC	ecological contaminant of concern
ECM	electrochemical machining
ED	exposure duration
EDI	estimated daily intake
EE/CA	Engineering Evaluation/Cost Analysis
EF	exposure frequency
ELCD	electrolytic conductivity detector
ELCR	excess lifetime cancer risk
ELUR	environmental land use restriction
EMI	electromagnetic induction
EPC	exposure point concentration
ERA	ecological risk assessment
ERDC	U.S. Army Corps of Engineers Engineer Research and Development Center
ER-L	effects range-low
ER-M	effects range-median
ET	exposure time
ET	ecotoxicity thresholds
^{o}F	degrees Fahrenheit
$FeSO_4$	ferrous sulfate
FS	Feasibility Study
ft bgs	feet below ground surface
ft/day	feet per day
ft^2/day	square feet per day
ft^3/day	cubic feet per day
ft/ft	feet per foot

FWENC	Foster Wheeler Environmental Corporation
g	grams
g/cc	grams per cubic centimeter
GB PMC	GB Classified Aquifer Pollutant Mobility Criteria
GC	gas chromatograph
GIS	geographical information system
GMS	Groundwater Modeling System
gpd	gallons per day
gpm	gallons per minute
GPR	ground penetrating radar
GWPC GA	CTDEP Groundwater Protection Criteria for a GA-classified aquifer
Harding ESE	Harding ESE, A MACTEC Company
HDPE	high density polyethylene
HEAST	Health Effects Assessment Summary Tables
HHBRA	human health baseline risk assessment
HI	hazard index
HLA	Harding Lawson Associates
hr	hour
HQ	hazard quotient
HSA	hollow-stem auger
I/C	industrial/commercial
IAF	inhalation adjustment factor
IATC	Indoor Air Target Concentrations
ID	inside diameter
INEL	Idaho National Engineering Laboratory
IRIS	Intergrated Risk Information System
K	hydraulic conductivity
Kd	partitioning coefficient
Koc	organic carbon partitioning coefficient
Ko/w	octanol/water partitioning coefficient
kg	kilograms
$KMnO_4$	potassium permanganate
LADD	lifetime average daily dose
L/day	liters per day
L/kg	liters per kilogram
LEL	lowest effect level
LNAPL	light non-aqueous phase liquid
LOAEL	Lowest Observable Adverse Effect Level
LOEC	Lowest Observered Effects Concentration
LRA	Local Redevelopment Authority
µg/dl	micrograms per deciliter
µg/kg	micrograms per kilogram

μg/L	micrograms per liter
μmhos/cm	micromhos per centimeter
MCL	Maximum Contaminant Level
MEK	methyl ethyl ketone
MF	modifying factor
mg/cm^2	milligrams per square centimeter
mg/kg	milligrams per kilogram
mg/L	milligrams per liter
MHz	megahertz
MNA	monitored natural attenuation
m^2/s	square meters per second
mrem/hr	millirem per hour
MSL	mean sea level
mV	millivolts
NAD	North American datum
NAE	U.S. Army Corps of Engineers New England District
NaOH	sodium hydroxide
NAPL	non-aqueous phase-liquid
NCEA	National Center for Environmental Assessement
NCP	National Oil and Hazardous Substances Pollution Contingency Plan
NCRA	Non-Time Critical Removal Action
NGVD	National Geodetic Vertical Datum
NOAA	National Oceanic and Atmospheric Administration
NOAEL	No Observable Adverse Effect Level
NPDES	National Pollutant Discharge Elimination System
NPL	National Priorities List
NPW	Net Present Worth
NRC	National Research Council
NTU	nephelometric turbidity unit
O&M	operation and maintenance
OATP	Oil Abatement Treatment Plant
OD	outside diameter
OF	Outfall
OME	Ontario Ministry of the Environment
ORP	oxidation-reduction potential
OSHA	Occupational Safety and Health Administration
OSWER	Office of Solid Waste and Emergency Response
OU	operable unit
PAH	polynuclear aromatic hydrocarbon
PAS	Preliminary Assessment Screening
PCB	polychlorinated biphenyl
PCE	tetrachloroethene
PCPT	piezometric cone penetrometer testing
PDIR	Pre-Design Investigation Report
PEC	probable effects concentration

PEF	particulate emission factor
PID	photoionization detector
PMC	Pollutant Mobility Criteria
PP	priority pollutant
ppb	parts per billion
ppbv	parts per billion by volume
ppm	parts per million
ppmv	parts per million by volume
PRG	Preliminary Remediation Goal
PRTV	peer reviewed toxicity values
psi	pounds per square inch
PVC	polyvinyl chloride
Q	discharge rate; flow rate
QA/QC	quality assurance/quality control
QA	quality assurance
QAPP	Quality Assurance Project Plan
RAGS	Risk Assessment Guidance for Superfund
RBC	risk-based concentration
RCRA	Resource Conservation and Recovery Act
RFA	RCRA Facility Assessment
RfC	reference concentration
RfD	reference dose
RI	Remedial Investigation
RIWP	Remedial Investigation Work Plan
RME	reasonable maximum exposure
RQD	rock quality designation
RSR	Remediation Standard Regulation
RTV	reference toxicity value
SA	surface area
SAEP	Stratford Army Engine Plant
SARA	Superfund Amendments and Reauthorization Act
SEM/AVS	simultaneously extracted metals/acid volatile sulfide
SERA	Screening-Level Ecological Risk Assessment
SES	Sound Environmental Solutions
SF	slope factor
SFF	site foraging frequency
SOP	standard operating procedure
SOW	statement of work
SPLP	Synthetic Precipitate Leaching Procedure
SPT	standard penetration test
SQC	sediment quality criteria
SQL	sample quantitation limit
S_s	specific storage
STSC	Superfund Technical Support Center
SVE	soil vapor extraction

SVOA	semi-volatile organic analysis
SVOC	semi-volatile organic compound
SWCHRON	chronic saltwater aquatic toxicity criteria
SWPC	Surface Water Protection Criteria
SWQS	surface water quality standards
Sy	specific yield
T	transmissivity
TACOM	U.S. Army Tank-automotive and Armaments Command
TAL	Target Analyte List
TBC	information to be considered
TCE	trichloroethene
TCL	target compound list
TCLP	toxicity characteristics leaching procedure
TDEMI	time domain electromagnetic induction
TEC	threshold effect concentration
TERC	Total Environmental Restoration Contract
TIC	tentatively identified compound
TOC	total organic carbon
TPH	total petroleum hydrocarbons
trans-1,2-DCE	trans-1,2-dichloroethene
UCL	upper confidence limit
UF	uncertainty factor
UR	unit risk
URSGWC	URS Greiner Woodward Clyde
USACE	United States Army Corps of Engineers
USAEC	United States Army Environmental Center
U.S. Army	U.S. Department of the Army
USCS	Unified Soil Classification System
USEPA	United States Environmental Protection Agency
USFWS	United States Fish and Wildlife Service
USGS	United States Geological Survey
UST	underground storage tank
UV	ultraviolet
VC	Volatilization Criteria
VOA	volatile organic analysis
VOC	volatile organic compound
W-C	Woodward Clyde Consultants
WQS	Water Quality Standards

ABB Environmental Services, Inc. (ABB-ES), 1996. “Final Environmental Baseline Survey Report, Stratford Army Engine Plant, Stratford, Connecticut”. Prepared for U.S. Army Environmental Center, 1996.

Agency for Toxic Substances and Disease Registry (ATSDR, 2002). www.atsdr.cdc.gov

AlliedSignal, 1995. “Facilities Information, Stratford Army Engine Plant”. January 1995.

American Cancer Society (ACS), 1997. “Cancer Facts and Figures”.

Associated Factory Mutual Fire Insurance Companies (AFM FIC), 1931. “Sikorsky Aviation Corporation, Stratford, Connecticut.” June 1, 1931.

Associated Factory Mutual Fire Insurance Companies (AFM FIC), 1937. “United Aircraft Corporation, et al, Stratford, Connecticut.” December 20, 1937.

Associated Factory Mutual Fire Insurance Companies (AFM FIC), 1941. “United Aircraft Corporation, Vought-Sikorsky Aircraft Division, Stratford, Connecticut.” April 2, 1941.

Associated Factory Mutual Fire Insurance Companies (AFM FIC), 1943. “United Aircraft Corporation, Chance Vought Aircraft Division, Stratford, Connecticut.” February 19, 1943.

ASTM. 1994. Standard Test Methods for Measuring the Toxicity and Bioaccumulation of Sediment Associated Contaminants with Freshwater Invertebrates. DRAFT: 1/7/94. ASTM E-4703, ASTM, Philadelphia, PA.

AVCO-Textron, 1992a. “AVCO-Textron Closure/Post-closure Plan”. Submitted to the CTDEP and USEPA in 1992.

AVCO-Textron, 1992b. “RCRA Closure Certification”. Submitted to the CTDEP and USEPA in 1992.

AVCO Lycoming, 1962. Untitled Facility Drawing Depicting Relocation of Fuel Tanks at Building B-53. January 17, 1962.

AVCO Lycoming, 1964. “Fuel Tanks and Distribution System for Building No. 6”. November 23, 1964.

Baerg, D.F., R.C. Starr, J.A. Cherry and D.J.A. Smyth, 1992. Performance Testing of Conventional and Innovative Downhole Samplers and Pumps for VOCs in a Laboratory Monitoring Well. In: Nat. Groundwater Sampling Symp. Proc. (Washington DC), Grundfos Pumps Corp., Clovis, CA, pp. 71-75.

Bhumbla, D.K. and R.F. Keefer, 1994. Arsenic Mobilization and Bioavailability in Soils. Arsenic in the Environment, Part 1: Cycling and Characterization, Jerome O. Nriagu, editor. John Wiley & Sons, Inc., New York.

Bower, H., R.C. Rice, 1976. “A slug test for determing conductivity of unconfined aquifers with completely or partially penetrating wells”. Water Resources Research. 12:423-28.

C.A. Rich Consultants, 1991. Supplementary Hydrogeologic Investigations. Prepared for Textron Lycoming. 1991.

C.A. Rich Consultants, 1995. Allied Signal Engines, Eleventh Year Annual Summary Report, Groundwater Assessment Monitoring Program. Volume 1. February, 1995.

Camp, Dresser, & Mckee Federal Programs Corporation (CDM FPC), 1992. RCRA Facility Assessment. Prepared for the U.S. Department of the Army, June 19, 1992.

Carapella, S.C. Jr., 1972. Arsenic: Element and Geochemistry. The Encyclopedia of Geochemistry and Environmental Sciences, Encyclopedia of Earth Sciences Series, Volume IVA, Rhodes W. Fairbridge, editor. Van Nostrand Reinhold Company, New York.

Chan, S.Y., Mohsen M.F.N, 1992. “Simulation of Intertidal Effects on Contaminant transport in Porous Media”. Groundwater. Vol.30, No.1, pp. 78-86.

Clement, T. P., and Jones, N. L., 1998. “RT3D Tutorials for GMS Users,” PNNL-11805, Pacific Northwest National Laboratory, Richland, WA.

Connecticut Department of Environmental Protection (CTDEP), 1990. Housatonic River Intensive Survey. Water Management Bureau. August.

Connecticut Department of Environmental Protection (CTDEP), 1996a. Remediation Standard Regulations. Section 22a-133k-1 through 22a-133k-3. 1996.

Connecticut Department of Environmental Protection (CTDEP), 1996b. Spatial and Temporal Composition of the Fisheries of the Lower Housatonic River, Connecticut. 1996.

Connecticut Department of Environmental Protection (CTDEP), 1998. Press Release: 1998 Piping Plover/Least Tern Nesting Season Results. October 28, 1998. Internet address: http://dep.state.ct.us/whatshap/press/1998/ps102898.htm

Connecticut Department of Environmental Protection (CTDEP), 1999. 1999 Piping Plover/Least Tern Nesting Season Results, November 19, 1999. www.dep.state.ct.us/whatshap/1999/ps1119b.htm

Connecticut Department of Environmental Protection (CTDEP), 2000a. Water Quality Standards, CTDEP Planning and Standards Division, Bureau of Water Management; Hartford, Connecticut.

Connecticut Department of Environmental Protection (CTDEP), 2000b. 2000 Piping Plover/Least Tern Nesting Season Results, November 19, 1999. www.dep.state.ct.us/whatshap/2000/ps1010b.htm

Connecticut Department of Environmental Protection (CTDEP), 2002. Comments on URSGWC 2000 Draft Remedial Investigation Report, Baseline Risk Assessment, Gary Ginsberg, January 7, 2002.

Connecticut Department of Public Health (CTDPH), 2002. State of Connecticut Department of Public Health Comments, dated April 2, 2002, on Indoor Air Monitoring and Revised Risk Screening Analysis, Sampling Rounds 1 through 24, Stratford Army Engine Plant, Stratford, Connecticut.

Crowley, W.P. The Bedrock Geology of the Long Hill and Bridgeport Quadrangles, 1968. The State Geological and Natural History Survey of Connecticut, Quadrangle Report No. 24, 81 pp. 1968.

CTDEP/USGS, 1995. Hydrogeology of Thick Fill Deposits/Stratigraphy and Hydraulic Properties of Tills. CT DEP Technical Publications Program. 1995.

Department of Defense, 1993. “BRAC Cleanup Plan Guidebook”. Prepared for the Department of Defense, Fall 1993.

Deutsch, W., 1997. Groundwater Geochemistry: Fundamentals and Applications to Contamination. Lewis Publishers, Boca Raton, FL.

Domenico, P.A., F.W. Schwartz, 1990. Physical and Chemical Hydrogeology. John Wiley & Sons. 1990.

Envgineer Research and Development Center (ERDC), 2003. “Natural Attenuation of Chlorinated Solvents at Stratford Army Engine Plant (SAEP) – Groundwater Flow and Reactive Transport Model Evaluation of Treatment Efficiency”. U.S, Army Engineer Research and Development Center. May 5, 2003.

Environmental Science and Engineering, Inc. (ESE), 1981. “Installation Assessment of Stratford Army Engine Plant, Connecticut, Report No. 313”. October 1981.

Environmental Science and Engineering, Inc (ESE), 1991. Seventh Year, Annual Summary Report, Groundwater Assessment Monitoring Program. Prepared for Textron Lycoming, Stratford, Connecticut, ESE Project No. 4904137. 1991.

Fetter, C.W., 1994. Applied Hydrogeology, 3rd Edition, Prentice Hall, Englewood Cliffs, New Jersey.

Fire Insurance Association (FIA), 1956. “AVCO Manufacturing Corporation, et al, Bridgeport-Lycoming Division, Stratford, Connecticut.” January 26, 1956.

Fire Insurance Association (FIA), 1986. “Textron, Inc., “AVCO”, Stratford, Connecticut.” January 6, 1986.

Flint, R.F., 1968. The Surficial Geology of the Ansonia and Milford Quadrangles. The State Geological and Natural History Survey of Connecticut, Quadrangle Report No. 23, 36 pp. 1968.

Foster Wheeler Environmental Corporation Harding Lawson Associates (FWENC/HLA), 1999. Final Work Plan for the Non-Time Critical Removal Action for Chromium and VOC Groundwater Contamination. Prepared for the U.S. Army Corps of Engineers, August 1999.

Foster Wheeler Environmental Corporation/Harding Lawson Associates (FWENC/HLA), 2000a. "Final Pre-Design Investigation Report for the Operable Unit 2 Groundwater NCRA". Prepared for the U.S. Army Corps of Engineers, May 2000.

Foster Wheeler Environmental Corporation/Harding Lawson Associates (FWENC/HLA), 2000b. "Final Pilot-Scale Treatability Study Report". Prepared for the U.S. Army Corps of Engineers, June 2000.

Foster Wheeler Environmental Corporation/Harding Lawson Associates (FWENC/HLA), 2000c. "Final Pre-Design Investigation Report for the Causeway and Dike". Prepared for the U.S. Army Corps of Engineers, April 2000.

Foster Wheeler Environmental Corporation/Harding Lawson Associates (FWENC/HLA), 2000d. "Final Engineering Evaluation/Cost Analysis for the Causeway and Dike". Prepared for the U.S. Army Corps of Engineers, September 2000.

Foster Wheeler Environmental Corporation/Harding Lawson Associates (FWENC/HLA), 2001. "Final Engineering Evaluation/Cost Analysis for the Operable Unit 2 Source Areas". Prepared for the U.S. Army Corps of Engineers, June 2001.

Freeze, A.R., Cherry, J.A., 1979. Groundwater, Prentice-Hall, Inc., Englewood Cliffs, NJ, 1979.

Fritts, Crawford E., 1965. Geologic Quadrangle Maps of the United States – Bedrock Geologic Map of the Milford Quadrangle. United States Geological Survey, Washington D.C. 1965.

Fuschman, P.C., D.J. Duda, and T.R. Barber, 1999. "A Model to Predict Threshold Concentrations for Toxic Effects of Chlorinated Benzenes in Sediment"; Environmental Toxicology and Chemistry, Vol. 18, No. 9, pp. 2100-2109.

Haley and Aldrich, Inc., 1987. Report on Geotechnical Engineering Investigation, Materials Process Center, Building No. 65. Prepared for Kasper Associates, Inc., Bridgeport, Connectictut. 1987.

Harding ESE, Inc. (Harding ESE), 2000. "Geotechnical Investigation Summary for Causeway Design, Stratford Army Engine Plant, Stratford, Connecticut." December 2000.

Harding ESE, Inc. (Harding ESE), 2001. "Causeway Non-time Critical Removal Action Basis of Design, Stratford Army Engine Plant, Stratford, Connecticut." August 2001.

Harding ESE, Inc. (Harding ESE), 2002a. "Draft Indoor Air Modeling Report, Stratford Army Engine Plant, Stratford, Connecticut." June 2002.

Harding ESE, Inc. (Harding ESE), 2002b. "Revised Draft RI Work Plan for Additional Groundwater Investigations". Prepared for the U.S. Army Tank-automotive and Armaments Command, March 2002.

Harding ESE, Inc. (Harding ESE), 2002c. "Draft Scope of Work for Baseline Risk Assessments, Stratford Army Engine Plant, Stratford, Connecticut." July 31, 2002.

Harding ESE, Inc. (Harding ESE), 2002d. "Indoor Air Monitoring and Revised Risk Screening Analysis, Sampling Rounds 1 through 24, Stratford Army Engine Plant, Stratford, Connecticut." March 25, 2002.

Harding Lawson Associates (HLA), 1998. "Building B-2 Chromium Plating Facility In-situ Soil and Groundwater Treatment Stratford Army Engine Plant Time-critical Removal Action Memorandum". Prepared for Allied Signal, Inc., September 1998.

Harding Lawson Associates (HLA), 1999. "Draft Data Package for the Chromium Plating Facility Investigation", Stratford Army Engine Plant, Stratford, Connecticut. 1999.

Honeywell, 2000. Letter from Robert J. Ford of Honeywell to Mr. Kevin Sullivan at CTDEP, entitled "RCRA Status Change Request, Storage Facility Status to Non-Handler Status, Stratford Army Engine Plant", November 16, 2000.

Huang Yan-Chu, 1994. Arsenic Distribution in Soils. Arsenic in the Environment, Part 1: Cycling and Characterization, Jerome O. Nriagu, editor. John Wiley & Sons, Inc., New York.

Hvorslev, M.J., 1951. Time Lag and Soil Permeability in Ground-Water Observations, Bull. No. 36, Waterways Experiment Station Corps of Engineers, U.S. Army, Vicksburg, Mississippi, pp. 1-50.

Idaho National Engineering Laboratory (INEL), 1991. Informal Report Phase III Stratford Army Engine Plant Process Optimization Survey (Draft). Prepared for United States Army Materiel Command.

Industrial Property Management (IPM), 2002. "Summary of Storage Tanks at SAEP Remedial Investigation Report, Stratford Army Engine Plant, Stratford, Connecticut." Updated: 12/30/2002 based on review comments from facility caretaker (Industrial Property Management).

Jones, D.S., G.W. Suter, and R.N. Hull, 1997. Toxicological Benchmarks for Screening Contaminants of Potential Concern for Effects on Sediment-Associated Biota: 1997 Revision; prepared for the U.S. Department of Energy Office of Environmental Management by Lockheed Martin Energy Systems, Inc. under contract DE-AC05-84OR21400; November, 1997.

Karickhoff, S. W., Brown, D. S., and Brown, D. S., and Scott, T. A., 1979. “Sorption of hydrophobic pollutants on natural sediments,” Water Res. 13:241-248.

LeBlanc, L. 1997. End of summer report: Piping Plover and shorebird monitoring in Stratford and Milford, Connecticut May 12 - August 15, 1997. Connecticut Audubon Coastal Center, Milford Point. Milford, Connecticut.

Long, E.R., and L.G. Morgan, 1990. "The Potential for Biological Effects of Sediment-sorbed Contaminants Tested in the National Status and Trends Program"; NOAA Technical Memorandum NOS OMA 52.

Long, E.R., D.D. MacDonald, S.L. Smith, and F.D. Calder, 1995. Incidence of Adverse Biological Effects within Ranges of Chemical Concentrations in Marine and Estuarine Sediments; Environmental Management; Vol. 19, No. 1, pp. 81-97.

MacDonald, D.D., C.G. Ingersoll, and T.A. Berger, 2000. Development and evaluation of Consensus-based Sediment Quality Guidelines for Freshwater Ecosystems. Arch. Environ. Contam. & Toxicol. 39:20-31.

Maughan, J.T., 1993. Ecological Assessment of Hazardous Waste Sites; New York, NY; Van Nostrand Reinhold; p. 352.

Melvin, R.L., Virginia de Lima, B.D. Stone, 1992. Stratigraphy and Hydraulic Properties of Tills in Southern New England: U.S. Geological Survey Open-File Report 91-481. 1992. 53.pp.

Metcalf & Eddy, 1987. Goundwater Monitoring, Assessment Programs. Prepared for Avco Lycoming Textron. March 1987.

Mok, W.M., and C.M. Wai, 1994. Mobilization of Arsenic in Contaminated River Waters. Arsenic in the Environment, Part 1: Cycling and Characterization, Jerome O. Nriagu, editor. John Wiley & Sons, Inc., New York.

Molz, F.J., M.A. Widdowson, and L.D. Benefield, 1986. “Simulation of microbial growth dynamics coupled to nutrient and oxygen transport in porous media,” Water Resources Research, 22(8): 1207-1216.

National Academy of Sciences (NAS), 1977. "Drinking Water and Health"; Safe Drinking Water Committee, Washington, D.C.

National Research Council (NRC), 1982. "Nutrient Requirements of Minks and Foxes. Second Revised Edition, 1982"; Subcommittee on Furbearer Nutrition, National Academy Press, Washington, D.C.

National Research Council (NRC), 1984. Nutrient Requirements of Poultry; Subcommittee on Poultry Nutrition; National Academy Press, Washington, D.C

Nikolaidis, N., et al., 1999. Methodology for Site-Specific, Mobility-Based Cleanup Standards for Heavy Metals in Glaciated Standards, Eviron. Sci. Technol., Vol. 33, No. 17, pp. 2910-2916.

Palmer, C. and R. Puls, 1994. Natural Attenuation of Hexavalent Chromium in Groundwater and Soils. EPA/540/5-94/505.

Pankow, J., and J. Cherry, 1996. Dense Chlorinated Solvents and Other DNAPLs in Groundwater, Waterloo Press, Portland, OR.

Perry, J.H., editor -in-chief, 1950. Chemical Engineers' Handbook, Third Edition. McGraw-Hill Book Company, Inc. New York.

Persaud, D., R. Jaagumagi, and A. Hayton, 1996. "Guidelines for the Protection and Management of Aquatic Sediment Quality in Ontario"; Ontario Ministry of the Environment, Queen's Printer for Ontario; 23 pp.

Rodgers, J., 1985. The Bedrock Geological Map of Connecticut, Connecticut Geological and Natural History Survey. 1985

SAEP Local Redevelopment Authority (SAEP LRA), 1997. "SAEP Redevelopment Plan and Implementation Strategy and Homeless Assistance Submission". 1997.

Schwille, F., 1988. Dense Chlorinated Solvents in Porous and Fractured Media: Model Experiments. Translated by J. Pankow, English Language Edition, Lewis Publishers, Chelsea, MI.

Serfes, M.E., 1991. “Determining the mean hydraulic gradient of ground water affected by Intertidal fluctuations”. Groundwater. Vol. 29, No. 4, pp. 549-555, 1991.

Sound Environmental Solutions (SES), 1999. Quarterly Groundwater Monitoring Report – RCRA Lagoons, Stratford Army Engine Plant, Stratford, Connecticut. 1999.

Sound Environmental Solutions (SES), 2002. 2001 Annual Summary Report – RCRA Groundwater Monitoring Program, Stratford Army Engine Plant, Stratford, Connecticut. February 18, 2002.

Stollenwerk, K., et al., 1985. Adsorption and Desorption of Hexavalent Chromium in an Alluvial Aquifer Near Telluride, Colorado. J. Environ. Qual., No. 14, pp. 150-155.

Stone, J.R., B.D. Stone, Lewis, R.S., 1985. “Late Quaternary Deposits of the Southern Quinnipiac-Farmington Lowland and Long Island Sound Basin: Their Place in a Regional Stratigraphic Framework”. In Guidebook for Fieldtrips in Connecticut and Adjacent Areas of New York and Rhode Island. New England Intercollegiate Geoogical Conference. 77th Annual Meeting. State Geological and Natural History Survey of Connecticut. Pp. 535-575.

Suter, Glen W., 1993. "Ecological Risk Assessment"; Lewis Publishers, Chelsea Michigan; 1993.

Suter, Glen Wl., and Tsao, C.L. 1996. Toxicological Benchmarks for Screening Potential Contaminants of Concern for Effects on Aquatic Biota: 1996 Revision. Prepared by Risk Assessment Program, Health Sciences Research Division, Oak Ridge, TN. Prepared for the U.S. Department of Energy, Office of Envt. Mgmt. ES/ER/TM-96/R2; June, 1996.

TetraTech (NUS), 1999. “Final Area 1 Remedial Investigation, Raymark-Ferry Creek – Operable Unit 3, Stratford, Connecticut.” October 1999.

Textron, 1991. File Memoranda.

Textron-Lycoming, 1993. SAEP site plan located in file room of Building B-1 of the Stratford Army Engine Plant, Stratford, Connecticut.

U.S. Army Assistant Chief Of Staff for Installation Management (ACSIM), 2004, Sampling and Analysis Plan, Remedial Investigation, Stratford Army Engine Plant, Stratford, CT. March 2004.

U.S. Army Corps of Engineers (USCOE), 1999. The BSAF Database. Waterways Experiment Station, Environmental Laboratory. www.wes.army.mil/el/bsaf/bsaf.html

U.S. Department of Agriculture, Soil Conservation Service, 1981. Soil Survey of Fairfield County, Connecticut. February, 1981.

U.S. Environmental Protection Agency (USEPA), 1988a. Superfund Exposure Assessment Manual. Office of Remedial Response; EPA/540/1-88/001; Washington, D.C.

U.S. Environmental Protection Agency (USEPA), 1988b. "Interim Sediment Criteria Values for Nonpolar Hydrophobic Organic Contaminants"; Office of Water Regulations and Standards; SCD No. 17; Washington, D.C.

U.S. Environmental Protection Agency (USEPA), 1989. “Risk Assessment Guidance for Superfund, Volume 1, Human Health Evaluation Manual (Part A)”; Office of Emergency and Remedial Response; EPA-540/1-89/002 (interim final); Washington, D.C., December.

U.S. Environmental Protection Agency (USEPA), 1990a. “Installation Assessment, Stratford Army Engine Plant, Stratford, Connecticut.” July 1990.

U.S. Environmental Protection Agency (USEPA), 1990b. Code of Federal Regulations, Title 40, Part 300, National Oil and Hazardous Substances Pollution Contingency Plan; Federal Register; March 8.

U.S. Environmental Protection Agency (USEPA), 1991a. “Risk Assessment Guidance for Superfund, Volume I: Human Health Evaluation Manual, Supplemental Guidance, Standard Default Exposure Factors"; Office of Emergency and Remedial Response, Toxics Integration Branch; OSWER Directive 9285.6-03 (interim final); Washington, D.C.

U.S. Environmental Protection Agency (USEPA), 1991b. Ambient Water Quality Criteria Summary. Office of Science and Technology. Health and Ecological Criteria Division.

U.S. Environmental Protection Agency (USEPA), 1991c. "Water Quality Criteria Summary"; Washington, D.C.; May 1, 1991.

U.S. Environmental Protection Agency (USEPA), 1992a. "Guidance for Data Useability in Risk Assessment (Parts A and B)"; Office of Emergency and Remedial Response; Publication 9285.7-09A; Washington, D.C.; April

U.S. Environmental Protection Agency (USEPA), 1992b. "Supplemental Guidance to RAGS, Calculating the Concentration Term"; Office of Solid Waste and Emergency Response; Intermittent Bulletin, Vol. 1, No. 1; Washington, D.C.; Pub. 9285.7-081; May 1992.

U.S. Environmental Protection Agency (USEPA), 1992c. Sediment Classification Methods Compendium; U.S. Environmental Protection Agency, Sediment OverSite Technical Committee of the Office of Science and Technology; EPA 823-R-92-006; Washington, DC; September, 1992.

U.S. Environmental Protection Agency (USEPA), 1993a. Wildlife Exposure Factors Handbook; Vol. I and II; Office of Research and Development; EPA/600/R-93/187a,b; Washington D.C.; December, 1993.

U.S. Environmental Protection Agency (USEPA), 1993b. Sediment Quality Criteria for the Protection of Benthic Organisms - Acenapthene; EPA-822-R-93-013; United States Environmental Protection Agency, Office of Water and Office of Research & Development; September, 1993.

U.S. Environmental Protection Agency (USEPA), 1993c. Sediment Quality Criteria for the Protection of Benthic Organisms - Dieldrin; EPA-822-R-93-015; United States Environmental Protection Agency, Office of Water and Office of Research & Development; September, 1993.

U.S. Environmental Protection Agency (USEPA), 1993d. Sediment Quality Criteria for the Protection of Benthic Organisms - Endrin; EPA-822-R-93-016; United States Environmental Protection Agency, Office of Water and Office of Research & Development; September, 1993.

U.S. Environmental Protection Agency (USEPA), 1993e. Sediment Quality Criteria for the Protection of Benthic Organisms - Fluoranthene; EPA-822-R-93-012; United States Environmental Protection Agency, Office of Water and Office of Research & Development; September, 1993.

U.S. Environmental Protection Agency (USEPA), 1993f. Sediment Quality Criteria for the Protection of Benthic Organisms - Phenanthrene; EPA-822-R-93-014; United States Environmental Protection Agency, Office of Water and Office of Research & Development; September, 1993.

U.S. Environmental Protection Agency (USEPA), 1994a. Understanding Variation in Partition Coefficient K_d Values. EPA/402-R-99-004A&B, August

U.S. Environmental Protection Agency (USEPA), 1994b. "Risk Updates", USEPA Region I, Waste Management Division; No. 2; August.

U.S. Environmental Protection Agency (USEPA), 1994c. Methods for measuring the toxicity and bioaccumulation of sediment-associated contaminants with freshwater invertebrates. EPA/600/R-94/024. Washington, DC.

U.S. Environmental Protection Agency (USEPA), 1994d. "Revised Interim Soil Lead Guidance for CERCLA Sites and RCRA Corrective Action Facilities", OSWER Directive 9355.4-12. July

U.S. Environmental Protection Agency (USEPA), 1995. "Risk Updates"; USEPA Region I, New England, Waste Management Division; No. 3; August.

U.S. Environmental Protection Agency (USEPA), 1996a. "USEPA Region 1 Low Stress Purging and Sampling Procedure for the Collecting of Groundwater Samples from Monitoring Wells"; Revision 2; July 30, 1996.

U.S. Environmental Protection Agency (USEPA), 1996b. "Eco Update", Intermittent Bulletin Volume 3, No. 3; Ecotox Thresholds; U.S. Environmental Protection Agency, Office of Remedial Response; EPA 540/F-95/038, Publication 9345.0-12FSI; January, 1996.

U.S. Environmental Protection Agency (USEPA), 1996c. "Risk Updates"; USEPA New England, Waste Management Division; No. 4; November.

U.S. Environmental Protection Agency (USEPA), 1997. Data Quality Evaluation Statistical Toolbox (DataQUEST) User's Guide. EPA QA/G-9D. QA96 Version. EPA/600/R-96/085. December 1997.

U.S. Environmental Protection Agency (USEPA), 1997a. "Exposure Factors Handbook, Volume 1; Office of Research and Development; EPA-600/P-95/002Fa; Washington, D.C.; August.

U.S. Environmental Protection Agency (USEPA), 1997b. Health Effects Assessment Summary Tables (HEAST), Annual Update: Office of Solid Waste and Emergency Response, EPA 540/R/97/036, PB97-921199

U.S. Environmental Protection Agency (USEPA), 1997c. Ecological Risk Assessment Guidance for Superfund: Process for Designing and Conducting Ecological Risk Assessment;

Interim Final U.S. EPA Document; U.S. EPA Environmental Response Team; Edison, NJ; June 2, 1997.

U.S. Environmental Protection Agency (USEPA), 1998a. Technical Protocol for Evaluating Natural Attenuation of Chlorinated Solvents in Ground Water. Wiedemeier, et al., National Risk Management Research Laboratory, Office of Research and Development, USEPA, Cincinnati, OH.

U.S. Environmental Protection Agency (USEPA), 1998b. Guidelines for Ecological Risk Assessment; Federal Register, Vol. 63, No. 93; May 14, 1998.

U.S. Environmental Protection Agency (USEPA), 1999a. "Use of Monitored Natural Attenuation at Superfund, RCRA Corrective Action, and Underground Storage Tank Sites," Directive Number 9200.4-17P, United States Environmental Protection Agency, Office of Solid Waste and Emergency Response, Washington, DC.

U.S. Environmental Protection Agency (USEPA), 1999a. "Risk Updates"; USEPA New England, Waste Management Division; No. 5; September 1999.

U.S. Environmental Protection Agency (USEPA), 1999b. Screening Level Ecological Risk Assessment Protocol for Hazardous Waste Combustion Facilities; United States Environmental Protection Agency; Office of Solid Waste; EPA/635/R-98/008; Washington, D.C.; August 1999.

U.S. Environmental Protection Agency (USEPA), 2000a. BIOCHLOR: Natural Attenuation Decision Support System. User's Manual Version 1.0. EPA/600/R-00/008. January 2000.

U.S. Environmental Protection Agency (USEPA), 2002. Preliminary Remediation Goals (PRG) Table. USEPA Region IX Office of Solid Waste and Emergency Response.

U.S. Environmental Protection Agency (USEPA), 2000c. Comments on URSGWC 2000 Draft Remedial Investigation Report, Baseline Risk Assessment, December 18, 2000.

U.S. Environmental Protection Agency (USEPA), 2001a. Evaluation of the Protocol for Natural Attenuation of Chlorinated Solvents: Case Study at the Twin Cities Army Ammunition Plant. EPA/600/R-01/025, March 2001.

U.S. Environmental Protection Agency (USEPA), 2001b. "Risk Assessment Guidance for Superfund, Volume 1, Human Health Evaluation Manual (Part D), Final"; Office of Emergency and Remedial Response; EPA-540/1-89/002 (interim final); Washington, D.C.; December 2001.

U.S. Environmental Protection Agency (USEPA), 2001c. "Risk Assessment Guidance for Superfund, Volume 1, Human Health Evaluation Manual (Part E, Supplemental Guidance for Dermal Risk Assessment, Interim)"; Office of Emergency and Remedial Response; EPA/540/R/99/005; Washington, D.C..

U.S. Environmental Protection Agency (USEPA), 2001d. "Frequently Asked Questions (FAQs) on the Adult Lead Model" Technical Review Workgroup for Lead (TRW). Guidance Document. August 2001.

U.S. Environmental Protection Agency (USEPA), 2002a. "Calculating Upper Confidence Limits for Exposure Point Concentrations at Hazardous Waste Sites:; Office of Solid Waste and Emergency Response/Office of Solid Waste and Remedial Response; OSWER 9285.6-10; December 2002.

U.S. Environmental Protection Agency (USEPA), 2002b. Preliminary Remediation Goals (PRG) Table. USEPA Region IX Office of Solid Waste and Emergency Response.

U.S. Environmental Protection Agency (USEPA), 2002c. "National Recommended Water Quality Criteria; Notice; Republication"; Federal Register, Vol. 63, No. 237; pp. 68354-68364; Thursday, December 10, 1998.

U.S. Environmental Protection Agency (USEPA), 2003a. Human Health Toxicity Values in Superfund Risk Assessments. OSWER 9285.7-53. December 2003.

U.S. Environmental Protection Agency (USEPA), 2003b. Recommendations of the Technical Review Workgroup for Lead for an Approach to Assessing Risks Associated with Adult Exposures to Lead in Soil. Final. EPA-540-R-03-001. January 2003.

U.S. Environmental Protection Agency (USEPA), 2004a. Risk-Based Concentration Table. USEPA Region III. April 14, 2004.

U.S. Environmental Protection Agency (USEPA), 2004b. "Integrated Risk Information System (IRIS)"; on-line data-base search; April, 2004.

U.S. Environmental Protection Agency (USEPA), 2004c. Software for Calculating Upper Confidence Limits. ProUCL Version 3.00. Technical Support Center for Monitoring and Site Characterization.

U.S. Fish and Wildlife Service (USFWS), 1995. Piping Plover (Charadrius melodur), Atlantic Coast Population, Revised Recovery Plan. Hadley, MA. 238pp.

U.S. Geological Survey, 1989. Water Resources Data: Connecticut, Water Year 1989. Report CT-89-1. Prepared by M.A. Cerrione, B.S. Davies, J.R. Bohr, and B.W. Hunter. Water Resources Division. USGS Connecticut Office, Hartford, CT.

URS Greiner-Woodward Clyde (URSGWC), 1999. Draft Remedial Investigation Ecological Problem Formulation Technical Memorandum, Stratford Army Engine Plant, Stratford, Connecticut. Contract Number DACW41-96D-8014. Prepared for US Dept. of Army, Corps of Engineers, NY District. September 30, 1999.

URS Greiner-Woodward Clyde (URSGWC), 2000. "Draft Remedial Investigation Report". Prepared for U.S. Army Corps of Engineers, September 2000.

URS Greiner-Woodward Clyde (URSGWC), 1998. Remedial Investigation Work Plan for the Stratford Army Engine Plant. Prepared for the U.S. Department of the Army, October 26, 1998.

Vogel, T.M., Criddle, C.S., and McCarty, P.L., 1987. "Transformations of halogenated aliphatic compounds," Environ. Sci. & Tech., 21(8): 722-736.

Vought, 2003. http://www.vought.com/heritage/peoplaces/html/lstratford.html

Weston, 1982. "Interim Report for Chemical Waste Treatment Plant Study, Stratford Army Engine Plant, Connecticut. Presented to Department of the Army, New York District Corps of Engineers".

Wiedemeier, T., at al., 1998. Technical Protocol for Evaluating Natural Attenuation of Chlorinated Solvents in Ground Water. EPA/600/R-98/128. Prepared for National Risk Management Research Laboratory, Office of Research and Development, USEPA, Cincinnati, OH.

Woodward-Clyde Consultants (W-C), 1991. "Final Preliminary Assessment Screening, Stratford Army Engine Plant, Connecticut". Prepared for the U.S. Army Corps of Engineers, Omaha District, 1991.

Woodward-Clyde Consultants (W-C), 1993. "Remedial Investigation Report (Phase I), Stratford Army Engine Plant, Connecticut". Prepared for the U.S. Army Corps of Engineers, Omaha District, 1993.

Woodward-Clyde Consultants (W-C), 1994. Phase II Remedial Investigation Work Plan. Stratford Army Engine Plant. Stratford, Connecticut, Prepared for USACE. November 1994.

Woodward-Clyde Consultants (W-C), 1995. Draft Baseline Ecological Risk. Stratford Army Engine Plant, Stratford, Connecticut, Prepared for USACE. November.

Woodward-Clyde Consultants (W-C), 1996. "Phase II Remedial Investigation Report, Stratford Army Engine Plant, Connecticut". Prepared for the U.S. Army Corps of Engineers, 1996.

Yim, C. and M. Mohsen, 1992. "Simulation of Tidal Effects on Contaminant Transport in Porous Media". Groundwater, Vol. 30, No.1. January-February, 1992, pp. 78-86.

Zachara, J., et al., 1987. Chromate Adsorption on Amorphous Iron Oxyhydroxide in the Presence of Major Groundwater Ions, Environ. Sci. Technol., Vol. 21, No. 6, pp. 589-594.

Zecco, Inc. Subsurface Investigation Report, Textron Lycoming. Prepared for Textron

FIGURES

TABLES