RI00523F

REMEDIAL INVESTIGATION

VOLUME I OF II
TEXT, TABLES, AND FIGURES

RAYMARK - OU2 - GROUNDWATER
STRATFORD, CONNECTICUT

RESPONSE ACTION CONTRACT (RAC), REGION I

For
U.S. Environmental Protection Agency

By
Tetra Tech NUS, Inc.

EPA Contract No. 68-W6-0045
EPA Work Assignment No. 126-RICO-01H3
TtNUS Project No. N4236

January 2005

TETRA TECH NUS, INC.

RI00523F

REMEDIAL INVESTIGATION

VOLUME I OF II
TEXT, TABLES, AND FIGURES

RAYMARK – OU2 - GROUNDWATER
STRATFORD, CONNECTICUT

RESPONSE ACTION CONTRACT (RAC), REGION I

For
U.S. Environmental Protection Agency

By
Tetra Tech NUS, Inc.

EPA Contract No. 68-W6-0045
EPA Work Assignment No. 126-RICO-01H3
TtNUS Project No. N4236

January 2005

Heather M. Ford
Project Manager

George D. Gardner, P.E.
Program Manager

RI00523F

REMEDIAL INVESTIGATION

VOLUME II OF II
APPENDICES

RAYMARK - OU2 - GROUNDWATER
STRATFORD, CONNECTICUT

RESPONSE ACTION CONTRACT (RAC), REGION I

For
U.S. Environmental Protection Agency

By
Tetra Tech NUS, Inc.

EPA Contract No. 68-W6-0045
EPA Work Assignment No. 126-RICO-01H3
TtNUS Project No. N4236

January 2005

TETRA TECH NUS, INC.

RI00523F

REMEDIAL INVESTIGATION

VOLUME II OF II
APPENDICES

RAYMARK – OU2 - GROUNDWATER
STRATFORD, CONNECTICUT

RESPONSE ACTION CONTRACT (RAC), REGION I

For
U.S. Environmental Protection Agency

By
Tetra Tech NUS, Inc.

EPA Contract No. 68-W6-0045
EPA Work Assignment No. 126-RICO-01H3
TtNUS Project No. N4236

January 2005

Heather M. Ford
Project Manager

George D. Gardner, P.E.
Program Manager

TABLE OF CONTENTS
REMEDIAL INVESTIGATION
RAYMARK – OU2 – GROUNDWATER
STRATFORD, CONNECTICUT

SECTION **PAGE**

ACRONYMS AND ABBREVIATIONS

E.0 EXECUTIVE SUMMARY E-1

1.0 INTRODUCTION 1-1
1.1 Purpose of Report 1-1
1.2 Report Organization 1-2
1.3 Study Area Background 1-3
1.3.1 History of the Raymark Facility 1-3
1.3.2 Study Area Description and Setting 1-6
1.3.3 Other On-Going Activities 1-8

2.0 FIELD INVESTIGATIONS 2-1
2.1 Sediment Sampling (1992) 2-1
2.2 Surface Water Sampling at the Raymark Facility (1993) 2-1
2.3 Expanded Site Inspection and Vertical Sampling Program (1993) 2-2
2.4 Phase I Remedial Investigation (1993 – 1995) 2-3
2.4.1 Soil Borings, Test Pits, and Soil Sampling (1994) 2-3
2.4.2 Groundwater Monitoring Well Installation (1994) 2-3
2.4.3 Ground Penetrating Radar Survey (1994) 2-4
2.4.4 Topographic Survey/Global Positioning Survey (1994) 2-4
2.4.5 Surface Water and Sediment Sampling (1994 – 1995) 2-4
2.4.6 Groundwater Sampling (1994 – 1995) 2-5
2.4.7 Wetlands Delineation (1994) 2-5
2.5 Comprehensive Site Investigation Sampling Program (1994 – 1995) 2-6
2.6 Removal Action Post-Excavation Program (1994 – 1996) 2-6
2.7 Phase 1 OU2 Groundwater Remedial Investigation (1996 – 1997) 2-7
2.7.1 Monitoring Well Installation (1996) 2-7
2.7.2 Groundwater Profiling (1997) 2-8
2.7.3 Monitoring Well Installation (1997) 2-8
2.7.4 Other Groundwater Evaluation Investigations (1997) 2-9
2.7.5 Groundwater Sampling (1997) 2-9
2.8 Ecological Risk Assessment (1996 – 1998) 2-10
2.9 Phase II Commercial Properties Site Investigation (1997) 2-10
2.10 Phase II Groundwater Remedial Investigation (1998 – 1999) 2-11
2.10.1 Monitoring Well Installation (1998) 2-11
2.10.2 Groundwater Sampling (1998 – 1999) 2-12
2.10.3 Seismic Refraction Survey (1998) 2-12
2.10.4 Hydraulic Conductivity Tests (1998 – 1999) 2-12
2.10.5 Tidal Influence Study (1999) 2-13
2.10.6 Ferry Creek Seepage Study (1999) 2-13

TABLE OF CONTENTS (cont.)
REMEDIAL INVESTIGATION
RAYMARK – OU2 – GROUNDWATER
STRATFORD, CONNECTICUT

SECTION **PAGE**

2.11 Residential Indoor Air Study (2000 through 2003) 2-14
2.12 Phase III Groundwater Remedial Investigation (2002 – 2003) 2-16
2.12.1 Groundwater and Soil Gas Profiling (2002) 2-16
2.12.2 Well Development/Redevelopment (2002 – 2003) 2-17
2.12.3 Monitoring Well Installation (2002 – 2003)..... 2-17
2.12.4 Resistivity and Very Low Frequency (VLF) Survey (2002)..... 2-19
2.12.5 Borehole Geophysics (2002)..... 2-19
2.12.6 Groundwater and Soil Gas Sampling (2002 – 2003)..... 2-19
2.12.7 Ferry Creek Seepage Study and Surface Water Sampling (2003) 2-20
2.12.8 Tidal Influence Study (2003)..... 2-21
2.12.9 Aquifer Test (2003) 2-21
2.13 Soil Sampling (2002 and 2003) 2-21

3.0 PHYSICAL CHARACTERISTICS OF THE STUDY AREA 3-1
3.1 Surface Features and Land Use 3-1
3.1.1 Study Area Infiltration 3-2
3.1.2 Groundwater Classification and Usage within the Study Area and Environs 3-2
3.2 Geology 3-4
3.2.1 Regional Geology..... 3-4
3.2.2 OU2 Study Area Geology 3-6
3.3 Surface Water 3-24
3.3.1 Ferry Creek 3-24
3.3.2 Housatonic River 3-24
3.4 Hydrogeology 3-25
3.4.1 Regional Hydrogeology 3-25
3.4.2 Study Area Hydrogeology 3-27
3.5 Soil Gas..... 3-51
3.6 Climate and Meteorology..... 3-52

4.0 NATURE AND EXTENT OF CONTAMINATION..... 4-1
4.1 Potential Groundwater Contaminant Sources 4-1
4.1.1 Former Raymark Facility 4-1
4.1.2 Off-facility Distribution of Raymark Waste 4-3
4.1.3 Other Non-Raymark Related Sources 4-4
4.2 Background Locations 4-4
4.3 Nature and Extent of Contamination in Groundwater 4-6
4.3.1 Volatile Organic Compounds in Groundwater..... 4-8
4.3.2 Semi-Volatile Organic Compounds in Groundwater 4-14
4.3.3 Pesticides and PCBs in Groundwater..... 4-17
4.3.4 Metals in Groundwater 4-19
4.3.5 Dioxins in Groundwater 4-22

TABLE OF CONTENTS (cont.)
REMEDIAL INVESTIGATION
RAYMARK – OU2 – GROUNDWATER
STRATFORD, CONNECTICUT

SECTION **PAGE**

4.4 Nature and Extent of Contamination in Indoor Air and Soil Gas 4-24
4.4.1 Volatile Organic Compounds in Indoor Air 4-25
4.4.2 Volatile Organic Compounds in Soil Gas 4-26

5.0 CONTAMINANT FATE AND TRANSPORT 5-1
5.1 Properties and Processes Controlling Contaminant Fate and Transport 5-1
5.1.1 Physical Properties of the Contaminants in Groundwater 5-1
5.1.2 Fate and Transport Processes for Contaminants in Groundwater 5-3
5.2 Contaminant Source Locations 5-7
5.3 Contaminant Fate and Transport Modeling 5-9
5.3.1 Transport Model Formulation 5-10
5.3.2 Model Assumptions and Limitations 5-12
5.3.3 Modeling Methodology 5-13
5.3.4 Pathways and Predicted Migration of Groundwater Plumes 5-14
5.3.5 Model Uncertainty 5-24
5.4 Contaminant Persistence 5-24
5.5 Summary of Fate and Transport of Contaminants in Groundwater 5-29
5.6 Fate and Transport of Contaminants in Soil Gas 5-31
5.6.1 Contaminant Transport from Groundwater to Soil Gas 5-31
5.6.2 Contaminant Transport from Groundwater and Soil Gas to Indoor Air 5-32

6.0 BASELINE HUMAN HEALTH RISK ASSESSMENT 6-1
6.1 Introduction – Overview of Risk Assessment Process 6-4
6.2 Data Evaluation 6-5
6.2.1 Selection of Chemicals of Potential Concern 6-5
6.2.2 Identification of COPCs 6-9
6.2.3 Exposure Point Concentrations 6-10
6.3 Exposure Assessment 6-16
6.3.1 Conceptual Site Model 6-16
6.3.2 Potential Routes of Exposure 6-20
6.3.3 Potential Receptors 6-22
6.3.4 Exposure Pathways 6-24
6.4 Toxicity Assessment 6-28
6.4.1 Non-carcinogenic Effects 6-30
6.4.2 Carcinogenic Effects 6-31
6.4.3 Toxicity Summaries for Major Chemicals of Potential Concern 6-33
6.5 Risk Characterization 6-36
6.5.1 Risk Characterization Methodology 6-36
6.5.2 Risk Characterization Results 6-38
6.6 Modeled Indoor Air Evaluation 6-40

TABLE OF CONTENTS (cont.)
REMEDIAL INVESTIGATION
RAYMARK – OU2 – GROUNDWATER
STRATFORD, CONNECTICUT

SECTION **PAGE**

6.7 Uncertainties Analysis 6-42
6.7.1 Uncertainty in Data Evaluation 6-43
6.7.2 Uncertainty in the Exposure Assessment 6-47
6.7.3 Uncertainty in the Toxicological Evaluation 6-49
6.7.4 Qualitative Evaluation of Potential Risks from Oyster Consumption 6-52
6.8 Conclusions 6-53

7.0 ECOLOGICAL RISK ASSESSMENT 7-1
7.1 Environmental Setting 7-1
7.1.1 Study Area Description 7-1
7.1.2 Water and Sediment Conditions 7-1
7.1.3 Habitats and Potentially Exposed Receptor Groups 7-2
7.2 Exposure Pathways 7-2
7.3 Chemicals of Potential Concern 7-3
7.4 Selection of Ecological Endpoints 7-4
7.5 Ecological Effects Evaluation 7-5
7.6 Exposure Estimate 7-6
7.7 Risk Calculation 7-6
7.7.1 Recent Exposure 7-6
7.7.2 Future Exposure 7-8
7.7.3 Uncertainty Analysis 7-9
7.8 Conclusions 7-11

8.0 SUMMARY AND CONCLUSIONS 8-1
8.1 Nature and Extent of Contamination Summary 8-2
8.1.1 Nature and Extent of Groundwater Contamination 8-3
8.1.2 Nature and Extent of Indoor Air and Soil Gas Contamination 8-9
8.2 Contaminant Fate and Transport Summary 8-10
8.3 Risk Assessment Summary 8-11
8.3.1 Human Health Risk Assessment 8-12
8.3.2 Ecological Risk Evaluation 8-16
8.4 Conclusions 8-16

TABLES

NUMBER

2-1 History of Activities Associated with Raymark Facility and Environs
2-2 Groundwater Samples Collected during 2002/2003 and Analyses Performed
3-1 Hydraulic Conductivity Values for Single Well Pump Tests

TABLE OF CONTENTS (cont.)
REMEDIAL INVESTIGATION
RAYMARK – OU2 – GROUNDWATER
STRATFORD, CONNECTICUT

TABLES (cont.)

NUMBER

2-1 History of Activities Associated with Raymark Facility and Environs
2-2 Groundwater Samples Collected during 2002/2003 and Analyses Performed
3-1 Hydraulic Conductivity Values for Single Well Pump Tests
3-2 Summary of Vertical Gradients
3-3 Soil Air Permeability Test Results
4-1 Chemicals or Materials Used or Handled at the Raymark Facility
4-2 Data Summary and Criteria Comparisons – Upgradient Overburden
4-3 Data Summary and Criteria Comparisons – Upgradient Bedrock
4-4 Criteria for Comparison - Groundwater
4-5 Data Summary and Criteria Comparisons – Overburden and Recovery
4-5a Data Summary and Criteria Comparison – Modeled VOCs and Metals in All Aquifers
4-6 Data Summary and Criteria Comparisons – Bedrock
4-7 Statistical Comparison of Temporal Changes in Groundwater VOC Concentrations throughout the OU2 Study Area
4-8 Statistical Comparison of Temporal Changes in Groundwater VOC Concentrations in the Residential area East of Ferry Boulevard
4-9 Summary of Indoor Air VOC Data
5-1 Fate and Transport Modeling Parameters
5-2 Fate and Transport Modeling Results
5-3 Simulated Versus Observed Concentrations
5-4 Spearman-Rho Correlation between VOC Concentrations in Groundwater and VOC Concentrations in Soil Gas
5-5 Biodegradation Screening for Monitoring Well MW-303B
5-6 Biodegradation Screening for Monitoring Well MW-307B
5-7 Biodegradation Screening for Monitoring Well MW-308B
5-8 Biodegradation Screening for Monitoring Well MW-501S
5-9 Biodegradation Screening for Monitoring Wells MW-501D and MW-501B
5-10 Biodegradation Screening for Monitoring Wells GW-J2 and GW-J5
5-11 Biodegradation Screening for Monitoring Wells RW-3 and RW-6
5-12 Biodegradation Screening for Monitoring Well RW-1
5-13 Biodegradation Screening for Monitoring Wells PC-14D and PC-14S
5-14 Biodegradation Screening for Monitoring Wells MW-302D and MW-302B
5-15 Biodegradation Screening for Monitoring Well MW-302S
5-16 Biodegradation Screening for Monitoring Wells PC-02D and PC-02B
5-17 Biodegradation Screening for Monitoring Wells MW-211D and MW-211B
5-18 Biodegradation Screening for Monitoring Well MW-211M
5-19 Biodegradation Screening for Monitoring Wells MW-212M and MW-212D
5-20 Biodegradation Screening for Monitoring Well MW-212S
5-21 Biodegradation Screening for Monitoring Well MW-503S
5-22 Biodegradation Screening for Monitoring Wells MW-111M and MW-111D
5-23 Biodegradation Screening for Monitoring Well MW-113M

TABLE OF CONTENTS (cont.)
REMEDIAL INVESTIGATION
RAYMARK – OU2 – GROUNDWATER
STRATFORD, CONNECTICUT

TABLES (cont.)

NUMBER

6-1 Summary of Groundwater Human Health Risk Assessment
7-1 Ferry Creek Surface Water Samples Previously Evaluated
7-2 Surface Water and Seep Samples Collected in 1999
7-3 Hazard Quotients for Future Sediment Pore Water and Surface Water Concentrations

FIGURES

NUMBER

1-1 Site Locus
1-2 Study Area – Potential Groundwater Sources
2-1 Groundwater and Soil Gas Monitoring Wells and Surface Water Gauges
2-2 Geophysical Surveys
2-3 Hydrologic Studies
2-4 2002 Direct-Push Soil Gas and Groundwater Locations
3-1 Land Use and Physical Features
3-2 Cross-Section Locations
3-3 A-A' Geological Cross-Section
3-4 B-B' Geological Cross-Section
3-5 C-C' Geological Cross-Section
3-6 Bedrock Contours
3-7 Water Table Contours and Flow Direction
3-8 Potentiometric Surface Contours from Overburden and Bedrock Wells Screened at Approximate Elevation – 40 feet NGVD
4-1 Spatial Distribution of Benzene in Groundwater
4-2 Spatial Distribution of Chlorobenzene in Groundwater
4-3 Spatial Distribution of Toluene in Groundwater
4-4 Spatial Distribution of 1,1,1-trichloroethane in Groundwater
4-5 Spatial Distribution of 1,1-dichloroethene in Groundwater
4-6 Spatial Distribution of Trichloroethene in Groundwater
4-7 Spatial Distribution of Vinyl Chloride in Groundwater
4-8 Spatial Distribution of Beryllium in Groundwater
4-9 Spatial Distribution of Thallium in Groundwater
4-10 Spatial Distribution of Arsenic in Groundwater
4-11 Spatial Distribution of Zinc in Groundwater
4-12 Spatial Distribution of Copper in Groundwater
4-13 Spatial Distribution of Lead in Groundwater
4-14 Spatial Distribution of Cadmium in Groundwater
4-15 Spatial Distribution of Nickel in Groundwater
4-16 1,1-Dichloroethene Concentrations in Soil Gas 8 Feet Below the Ground Surface
4-17 1,1-Dichloroethene Concentrations in Soil Gas along Cross Section line A-A'

TABLE OF CONTENTS (cont.)
REMEDIAL INVESTIGATION
RAYMARK – OU2 – GROUNDWATER
STRATFORD, CONNECTICUT

FIGURES (cont.)

NUMBER

4-18 Trichloroethene Concentrations in Soil Gas 8 Feet Below the Ground Surface
4-19 Trichloroethene Concentrations in Soil Gas along Cross Section Line A-A'
4-20 Indoor Air Area of Concern
5-1 Selected Sources and Flowlines
5-2 Predicted Concentration vs. Time

REFERENCES

APPENDICES

A Boring Logs and Well Construction Logs
B Analytical Data Disc
C Previous Hydraulic Study Data
D Geology and Hydrogeology
E Groundwater Modeling Supplemental Information
F Human Health Risk Support

ACRONYMS AND ABBREVIATIONS

1,1,1-TCA	1,1,1-trichloroethane
1,1-DCE	1,1-dichloroethene
1,2-DCA	1,2-dichloroethane
1,2-DCP	1,2-dichloropropane
2,3,7,8-TCDD	2,3,7,8 – tetrachlorodibenzo-p-dioxin
ABS	Absorption Factor
ACOE	United States Army Corps of Engineers
ARCS	Alternative Remedial Contracting Strategy
ASTM	American Society for Testing and Materials
ATSDR	Agency for Toxic Substances and Disease Registry
ATV	acoustic televiewer
AVS/SEM	Acid Volatile Sulfide/Simultaneously Extractable Metals
AWQC	Ambient Water Quality Criteria
BCF	biological concentration factor
Beta-BHC	beta-hexachlorocyclohexane
bgs	below ground surface
BHC	Bridgeport Hydraulic Company
B&RE	Brown & Root Environmental
°C	degrees Celsius
CAS	Chemical Abstracts Service
CERCLA	The Comprehensive Environmental Response, Compensation, and Liability Act of 1980. Amended by SARA in 1986. Also called the Superfund Law.
CFR	Code of Federal Regulations
CLP	(EPA) Contract Laboratory Program
cm	centimeters
cm^2	square centimeters
cm^3	cubic centimeters
CNS	central nervous system
COPC	chemicals of potential concern
CSF	Cancer Slope Factor
CSI	Comprehensive Site Investigation
CT	Connecticut
CTDEP	Connecticut Department of Environmental Protection
CTDOT	Connecticut Department of Transportation
CT DPHAS	Connecticut Department of Public Health and Addiction Services
CTE	central tendency exposure
DA_{event}	absorbed dose per event
DAS	Delivery of Analytical Services
DDT	dichloro diphenyl trichloroethane
Delta-BHC	delta-hexachlorocyclohexane
DNAPL	dense non-aqueous phase liquid
DPT	direct push technology
Eh	Observed Electrode Potential
ELI	Environmental Laboratories, Inc.
EM	electromagnetic
EPA	United States Environmental Protection Agency
EPC	exposure point concentration

ACRONYMS AND ABBREVIATIONS (cont.)

ERA	Ecological Risk Assessment
ESI	Expanded Site Inspection
°F	degrees Fahrenheit
FAWQC	Federal Ambient Water Quality Criteria
FEMA	Federal Emergency Management Agency
FID	flame ionization detector
f_{oc}	fraction of organic carbon
Foster Wheeler	Foster Wheeler Environmental Corporation
FS	Feasibility Study
ft	feet
ft^2	square feet
ft^3/hr	cubic feet per hour
g/cm^3	grams per cubic centimeter
GB	State of Connecticut classification for non-drinking water sources
GI	gastrointestinal
gpm	gallons per minute
GPR	Ground Penetrating Radar
GPS	Global Positioning System
HEAST	EPA's Health Effects Assessment Summary Tables
HHRA	human health risk assessment
HI	hazard index
HNUS	Halliburton NUS Corporation
hr	hours
HQ	Hazard Quotient
I-95	Interstate 95
ILCR	incremental lifetime cancer risk
IRIS	EPA's Integrated Risk Information System Database
JE	Johnson and Ettinger
K_d	Adsorption coefficient
kg	kilograms
K_{OC}	Adsorption partitioning coefficient
K_p	Permeability Coefficient from water through skin
K_{OW}	Octanol-water partitioning coefficient
L or l	liters
L/cm^3	liters per cubic centimeters
L/m^3	liters per cubic meter
LNAPL	light non-aqueous phase liquid
LOAEL	lowest-observed-adverse-effect-level
m	meters
m^3	cubic meters
m^3/day	cubic meters per day
μ	micro (prefix)
μg/dL	micrograms per deciliter
μg/l or μg/L	micrograms per liter
$\mu g/m^3$	micrograms per cubic meter
MEP	Multiple Extraction Procedure
mg/kg	milligrams per kilogram

ACRONYMS AND ABBREVIATIONS (cont.)

mg/kg/day	milligrams per kilogram per day
mg/l or mg/L	milligrams per liter
mg/m^3	milligrams per cubic meter
mg/μg	milligrams per microgram
ml/g	milliliters per gram
mph	miles per hour
MSL or msl	mean sea level
mV	millivolts
NAPL	non-aqueous phase liquid
NCP	National Oil and Hazardous Substances Pollution Contingency Plan
NERL	New England Regional Laboratory
NESHAP	National Emission Standards for Hazardous Air Pollutants
NGVD	National Geodetic Vertical Datum
NOAA	National Oceanic and Atmospheric Administration
NOAEL	no-observed-adverse-effect-level
NPL	National Priorities List
NTCRA	Non-Time Critical Removal Action
ORNL	Oakridge National Laboratory
ORP	Oxygen Reduction Potential
OSWER	(EPA's) Office of Solid Waste and Emergency Response
OU	Operable Unit
PAH	polynuclear aromatic hydrocarbon
PC	Post closure
PCB	polychlorinated biphenyl
PCDD	polychlorinated dibenzo dioxins
PCDF	polychlorinated dibenzo furans
PCE	tetrachloroethene
pg/l or pg/L	picograms per liter
pH	hydrogen-ion concentration
PID	photoionization detector
PMC	Pollutant Mobility Criteria
ppb	parts per billion
ppb/V	parts per billion by volume
ppm	parts per million
ppt	parts per trillion
PRG	Preliminary Remediation Goal
PVC	polyvinyl chloride
RAC	Response Action Contract
RAGS HHEM	Risk Assessment Guidance for Superfund, Human Health Evaluation Manual
Raymark Facility	Raymark Industries, Inc. Facility
RBC	EPA Region III Risk-Based Concentration
RBCA	Risk Based Corrective Action Process
RCRA	Resource Conservation and Recovery Act
Removal Action	Action taken by EPA to address immediate danger to public health and the environment
RfC	Reference Concentration
RfD	Reference Dose

ACRONYMS AND ABBREVIATIONS (cont.)

RI	Remedial Investigation
RI/FS	Remedial Investigation/Feasibility Study
RME	reasonable maximum exposure
ROD	(EPA's) Record of Decision. Documents the selection of a cost-effective Superfund remedy.
RSRs	State of Connecticut Remediation Standard Regulations
RVC	residential volatilization criteria
SARA	Superfund Amendments and Reauthorization Act of 1986. Amended CERCLA. Also known as the Superfund law.
SAS	Special Analytical Services
SPLP	Synthetic Precipitation Leaching Procedure
SPR	single-point resistance
SVOC	semivolatile organic compound
SWP/SWPC	Surface Water Protection Criteria
TAC	Target Air Concentration
TAL	(CLP) Target Analyte List for Inorganics
TCE	trichloroethene
TCL	(CLP) Target Compound List for Organics
TCLP	Toxicity Characteristic Leaching Procedure
TEF	Toxicity Equivalence Factor
TEQ	Toxicity Equivalency
TERC	Total Environmental Restoration Contract
TOC	total organic carbon
TtNUS	Tetra Tech NUS, Inc.
UCL	upper confidence limit
USDOI	United States Department of the Interior
USGS	United States Geological Survey
VLF	very low frequency
VOC	volatile organic compound
VSP	Vertical Sampling Program
Weston	Roy F. Weston, Inc.
WQC	State of Connecticut water quality standards
XRF	X-Ray fluorescence
Yr	year

E.0 EXECUTIVE SUMMARY

This Remedial Investigation (RI) report for the Raymark Industries, Inc. Superfund Site – Operable Unit (OU) 2 - Groundwater has been prepared by Tetra Tech NUS, Inc. (TtNUS), as authorized by the Unite States Environmental Protection Agency (EPA) under Work Assignment No. 126-RICO-01H3, Contract No. 68-W6-0045. See Figure 1-1 for the OU2 study area (study area) location. This RI report presents the findings of groundwater field investigations undertaken over a 9-year period (1994-2003), with one limited data set from 1984 for a property located north of the former Raymark Facility (see Figure 2-1), spill information from the former Raymark Facility itself from the 1980s, and dense non-aqueous phase liquid (DNAPL) data from 1993, also from the former Raymark Facility. The results of these field investigations have identified a groundwater plume from the former Raymark Facility. A Feasibility Study (FS) identifying the groundwater cleanup options being considered will be issued as a separate document.

This document incorporates information collected in 2002 and 2003 in order to fill data gaps identified in the November 2000 OU2 Draft Final RI. Tasks completed included: 1) groundwater and soil gas profiling to assist in locating soil gas and groundwater monitoring wells and further identification of “at risk” homes and buildings; 2) well repair and redevelopment; 3) installation of soil gas and groundwater monitoring wells in order to define bedrock topography, groundwater flow, and the interaction between soil gas and shallow groundwater; 4) geophysical studies to identify suspected fracture zones; 5) a sampling round of all new and previously existing monitoring wells; 6) a seepage study in Ferry Creek upstream of the original 1999 study; 7) a tidal study to determine impacts of the Housatonic River on the northern portion of the study area; and 8) an aquifer test to evaluate hydraulic conductivity and bedrock/groundwater interactions.

This RI was developed for groundwater contamination and does not include assessment of other media, such as sediments, surface water, or soils. The interpretation of the data and information compiled for this RI indicates that:

- Raymark Facility waste was disposed of on the former Raymark Facility and throughout properties located within and in close proximity to the OU2 study area. This fill material has the potential to leach metals into the shallow groundwater, as evidenced by the synthetic precipitation leaching procedure (SPLP) data. The SPLP analysis indicates the potential for contamination to leach from soils into the groundwater; however, most of the properties with

Raymark waste do not have corresponding groundwater data that conclusively indicate that Raymark waste has impacted groundwater quality. Refer to Appendix B for SPLP analytical results within the OU2 study area.

- Volatile organic compounds (VOCs) are present in both shallow and deep fill materials at the former Raymark Facility (HNUS 1995). VOCs were identified within the fill materials, soils, and groundwater at the former Raymark Facility on a frequent basis. However, given the irregular filling pattern, there is not always a direct correlation between VOCs in soil and groundwater samples. VOCs were used at the former Raymark Facility and, based on filling on-site, waste management practices, and groundwater sampling results, VOCs were released into the groundwater.

- Analysis of groundwater samples reveals there is contamination in Ferry Creek from discharge of contaminated groundwater from the former Raymark Facility and other sources. Although groundwater contamination is ubiquitous through most of the study area, the groundwater contaminants and concentrations are not distributed evenly. Six discrete source areas for groundwater contamination were identified and analyzed for fate and transport. One area (Source E) was associated with 540 Longbrook Avenue, which is upgradient from the former Raymark Facility; one area (Source F) was associated with Raymark waste disposed of at the 600 East Broadway property; and the remaining four areas (Sources A, B, C, and D) were on the former Raymark Facility itself. This report documents that the ultimate fate of the contaminant plumes from these sources is to enter Ferry Creek or the Housatonic River. See Figure 5-1 for the location of these source areas.

- Modeling results suggest that 1) the concentration of 1,1-dichloroethene (1,1-DCE) and trichloroethene (TCE) entering Ferry Creek and the Housatonic River will remain at their present level until their sources are contained, destroyed, or depleted, and 2) the concentration of all modeled inorganic contamination discharging to Ferry Creek is expected to increase over time.

- The Human Health Risk Assessment (HHRA) estimates the potential for human health risk from exposures to groundwater in the study area. Groundwater in the study area and surrounding areas is not used as a drinking water source, therefore the primary pathways of potential concern for human health exposures are inhalation of volatiles present in indoor air

due to volatilization of groundwater contaminants through building foundations, direct contact with surface water contamination resulting from migration of groundwater to Ferry Creek, and ingestion of shellfish (oysters), which may be contaminated as a result of migration of groundwater to Ferry Creek and subsequent uptake of surface water contaminants by oysters.

- Risks to human health from inhalation of indoor air were evaluated through both direct measurements of indoor air and indoor air concentrations modeled from groundwater contamination data. The conclusion of both methods is that potential risk to human health from inhalation of indoor air is of concern for both residents and industrial/commercial workers. The results of the risk evaluation of indoor air data indicate risk at levels of concern from some contaminants present in groundwater, but also risks at levels of concern from other contaminants not detected in groundwater at levels expected to impact indoor air. It is presumed that groundwater is the source of some indoor air contaminants; however, the source for indoor air contaminants that were not detected in groundwater is unclear. The risk from volatilization of contaminants present in groundwater has decreased with the installation of sub-slab ventilation systems at over 100 homes located within the study area.

- Based on predicted models of groundwater contamination reaching Ferry Creek, surface water exposures are not expected to present a risk for future recreational users of Ferry Creek.

- Potential future human exposures through consumption of oysters were evaluated qualitatively. Predicted future oyster contamination indicates that potential future risks from human consumption of oysters may be higher than 10^{-6}.

- Recent surface water sampling indicates a potential risk for sediment and surface water-dwelling organisms that live in the study area.

- Predicted future surface water concentrations of arsenic, zinc, and 1,1-DCE in Ferry Creek and TCE in the Housatonic River may pose negligible risks to aquatic receptors.

1.0 INTRODUCTION

This Remedial Investigation (RI) Report (Report) evaluates the nature and extent of contamination in the groundwater resulting from past disposal practices at the Raymark Industries, Inc. Facility (former Raymark Facility or Facility), located in Stratford, Fairfield County, Connecticut (CT). The United States Environmental Protection Agency (EPA) has designated groundwater contamination associated with the former Raymark Facility as Operable Unit No. 2 (OU2) (see Figure 1-1). Tetra Tech NUS, Inc. (TtNUS) prepared this RI Report for the EPA under a Response Action Contract (RAC), Contract No. 68-W6-0045, Work Assignment No. 126-RICO-01H3, to fulfill the requirements for Raymark - OU2 - Groundwater. This RI Report was developed based on the approved Draft Work Plan, February 2002, and subsequent Amendments.

As requested by EPA, this OU2 RI Report was prepared in accordance with the *Interim Final Guidance for Conducting Remedial Investigations and Feasibility Studies under CERCLA* (EPA 1988) and is consistent with the requirements of the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) of 1980, as amended by the Superfund Amendments and Reauthorization Act (SARA); and the National Oil and Hazardous Substances Pollution Contingency Plan (NCP).

1.1 Purpose of Report

This Report documents the investigations performed; and evaluates the nature and extent of groundwater contamination, public health and environmental risks, and the movement of contaminants associated with the former Raymark Facility (Figure 1-2 identifies the OU2 study area). The purpose of this Report is to provide the documentation necessary to support a Feasibility Study (FS) and the selection of a remedy in a Record of Decision (ROD). The overall objectives of this RI Report are to:

- Compile and evaluate available data needed to characterize the OU2 study area conditions pertinent to groundwater contamination, and to determine the nature and extent of contamination in the groundwater within the study area;

- Evaluate the risks to human health and the environment posed by contaminated groundwater within the OU2 study area; and

- Serve as the data resource for developing, screening, and evaluating a range of potential remedial alternatives that will address the groundwater contamination within the OU2 study area and support the FS.

1.2 Report Organization

This OU2 RI Report is comprised of three volumes. Volume I presents the text and discussion of investigation activities, results, interpretations, and references. Volume I also includes the tables and figures referenced in the text. Volumes II and III present the appendices. Appendix A contains the boring logs; Appendix B is comprised of a disk that contains all the analytical data used to produce this RI Report; Appendix C is the Hydraulic Assessment prepared by the United States Army Corps of Engineers (ACOE) and flood insurance maps published by the Federal Emergency Management Agency (FEMA); Appendix D presents the details on the geological investigation studies, including pump tests and tidal studies performed in 2003; Appendix E is the backup information for the groundwater modeling; and Appendix F provides the backup tables and calculations for the Human Health Risk Assessment.

This RI Report is organized as follows:

- Section 1.0, *Introduction*, discusses the purpose and scope of the RI, summarizes the background and history of the Raymark Facility, and describes the OU2 study area.

- Section 2.0, *Study Area Investigations*, presents a summary of the field investigation activities conducted within the OU2 study area.

- Section 3.0, *Physical Characteristics of the Study Area*, presents descriptions of surface features and land use, geology, surface water, hydrogeology, soil gas, and climate meteorology.

- Section 4.0, *Nature and Extent of Contamination*, discusses the potential sources, background locations, contaminant presence, and contaminant distribution within the groundwater, indoor air, and associated soil gas in the OU2 study area.

- Section 5.0, *Contaminant Fate and Transport*, presents an interpretation of potential contaminant migration in groundwater.

- Section 6.0, *Human Health Risk Assessment*, includes the identification of human receptors and exposure pathways, selection of contaminants of potential concern (COPCs), and a discussion of the human health effects associated with the COPCs. The results of the evaluation are used to characterize the human health risk.

- Section 7.0, *Ecological Risk Assessment*, presents a summary of the environmental setting and identifies areas of potential ecological concern. The results are used to characterize ecological risk.

- Section 8.0, *Summary and Conclusions*, summarizes the nature and extent of contamination, the fate and transport of contaminants, and the risks to human health and the environment associated with the groundwater in the OU2 study area.

1.3 Study Area Background

This section summarizes the history of the Raymark Facility, describes the study area, and identifies other activities associated with the former Raymark Facility. Refer to the *OU1 Final Remedial Investigation Report* (HNUS 1995) for further details on Facility operating history, environmental activities, permits, and compliance history.

1.3.1 History of the Raymark Facility

The Raymark Facility, formerly named Raybestos - Manhattan Company, was located at 75 East Main Street in Stratford, Fairfield County, CT, at latitude 41°12′02.5″N and longitude 73°07′14.0″W (see Figure 1-1). The Raymark Facility operated from 1919 until 1989, when the plant was shut down and permanently closed; however the property was not cleaned up until

1997. Subsequent to the completion of an RI/FS, EPA designated the Facility as Operable Unit No. 1 (OU1). In 1996 and 1997, as part of the property cleanup activities, the Facility buildings were demolished and a permanent cap was placed over the contaminated areas on the property. Based on Stratford tax map information, the Facility occupied 33.4 acres. Raymark manufactured friction materials containing asbestos and non-asbestos components, metals, phenol-formaldehyde resins, and various adhesives. Primary products were gasket material, sheet packing, and friction materials including clutch facings, transmission plates, and brake linings. As a result of these activities, groundwater at the Facility became contaminated with volatile organic compounds (VOCs), semi-volatile organic compounds (SVOCs), and metals.

During the Facility's 70 years of operation, it was common practice to dispose of manufacturing waste as "fill" material both at the Raymark Facility, as well as at various locations in Stratford. The manufacturing wastes from different plant operations were used to fill low-lying areas on-site to create additional space for Facility expansion. Based on aerial photographs and reported knowledge of site activities, on-site disposal occurred between 1919 and 1984, and progressed essentially from north to south, across the Raymark Facility. As a result of the disposal of these manufacturing wastes on the property, soils at the Facility became contaminated primarily with asbestos, lead, copper, and polychlorinated biphenyls (PCBs). New buildings and parking areas were constructed over these filled areas as the manufacturing Facility expanded. Raymark also offered manufacturing wastes as "free fill" to employees, residents, and the town.

The Raymark Facility was underlain by an extensive manmade drainage system network. Water and wastes from the manufacturing operations were collected and diverted into the Facility drainage system, which also collected stormwater runoff. These liquids were transported through the drainage system network, mixed with lagoon wastewaters, and discharged to Ferry Creek.

During peak operations at the Raymark Facility, approximately two million gallons of water were used for plant processes each day. Municipal water was used for both contact and non-contact cooling water. During the 1970s, to supplement this source, Raymark installed an additional on-site supply well. The well, located in the northeastern corner of the Facility, was used for non-contact cooling water. Facility water was recirculated, with some percentage reinjected into the on-site well; the remaining water and municipal water were discharged through the Facility's

drainage system. Wastewater from Facility operations was collected and discharged to a series of four settling lagoons located in the southwestern corner of the Facility, and along the southern property boundary near Longbrook Avenue and the Barnum Avenue Cutoff. The wastewater consisted of wastewater from the acid treatment plant, wet dust collection, paper making processes, non-contact cooling water, and wastewater from the solvent recovery plant operations. The lagoons also received stormwater drainage and surface water runoff.

Solids were allowed to settle in Lagoon Nos. 1, 2, and 3 prior to the discharge of clarified wastewater and unsettled solids to Lagoon No. 4. Lagoon No. 4 discharged into Ferry Creek. Discharge of wastewater to Lagoon Nos. 1, 2, and 3 ceased in 1984. These lagoons were closed in December 1992 and January 1993. During the fall of 1994, stormwater drainage that exited the Raymark Facility through Lagoon No. 4 was diverted around this lagoon and connected directly to the storm sewer. The storm sewer ultimately discharged to Ferry Creek. Lagoon No. 4 was closed in early 1995, prior to the placement of the permanent cap over the property.

During the operation of the lagoons, the settled material in the lagoons was periodically removed by dredging. During the Facility's 70 years of operation, it was common practice to dispose of both this dredged lagoon waste and other manufacturing waste as fill material (referred to as waste in this Report) both at the Raymark Facility and at various locations in Stratford. Several of the locations that received waste are included within the area designated as the OU2 study area for this RI Report (Figure 1-2).

EPA listed the former Raymark Facility and properties that contain waste from the Raymark Facility on the National Priorities List (NPL) on January 18, 1994. Listing on the NPL authorizes the expenditure of CERCLA funds. The listing was final on April 25, 1995. This OU2 study area is included under this listing.

A number of the non-Facility locations where Raymark waste was disposed were contaminated with asbestos, lead, copper, and/or PCBs at levels that posed a potential threat to public health. To abate the potential health threat to residential properties, residential locations were cleaned up under CERCLA time-critical removal actions from 1993 to 1996. The excavated material from these residential locations was placed under a permanent cap at the Raymark Facility.

Raymark waste identified at one municipal property, Wooster Middle School, was also excavated, stored, and ultimately placed under a permanent cap at the Raymark Facility.

A substantial number of field investigations relating to soil, sediment, surface water, biota, and groundwater have been conducted at the former Raymark Facility and its environs. A discussion of investigations that are pertinent to this OU2 study area is included in Section 2.0.

1.3.2 Study Area Description and Setting

Contaminated areas associated with the Raymark Superfund Site have been divided into nine operable units. EPA created these nine operable units (OUs) to help manage the cleanup process. The nine operable units are as follows:

- OU1 Former Raymark Facility
- OU2 Groundwater
- OU3 Upper Ferry Creek and Surrounding Wetlands
- OU4 Raybestos Memorial Field
- OU5 Shore Road Area
- OU6 Additional Properties Study Area
- OU7 Lower Ferry Creek, Selby Pond, and Housatonic River Wetlands
- OU8 Beacon Point Area and Elm Street Wetlands
- OU9 Short Beach Park and Stratford Landfill

The area identified as the study area for this OU2 RI encompasses approximately 500 acres (see Figures 1-1 and 1-2). Approximately half of these acres are commercial in nature, containing highways and business activities; the remaining area includes residences and water bodies. This study area includes businesses that have in the past handled, or continue to handle, hazardous materials. This OU2 RI report focuses only on those groundwater contaminants that appear to be attributable to the former Raymark Facility. Contamination from non-Raymark sources has not been evaluated in this Report.

Background wells that are upgradient from the former Raymark Facility contain contaminants. For this reason, background wells will be referred to as “upgradient/cross-gradient wells” in this

report. The origin of the upgradient contaminants could be from other sources, from the industrialized nature of Stratford, or from Raymark wastes that may have been deposited in this area and impacted the wells. For purposes of this RI, it is assumed that the contaminants in the upgradient/cross-gradient wells are not from the former Raymark Facility (OU1). Most of the 500-acre OU2 study area is downgradient of the former Raymark Facility and includes areas that may have been affected by wastewater discharge, surface water runoff, direct deposition of manufacturing waste, and groundwater contaminant migration from the former Raymark Facility.

The OU2 study area is bounded by the Housatonic River to the east; just above Selby Pond to the south; Interstate-95 (I-95)/Blakeman Place to the southwest; Patterson Avenue to the northwest; and the East Main Street/Dock Shopping Center to the north. The study area boundaries were set based on current monitoring well data that delineate groundwater flow directions from the former Raymark Facility, and groundwater contamination attributable to the Facility. Areas showing little or no groundwater contamination attributable to the former Raymark Facility were excluded from the study area. The OU2 indoor air area of interest is downgradient of the facility, within the groundwater study area. Properties in this Report are indicated by street address where possible. This report focuses only on contamination of the groundwater within the study area.

The drainage network throughout the OU2 study area is extensive. It appears that a portion of the drainage network connects to an outfall from the former Raymark Facility, which discharges to Ferry Creek. The CT Department of Transportation (CTDOT) has made improvements to the drainage network from U.S. Route 1 to Ferry Creek. These plans indicate that stormwater drainage from the former Raymark Facility, exiting from the southeast corner of the former Facility along Barnum Avenue, connects to the drainage network and discharges into Ferry Creek (CTDOT 1995).

In addition, a sanitary sewer interceptor runs through the study area from north to south. The sanitary sewer interceptor is located at the northwest corner of the Facility; it intersects the lower west portion of the former Raymark Facility and continues south crossing the Barnum Avenue cutoff and intersecting the former Synthetic Products Facility, and exits out towards West Ferry Boulevard. The sanitary sewer interceptor consists of a 15-foot sewer easement filled with gravel. Refer to Figure 1-2 (Facility Description) from the OU1 Final Remedial

Investigation Report (HNUS 1995) for location of the sanitary sewer interceptor through the former Raymark Facility. Refer to Figure 3 (Site Drainage Map) from the *Closure Plan for Synthetic Products Inc.* (HRP Associates 1997) for location of the line through the former Synthetic Products Facility.

1.3.3 Other On-Going Activities

Activities undertaken in the vicinity of the OU2 study area that are related to the investigations conducted to support this RI include:

- Raymark Facility Closure - The former Raymark Facility, designated by EPA as Operable Unit No. 1 (OU1), has been permanently capped by EPA under the ACOE Total Environmental Restoration Contract (TERC) (Foster Wheeler 1996). This property has been sold to a private entity that built a shopping center on the property. A soil vapor extraction system is in place to capture the bulk of the toluene remaining under the cap, and a dense non-aqueous phase liquid (DNAPL) extraction system is removing free non-aqueous phase liquid (NAPL) (predominantly 1,1,1-trichloroethane [TCA]), which is collected in the sump portion of the wells. Operation and maintenance activities are being conducted by the CT Department of Environmental Protection (CTDEP). A Five-Year Review report was issued for OU1 in 2000 (TtNUS 2000a). Another five-year review will be performed during 2005.

- Residential area downgradient of the former Raymark Facility - After analyzing data from soil gas, groundwater, and indoor air sampling events from homes within the residential area downgradient of the former Raymark Facility, the Connecticut Department of Public Health and Addiction Services (CT DPHAS), with the concurrence of the Agency for Toxic Substances and Disease Registry (ATSDR), recommended that subslab ventilation systems be considered as a prudent public health action for each residential home located above the groundwater plume contaminated with VOCs from the former Raymark Facility. The EPA offered to fund the installation of sub-slab ventilation systems in any residential home within this area. The CTDEP assumed responsibility for overseeing the installation and for the operation and maintenance of the sub-slab ventilation systems.

- Ferry Creek Area Activities – The Ferry Creek area has been designated by EPA as Operable Unit No. 3 (OU3), Operable Unit No. 7 (OU7), and Operable Unit No. 8 (OU8). All three areas are included in the OU3 RI Report. Area I consists of Upper Ferry Creek and associated wetlands, and a Final RI has been completed (TtNUS 1999a). The Draft Final RIs for Area II, Lower Ferry Creek, Selby Pond, and the Housatonic River wetlands (referred to as OU7) and Area III, Beacon Point Boat Launch Area and Elm Street wetlands (referred to as OU8), have been released (TtNUS 2000b and 2000d). A comprehensive FS for all three areas of OU3 and OU4, OU5, and OU6 will be issued pending completion of the evaluation of other Raymark operable units.

- Raybestos Memorial Ballfield Activities – The ballfield received Raymark wastes, and was designated as Operable Unit No. 4 (OU4) by EPA. EPA has issued the Final RI report for OU4, (TtNUS 1999b). A comprehensive FS for OU3, OU4, OU5, and OU6 will be issued pending completion of the evaluation of other Raymark operable units.

- Shore Road Area – The Shore Road area has been designated as Operable Unit No. 5 (OU5) by EPA. EPA has issued an Engineering Evaluation/Cost Analysis Report (TtNUS 1999c), and two Addenda to that Report (TtNUS 1999d and 1999e). EPA issued an Action Memorandum in September 1999 that described the cleanup remedy. EPA received significant public comment on the selected remedy and in January 2000 decided to delay final cleanup until decisions on the cleanup of the other operable units have been made. An interim action was completed on the site in July 2000. This OU will be included in the comprehensive FS for OU3, OU4, OU5, and OU6.

- Additional Properties – Various properties within Stratford have been designated as Operable Unit No. 6 (OU6) by EPA. EPA has identified several properties that have received Raymark waste primarily through disposal activities. EPA has released the Final OU6 RI (TtNUS 2004). A comprehensive FS for OU3, OU4, OU5, and OU6 will be issued pending completion of the evaluation of other Raymark operable units.

- Short Beach Park and Stratford Landfill Activities – EPA has designated these parcels as Operable Unit No. 9 (OU9). A field investigation of whether these areas received Raymark waste was completed and an RI and FS is expected to be released in 2005.

2.0 FIELD INVESTIGATIONS

Section 2.0 presents a brief description of the investigations performed to evaluate and characterize the nature and extent of contamination associated with the former Raymark Facility and environs. Investigations were performed by TtNUS (also operating as Brown & Root Environmental (B&RE) and Halliburton NUS Corporation (HNUS)); Environmental Laboratories, Inc. (ELI); Roy F. Weston, Inc. (Weston); Foster Wheeler Environmental Corporation (Foster Wheeler); CTDEP; Connecticut Department of Public Health and Addiction Services (CT DPHAS) under a cooperative agreement with the Agency for Toxic Substances and Disease Registry (ATSDR); and the National Oceanic and Atmospheric Administration (NOAA). Table 2-1 summarizes information collected from these investigations, which is used in this Report as appropriate. Many dates overlap because contractors were hired by a variety of entities (EPA, State of Connecticut, ACOE) to perform specific tasks. Information collected during these investigations was used to meet the Remedial Investigation objectives presented in Section 1.1. Brief descriptions of previous investigations, which have been used in the development of this Report are presented below in chronological order.

2.1 Sediment Sampling (1992)

In 1992, Weston collected sediment samples as part of an EPA Site Inspection for the Raymark Facility. Fifteen samples were collected along Ferry Creek and the Housatonic River. Samples were submitted to EPA-approved laboratories for analysis of VOCs, SVOCs, PCBs, metals, cyanide, dioxins/furans, and asbestos. Numerous organic and inorganic contaminants were detected at elevated levels. The sampling locations and analytical results are summarized in Roy F. Weston's Final Site Inspection Report (Weston 1993).

2.2 Surface Water Sampling at the Raymark Facility (1993)

Five surface water samples were collected in July 1993 by Raymark Industries, Inc. to characterize both the quantity and quality of drainage discharges into and out of Lagoon No. 4. After installation of the surface stormwater drainage diversion system around Lagoon No. 4, the outlet to this lagoon (Station No. 5) was resampled in October 1993. Samples were submitted for laboratory analysis of VOCs, SVOCs, PCBs, metals, cyanide, sulfide, chlorinated herbicides, organophosporous pesticides, dioxin/furan, and asbestos (ELI 1994). These sampling rounds

confirmed that the Raymark Facility had discharged contaminated materials/water into Ferry Creek. The results from subsequent sampling rounds indicated that similar contaminants were detected both on-site and in the Ferry Creek sediments (HNUS 1995).

2.3 Expanded Site Inspection and Vertical Sampling Program (1993)

Between July and October 1993, soil samples were collected from two properties located near Ferry Boulevard, residential properties on Patterson and Clinton Avenues, properties along Elm Street, and properties along 3rd/4th/5th Avenue as part of the Expanded Site Inspections (ESIs)/Vertical Sampling Program (VSP).

The soil sampling was conducted to provide information regarding the presence or absence of waste, waste characteristics, and extent of contamination. Soil horizons and individual sample collection locations were selected based on EPA recommendations, visual field observations, and data from Ground Penetrating Radar (GPR) interpretations. Samples were collected from soil borings ranging in depth from 0 to 16 feet below ground surface (bgs). The borings were advanced primarily using a Model 8-M Soil Probe Unit developed by Geoprobe Systems. Soil samples collected from various locations and depths at each property were screened for lead, asbestos, and PCBs using EPA-approved screening methods. Approximately 15 percent of the samples were submitted for confirmatory analysis through the EPA Contract Laboratory Program (CLP); these samples were analyzed for Target Compound List (TCL) VOCs, TCL SVOCs, TCL pesticides/PCBs, and target analyte list (TAL) metals. Selected samples were also analyzed under the EPA Special Analytical Services (SAS) program for dioxins/furans, PCBs, Toxicity Characteristic Leaching Procedure (TCLP), and Multiple Extraction Procedure (MEP).

Summary reports for these investigations were completed by Weston in 1995, and were presented in the *Final Technical Memorandum, Compilation of Existing Data, RI/FS, Raymark - Ferry Creek* (B&RE 1997). Field observations, soil boring logs, and analytical results summarized in these reports were used to evaluate the presence, and location of, Raymark-type waste on the various properties investigated. The sample results were used to identify the most contaminated residential properties; these properties have been excavated, and the contaminated material transported to the former Raymark Facility and placed under the cap.

2.4 Phase I Remedial Investigation (1993 - 1995)

TtNUS (as HNUS) conducted an investigation within several areas of Stratford from 1993 through 1995. This investigation was conducted under the Alternative Remedial Contracting Strategy (ARCS), Contract No. 68-W8-0117, Work Assignment No. 42-1LH3. The field investigation included a soil boring and sampling program, monitoring well installation and sampling program, surface water and sediment sampling, GPR survey, topographic survey, and wetlands delineation.

2.4.1 Soil Borings, Test Pits, and Soil Sampling (1994)

Soil borings were advanced on the 600 East Broadway property, Housatonic Avenue property, Housatonic Boat Club properties, and the Vacant Lot on Housatonic Avenue; and four test pits were excavated at 600 East Broadway. Individual boring and test pit locations were selected in the field by EPA based on preliminary GPR survey interpretations and other available information. Twenty-seven soil borings were advanced to depths up to 22 feet bgs using hollow stem auger or rotary methods. Six additional borings were advanced at the Housatonic Boat Club property using a slide-hammer to depths of 4 feet bgs. The borings used for this OU2 RI Report are located on Figure 3-6.

Continuous split-barrel sampling was conducted throughout the advancement of each boring. Representative samples from borings and test pits were screened for asbestos, lead, copper, and PCBs using EPA-approved screening methods. Selected samples were also submitted for confirmatory analysis of TCL VOCs, TCL SVOCs, TCL pesticides/PCBs, TAL metals, and dioxins/furans.

2.4.2 Groundwater Monitoring Well Installation (1994)

From March through May 1994, TtNUS (as HNUS) installed 23 monitoring wells at eight cluster locations and one single well location. These well clusters, designated the 100-series wells, are located at 600 East Broadway and commercial properties along Ferry Boulevard. A single well was also located at the Housatonic Boat Club on Shore Road. This well was destroyed in 2000 during on-site removal activities and replaced in 2003. Additional 100-series wells were installed in 1999 at pre-existing cluster locations. Refer to Figure 2-3 for well locations.

The 100-series wells were installed using drive and wash drilling methods. For each well cluster, the deepest boring was drilled first. Soil samples were collected using split spoons, logged, and field screened with a photoionization detector (PID) or flame ionization detector (FID). Soil samples were sent to an off-site laboratory for analysis of VOCs, SVOCs, pesticides, PCBs, metals and asbestos. The well-screen intervals of the subsequent wells drilled in each cluster were selected based on levels of contamination, using VOC field screening data, and the highest estimated hydraulic conductivity, using the grain-size data.

2.4.3 Ground Penetrating Radar Survey (1994)

A geophysical survey using GPR was performed in portions of the OU2 study area in March 1994 by Hager-Richter Geoscience, Inc., a subcontractor to TtNUS (Hager-Richter 1994). The GPR survey was conducted within three areas: 600 East Broadway, properties along Ferry Boulevard, and the Housatonic Boat Club Property. Twenty-three traverses totaling approximately 9,100 feet of profile were completed. The purpose of the GPR survey was to determine the presence, location, and character of wastes disposed of as fill at each area, including the location of potential buried vessels and subsurface utilities. This information was used to select soil boring and monitoring well locations, as well as to develop estimates of the thickness of Raymark waste on each property. See Figure 2-2 for GPR survey locations.

2.4.4 Topographic Survey/Global Positioning Survey (1994)

A topographic survey was conducted in September and October 1994 by Diversified Technologies Corporation, as a subcontractor to TtNUS. The survey was performed to generate a base map and locate soil borings, test pits, monitoring wells, and GPR survey lines. A survey of the known bedrock outcrops in the area was also conducted at the same time. TtNUS used Global Positioning Survey (GPS) methods to pinpoint surface water and sediment sampling locations during the four sediment/surface water sampling rounds (see Section 2.4.5). GPS was also used to delineate the boundary separating wetland and upland environments.

2.4.5 Surface Water and Sediment Sampling (1994 - 1995)

Four rounds of surface water and sediment sampling were conducted at selected locations to evaluate potential contaminant migration from the former Raymark Facility. In the course of the

four sampling rounds, 140 locations (both within and outside the OU2 groundwater study area) were sampled, including streams, ponds, wet areas, and leachate outbreaks identified by EPA. Surface water samples were collected and submitted to laboratories for analysis of TCL VOCs, TCL SVOCs, TCL pesticides/PCBs, and TAL metals. Field measurements included pH, temperature, specific conductivity, dissolved oxygen, and salinity. Sediment samples were submitted to laboratories for analysis of TCL VOCs, TCL SVOCs, TCL pesticides/PCBs, TAL metals, asbestos, dioxin/furans, total organic carbon (TOC), and grain size. In the fourth sampling round, some sediment samples were also submitted for acid volatile sulfide/simultaneously extractable metals (AVS/SEM) analysis.

A limited salinity survey was conducted along Ferry Creek from Broad Street to the point where Ferry Creek and Long Brook Creek intersect. The study defined the saline/freshwater interface in Ferry Creek. Temperature and salinity were recorded at predetermined locations along Ferry Creek at both high and low tide, and sediment samples were collected at each monitoring location during low tide.

2.4.6 Groundwater Sampling (1994 - 1995)

Four quarterly rounds of groundwater sampling were performed at the twenty-three 100-series wells from May 1994 through March 1995. Samples were collected using EPA Region I Low-Stress/Low-Flow Sampling Methods. Groundwater samples were analyzed for VOCs, SVOCs, pesticides, PCBs, and metals. See Appendix B for sample results from all groundwater sampling rounds in the OU2 study area.

2.4.7 Wetlands Delineation (1994)

A team of wetland specialists comprised of personnel from EPA, United States Fish and Wildlife Service, and HNUS (now TtNUS) delineated the boundary separating wetlands from upland environments using the methodology described in the 1987 United States Army Corps of Engineers Wetland Delineation Manual. The locations studied included Ferry Creek and 600 East Broadway, properties along Housatonic Avenue, the Housatonic Avenue property, the Housatonic Boat Club, Beacon Point Road, and properties along Lockwood Avenue. The wetland boundary was staked and later surveyed by HNUS with GPS.

2.5 Comprehensive Site Investigation Sampling Program (1994 - 1995)

Comprehensive Site Investigation (CSI) reports were prepared by multiple EPA contractors in 1994 and 1995 for properties under investigation as part of the Stratford Superfund Sites program. The purpose of the CSIs was to determine the extent and magnitude of lead, PCB, and asbestos contamination associated with Raymark Facility waste disposal practices in areas of Stratford beyond the Facility. The CSI reports were designed to provide site-specific data necessary to proceed with the Stratford Superfund Sites Removal Action Program. The information contained in the reports was based on the subsurface samples collected during the vertical sampling program described in Section 2.3. This program was terminated prior to the completion of all CSI reports.

Sample locations for properties investigated during the EPA removal program were selected based on a systematic grid approach for each property investigated. Grid intersections were set at 25-foot intervals and sampling was conducted at each grid intersection. Surface soil samples were collected from depths of 0 to 12 inches bgs using a stainless steel trowel. Subsurface soil samples were obtained from depths of 1 to 12 feet bgs using a hand-operated Geoprobe® slide-hammer piston rod apparatus advanced hydraulically using a Terraprobe® truck-mounted unit. Soil samples from each boring were visually classified and logged. Constituents of all soils were characterized using the Burmister soil classification ranges, and soil color was described using Munsell color charts. Samples were composited from 1-foot intervals and screened at the on-site laboratory for asbestos, lead, and PCBs. Approximately 10 percent of the samples were submitted for confirmatory analysis at an off-site laboratory.

Site-specific data for numerous properties have been generated through the CSI program. Final CSI Reports for the properties were completed in 1995, and report sections relevant to OU3 were presented in *the Final Technical Memorandum, Compilation of Existing Data, RI/FS, Raymark - Ferry Creek* (B&RE 1997).

2.6 Removal Action Post-Excavation Program (1994 - 1996)

Site-specific removal actions were performed at residential properties contaminated with Raymark waste based on the results of the CSI sampling program discussed in Section 2.6. Upon completion of the removal at each property, samples were collected to ensure that the

contaminated materials were removed. Removal action soil sample locations were selected based on a systematic grid approach for each property excavated. Grid intersections were set at 15-foot intervals; samples were collected at depths of 0 to 3 inches from each exposed trench wall, base, and perimeter using a pre-cleaned steel shovel or hand trowel. Samples were composited from each exposed surface and screened at the on-site laboratory for asbestos, lead, and PCBs. Approximately 10 percent of the samples were submitted for confirmatory analysis at an off-site laboratory. Once the contaminated materials were removed, the areas were backfilled with clean materials and seeded.

Post-Excavation Record Plans were prepared for these properties. The Post-Excavation Record Plans documented the soil removal action clean-up activities conducted at each property and showed that the established clean-up criteria had been achieved.

2.7 Phase 1 OU2 Groundwater Remedial Investigation (1996 - 1997)

Five activities related to the OU2 Study Area were performed by TtNUS (as B&RE) under the Phase I investigation and are described below. The five tasks included monitoring well installation (1996), groundwater profiling (1997), monitoring well installation and groundwater sampling (1997), other groundwater evaluation investigations (1997), and groundwater sampling (1997). The objective of the investigation was to determine the nature and extent of groundwater contamination.

2.7.1 Monitoring Well Installation (1996)

In March 1996, under EPA direction and approval, TtNUS installed seven 200-series monitoring wells, MW-201D through MW-207D, at seven locations due east of the former Raymark Facility. Refer to Figure 2-3 for the well locations. The wells were installed in order to evaluate the extent of groundwater contamination. Prior investigations had indicated groundwater was flowing east from the former Raymark Facility toward the Housatonic River. The borings for the wells were advanced using the drive and wash drilling method. Refer to Appendix A for boring logs and well construction logs. Soil samples were collected and analyzed for VOCs, SVOCs, pesticides, and metals by the EPA New England Regional Laboratory (NERL) in August 1996.

2.7.2 Groundwater Profiling (1997)

From July through September 1997, TtNUS conducted horizontal and vertical groundwater profiling. Small diameter well points were advanced using vibrating direct push technology (DPT) at 77 locations north, south, and east of the former Raymark Facility (OU1). This method obtains samples from groundwater, not subsurface soils. The objective of this investigation was to determine the extent of groundwater contamination down gradient and cross gradient of the former Raymark Facility, and to aid in the selection of locations for permanent groundwater monitoring wells. Refer to Appendix D for the DPT sample locations and Appendix B for sample results from all groundwater sampling rounds in the OU2 study area.

During the installation of the well points, groundwater samples were collected using a peristaltic pump every 10 feet and at the point of refusal. Samples were collected and sent off site for rapid turnaround analysis of 1,1,1-trichoroethane, 1,1-dichoroethane, 1,1-dichoroethene, vinyl chloride, toluene, benzene, ethylbenzene, and xylene. Approximately 10 percent of all the samples collected were sent for confirmatory EPA CLP analysis. Samples under the CLP were analyzed for VOCs and metals. The well points were abandoned after sample collection by filling with a bentonite slurry mixture.

The groundwater profiling confirmed the highly variable bedrock topography. Both the historical groundwater sampling and groundwater profiling identified the general extent of the VOC contamination.

2.7.3 Monitoring Well Installation (1997)

During September and October 1997, TtNUS constructed 30 monitoring wells, also designated as the 200-series wells, at 11 clusters using the Vibra-Sonic drilling method. Additional 200 series wells were later installed in 1999 (201B, 208B, and 216DB) and 2002 (202S, 203S, 207B, 210B, 213DB, 214B, 215DB, and 217S). In addition, wells were added to single 100-series wells installed earlier to form clusters, which would fill data gaps identified during previous groundwater sampling. Refer to Figure 2-1 for the monitoring well locations, and Appendix A for well construction logs.

Soil samples from the deepest boring at each well cluster were logged, field screened with a PID, and sent to a laboratory for VOC analysis. Subsequent overburden well screen intervals were selected based on maximum VOC concentrations noted or direct observation of contamination. If VOCs were not detected by the PID or in the laboratory screening analysis, then zones with the highest estimated hydraulic conductivity (based on visual observations of the grain-size of the subsurface soils) were selected.

2.7.4 Other Groundwater Evaluation Investigations (1997)

During August 1997, TtNUS installed three piezometers in upper Ferry Creek. The piezometer screens were set at intervals from 11 feet to 13 feet bgs. See Figure 2-3 for piezometer locations. The water levels in the piezometers were compared to water level data at nearby wells; the piezometers were sampled later in the Phase I groundwater round. See Section 2.7.5 for details.

During November 1997, TtNUS observed the installation and sampling of eight wells on the Dock property, located adjacent to and east-northeast of the former Raymark Facility. The work was performed, based on an agreement worked out with EPA, by a contractor hired by the property owner.

2.7.5 Groundwater Sampling (1997)

TtNUS conducted the final Phase I activity during November and December of 1997. The work consisted of one round of groundwater sampling from selected monitoring wells. Groundwater samples were collected from the following wells using the EPA Region I low-stress/low flow sampling method:

- 100-Series wells (except 104B, installed in 1999)
- 200-Series wells (except 201B, 208B, and 216DB, installed in 1999; and 202S, 203S, 207B, 210B, 213DB, 214B, 215DB, and 217S installed in 2002)
- Three piezometers (FCP 1-3)
- All Dock wells (MW-1 cluster, MW-2, MW-3, and MW-4 cluster).
- Selected wells at 540 Longbrook Avenue (MW-1-10, BR-1&2, MW-X, Y, Z, and PZ-12, 13, 14)

- Five wells at 375 Barnum Avenue (CRA-2D, 4S, 6D, 6S, 8)
- OU1 Post Closure (PC) wells sampled by the ACOE

Groundwater samples from this phase were analyzed for VOCs, SVOCs, pesticides/PCBs, metals and/or water quality parameters including chloride, sulfate, alkalinity, and nitrate/nitrite. Refer to Appendix B for a complete list of the sample results in the OU2 study area.

2.8 Ecological Risk Assessment (1996 - 1998)

An Ecological Risk Assessment (ERA) report was prepared for EPA Region I by NOAA and its contractor (NOAA 1998). This assessment addressed the risks to ecological receptors posed by contaminants present in Ferry Creek, portions of the Housatonic River, and associated wetlands. The report is presented in its entirety in Appendix D of the Final Area I Remedial Investigation, Raymark-Ferry Creek OU3 (TtNUS 1999a).

In addition, supplements to the ERA report were prepared for support of OU3, Areas II and III. These are presented in their entirety in Appendix D of the Raymark – Ferry Creek OU3, Area II RI (TtNUS 2000b) and in Appendix D of the Raymark – Ferry Creek OU3, Area III RI (TtNUS 2000d).

2.9 Phase II Commercial Properties Site Investigation (1997)

Following a review of all the data from 1992 through 1996, TtNUS (as B&RE) identified data gaps. Data gaps were identified for each area, except for the Selby Pond site (Area F), which had been investigated previously to determine the need for a non-time critical removal action (NTCRA) (B&RE 1997). Field investigations and sample collection were conducted by TtNUS during July and August 1997. Field activities included advancing soil borings and collecting soil samples, and collecting surficial soil and sediment samples.

Approximately 35 soil borings were advanced to a depth of 16 feet bgs. The intent was to advance the boring until “natural” soil was encountered. At the direction of EPA, no borings were advanced to depths greater than 16 feet bgs. All soil borings used in this RI are located on Figure 3-6.

Samples were collected using split-barrels throughout the advancement of each boring, and soil samples were field screened using a PID or FID. Based on these field screening results, selected samples were sent for laboratory analysis of VOCs. Soils from each sampled interval were sent to the Connecticut Department of Health laboratory for analysis of asbestos. Soil samples were also sent to an off-site laboratory for screening of lead and copper using x-ray fluorescence (XRF). Based on the XRF screening results, approximately two samples were selected from each borehole for confirmation analysis. Analyses included TCL VOCs, TCL SVOCs, TAL metals, dioxin/furans, and/or TCL pesticides/PCBs (plus Aroclor 1262 and 1268). Selected soil samples were also analyzed for synthetic precipitation leaching procedure (SPLP) metals, based on the amount of soil recovered from the sampled interval, direction from EPA in the field, and the XRF field screening results.

2.10 Phase II Groundwater Remedial Investigation (1998 – 1999)

Phase II was conducted from November 1998 through February 1999. The tasks completed include: monitoring well installation and sampling (1998/1999), GPR and electromagnetic survey (EM) in 1998; hydraulic conductivity tests (1998/1999); tidal study (1999); and a seepage study in Ferry Creek (1999).

2.10.1 Monitoring Well Installation (1998)

TtNUS constructed 34 wells at 13 clusters (300-series), and four wells at two clusters (400-series) using the drive and wash drilling method. The 300-series wells were installed throughout the OU2 study area. The 400-series wells were installed at the former Raybestos Memorial Ballfield (OU4) located north of the former Raymark Facility. Locations of well clusters were selected based on a review of existing data and the results of the seismic refraction survey, described below. The objective was to intentionally target the deepest portions of bedrock valleys with the thickest areas of overburden. Additional wells were also added to existing well clusters to fill data gaps identified during Phase I groundwater sampling. Refer to Figure 2-1 for the monitoring well locations and Appendix A for well construction logs.

Soil borings were advanced using the drive and wash drilling method. Soil samples from the deepest boring at each cluster were logged, field screened with a PID, and sent for off-site VOC screening. Subsequent overburden well-screen intervals were selected based on maximum

VOC concentrations from screening data or direct observation of contamination. If VOCs were not detected by the PID in the field or laboratory screening, then zones with the highest estimated hydraulic conductivity (based on visual observations) were selected.

2.10.2 Groundwater Sampling (1998 – 1999)

Groundwater sampling was conducted from November 1998 to March 1999 using the EPA Region I procedure for low-stress/low-flow sampling for the following wells:

- All 300-series wells
- All 400-series wells
- Selected OU1-PC wells
- Selected 100-series wells
- Selected 200-series wells

Groundwater samples for this phase were analyzed for VOCs, SVOCs, pesticides/PCBs, metals, cyanide, water quality parameters (including chloride, sulfate, alkalinity, and nitrate/nitrite), and/or natural attenuation parameters (including carbon dioxide, ferrous iron (iron II), methane, ethane, and ethene). Refer to Appendix B for a complete list of sample results in the OU2 study area.

2.10.3 Seismic Refraction Survey (1998)

The subsurface bedrock relief was known to be highly variable. Further details regarding the bedrock relief were obtained in November 1998 using a seismic refraction survey, consisting of eight separate lines totaling 0.85 miles (Hager-Richter 1999a). This data was used to interpret the depth to bedrock between borings, and identify the geometry of the top of bedrock. Refer to Figure 2-2 for the locations of the seismic refraction lines.

2.10.4 Hydraulic Conductivity Tests (1998 – 1999)

Hydraulic conductivity data were collected at 55 locations from November 1998 to March 1999. Hydraulic conductivity in bedrock was measured in 28 wells using injection packer tests and/or

single well pump tests. Hydraulic conductivity in overburden was measured using single well pump tests (static discharge/drawdown tests) after low-stress samples had been collected. The purpose of the hydraulic conductivity tests was to provide estimates of the hydraulic conductivity of the aquifer materials within the OU2 study area in order to refine the site conceptual model and divide the study area into hydrostratigraphic units. See Figure 2-3 for test locations, Section 3.0 for additional discussion, and Appendix D for supporting tables.

2.10.5 Tidal Influence Study (1999)

In February 1999, a 24-hour tidal influence study was conducted within Ferry Creek, the Housatonic River, and 11 monitoring wells at four clusters in proximity to Ferry Creek and the Housatonic River (MW-112, MW-213, MW-302, and MW-304). Electronic transducers, which measured varying water pressures as the tides and groundwater changed elevation, were used to monitor both overburden and bedrock groundwater elevation changes. The objective was to determine what impact, if any, tidal fluctuations have on groundwater elevations. See Figure 2-3 for the tidal monitoring locations, Section 3.0 for additional discussion, and Appendix D for supporting information.

2.10.6 Ferry Creek Seepage Study (1999)

In March 1999, a seepage study was performed in Ferry Creek from the head wall just south of I-95 to the bridge at Broad Street. The seepage study was conducted to estimate contaminant concentrations and the rate of groundwater recharge from groundwater into the creek.

The study to investigate the flux between the groundwater and surface water was divided into two phases: thermal mapping of Ferry Creek to assist in determining areas of groundwater discharge into the creek; and collection of groundwater discharging to the creek using seepage meters. The thermal mapping was conducted during the approach of low tide (when the surface water velocity is low and mixing of groundwater with the colder surface water is minimized) and high tide. During thermal mapping, the creek bank was visually inspected for evidence of groundwater seeps. The high and low tide thermal maps were compared to determine if groundwater discharges to the creek during the complete tidal cycle or only during periods of low tide. Seepage meter locations were selected based on the comparison of

thermal mapping during high and low tides, as well as the visual inspection of the creek. Refer to Figure 2-3 for seepage meter locations.

Twelve seepage meters were installed in the areas identified through thermal mapping as groundwater discharge areas, and three meters were installed in areas less likely to be affected by groundwater discharge. The seepage meters were installed in the locations selected and left in place until a sufficient quantity of water was collected. The sample collection bag was then removed and the volume of water in the bag was measured. Refer to Appendix D for a summary of collection times and discharge rates for each of the seepage meters. A groundwater sample was collected directly from the sample collection bag for off-site VOC analysis. To obtain a SVOC sample, a peristaltic pump was attached to the sample collection bag and water was pumped into an appropriate sample container for off-site analysis. Surface water samples were also collected from four locations within Ferry Creek and analyzed for VOCs and SVOCs to provide a comparison to the groundwater discharge contaminant concentrations. A discussion of the results of this investigation is provided in Sections 3.0 and 4.0. Raw data collected during the study are presented in Appendix D.

2.11 Residential Indoor Air Study (2000 through 2003)

Contaminants in groundwater may potentially volatilize through soil gas and into indoor air spaces located above contaminated groundwater. EPA conducted soil gas and indoor air sampling within the indoor air area of interest over a 4-year period. Sampling locations were based on sampling of the groundwater in prior years in and around the former Raymark Facility and knowledge of the potential flow of the groundwater. The purpose of these sampling efforts was to evaluate groundwater impacts on residential properties from contamination from the former Raymark Facility. Refer to Figure 2-4 for sampling locations. Indoor air sampling was performed at numerous homes and light commercial buildings over this 4-year period. Properties where sampling was performed are not specifically indicated on the figure to avoid presenting individual property results.

Indoor air and soil gas samples were collected during April 2000, February and March 2001, February and March 2002, July 2002, February 2003, and March 2003. Indoor air sampling was performed at six properties in April 2000. In February and March 2001, indoor air samples were collected in eight homes, including three homes that had been sampled the previous year. In

February and March 2002, indoor air samples were collected in four homes where sub-slab ventilation systems had been installed in December 2001. Indoor air samples were also collected from one other home previously sampled and ten homes not previously sampled. In July 2002, indoor air samples were collected at five homes identified in the February and March 2002 sampling as candidates for sub-slab ventilation systems. The February 2003 sampling was conducted at two homes located outside the residential area of interest to determine average background concentrations of indoor air contaminants and five homes where sub-slab ventilation systems had been installed in the fall of 2002. In March 2003, indoor air samples were collected at 11 homes, including one home previously sampled in April 2000. Soil gas samples were collected from areas adjacent to building foundations, below the basement floor, and from locations along sidewalks. Between April 2000 and May 2003, soil gas samples were collected at approximately 120 properties.

The data were used to determine if contaminant levels are a health risk to the occupants of the buildings tested. Indoor air and soil gas grab samples were field-screened for the following target VOCs: 1,1-dichloroethene, vinyl chloride, trichloroethene, benzene, 1,1,1-trichloroethane, chlorobenzene, and toluene. These target VOCs were selected based on the contaminants detected in the groundwater throughout the OU2 Study area at concentrations in excess of CT groundwater volatilization criteria. Ambient air samples were also collected from an area adjacent to the residential sampling area to be used for background data. Field-screening analyses provided immediate data to aid in the determination of further sampling locations for fixed laboratory analyses.

Soil gas samples were collected through a variety of techniques and analyzed on-site by the EPA mobile laboratory. Canister grab samples were also collected from some locations and analyzed off site at the EPA laboratory.

Indoor air samples were collected as both grab and 8-hour integrated samples using SUMMA™ canisters. Grab samples were analyzed on-site by the EPA mobile laboratory. SUMMA canister samples were collected over an 8-hour period. Canisters were placed in basement areas expected to have the highest levels of target compounds, in basement areas used as living areas, as well as in the first floor living area. Air samples were also collected at locations adjacent to the residential area for representative background concentrations. All canisters were analyzed off site at the EPA laboratory for VOCs.

2.12 Phase III Groundwater Remedial Investigation (2002 - 2003)

Phase III was conducted from February 2002 to February 2003. This phase was conducted to fill data gaps identified in the November 2000 Draft Final OU2 RI. The tasks completed include: groundwater and soil gas profiling (2002); well repair and redevelopment as needed (2002); monitoring well installation (2002/2003); resistivity and VLF (Very Low Frequency) survey (2002); borehole geophysics survey for three bedrock wells (2002); groundwater and soil gas sampling (2002-2003); a seepage study in Ferry Creek (2003); tidal study (2003); and an aquifer test (2003).

2.12.1 Groundwater and Soil Gas Profiling (2002)

TtNUS conducted horizontal and vertical groundwater profiling in tandem with soil gas sampling in two phases, in February and March 2002 and from May to July 2002. The objective of this investigation was to determine the extent of contamination within the residential area downgradient of the Raymark Facility. This information was used to support decisions on the locations of permanent soil gas monitoring stations, identification of “at-risk” homes and buildings from potential indoor air vapors, and future placement of permanent groundwater monitoring wells. Refer to Figure 2-4 for the 2002 soil gas and groundwater profiling locations.

Small diameter well points were advanced using a track- or truck-mounted geoprobe in the residential neighborhoods downgradient of the former Raymark facility. The profiling took place in two phases: the first phase concentrated on the neighborhood adjacent to Housatonic Avenue; and the second phase included locations upgradient of the neighborhood and other residential neighborhoods nearby.

During the installation of the groundwater well points, samples were collected at the water table, 5 feet below the water table, and every 10 feet thereafter until the point of refusal. Samples were collected and sent to the EPA Region I on-site mobile laboratory for rapid turnaround analysis of target VOCs. Approximately 10 percent of all the samples collected were sent to the EPA NERL for confirmatory analysis.

Soil gas samples were collected using the geoprobe Post Run Tubing System® or equivalent. Teflon-lined tubing was advanced and connected to the sampling point after the geoprobe rods

were advanced. The Teflon tubing was connected to a system developed by TtNUS consisting of stainless-steel tubing with pressure and vacuum gauges as well as an exhaust port and a vacuum box containing a tedlar bag for the VOC sample collection. Soil gas samples were collected from 3 feet bgs, 8 feet bgs, and just above the water table. The samples were analyzed for VOCs, and a single separate sample was collected for screening of oxygen, carbon dioxide, and methane immediately following collection of the VOC sample. At five locations, a time series of three samples was collected for oxygen, carbon dioxide, and methane screening. Confirmatory soil gas samples were also collected for five percent of the samples using SUMMA canisters and sent to the EPA NERL. TtNUS personnel also conducted air permeability tests of the vadose zone in February and March 2002 using an extraction well and one observation point. Air permeability tests were performed at a depth of 8 feet (just below the foundation of most basements) in areas where elevated concentrations of contaminants were detected in soil gas.

2.12.2 Well Development/Redevelopment (2002 – 2003)

All of the existing monitoring wells pertinent to the OU2 study area were inventoried from June to November 2002, and wells needing repairs and/or redevelopment were noted. Wells were repaired in October 2002. Redevelopment of pre-existing wells and development of new wells began in November 2002 and ended in January 2003. Single well pump tests were performed during development to determine hydraulic conductivity. The purpose of the tests was to provide estimates of the hydraulic conductivity of the aquifer materials within the study area in order to further refine the site conceptual model and hydrostratigraphic units. Data collected from low-flow purging (see Section 2.12.3) were also used as a single-well pump test for some wells set in low-conductivity material. See Figure 2-1 for well locations, Section 3.0 for additional discussion, and Appendix D for supporting tables.

2.12.3 Monitoring Well Installation (2002 – 2003)

From August 2002 to January 2003, TtNUS constructed 59 wells at 36 clusters (500-series), replaced eight wells at three clusters, and constructed eight additional wells at existing well clusters. These wells were installed in order to fill data gaps identified in the November 2000 Draft Final OU2 RI. They included clusters in the vicinity of 250 East Main Street (MW-501 and MW-533 in the northeastern portion of the site), as well as other locations (MW-502 through

MW-504, MW-507, MW-508, and MW-514) in order to better define the bedrock topography and groundwater flow. Wells were installed around 375 Barnum Avenue (MW-502, MW-506, and MW-529) to better define groundwater contamination. Wells were installed in the bedding of a sewer line running through the study area (MW-521S, MW-522S) to ensure that the bedding materials were not acting as a preferential pathway for contamination. Deep bedrock wells (MW-213, MW-215, and MW-524) were installed to study the orientation and permeability of water-bearing fractures in the bedrock. Finally, a network of shallow groundwater wells and soil gas wells was constructed in the residential area downgradient of the former Raymark facility, based on the results of soil gas and groundwater profiling described in Section 2.12.1.

Thirty soil gas wells were advanced using either DPT (geoprobe) or drive and wash drilling method. Soil gas well screens were placed at 7 to 8 feet bgs or 1 foot above the water table, whichever was higher. At two locations, soil gas well clusters were installed with the bottom of the screen at 3 feet bgs, 8 feet bgs, and 2 feet above the water table. The soil gas monitoring wells were paired with selected shallow groundwater monitoring wells in order to better define the interaction between contaminants in the water table and soil gas from the vadose zone. Refer to Figure 2-1 for the monitoring well locations, and Appendix A for well construction details.

All of the groundwater monitoring wells were advanced using the drive and wash drilling method, with the exception of three bedrock wells, which were advanced using the air rotary method. Soil samples from the deepest boring at each cluster were logged, field screened with a PID and FID, and sent for VOC screening analysis. Overburden well-screen intervals were selected based on maximum VOC concentrations from screening data or direct observation of contamination. If VOCs were not detected by the PID in the field or laboratory screening, the zones with the highest estimated hydraulic conductivity (based on visual observations) were selected. Soil samples considered to be representative of the grain size at the water table and vadose zone were collected from each of these wells and analyzed for total organic carbon (TOC) and grain size. TOC and grain size analyses were used to support groundwater modeling efforts.

2.12.4 Resistivity and Very Low Frequency (VLF) Survey (2002)

A geophysical survey was conducted at the former Raybestos Memorial Ballfield (OU4) in September 2002 using two techniques, multi-electrode resistivity and VLF. The objective of the geophysical survey was to help locate suspected bedrock fracture zones within OU4. Usable VLF signals could not be detected in the line orientation of the planned traverses; however, the resistivity data indicated a potential fracture zone bisecting one transverse (Geophysical Applications 2002). See Figure 2-2 for the survey location.

2.12.5 Borehole Geophysics (2002)

In October and November 2002, three wells (MW-213DB, MW-215DB, and MW-524B) were advanced using air-rotary methods to 105 feet below bedrock, with the exception of MW-215DB, which stopped at 45 feet below bedrock. The open boreholes were surveyed using natural gamma, single-point resistance (SPR), caliper, fluid temperature, fluid resistivity, borehole video, acoustic televiewer (ATV), and heat-pulse flowmeter testing in order to determine water-bearing fracture size and orientation (Geophysical Applications 2003). This information, along with the descriptions from the boring logs, was used to determine the characteristics of the water bearing fractures in the bedrock, as well as placement of well screen intervals in the individual wells. Refer to Figure 2-2 for the locations of wells tested using borehole geophysics.

2.12.6 Groundwater and Soil Gas Sampling (2002 – 2003)

Groundwater sampling was conducted using the EPA Region I low-stress/low-flow method for all known groundwater monitoring wells in the study area. The groundwater sampling round was conducted from December 2002 to February 2003, with the exception of wells at one property, which were sampled during July 2003 for access reasons. Wells sampled included:

- All 100 - 500 series wells, with the exception of MW-120S (destroyed)
- All Dock wells (MW-1 cluster, MW-2, MW-3, and MW-4 cluster) sampled by EMS
- Selected wells at 540 Longbrook Avenue (MW-1-4, MW-6-10, BR-1&2, PZ13-14, and MW-Z)
- Selected wells at 375 Barnum Avenue (CRA-2 and CRA-5 through CRA-8 clusters)

- OU1 Post Closure (PC) wells
- OU1 Recovery wells (RW)
- Wells at the vacant lot on Housatonic Avenue (SPADA-E222, 5132 and WA120AG-E512-540)

Groundwater samples for this phase were analyzed for VOCs, SVOCs, pesticides/PCBs, and metals. Selected samples were also analyzed for dioxins, cyanide, water quality parameters (including chloride, sulfate, alkalinity, and nitrate/nitrite), and/or natural attenuation parameters (including carbon dioxide, ferrous iron, methane, ethane, and ethene). A synoptic water level round of all the wells was conducted within 4 hours on April 15, 2003.

Soil gas sampling of permanent monitoring wells and associated shallow groundwater monitoring wells was conducted in January 2003 by TtNUS personnel and in May 2003 by EPA personnel. Soil gas sampling methods were the same as those used for soil gas profiling; see Section 2.12.1 for details. Soil gas samples were analyzed for VOCs and screened for oxygen, carbon dioxide, and methane. The associated groundwater samples were analyzed for VOCs, SVOCs, pesticides/PCBs, and metals. Selected groundwater samples were also analyzed for dioxins, cyanide, water quality parameters (including chloride, sulfate, alkalinity, and nitrate/nitrite), and/or natural attenuation parameters (including carbon dioxide, ferrous iron, methane, ethane, and ethene). See Table 2-2 for a complete list of samples collected.

2.12.7 Ferry Creek Seepage Study and Surface Water Sampling (2003)

In January and February 2003, a seepage study was performed in Ferry Creek north of Interstate 95. This study was upstream of the seepage study area examined in March 1999. The creek bank was inspected for evidence of groundwater seeps and other zones of potential recharge in order to select seepage meter locations. Refer to Figure 2-3 for seepage meter locations. Four seepage meters were installed in the areas identified as potential groundwater discharge areas. The seepage meters were left in place and the collection bags checked at low tide, when the creek had the most potential to gain water from groundwater sources. The bags were empty at low tide (about 6 hours after installation) and again at 24 and 48 hours; no samples were collected from the meters. After installation of the seepage meters, surface water samples were collected upstream and downstream of the study area and analyzed for VOCs, SVOCs, and metals. After removal of the seepage meters, five piezometers were installed in

Ferry Creek. The piezometers gave little to no water because of the high percentage of silt and clay and were not sampled. Five piezometers (MW-509 through MW-512 and MW-516) were also installed north of Ferry Creek below the organic layer and were sampled. A discussion of this investigation is provided in Section 3.0.

2.12.8 Tidal Influence Study (2003)

In April 2003, a 24-hour tidal influence study was conducted within the Housatonic River at the Dock property launching area and 20 monitoring wells at eight clusters in proximity to Ferry Creek and the Housatonic River (MW-201D and B, the MW-309 cluster, the MW-533 cluster, the PC-04 cluster, and the Dock wells MW-4B, M, and S). Electronic transducers that measured varying water pressures as the water levels changed were used to monitor both groundwater and surface water (Housatonic River) elevation changes. The objective was to determine what impact tidal fluctuations have on groundwater elevations in the northern end of the study area. See Figure 2-3 for the tidal monitoring locations, Section 3.0 for additional discussion, and Appendix D for supporting information.

2.12.9 Aquifer Test (2003)

In May 2003, a step-drawdown aquifer pumping test was performed. Data from the aquifer test was used to evaluate hydraulic conductivity and the interaction between bedrock and the overburden in the area. First, an 8-inch aquifer test well was drilled to a depth of 68 feet using a mud-rotary method. Once well construction and development were complete, transducers were installed in the MW-210, MW-304, MW-502, MW-504, and MW-505 well clusters to measure changes in water elevations during the pumping test. The test well was pumped at 110 to 75 gpm for 48 hours while the drawdown in the selected observation wells was measured using the transducers. Refer to Figure 2-3 for the locations of the test well and observation wells. See Section 3.0 for additional discussion and Appendix D for supporting information.

2.13 Soil Sampling (2002 and 2003)

From June to August 2002 and from May to June 2003, properties throughout Stratford were sampled to identify the extent of soil contamination from disposal of Raymark waste.

Residential, municipal, and commercial properties were sampled. The universe of properties sampled during the effort included all properties that had been identified over a 10-year period where the presence of Raymark waste was suspected. These locations were identified by a number of sources including, but not limited to, officials of the Town of Stratford, former Raymark records and/or employees, town records and neighbors/citizens. Each property was evaluated to determine if sampling was required, and if so, whether adequate sampling had been conducted to determine the potential presence of Raymark waste. Both EPA and the CTDEP participated in these evaluations.

Phase I soil samples were collected from shallow locations (0-4 feet deep) and were screened for the four primary contaminants that have been identified as indicators of Raymark waste: lead, copper, PCBs (Aroclor 1268), and asbestos. Phase II sampling was conducted at properties where potentially asbestos-containing materials were noted during the drilling program and where screening results from Phase I and earlier efforts indicated potential Raymark waste. Phase II samples were collected to native material. The samples from one boring on each property identified as Phase II were sent for a more extensive set of analysis, including VOCs, SVOCs, pesticides, PCBs, total metals, dioxin, and/or SPLP metals. The borings from this investigation that are used in the OU2 RI are included in Figure 3-6. Boring logs are included in Appendix A. Results from this field investigation were included in the 2004 OU6 Final Remedial Investigation (TtNUS 2004).

3.0 PHYSICAL CHARACTERISTICS OF THE STUDY AREA

This section presents a summary of the physical characteristics of the OU2 study area. This section includes:

- Section 3.1 - Surface Features and Land Use
- Section 3.2 - Geology
- Section 3.3 - Surface Water
- Section 3.4 - Hydrogeology
- Section 3.5 – Soil Gas
- Section 3.6 – Climate and Meteorology

Throughout this report, all elevations are stated in feet with respect to the National Geodetic Vertical Datum (NGVD) 1929.

3.1 Surface Features and Land Use

The OU2 study area is part of the Housatonic River Basin, a tidally influenced system. The study area includes approximately 500 acres of residential and commercial properties, highways, streets, wetlands, and water bodies such as Ferry Creek and the Housatonic River. The study area is considered urban with approximately 50 percent covered with pavement or buildings. The OU2 study area is described in Section 1.3 and shown on Figure 1-2. See Figure 3-1 for details of land use within the study area.

The lower reaches of Ferry Creek and the Housatonic River are used for recreational fishing and boating. The mouth of the Housatonic River is considered to be a recreational fishery and a source of human food chain organisms. The lower Housatonic River, near the mouth of Ferry Creek, contains important commercial seedbeds for oyster cultivation. EPA representatives have observed people crabbing from numerous locations along Ferry Creek including the flood control barrier located on Broad Street.

With the exception of the northwestern portion of the study area, the topography is relatively flat, with gentle slopes to Ferry Creek and the Housatonic River. Based on a review of USGS

topographic maps, the majority of the study area, with the exception of the northwest portion, lies at a topographic elevation of approximately 10 feet. The northwestern portion lies at an elevation of approximately 20 feet, but rises quickly to greater than 50 feet in elevation along a northeast to southwest trending ridgeline (USGS 1984a and 1984b).

With the exception of the northwestern portion, most of the study area is located within the 100-year floodplain, as observed from Federal Emergency Management Agency (FEMA) Flood Insurance Rate Maps for Stratford, Connecticut (FEMA 1992). The 100-year frequency base flood elevation ranges from 10 - 13 feet in the vicinity of the Housatonic River, from 15 - 32 feet around Tanners Brook, and 10 feet around Ferry Creek. See Appendix C for a copy of the Flood Insurance Rate Maps.

The principal industries within the Stratford community include manufacturing of aircraft, air conditioning units, chemicals, plastic, paper, rubber goods, electrical and machine parts, and toys. There are approximately 2,200 business establishments in Stratford. The Stratford, Connecticut web page (www.townofstratford.com) states the 2003 population as 49,389 people within the 18.7 square miles of the Town. The median income of the Town is $59,750 and the median housing cost is $133,000.

3.1.1 Study Area Infiltration

The study area receives approximately 44 inches of precipitation (rainfall, snow, etc.) per year (NOAA 2002). The 500-acre study area consists of approximately 240 acres of buildings and paved areas including roads and parking lots, 10 acres of open water, and 250 acres of gravel, grasses, and bushes. This implies that a substantial amount of surface runoff from precipitation is diverted from the study area into drainage collection systems. Figure 3-1 depicts the distribution of land use features.

3.1.2 Groundwater Classification and Usage within the Study Area and Environs

Groundwater within and surrounding the study area has been classified as GB (unsuitable for drinking without treatment) (CTDEP 1997a).

The Town of Stratford, Connecticut has been a customer of the Aquarian Water Company [formerly Bridgeport Hydraulic Company (BHC)] for over 100 years. BHC received authorization to provide Stratford with drinking water on April 29, 1897. Currently, the Aquarian Water Company supplies Stratford with water predominately from the Trap Falls Reservoir, located approximately 4.0 miles north of the OU2 study area. The water company also provides Stratford with water from the Hemlocks Reservoir, located approximately 8.5 miles west of the OU2 study area, and the Easton Reservoir, located approximately 8.0 miles northwest of the study area. The Hemlocks and Easton Reservoirs are only utilized during drought conditions, low flow conditions, or high water demands. However, the Hemlocks Reservoir is also used in conjunction with the Trap Falls Reservoir to provide water to the Lordship area of Stratford, which is the point of the Stratford distribution system furthest from the supply source (Bridgeport 1999a and 1999b, Aquarian 2003).

Although most of the Stratford drinking water is supplied from the water company, 15 percent of the town presently uses private wells. Most of these wells are located north of the study area. According to well inventories from the Stratford Health Department conducted in 1994 and updated in 2003, there are approximately 289 private wells located and used in Stratford. The well inventories were developed by comparing Stratford's town property list to the water company's billing list. Properties that were not on the billing list were considered to be on well water. These locations were verified in the field. The inventories show no private wells in use located within the OU2 study area, and few private wells located close to the study area (Stratford 1999a, 1999b, and 2003). In addition, a public health survey concluded that no private wells were located within 0.5 miles of any known Raymark waste disposal sites (ATSDR 1996).

According to the Town of Stratford, there were two commercial wells in the town, one at the former Tilo Industry Facility (411 Barnum Avenue cutoff) and a second at an operating industrial property located at the intersection of West Broad and Stratford Avenue. Water from both wells was used for cooling purposes and not for drinking water supply. Both wells have been abandoned (date unknown). Also, two recovery wells for pumping and treatment of groundwater existed at 375 Barnum Avenue. These wells removed 2,500 gallons of free phase-hydrocarbon during operation of the recovery system (B&RE 1998 and CDM 1994). A supply well was also used during the early 1970s at this property. The well has since been abandoned and no records are available (HRP 1997).

At the former Raymark Facility, an on-site well in the northern portion of the Facility was used for non-contact cooling water. During remedial activities at the Facility, the area was capped and the supply well abandoned. A DNAPL extraction system with five recovery wells (see Figure 2-1 for locations) was installed in the northwest portion of the site. The extraction system is maintained and operated by the CTDEP. The DNAPL system operation was automatic and the system holding tank was only sampled for disposal purposes. The tank has been emptied on three occasions: In 1997, when the EPA turned the site over to the CTDEP, the tank was about half full with a DNAPL/water mix and was emptied. In July 2001, 460 gallons were removed from the tank. In June 2002, 1,000 gallons were removed from the tank; however, this water was from the redevelopment of the recovery wells and included a water/DNAPL mix. Additional amounts of recovered mixed DNAPL and water were stored in drums and later removed. In March 2000, 165 gallons were removed, and in November 2003, 55 gallons were removed. Refer to Appendix B for an analysis of DNAPL recovered from the former Raymark Facility in 2003. For DNAPL system design details, refer to the *Final Basis of Design/Design Analysis Report* (Foster Wheeler 1996).

3.2 Geology

This section provides a brief overview of the geology of the region and the study area. Each geology section is divided into two subsections: overburden and bedrock. The discussion of the study area geology is based on data collected during the subsurface investigations identified in Section 2.0.

3.2.1 Regional Geology

This discussion of the regional geology is divided into overburden and bedrock subsections. The regional overburden geology is defined as the unconsolidated deposits of sand, silt, gravel, clay, and peat. The overburden is underlain by bedrock consisting of metamorphic rock, mainly schist and gneiss.

3.2.1.1 Regional Overburden Geology

The regional surficial geology within the Housatonic River basin was shaped by glaciation. The State of Connecticut was covered by glacial ice at least twice. During the last retreat of the ice,

glaciers deposited a thin mantle of till overlying bedrock. Glacio-fluvial outwash deposits formed thicker, highly stratified sequences of silty sand to gravelly sand that overlies till, and filled bedrock valleys. Windblown sand and silt were also deposited on valley floors, however, these deposits are indistinguishable from present day organic topsoil deposits (Flint 1968). The Stratford, Connecticut region is generally located in the Stratford outwash plain, on the western Housatonic River valley floor, within the Housatonic River Basin. Natural overburden deposits in the vicinity of Stratford consist of glacial deposits (outwash sediments, ice-contact stratified drift, and till) and recent swamp and marsh deposits.

Glacial till was deposited by glacier ice, is variable in thickness, and forms a discontinuous mantle over bedrock. The till consists of a non-stratified, poorly sorted mixture of coarse (pebbles/cobbles/boulders) and fine (sand/silt/clay) fractions, with the coarse fraction generally not exceeding 20 percent. In the lower Housatonic River Basin, the median till thickness is 30 feet (Wilson et al. 1974). Till is commonly exposed in areas of relatively high elevation, and is generally covered by sediments of post-till age within the valleys.

Most of the sediments overlying till consist of stratified drift and outwash materials. Ice-contact stratified drift includes sand, gravel, silt, and clay, frequently poorly sorted with abrupt changes in grain size. These deposits were formed during glaciation in streams and local ephemeral lakes in close proximity to melting glacier ice, and often grade into outwash sediments. Stratified drift covers 16 percent of the area of the lower Housatonic River Basin, and generally occurs as narrow belts in stream valleys and lowlands (Wilson et al. 1974). Glacial outwash deposits are predominant in the stream valleys, and consist of highly stratified sand, silty sand, and gravelly sand. Beds are not persistent, and individual lenses attain thicknesses of tens of feet, and thin out or are truncated over short distances. Outwash units in the vicinity of the study area generally consist primarily of sands with up to 50 percent gravel.

Swamp and marsh deposits are present in lowlands and in proximity to the Housatonic River. These deposits consist of silt, sand, and clay-sized particles interbedded with organic fragments and peat deposits up to 10 feet thick. Swamp and marsh deposits, where present, commonly overlie stratified drift. As a result of the practice of filling in lowland areas, fine-grained swamp and marsh deposits, including peat, are commonly found underlying fill deposits.

Fill deposits consist of a mixture of natural and manmade materials. A large portion of the Stratford, Connecticut region is composed of manmade fill (Flint 1968). These deposits are often found at lowland areas of the region and are often overlying swamp and marsh deposits. These deposits also overlie stratified drift and ice contact deposits, such as sands and gravels. Fill materials frequently include manufacturing, household, and construction debris usually mixed with natural materials. Natural materials include various amounts of clay, silt, sand, and gravel, as well as organic soils. Other fill materials that do not contain visual evidence of manmade debris are present throughout the region, including soils from dredging operations and other soils brought in to fill low-lying areas. This fill is frequently more difficult to distinguish from natural deposits.

3.2.1.2 Regional Bedrock Geology

The Stratford, Connecticut region is located in the Connecticut Valley Synclinorium of Connecticut's Western Uplands. The regional bedrock setting consists of a series of generally foliated meta-sedimentary and meta-volcanic rocks of the Early and Middle Paleozoic Age, with foliation trending northeast-southwest, in a large syncline. These rocks are mainly schists and gneisses. The sequence was tightly folded and subjected to progressive regional metamorphism ranging from chlorite to kyanite metamorphic grade. A high angle fault is mapped approximately 1 mile southeast of the study area, across the Housatonic River, generally trending southwest to northeast (CT GNHS 1985). The implication of this fault and any related splay faulting to local geology and contaminant transport was not evaluated.

3.2.2 OU2 Study Area Geology

The geology of the OU2 study area is divided into two subsections: overburden and bedrock. The discussion of the study area geology is based on overburden and bedrock data collected during soil boring activities conducted during the investigations summarized in Section 2.0.

3.2.2.1 Study Area Overburden Geology

This section discusses the overburden materials beginning with the surficial soils and proceeding downward to the materials in contact with bedrock. For the purposes of linking the OU2 study area geology to the hydraulic properties of the overburden materials, the overburden

materials have been separated into hydrostratigraphic units based on similar hydraulic and geologic properties within each unit rather than the glacial origin of the deposits. The information presented in this section is based on available boring logs and a review of the three cross sections chosen to represent the OU2 study area (see Section 3.2.2.2). The boring logs are supplied as Appendix A.

Based on borings advanced in the OU2 study area, the overburden deposits are characterized as till, a variety of glacial outwash deposits, ice contact deposits, alluvial deposits, swamp and marsh deposits, and fill materials. These surficial deposits identified within the OU2 study area are consistent with deposits mapped by Flint (1968). Glacial till is present locally, but was found to be discontinuous within the OU2 study area. The predominant overburden material noted within the OU2 study area consists of a complex sequence of glacial outwash deposits (sand and gravel) ranging from silty sand to coarse gravel. Finer grained layers, predominantly consisting of silt and clay-sized particles, were identified within the OU2 study area at shallow depths, which are thought to be former swamp and marsh deposits, and also at lower elevations, which may be ice contact deposits or lenses within the glacial outwash. As a result of the practice of filling lowland areas, fine-grained swamp and marsh deposits, including peat/organic silt, are commonly found underlying fill.

The overburden thickness in the OU2 study area varies from nonexistent at bedrock outcrops to 150 feet at MW-211B. The thickness of the overburden exceeds 100 feet throughout large portions of the OU2 study area where bedrock valleys have been identified. This wide range of overburden thickness is a result of the variation in bedrock elevation beneath a relatively flat topographic land surface. The monitoring wells referenced in this section are shown on Figure 2-1.

The hydrostratigraphic units identified for the overburden in the OU2 study area include: fill, peat, organic silt, inorganic silt and clay, sand and gravel, gravel, and till. Each of these units is described below.

Fill

Fill consists of a mixture of natural and manmade materials. Fill materials frequently include manufacturing, household, and construction debris usually mixed with natural materials.

Natural materials include various amounts of clay, silt, sand, and gravel. Manmade materials consist of charcoal, asphalt, metal, brick, tile, glass, and miscellaneous manufacturing debris. Both the former Raymark Facility and other local manufacturers generated waste used to fill in low-lying areas in the town of Stratford. Fill materials that do not contain visual evidence of manmade debris are present throughout the OU2 study area, generally consisting of sands with varying amounts of silt and gravel. This fill is frequently more difficult to distinguish from natural deposits. Specific information on the presence or absence of manmade fill materials and contaminated soil intervals are noted on the boring logs (see Appendix A), but is not discussed in this Remedial Investigation.

Fill was identified by visual descriptions of soil samples collected during the field investigations. Peat, silt, and clay layers commonly underlie fill throughout much of the OU2 study area, especially in present or former wetland locations. The actual thickness of fill varies over short distances, as illustrated on the cross sections discussed in Section 3.2.2.2. Fill thickness at the former Raymark Facility varied from depths of less than one foot to more than 20 feet bgs based on the boring logs.

The hydraulic conductivity of the fill is expected to vary depending on the type of fill placed. Fill that consists of finer grained materials such as silt and clay is expected to have a lower hydraulic conductivity compared to fill that consists of sand and gravel.

This OU2 RI does not discuss in detail the specific chemical characteristics of the fill material located on properties throughout the study area, or the potential for contaminants leaching from these fill materials into the groundwater. Previous RIs have addressed the details of these materials as potential sources of contamination (HNUS 1995; TtNUS 1999a; TtNUS 1999b; TtNUS 2000b; TtNUS 2000d; and TtNUS 2004). SPLP data for properties in the OU2 study area are included in Appendix B of this report as a reference.

Peat

Deposits of peat and associated organic-rich silt and clay-sized particles related to present or former swamp and marsh deposits were identified in numerous borings throughout the OU2 study area. The peat layers are discontinuous and vary greatly both in thickness and composition. The peat units frequently grade into organic silts with clay-sized particles over

very short distances. Sand stringers, usually fine sand, were noted within the peat units. No effort was made to differentiate the type of peat units (humic, fibric) identified in the subsurface soil borings. As a result of the historic practices of dumping fill materials into wetland areas throughout Stratford, the peat and finer grained deposits directly underlie fill at many areas within the OU2 study area.

Significant areas of peat are located beneath the former Raymark Facility along the eastern edge of the property and extending eastward towards I-95. This peat layer appears to have a northern limit located beneath East Main Street between borings PC-01 and MW-3S, as it does not appear at MW-533. The peat is not found at boring PC-08 or PC-09, but is located just south of those borings at MW-207 and MW-211 at a slightly deeper elevation. This peat unit does not appear to extend to the east of I-95. The upper contact of the peat gradually slopes to the south. The fill is in direct contact with the organic silts and peat at PC-01, and in contact with sands and gravel to the south as noted at borings PC-02, MW-207 and MW-211.

Additional peat deposits are noted at PC-06, located at the northern corner of the former Raymark Facility, but were not found at borings PC-05, MW-309, MW-402, or PC-10, which surround PC-06, indicating the discontinuous nature of these deposits. The thickness of the peat unit along the eastern portion of the OU1 property has been modified through surcharging and dynamic compaction. Data from the post-closure boring logs pre-dates the capping activities.

Boring logs from soil borings SB-425 and MW-401, located at the OU4 Memorial Ballfield property, indicate a small area of peat. This deposit is related to the Former Frog Pond, which was filled with Raymark process fill. Peat was also identified in borings D1-203 and D1-210 in the northern portion of the site, but not at D1-201, indicating the discontinuous nature of the deposits.

Significant areas of swamp and marsh deposits, consisting of peat and organic silt and clay-sized particles, are located south of the OU1 property. At the 411 Barnum Avenue cut-off property (borings MW-209 and MW-305), a 5-foot peat layer was encountered. This unit grades into organic silts to the east along Ferry Creek (MW-S12, MW-S16, and MW-S09). Organic-rich fines and peat related to the present and ancestral Ferry Creek and its associated wetlands underlie the majority of the 600 East Broadway property and areas to the south and

east. The eastern boundary of these units appears to be Willow Avenue (MW-215) and Housatonic Avenue (MW-302 and MW-217). These materials are discontinuous and highly variable in the upper reaches of present day Ferry Creek (west of Ferry Boulevard) as noted in boring series A1-SD01 through A1-SD06.

Peat tends to have a relatively high permeability. The peat in the OU2 study area, however, if frequently interbedded with other materials. Overall, the peat units containing discontinuous layers of organic silt and clay should have relatively high permeability, whereas peat underlain by a continuous organic silt or clay layer generally has low vertical hydraulic conductivity.

Organic Silt

Deposits of organic silt related to the present and former swamp and marsh deposits were identified in several borings throughout the OU2 study area. The organic silt layers are discontinuous and have varying thickness.

Significant areas of organic silt are located south of the former Raymark Facility near the present and past locations of Ferry Creek. A 14-foot thick layer of organic silt was identified at boring MW-211, located just south of the former Raymark Facility. The organic silt was not encountered in adjacent borings and therefore is not believed to extend a significant distance beyond MW-211. A 5-foot thick layer of organic silt was identified at monitoring well clusters MW-113 and MW-312, adjacent to Ferry Creek. A 20-foot thick layer of organic silt was encountered at MW-113 and appears to continue to MW-312, where the unit thins to 7 feet. A 6-foot thick layer of organic silt was also encountered at MW-313. The extent of this unit cannot be determined because of insufficient subsurface information in this area.

Organic-rich silts and clay-sized particles grade into inorganic silts to the south as noted at borings MW-102, MW-310, MW-311, and MW-313. A mix of organic and inorganic fines is noted beneath the lower portions of Ferry Creek. The thickness of these former swamp and marsh deposits increases to the south, exceeding 20 feet in the area north of the Broad Street tide control structure.

The organic silts have significantly lower hydraulic conductivities than the peat that they are often associated with. They may act as a potential barrier to groundwater flow, especially in the shallow overburden aquifer.

Inorganic Silt and Clay

Silt and clay-sized particles not related to the more recent swamp and marsh deposits were found at deeper elevations within the study area. Small lenses and stringers of finer-grained sediment are dispersed throughout the predominant glacial outwash sand and gravel. However, some areas contained layers of fines up to 21 feet thick, which may affect groundwater movement. Inorganic fine-grained deposits, which consist primarily of silt with fine sand and clay-sized particles, are noted within the intermediate and deep overburden of the bedrock valleys underlying the former Raymark Facility. These beds are discontinuous and are elongated in a northeast - southwest direction parallel to the orientation of the bedrock valley underlying the study area. Multiple alternating layers of the silts and sand are noted beneath the OU1 property at boring PC-02 and extending southward to MW-207 and MW-211. Similar interbedded layers of the silts and sand are noted at borings PC-10 and PC-13, and PC-14. Approximately 25 feet of silt overlies bedrock at PC-16 at the southwest border of OU1.

Over 20 feet of rhythmically bedded silt and clay-sized particles were found between 68 and 90 feet bgs at MW-305 (located southwest of OU1 and the 411 Barnum Avenue cut-off Property). The horizontal extent of this unit is not fully known because of the horizontal distances between deep borings in this portion of the OU2 study area, but appears to be truncated by a bedrock rise at MW-209 to the northeast and to the north at MW-303. However, about 16 feet of silt was encountered east of MW-209 at MW-502. Further south, at MW-311, silt was encountered from 18 to 52 feet bgs overlying bedrock. A 3-foot thick clay layer was encountered at monitoring well cluster PC-15. A clay layer was not encountered in any of the adjacent borings and it appears that the clay layer is localized. The only other significant deposit of clay was encountered at monitoring well cluster MW-305 and is associated with the silt layer encountered at this location. Clay is noted on several of the boring logs, but usually as a minor component of the deposits encountered.

The fine-grained sediments have significantly lower hydraulic conductivities than the surrounding sands and may act as a potential barrier to groundwater flow and contaminant transport from the former Raymark Facility.

Sand and Gravel

The most prevalent overburden material is glacial outwash and ice-contact deposits consisting of predominantly fine to coarse sand with variable amounts of silt and gravel. Differentiation was not made between the finer-grained silty sands and the coarser-grained gravelly sands because of the highly variable nature of glacial outwash sediments. Glacial outwash sediments are typically highly variable over short horizontal and vertical distances because of the cut and fill stratification created in the depositional environment. The sand and gravel deposits were treated as a single hydrostratigraphic unit because on the large scale, these deposits have similar hydraulic characteristics.

As a result of the highly variable bedrock relief throughout the study area and the relatively uniform ground surface elevation, the overburden thickness varies greatly. Because the sand and gravel deposits are the most prevalent overburden materials, this variation in overburden thickness affects the sand and gravel units the most. In general, the volume of sand and gravel is diminished in areas where bedrock depths are shallow compared to the deeper portions of the bedrock valley.

Sand and gravel units commonly exceed 40 feet in thickness throughout the study area. At PC-06 the sand and gravel is 74 feet thick. At MW-211 and MW-533, close to 100 feet of saturated sand and gravel are present. The coarser sand and gravel materials frequently lie beneath finer-grained swamp and marsh deposits, but large areas within the study area have sand and gravel in direct contact with the overlying fill materials. The central portion of the former Raymark Facility is one such example.

Boring logs from the majority of the former Raymark Facility (PC-03, PC-04, PC-05, PC-07 through PC-16) indicate that the fill materials are in direct contact with the sand and gravel overburden. This is also the case along the northern edge of the study area at 540 Longbrook Avenue and the former Memorial Ballfield (OU4), with the exception of the small quantity of peat noted at SB-425. Significant thicknesses of sand and gravel are found south of the OU1

property where the bedrock depth increases. An example is at boring MW-304, located at the 411 Barnum Avenue cut-off south of many potential source areas, where approximately 50 feet of sand and gravel lies between fill and bedrock with no potential low permeability layer present. Further to the south, the sand and gravel thickness increases to approximately 60 feet, but is at least partially separated from the surficial fill by the fine-grained swamp and marsh deposits noted in the areas of 600 East Broadway and Ferry Creek.

The sand and mixed sand and gravel units in general have higher hydraulic conductivities than the other hydrostratigraphic units and may act as a pathway for groundwater flow and contaminant transport.

Gravel

Gravel has been encountered both within the sand unit and as a separate unit in the overburden. Gravel units that are considered thick enough to be mapped generally occur in the deep bedrock valleys. An example of the deep gravel units was noted at well cluster MW-211, where several layers of gravel were encountered below the fine-grained material. These gravel units have a cumulative thickness of about 50 feet at this location. Other significant gravel units were encountered at well clusters PC-02 and PC-01. Localized gravel layers were encountered in the sand unit, but these gravel units were considered too small to be mapped.

Gravel is expected to have the highest hydraulic conductivity of all the overburden deposits in the study area. For this reason the gravel deposits represent a significant groundwater flow path and a significant groundwater contamination migration pathway.

Till

Glacial till consists of various amounts of clay, silt, sand, and gravel. This unit is generally dense and was encountered just above the top of bedrock. The identification of the till is based on the presence of clay to gravel-sized soil particles, the density of the unit, and the position of the unit in the stratigraphic column.

Glacial till was encountered at locations MW-104, MW-201, MW-305, MW-514, and PC-14, but is not considered to have a significant role in the groundwater flow or contaminant migration

within the study area. The till unit where encountered was generally thin and was either not encountered in adjacent borings or the bedrock topography does not support connecting the till between borings.

3.2.2.2 Geologic Cross Sections

Three geologic cross sections, A-A' through C-C', were created to interpret the hydrogeology of the OU2 study area. The cross sections were selected to highlight important geological features, especially in areas with high concentrations of groundwater contaminants (see Section 4.0). The geologic cross sections were prepared using the information presented on boring logs and a seismic survey within the study area (Hager-Richter 1999a). Bedrock depths between the soil borings were taken from the contour lines of the bedrock map; refer to Figure 3-6. See Figure 3-2 for cross section locations and Figures 3-3 through 3-5 for the cross sections. Boring logs and monitoring well construction logs are included in Appendix A. The seismic survey is discussed in Section 2.10.3 and Section 3.2.2.4.

Boring logs were reviewed and summarized into seven overburden hydrostratigraphic units; fill, peat, organic silt, inorganic silt and clay, sand and gravel, gravel and till as described in Section 3.2.2.1; and bedrock as described in Section 3.2.2.4. Differentiation was not made between the finer silty sands and coarser gravelly sands because of the highly variable nature of the glacial outwash sediments and the cut-and-fill stratification, which is typical of such deposits. Typically glacial outwash sediments are highly variable over short horizontal and vertical distances because of the dynamic environment of deposition. Sediments deposited in riverbed channels form thin discontinuous beds or lenses as the river channel meanders across the flood plain. Glacial till is present in minor, discontinuous layers overlying bedrock at various locations.

Monitoring well screen intervals and water-level head measurements from the most recent synoptic water level measurement round, conducted on April 15, 2003, are presented on each cross section.

Cross Section A - A'

Cross section A - A' transects the OU2 study area in a north-northeast direction and includes soil data from the following monitoring wells: MW-312B, MW-113B, MW-111D, MW-503B,

MW-212B, MW-304B, MW-211B, MW-207D, PC-02B, PC-01B, MW-533B, and MW-4B. Refer to Figure 3-3 for cross section A - A'. This cross section was selected because it presents a view through the entire OU2 study area along the southeast bedrock valley and includes the southeast border of the former Raymark Facility, as well as Ferry Creek. Cross section A - A' shows that the bedrock surface topography is highly variable. Bedrock highs occur at MW-4 at the northern end of cross section A - A' and north of MW-212B. Bedrock lows are at MW-533B, PC-02B, and MW-212B, and from MW-503B south. The deepest portion of the bedrock valley is located at MW-211B at an elevation of –130.42 feet.

Most of the surficial soils in cross section A - A' consist of fill. Fill, as defined in Section 3.2.1.1, was identified at boring locations throughout the study area. Areas of maximum fill thickness noted on cross section A - A' are located beneath I-95 and at PC-01B at the former Raymark Facility, where approximately 20 feet of fill is noted. Fill thickness varies across the OU2 study area depending on land use and history. Based on the water level elevations indicated on cross section A - A', some portion of the fill underlying the former Raymark Facility and the fill to the south of MW-111D is saturated for at least some portion of the year.

Approximately 10 feet of peat, organic silt and clay, and former swamp and marsh deposits are overlain by fill, as identified at MW-312. Based on similar elevations, thickness, and contacts with overlying fill, the organic silt and clay units noted at MW-113B and MW-111D are interpreted as one continuous layer in the shallow subsurface in the southern portion of cross section A - A'. This fine-grained unit is exposed at Ferry Creek east of Ferry Boulevard and may have been deposited by ancestral Ferry Creek and its associated wetlands. Cross section A - A' also shows that Ferry Creek is underlain by silt and organic silt estimated to be about 10 feet thick at this location.

Alternating layers of former swamp and marsh deposits, sand, and silt and clay lie beneath fill that extends south beneath the former Raymark Facility. Although the silt and peat is shown to be continuous between PC-01 and PC-02 on cross section A - A', the layers are not continuous beneath the entire former Raymark Facility, as shown on cross sections B - B' on Figure 3-4, and C - C' on Figure 3-5. Silt noted at depth at MW-113B may represent ice-contact sediments or an isolated lens within the outwash sands and gravel. The silt layer has a lower hydraulic conductivity than the sand and gravel and may restrict vertical movement of groundwater at this location.

A significant thickness of saturated glacial outwash sands and gravel is indicated throughout cross section A - A'. Approximately 100 feet are present below finer grained units at MW-211B in the deeper portions of the bedrock valley. Over 40 feet of saturated gravel and sand is shown beneath the silt layer at MW-111D. These units may contain discontinuous layers of silt and gravel as well as cobbles and boulders. Several gravel layers can be seen in this cross section; however, they are discontinuous and make up a small fraction of the generally sandy units. The most extensive gravel layer begins at MW-533B and is found in borings as far south as MW-211B. The units with a higher percentage of gravel have a higher hydraulic conductivity and represent a potential contaminant pathway.

Cross Section B - B'

Cross section B - B' transects the former Raymark Facility and the OU2 study area in a northwest - southeast direction. The cross section includes soils data from locations MW-308B, SB-427, MW-401B, MW-10, PC-10B, PC-08B, PC-02B, MW-202D, and ends at Bedrock Outcrop "Q". Refer to Figure 3-4 for cross section B - B'. Cross section B - B' originates hydraulically upgradient from the former Raymark Facility, crosses the OU4 Memorial Ballfield, the northern edge of 540 Longbrook Avenue, and the former Raymark Facility, and ends on a bedrock outcrop located between the former Raymark Facility and the Housatonic River. Cross section B - B' shows that the predominant bedrock topographic feature is a set of two valleys. The southeast valley is deeper and wider, with a depth of 116 feet bgs at PC-02B, which is located very close to the former Lagoon 4 at the Raymark Facility.

Bedrock lies close to the surface on the west side and outcrops on the eastern side of the buried valleys. The bedrock ridge between the valleys at PC-08B at the OU1 property results in a significant reduction in the thickness of the saturated glacial outwash sediments. In cross section B - B', areas of maximum saturated thickness of overburden soils are located at PC-02 and west of PC-08.

Fill, as defined in Sections 3.2.1.1 and 3.2.2.1, covers most of the shallow surface in cross section B - B' and was identified in borings throughout the OU2 study area. Areas of maximum fill thickness noted on cross section B - B' are beneath I-95, at the former Raymark Facility where more than 20 feet of fill is noted, and at the former Raymark Memorial ballfield (OU4) where approximately 17.5 feet of fill is found at SB-427. Fill thickness varies across the OU2

study area depending on land use and history. Fill is underlain mostly by sand and gravel, with some layers of silt and peat; however, the fill lies directly on bedrock at the western edge of the ballfield property. Based on the water level elevations indicated on cross section B - B', portions of the fill underlying the former Raymark Facility and the ballfield are saturated during parts of the year.

A complex sequence of alternating layers of silt, peat, and sand with abundant localized gravel underlie the former Raymark Facility. The silt layer encountered at PC-10 is interpreted to continue across the facility to PC-02. The silt layer was encountered at similar elevation in PC-10, PC-08, and PC-02. The peat and associated sand at PC-02 indicate former marsh and swamp deposits. Silt noted at depth at PC-02, PC-08, and PC-10 may represent ice-contact sediments within the outwash sands and gravel. The silt and silty units have lower relative hydraulic conductivities than the peat and sand and gravel, and may cause retardation of downward contamination migration beneath portions of the former Raymark Facility. However, these layers are probably not an effective barrier to downward migration over the Facility.

Saturated glacial outwash sands and gravel are indicated throughout the cross section, with close to 40 feet present below finer grained units at PC-02B in the deeper portions of the bedrock valley. Over 20 feet of saturated gravel and sand are indicated beneath the fill from MW-401B extending eastward to PC-08B. These units may contain discontinuous layers of silt and gravel as well as cobbles and boulders. The sand and gravel units have a relatively high hydraulic conductivity and are a viable route of a potential groundwater contaminant transport.

Cross Section C - C'

Cross section C – C' roughly follows the water table groundwater flow direction beginning at the bedrock ridge upgradient of the former Raymark Facility, and transects many non-Raymark-related potential source areas. Refer to Figure 3-5 for cross section C – C'. This cross section traverses generally to the southeast across the bedrock valleys, which run northeast-southwest. Cross section C – C' includes soils data from the following locations: MW-307DB, PC-16B, PC-15B, SB-210 (old), MW-304B, MW-514B, MW-508B, and ends at MW-302B.

Cross section C - C' indicates that the buried bedrock topography is highly variable. The western side of the buried bedrock valley lies just below the surface in this cross section and is

part of a northeast-southwest trending ridge, with a maximum elevation of 51.72 feet. The boring data indicates bedrock highs within the valley at PC-15, SB-210 (old), and north of MW-302B. These high spots result in a significant reduction in the thickness of the glacial outwash sediments.

Fill covers the shallow subsurface through most of the cross section, but pinches out on either end. Areas of maximum fill thickness noted on cross section C - C' are beneath I-95 and at the former Raymark Facility, where more than 20 feet of fill is noted. Fill thickness varies across the study area depending on land use and history. Fill is underlain by sand and gravel throughout the majority of the study area, however at the western edge of the Contract Plating property, fill lies directly on bedrock.

Based on the water elevations indicated on cross section C - C', portions of the fill located at 540 Longbrook Avenue and the former Raymark Facility are saturated for at least some portion of the year.

Multiple layers of silt and sandy silt up to 60 feet thick are noted within cross section C - C'. At PC-16B, fill overlies almost 60 feet of sandy silt and silt. However, these materials do not extend to PC-15 to the southeast. This area has a lower hydraulic conductivity in general than the rest of the cross section and may retard migration of groundwater and associated contaminants, especially at depth. Discontinuous lenses of silt and clay are located at PC-15B, MW-304B, and MW-508B, and may retard groundwater flow to a much smaller extent.

Glacial outwash sands and gravel are indicated primarily east of PC-16, with over 40 feet present below fill at MW-304D, in the deeper portions of the southeast bedrock valley. These units contain discontinuous layers of silt and gravel as well as cobbles and boulders. The most extensive gravel layer in this cross section underlies the fill at MW-508B and MW-302B. The sand and gravel units have a higher general hydraulic conductivity and represent a potential contaminant pathway.

Glacial till overlies the bedrock at the bottom of the two deeper valleys southeast of PC-16B and at MW-514B. The till may act as a confining unit, preventing groundwater from the bedrock and overburden aquifers from mixing.

3.2.2.3 Study Area Bedrock Topography

Bedrock drilling and a seismic refraction study (Hager-Richter 1999a) were conducted to help define the topography of the bedrock surface beneath the OU2 study area. The contour map of the bedrock surface was created by combining the seismic refraction data with the bedrock elevation data compiled by TtNUS from numerous sources throughout the OU2 study area. In areas with few bedrock data points, shallower soil borings were used to constrain the minimum depth to bedrock. Bedrock elevation sources include data from the OU2 site investigations, boring logs, and outcrop mapping, presented in Appendix D. Location of the seismic lines are presented on the bedrock topographic map. Additional data was gathered from previous investigations conducted at OU1, geotechnical borings for the CT DOT for I-95, geophysical surveys and environmental site investigations at 540 Longbrook Avenue, 375 Barnum Avenue, and the Raybestos Memorial Ball Field. The map was constructed by hand using standard contouring techniques. Refer to Figure 3-6 for the bedrock topographic map.

The interpretation of the bedrock topography is indicated through the use of contour lines for every 10 feet of vertical change. The actual bedrock elevation between data points may be different than indicated on the contour map. Areas with a higher density of data points have more detail compared to areas with fewer data points.

A prominent feature of the bedrock topography within the study area is a set of bedrock valleys and ridges with more than 180 feet of vertical relief. The main valley starts near the intersection of Broad Street and Housatonic Avenue and continues toward the north. It splits into a northwest and southwest valley at MW-310, located near the corner of East Broadway and White Street. From MW-310, the north valley continues due north along the west edge of the 600 East Broadway property and the 411 Barnum Avenue cut-off property; then continues along the northwest edge of the former Raymark Facility, adjacent to the railroad tracks; and exits the former Raymark Facility property near the intersection of the railroad tracks and East Main Street. The valley is believed to continue to the north of a large bedrock outcrop observed at the end of Peck Street and northeast of 250 East Main Street. However, no borings deeper than 12 feet bgs exist in the area between MW-4B, MW-309B, MW-501B, and bedrock outcrop “A”.

The south valley starts near MW-310 and continues adjacent to I-95. The valley crosses I-95 in the vicinity of MW-517, then passes under the southeast corner of the former Raymark property and crosses East Main Street near the Sidney Street intersection. The valley then continues to the northeast, between the bedrock outcrop observed along I-95 that defines the southeast wall of the valley and the bedrock outcrop located to the northeast of 250 East Main Street that defines the northwest edge of the valley. However, this area contains few borings at depth and the bedrock topography is not well-constrained. The southern bedrock valley is narrow at depth, but widens at Homestead Avenue in the vicinity of MW-508B. This widening of the bedrock valley creates a bedrock depression in the residential neighborhood downgradient of the former Raymark Facility.

The bedrock ridge that separates the north and south valleys starts as a knob at MW-310B on East Broadway. The higher relief continues to MW-210, located behind the 411 Barnum Avenue cut-off property. The ridge then continues under the 411 Barnum Avenue cut-off and across the southeast portion of the former Raymark Facility, then crosses East Main Street near the entrance to the Dock Shopping Center, and continues to the bedrock outcrop located northeast of 250 East Main Street.

A depression in the bedrock ridge relative to the bedrock elevation in adjacent portions of the bedrock ridge was identified under the southeast portion of OU1. This bedrock depression runs between monitoring well clusters PC-12 and PC-14. At these locations the bedrock elevation is about -30 feet NGVD, which is 20 to 30 feet lower than bedrock elevations observed at monitoring well locations PC-15 and MW-524, respectively, located southwest of the PC-12 and PC-14 well clusters.

3.2.2.4 Study Area Bedrock Geology

This section provides an evaluation of the OU2 study area bedrock geology based on the review of referenced geologic maps, 69 boring logs from within the OU2 study area, and four test pits where bedrock was encountered. Bedrock outcrops (surficial exposures of bedrock) were identified and surveyed at 17 separate locations in the northern portion of the study area. The bedrock outcrops are shown on Figure 3-6. Additional outcrops were visually noted beneath I-95 at the Dock Property, but were not surveyed because of access issues. No

bedrock outcrops were found south of the property at 540 Longbrook Avenue, or at the Housatonic Avenue Extension and Ferry Boulevard.

The highest surveyed elevation of the bedrock surface is 51.72 feet NGVD at the western edge of the OU2 study area. The lowest bedrock elevation found within the OU2 study area is at MW-211B at –130.4 feet NGVD (150 feet below grade) located southeast of OU1 near the center of the OU2 study area. Bedrock underlying the OU2 study area is mapped as the Derby Hill Schist, a mainly medium- to fine-grained, thinly laminated, greenish-gray to medium dark-gray chlorite muscovite schist, which is Lower to Middle Ordovician in age. This rock type is composed mainly of quartz, muscovite, chlorite, and sodium plagioclase, with accessory minerals (Fritts 1965). The bedrock cores from within the OU2 study area were typically described as foliated, quartz-rich, chlorite-mica-schist with variable amounts of garnet and sulfide minerals such as pyrite. Typically, the bedrock is medium-grained and usually dark green or dark gray. Veins composed of both quartz and calcite were frequently observed.

High-angle schistosity and foliation were observed to be common; weathered fractures ranging from low-angle or horizontal up to high-angle and vertical were also noted. Many of the fractures noted were oriented parallel to foliation planes. Oxidation staining was common and zones of soft degraded bedrock were also noted. In addition to the bedrock fractures, vugs and pitting were noted in the bedrock. Slickensides (possible evidence of faulting) were noted at MW-216DB and at MW-306B along the western edge of the bedrock valley. At MW-214B and MW-401B, large open fractures were encountered, which consumed all drilling fluids used during the coring process.

The orientations of the bedrock fractures were determined from mapping bedrock outcrops and borehole geophysical methods. Based on rock outcrop observations, the strike of the foliation ranges from N35E to N55E. The foliation measured at several outcrops dips at high angles, ranging from 80° to near vertical. The predominant rock type in the bedrock outcrops was schist, with the primary fracture orientation parallel to foliation. Secondary fracture sets were noted at several outcrops with strikes of N30W and N35W, which is nearly perpendicular to the foliation and primary fracture orientation.

The nature and orientation of the bedrock fractures were also evaluated using borehole geophysical methods. These methods included, natural gamma, single-point resistance,

caliper, fluid temperature, fluid resistivity, borehole video, acoustic televiewer, and heat pulse flow meter. The heat pulse flow meter was conducted for both a static condition and a stressed condition. The borehole video and acoustic televiewer for the geophysical survey indicated that the bedrock encountered in the boreholes has been extensively folded, resulting in a complex pattern of bedrock fractures. The water transmitting fractures were identified from the geophysical survey. The borehole geophysical logging data is presented in the borehole geophysical report (Geophysical Applications 2003). A summary of the survey results is presented below.

Monitoring Well MW-213DB

Water-bearing fractures were identified at depths of 78 and 90 feet bgs. The fractures at 78 feet bgs have a dip angle of 10° to 40° to the southeast and northwest and a strike of north 23° to 40° east. The fractures at 90 feet bgs have a dip angle of 10° to 15° to the south to southwest and a strike of north 50° west.

Monitoring Well MW-215DB

Water-bearing fractures were identified at a depth of 33 feet bgs. These fractures have a dip angle of 65° to 78° to the southeast and northwest. The strike of these fractures is north 60° east. A set of potentially open fractures that did not produce very much water was identified at a depth of 45 to 54 feet. These fractures have a dip angle of 10° and a strike of north 40° to 60° east.

Monitoring Well MW-524DB

Water-bearing fractures were identified during the stressed flow test at depths of 44, 47.5, 58, and 79 feet bgs. The fracture at a depth of 44 feet bgs has a dip angle of about 45° to the south east and a strike of about north 45° east. The fracture at a depth of 47.5 feet bgs has a dip angle of about 70° to the south east and a strike of about north 45° east. The fracture at a depth of 58 feet bgs has a dip angle of about 40° to the south and a strike of about north 90° east. The fractures at a depth of 79 feet bgs have a dip angle of about 20° to 25° to the northeast with a strike of about north 45° west.

Surface Geophysical Methods

Surface geophysical methods (VLF survey and resistivity) were also used to determine the orientation of the bedrock fractures. The VLF survey uses low frequency radio transmissions broadcast by the United States Navy as part of the old communication system for submarines. Signals from this system can be used to locate bedrock fractures that are aligned with the source of the radio transmitter. However, usable VLF signals could not be detected at the site. The lack of usable detections could be the result of the cultural noise at the site, including overhead electrical power lines and buried electrical lines. The VLF survey was attempted at Raybestos Memorial Ball Field and along I-95 without any success.

Electrical resistivity imaging was also performed at the OU4 Raybestos Memorial Ball Field (see Figure 2-2). This method applies an electrical current into the ground using a series of spaced electrodes and measures the resistance between the electrodes. This data is then evaluated using a computer program to identify locations where the resistance is lower than adjacent areas. Areas of bedrock with lower resistance generally are the result of water-filled fractures. Therefore, areas of lower resistance indicate locations of potential water-bearing fractures.

The resistivity survey was conducted along a 690-foot long line that started near the access gate to the ball field and extended across the site almost to the opposite side of the site. The survey identified an area that potentially contains water-bearing fractures at a distance of about 320 to 360 feet from the west end of the survey line (Hager-Richter 1999b).

In summary, the bedrock fractures were identified within the OU2 study area using direct observation at bedrock outcrops and of cores retrieved during the drilling program, as well as using surface and borehole geophysical methods. The fractures identified at bedrock outcrops consist of two sets of fractures. The first or primary set of fractures are associated with the foliation in the bedrock and have a strike of N35E to N55E and have a high dip angle, 80° to near vertical. The secondary set of fractures has a strike of N30W and N35W, which is nearly perpendicular to the foliation and primary fractures. The observations made at the bedrock outcrops are consistent with the analysis of the borehole geophysical data indicating two sets of bedrock fractures; a northeast striking set and a northwest striking set.

3.3 Surface Water

The OU2 study area is located in the Housatonic Regional Drainage Basin. Long Island Sound receives the area's entire surface drainage via the Housatonic River. Ferry Creek and the Housatonic River are the major surface water features that lie within the OU2 study area; they are discussed in Sections 3.3.1 and 3.3.2, respectively. Large areas of wetlands are also included within the OU2 study area. The wetlands are discussed in detail in the Ferry Creek Remedial Investigation Reports (TtNUS 1999a, 2000b, and 2000d).

Both Ferry Creek and the Housatonic River receive water from the aquifers in the OU2 study area. Interactions between these surface water bodies and the groundwater are discussed in detail in Section 3.4.2.6.

3.3.1 Ferry Creek

Ferry Creek bisects the eastern portion of the OU2 study area. Although tide gates are present at the Broad Street crossing of Ferry Creek, these gates are ineffective at preventing backwater from high tides from passing upstream into Ferry Creek. Therefore, Ferry Creek is tidally influenced within the OU2 study area (ACOE 1998); see Appendix C. Ferry Creek is connected to Selby Pond by a pipe and open channel; Selby Pond is also tidally influenced. Ferry Creek is listed as an inland surface water of Class B/A (CTDEP 1997b). The B/A classification indicates its present condition (B), with designated uses including recreational; and the goal (A), with designated uses including potential drinking water supply. It is a tidally influenced, brackish stream with a fresh and salt water mix from the mouth of the Ferry Creek/Housatonic River to north of the confluence of Ferry Creek and Long Brook Creek. As a result of tidal influences, the surface water elevation in the creek changes by approximately 1 to 5 feet.

Based on a groundwater seepage study conducted in 1999 (see Section 3.4.2.6), it was determined that contaminated groundwater is discharging to Ferry Creek.

3.3.2 Housatonic River

The Housatonic River is listed as coastal and marine surface water, Class SC/SB, with an average discharge of 3,400 cubic feet per second at its mouth. The SC/SB classification

indicates its present condition (SC), indicating unacceptable quality because of problems that can and will be corrected by normal CTDEP programs, and goal condition (SB), with uses including recreational (CTDEP 1997a; Weston 1993). The Housatonic River runs north to south through the State of Connecticut and discharges to Long Island Sound, located south of the OU2 study area. The Housatonic River is extensively used for recreational activities and some light commercial and tourist operations. Tidal influences extend about 11 miles upstream from the mouth of Ferry Creek, to the Derby Dam in Derby, Connecticut (Weston 1993).

3.4 Hydrogeology

This section provides a description of the regional and OU2 study area hydrogeology focusing on groundwater under, in the vicinity of, and downgradient of the former Raymark Facility.

3.4.1 Regional Hydrogeology

Regional hydrogeologic units consist of both overburden deposits and bedrock. The overburden units include fill, till, stratified outwash, and swamp and marsh deposits. The bedrock aquifer consists of fractured schist. The regional groundwater flow direction is generally toward the Housatonic River.

3.4.1.1 Regional Overburden Aquifer

Based on the subsurface geology, the overburden aquifer was divided into the stratified drift aquifer and the till aquifer in the water resources investigation conducted by Wilson et al. (1974). On a regional basis, the stratified drift aquifer consists of the “ice-contact stratified drift” and “glacial outwash deposits” described in Section 3.2.2.1, and is volumetrically the most significant aquifer in terms of water supply. As stated in Section 3.1.2, the overburden aquifer is not utilized for drinking water within the OU2 study area.

Within the lower Housatonic River Basin, the thickness of the stratified drift aquifer varies from about 10 feet in many small valleys to 200 feet within larger valleys. The boundaries of the stratified drift aquifer generally consist of underlying till and/or bedrock and, occasionally, overlying peat and organic silt deposits, which can locally serve as confining layers. Estimated values for the transmissivity of the stratified drift aquifer within the lower Housatonic River Basin

range from 2,700 $feet^2$/day in headwater areas, small valleys, and along the margins of larger valleys, to 20,000 $feet^2$/day in parts of the Naugatuck and Housatonic River valleys (Wilson et al. 1974).

The absence of stratification and sorting gives the till aquifer its characteristic low hydraulic conductivity and limits the use of this aquifer as a water source. The median till thickness in 240 bedrock wells within the lower Housatonic River was 30 feet (Wilson et al. 1974).

The primary source of recharge to the overburden aquifer is through the infiltration of water from precipitation. Between 1931 and 1960, annual precipitation on the lower Housatonic River Basin ranged from 33 inches to 64 inches and averaged 47 inches. Approximately half of this precipitation returns to the atmosphere by evapotranspiration (Wilson et al. 1974). The remainder is divided between surface water runoff and infiltration. Surface infiltration is impeded throughout much of the region because a large portion of the area is paved or built on, with surface flow diverted through storm drains. Infiltration characteristics within the OU2 study area are discussed in Section 3.1.1.

3.4.1.2 Regional Bedrock Aquifer

The bedrock characteristics of the regional bedrock aquifer are discussed in Section 3.2. A total of 294 bedrock wells located within the lower Housatonic River Basin were studied by Wilson et al. (1974) to determine the range of yields of bedrock wells within the basin. Yields varied from less than 1 gallon per minute (gpm) to more than 20 gpm. The median yield in bedrock wells directly overlain by stratified drift was 7 gpm. Bedrock wells overlain by till, which has a lower hydraulic conductivity than stratified drift, had a median yield of only 5.5 gpm. This suggests that a hydraulic connection exists between the overburden and bedrock within the basin, and that the hydraulic conductivity of the material overlying the bedrock has an effect on the amount of water available to bedrock wells.

Groundwater flow in the bedrock generally occurs within fractures (secondary porosity). The magnitude and direction of groundwater flow within the bedrock depends on the size, spacing, connection, and orientation of fractures and/or faults within the bedrock.

3.4.2 Study Area Hydrogeology

This section presents the site hydrogeology based on the site-specific information gathered from the overburden and bedrock aquifers during the OU2 site investigation, including, groundwater flow directions, groundwater/surface water interactions, hydraulic conductivity data, interaction of the overburden and bedrock aquifers, and generalized groundwater flow directions in the bedrock.

Groundwater in the vicinity of the OU2 study area is classified as GB, or unsuitable for drinking without treatment (CTDEP 1997a). The Town of Stratford's public drinking water is supplied primarily by the Aquarian Water Company (formerly BHC). The source of the public drinking water is Trapp Falls Reservoir in Shelton, Connecticut, located approximately five miles from the study area (Aquarian 2003). The remainder of the drinking water is supplied by private drinking water wells within Stratford, none of which is known to exist near the study area (Stratford 2003). Current public water supply for the OU2 study area is described in greater detail in Section 3.1.2.

3.4.2.1 Hydraulic Conductivity

The hydraulic conductivity of the overburden and bedrock aquifers was determined using constant discharge single well testing, packer testing, and a two-day aquifer test. Each of these tests is discussed below.

Single Well Pump Tests

The single well constant discharge tests were conducted in both overburden and bedrock monitoring wells using high and low pumping rates. The low pumping rate single well pumping test was performed after low stress groundwater sampling was completed at a monitoring well. In 1999, peristaltic and Grunfos pumps were used; in 2003, bladder and, rarely, peristaltic pumps were used. The procedure for the test was to increase the pumping rate at the monitoring well after the groundwater samples had been collected and measure the drawdown in the monitoring well until stabilized drawdown was reached in the well. If the monitoring well was installed in a unit with very low conductivity, the hydraulic conductivity was calculated from the drawdown that occurred during the course of low-stress sampling. This testing method

provides an estimate of the hydraulic conductivity of the aquifer by removing a larger volume of water compared to a slug test. The larger stress on the aquifer from the larger volume of water removed results in an estimate of hydraulic conductivity that is representative of a larger portion of the aquifer.

The second single well test method used a centrifugal pump, whale pump, or Waterra pump. The tests were conducted as part of the monitoring well development process. The three pumps used were capable of pumping water at a much higher rate than the pumps used for the low stress sampling. The centrifugal pump was used when the aquifer materials could supply water to the well at a rate that would prevent the well from being pumped dry or the water level from dropping below the maximum depth that the pump could lift. The whale pump was used instead of the centrifugal pump at locations where the depth to water was greater than 20 feet or when a lower pumping rate was necessary. The Waterra pump was used at locations with extremely low water levels and/or extremely low hydraulic conductivities.

The single well hydraulic conductivity tests were analyzed using methods presented by Lambe and Whitman (1969). A summary of hydraulic conductivity values is presented in Table 3-1. The single well hydraulic conductivity values ranged from a high of 518.33 feet/day (MW-217S) to a low of 0.02 feet/day (MW-211B). Table 3-1 shows that the higher hydraulic conductivity values, with the exception of MW-214B, a bedrock well, were typically measured in monitoring wells screened in sand and sandy gravel. A table of the pumping test observations including pumping rates and observed drawdown is presented in Appendix D.

The lower hydraulic conductivity values (less than 5 feet/day) were measured in the wells screened in the finer grained aquifer materials such as silty sands and bedrock. These values for hydraulic conductivity are within the typical range of values for these types of aquifer materials.

Packer Tests

Selected borings in bedrock were tested for hydraulic conductivity using double-packer injection methods. The data gathered during the packer injection tests was evaluated using methods presented in the Ground Water Manual (USBOR 1977). The packer test data included three or four tests at each depth interval using different injection pressures.

The results of the packer injection tests and the calculated hydraulic conductivity values should be used as estimates of hydraulic conductivity in the bedrock, and not the actual conductivity of the individual fractures. The analytical methods used to calculate the hydraulic conductivity from the injection test data assume that the entire length of the test interval accepts the injected water. However, only a few very small fractures in the bedrock actually transmit groundwater. These fractures have a much higher hydraulic conductivity than the calculated value. Therefore, caution should be used when using the calculated hydraulic conductivity values to estimate movement rates of both the groundwater and the groundwater contaminants through the fractured rock.

The bulk hydraulic conductivity calculated from the packer tests ranged from a low of 0.0 to a high of 6.5 feet per day, calculated for the bedrock at monitoring well cluster MW-308. Bulk hydraulic conductivity values of approximately 1 foot per day were calculated at four locations, monitoring well clusters MW-302, MW-307, MW-309, and MW-312. The other hydraulic conductivity values calculated were below 1 foot per day. The packer test hydraulic conductivity values are presented in Appendix D.

The hydraulic conductivity of the bedrock measured through the vertical borings may also result in an underestimation of the actual conductivity of the bedrock. As described in Section 3.2.2.4, the fractures developed in the foliation and cross-foliation tend to be at high, near vertical angles. This means that vertical borings may miss many fractures that convey groundwater, resulting in an underestimate of the bulk conductivity of the bedrock mass.

Aquifer Test

An aquifer test was conducted behind the 411 Barnum Avenue cut-off property in May 2003 (see Figure 2-1). The aquifer test was conducted to test the response of surrounding overburden and bedrock wells to the removal of water from a primary well. First, a 14-inch boring was drilled to the top of bedrock. An 8-inch inside diameter, 20-foot long stainless steel wire wound well screen with a 0.03-inch slot opening was installed. Once the well was developed, transducers were installed in the test well and 25 observation wells at varying distances from the test well. The observation wells included bedrock as well as shallow and deep overburden wells. See Figure 2-3 for well locations.

Prior to the actual aquifer test, the transducers were allowed to run for a minimum of 24 hours to determine the pre-pumping groundwater elevation readings. Conditions that could influence the observation wells, such as tidal influences (indicated by groundwater changes on a 6-hour cycle), and precipitation (indicated by a rise in groundwater elevation), were recorded.

After the collection of the pre-pumping data, the test well was pumped and the water levels in the observation wells were recorded for 48 hours of pumping. After pumping stopped, the water levels were recorded until they recovered to near pre-pumping levels. Once the aquifer test was complete, the data were graphed (water elevation vs. time) for each observation well. A decline in water levels during the test greater than the background decline in water levels, if any, indicated that the observation well had responded to the stress in the aquifer caused by the pumping test. Graphs of changes in water levels over time for the observation wells are presented in Appendix D.

A review of the monitoring well hydrographs indicates that several of the monitoring wells did respond to the stress caused by pumping the test well, including the MW-505 cluster (25 feet away from the test well), the MW-504 cluster (50 feet away), the MW-304 cluster (125 feet away), the MW-502 cluster (350 feet away), and the MW-210 cluster (500 feet away).

The primary objective of the aquifer test was to provide better estimates of the hydraulic conductivity and storage in the overburden aquifer. A secondary objective was to determine the nature of the hydraulic connection between the overburden and bedrock aquifers, if any. For details about the connection between the bedrock and overburden, refer to Section 3.4.2.5.

The analysis of the aquifer tests indicated that the transmissivity of the overburden aquifer is between 30,500 and 45,700 ft^2 per day with a hydraulic conductivity of between 600 and 700 feet per day. These values compare favorably with the single well test data from MW-304D and MW-505D that indicate a deep overburden hydraulic conductivity in the vicinity of the aquifer test well in the order of hundreds of feet per day. The hydraulic conductivity values also compare favorably with the type of aquifer material encountered in the location of the aquifer test well. For a detailed explanation of the aquifer test, graphical results, and analysis, see Appendix D.

The storativity of the overburden aquifer is estimated at 7.9E-02 to 4.7E-01. These storativity values indicate that the overburden aquifer is unconfined. The storativity value is within the range of values for an unconfined aquifer. This is not an unexpected result given that the coarse sands and gravel in the bedrock valley where the aquifer test was conducted are generally overlain by finer sands and in some cases silty sands.

3.4.2.2 Water Elevations and Lateral Overburden Groundwater Flow

Synoptic water level rounds were completed during previous site investigations. However, these rounds did not include all of the wells on the site. The water level round conducted in 1999 included all wells available, but new wells were constructed after the completion of that study. The April 2003 synoptic round is discussed in detail since it represents the most complete data set and utilized the largest number of monitoring wells. The water table map constructed from the 1999 synoptic round of water levels is presented in Appendix D. Refer to Figure 3-7 for the water table map from the April 2003 synoptic water level round.

During the April 2003 synoptic round, water level measurements were taken in all available monitoring wells in the study area, including wells on commercial properties. The water level measurements were taken in as short a period of time as possible, starting near the Housatonic River and Ferry Creek to attempt to lessen the impacts of tidal influence, if any, on the data. Table 3-2 summarizes the water level elevations for the well clusters and indicates the vertical gradients. The March 1999 groundwater elevation and vertical gradient data are also presented in Table 3-2 for comparison. A table with the 1999 and 2003 groundwater elevations, as well as data from previous water level rounds, is presented in Appendix D.

The data gathered during the April 2003 synoptic water level round were used to construct two maps that depict ground water flow within the study area. These maps include a water table map and a -40 potentiometric map. Both maps show slices of the OU2 study area and are indicators of horizontal flow only. For a discussion of vertical flow, refer to Section 3.4.2.3.

Water Table Map

The water table map was constructed using monitoring wells screened across the water table or near the water table. In some cases the water table is located very close to the ground surface,

which makes installation of a well screen difficult. At these locations well screens had to be set just below the water table to allow for installation of the protective casing to secure the monitoring well from unauthorized entry or the introduction of surface water into the monitoring well. Water level elevations of selected bedrock monitoring wells (MW-216B, MW-303B, MW-307B, and MW-308B) were used where the water table is located below the bedrock surface.

A review of the water table map (Figure 3-7) indicates that groundwater flows from the area northwest of the former Raymark Facility to the southeast and south across the property. From the Facility, the groundwater flow directions vary depending on which portion of the OU2 study area is being evaluated. Therefore, the groundwater flow directions at the water table are discussed for various portions of the OU2 study area.

Groundwater from the northeast end of the former Raymark Facility along East Main Street flows east to southeasterly under East Main Street and toward the Housatonic River. The water table in this portion of the study area is flat and the groundwater is expected to move very slowly toward the river.

Groundwater at the northeast corner of the former Raymark Facility near East Main Street and Route 1 flows generally southeast toward the Housatonic River.

Groundwater in the central portion of the former Raymark Facility flows generally south under I-95 and continues under the residential neighborhood southeast of the Facility. Most of the groundwater under the neighborhood flows to the Housatonic River.

Groundwater from the southwest portion of the former Raymark Facility flows southeast to 375 Barnum Avenue and then turns south to flow under I-95. Past I-95, the groundwater flow direction splits, with a portion flowing under the southern end of the residential neighborhood near Willow Avenue and Homestead Avenue toward the Housatonic River, and the rest flowing southwest toward Ferry Creek.

At MW-110S, the water level is consistently higher than in the surrounding monitoring wells, indicating an area of groundwater mounding believed to be caused by artificial recharge from a leaching field. The groundwater mound is expected to extend some distance from the leaching

field. The leaching field's area of influence is not known and therefore the data from this monitoring well were not used to construct the water table map.

As shown by the water table contours, Ferry Creek has a strong influence on local groundwater levels. The type and thickness of materials that underlie Ferry Creek, and the difference between surface water and groundwater levels, control the actual rate of groundwater discharge into the creek. Discharge of groundwater to Ferry Creek was confirmed by a seepage study conducted in March 1999, which estimated a discharge rate of groundwater from the shallow overburden into the creek throughout a tidal cycle. This study is discussed in Section 3.4.2.6.

The variations in the shallow groundwater flow directions in the OU2 study area are in part caused by the variations in the thickness of the overburden aquifer materials and the effects of deep overburden flow.

– 40 Potentiometric Groundwater Map

The flow directions in the deep overburden aquifer were determined with the use of a –40 foot NGVD potentiometric map (see Figure 3-8). The map was constructed using groundwater elevation data from monitoring wells with well screens set at or about elevation –40 feet NGVD. This elevation was chosen because numerous monitoring wells in the OU2 study area have screens set at or near –40 feet NGVD.

The screen interval for a monitoring well intended for placement in the deep overburden was determined by evaluating the results of chemical analysis of soil samples collected during the advancement of the bedrock (first) boring at a well cluster. The interval with the highest VOC concentration as determined from the laboratory analysis was selected for monitoring, and the well screen was installed at this location. If VOCs were not detected in the soil samples, the well screen interval was selected from the portion of the overburden aquifer expected to have the highest hydraulic conductivity, based on the visual description of the soils encountered. Therefore, the large number of wells screened at or near –40 feet NGVD indicates that this elevation has in general a higher concentration of VOCs and/or has a higher hydraulic conductivity and is likely a pathway for deeper overburden groundwater flow.

The –40 potentiometric map was drawn by first defining the edges of the overburden aquifer at the –40 foot NGVD elevation. This was done by using the –40 foot topographic contour line from the bedrock topographic map. Refer to Figure 3-6 (Bedrock Topographic Map) for the bedrock contours. The groundwater elevations measured in overburden monitoring wells with well screens at or near elevation –40 feet NGVD were then plotted on the figure. In addition, groundwater elevations in bedrock wells located outside the bedrock valleys were also used to construct the map. The bedrock groundwater elevations were used to help determine where the overburden potentiometric contours would intersect the bedrock valley wall. The construction of Figure 3-8 assumed that the hydraulic conductivity of the bedrock aquifer is much less than the hydraulic conductivity of the overburden aquifer. This assumption is supported by the hydraulic conductivity data gathered from the bedrock monitoring wells and packer tests. There is one notable exception at bedrock monitoring well MW-214B, which has a bulk hydraulic conductivity of 475 feet per day.

The –40 potentiometric map presents the two bedrock valleys as described on the bedrock topographic map. These northeast trending valleys are separated by a bedrock high that was encountered under the former Raymark Facility and part of the 411 Barnum Avenue cut-off property. The two valleys converge near monitoring well MW-210, located behind the 411 Barnum Avenue cut-off property, and continue toward the mouth of Ferry Creek. These valleys are believed to control the bulk of the overburden groundwater flow.

A review of Figure 3-8 indicates that groundwater from the north valley, located parallel to the railroad tracks north of the former Raymark Facility, flows southeasterly across the Facility. Groundwater flows through both the overburden and bedrock aquifers. For a detailed discussion of flow in bedrock, refer to Section 3.4.2.4. Both the water table map (Figure 3-7) and the –40 potentiometric map indicate that groundwater flows southeast across the southwest portion of the former Raymark Facility. The bedrock topographic map (Figure 3-6) indicates that a depression in the bedrock ridge that separates the north and south bedrock valleys is located at monitoring wells PC-12 and PC-14. The bedrock elevation at these locations is about –30 feet NGVD, which is only slightly above the screen elevations used to construct the –40 potentiometric map. The groundwater elevations from PC-12D (4.79 feet NGVD) and PC-14D (5.95 feet NGVD), which have screen midpoints at about 25 feet NGVD, indicate that groundwater in the overburden at the top of bedrock is flowing in a general southeast direction within this bedrock depression. Therefore, the bedrock depression in the

bedrock ridge allows overburden groundwater to flow toward the southeast across this portion of the former Raymark Facility.

The monitoring wells located in both the north valley and the bedrock ridge under the former Raymark Facility indicate that groundwater in the bedrock flows toward the southeast across the ridge; however, the north bedrock valley continues to the southwest. This groundwater flow direction is also supported by the groundwater chemistry in the monitoring wells within the former Raymark Facility and at the 411 Barnum Avenue cut-off property. Refer to Section 4.0 for a more complete discussion of the nature and extent of contaminants.

Groundwater in the bedrock aquifer southwest of the former Raymark Facility at an elevation of –40 NGVD flows generally southeast into the southern valley. Once the groundwater enters the south valley, the flow direction turns to the southwest and toward Ferry Creek. Figure 3-8 also indicates that some groundwater discharges from the overburden valley into the bedrock aquifer along the walls of the valley and flows across the residential neighborhood to the Housatonic River. The actual rate of groundwater movement into the bedrock is not known. However, the distribution of groundwater contaminants in the bedrock aquifer supports this interpretation; refer to Section 4.0 for a further discussion.

Some of the groundwater in the southern valley flows into a small valley in the vicinity of Homestead Avenue. However, the contours from groundwater elevations observed in monitoring wells MW-214B, MW-302B, and MW-508B indicate that this side valley is not a main flow path for the groundwater in the south valley. The bulk of the groundwater in the south valley continues to the south-southwest toward Ferry Creek.

The groundwater in the south valley continues flowing to the south-southwest and toward Ferry Creek. However, the groundwater elevation data from monitoring wells MW-503M, MW-104D, MW-110D, and MW-111D indicates that the groundwater flows past the intersection of Ferry Creek and the bedrock valley. The data from these wells and surrounding wells indicate that the deep overburden groundwater flow from the south valley continues south and converges under Ferry Boulevard with water entering the bedrock valley from the west and from the east. From this point, groundwater continues flowing south under Ferry Boulevard.

3.4.2.3 Vertical Gradients

The vertical gradients at well clusters were calculated for both the March 1999 and April 2003 synoptic groundwater monitoring rounds between vertically adjacent wells in each well cluster. For example, the gradients were calculated for the bedrock well and deep overburden well and for the deep and intermediate well. The gradient was calculated by dividing the difference in groundwater elevation by the vertical distance between well screens as measured by the elevation of the center of the well screen. Refer to Table 3-2 for details.

The direction of the vertical gradient is caused by a number of conditions, including the location of the well cluster and the nature of the bedrock and overburden aquifers at each location. The vertical gradient can be the result of changes in the saturated thickness of the overburden and the nature of the bedrock fractures. As indicated on Figure 3-6, the bedrock topography varies over the OU2 study area. Areas with strong positive gradients (greater than 0.01) from bedrock to deep overburden (recharge upward) include MW-4, MW-304, MW-501, and MW-506. All of the wells in these clusters are located on the sides of bedrock ridges (see Figure 3-6 for details), indicating that groundwater is potentially moving up into the overburden from the bedrock as it encounters the more resistant bedrock material, which makes up the ridges. See Section 3.2.2.3 for a discussion of bedrock topography. The maximum vertical gradient was seen in PC-11 (0.81), which was much higher than any of the other gradients. This strong upward gradient may indicate that the bedrock monitoring well intersected a bedrock fracture or fracture set that is receiving groundwater recharge from an area upgradient of the former Raymark Facility.

Areas where the bedrock and deep overburden wells have strong negative gradients (less than – 0.01), indicating that groundwater has the potential to recharge from the overburden into the bedrock, include MW-507, MW-514, PC-12, and PC-15. Till is present at MW-514, indicating that the high vertical gradient may be the result of the till acting as a confining layer. All of the other well clusters with strong negative gradients are located in the central portion of the site, roughly in line with the saddle in the center of the middle bedrock ridge.

Several of the well clusters in the OU2 study area also have strong positive gradients between the deep and shallow overburden wells. Clusters with strong positive gradients, indicating the potential for upward movement, include MW-112, MW-217, MW-313, MW-503, MW-505,

MW-506, PC-09, and PC-11. Two of these locations (MW-506 and PC-11) also have strong positive gradients from the bedrock to the deep overburden, indicating that recharge from the bedrock may be significant enough to affect the groundwater in the shallow overburden. Other areas with strong positive overburden gradients are scattered throughout the OU2 study area and may reflect changes in local topography.

Areas with strong negative gradients, indicating downward movement between the deep and shallow overburden wells, include MW-101, MW-104, MW-111, MW-311, MW-312, MW-501, and PC-14. The high downward gradient at the MW-110 cluster is caused by artificial recharge into the shallow overburden from the leaching field located adjacent to the monitoring wells. Two other clusters (MW-313 and MW-501) are located on topographic highs on the southwestern and northern portions of the study area. The other clusters are located within the southern portion of the OU2 study area where the bedrock valleys widen. This expansion increases the volume of overburden material substantially. Groundwater entering this area tends to spread out, causing the groundwater to move downwards between the shallow and deep overburden, barring the presence of confining layers that would prevent groundwater movement.

For a discussion of groundwater flow in three dimensions, refer to Section 3.4.2.7.

3.4.2.4 Bedrock Groundwater Flow

Information in this section is qualitative in nature, as the actual path that groundwater takes through the bedrock is determined by the nature and orientation of discrete fractures. To determine the orientation of the bedrock fractures, surface and borehole geophysical investigation methods were used. Bedrock cores were also used to support a qualitative description of the fractures in the bedrock; fracture orientations could not be preserved when the cores were removed.

The surface geophysical methods used included electrical resistivity imaging (Geophysical Applications 2002). Resistivity data was obtained using electrodes placed at a five-meter spacing at the Raybestos Memorial Field (OU4), located north of the former Raymark Facility. The electrical resistivity survey identified a potential water bearing zone in the bedrock. However, additional resistivity surveys could not be performed to trace this potential fracture

zone across the OU2 study area because of cultural interference, including buried and overhead utilities. The location of the resistivity survey is presented on Figure 2-2.

As discussed in Section 3.2.2.4, the bedrock within the OU2 study area is highly fractured. Rock core samples show evidence of groundwater movement through the fractures, such as oxidation staining in the rock cores. In several locations, such as MW-213 and MW-215, the drill rig observations such as drops when drilling (indicating voids or soft zones in the bedrock) or quantities of water lost to the formation indicate the presence of fractures. Refer to the boring logs presented in Appendix A. Hydraulic conductivity for the bedrock aquifer was estimated using data from single well pump tests and packer tests. Hydraulic conductivities are discussed in Section 3.4.2.1 (Hydraulic Conductivity), and a summary of hydraulic conductivity values is presented in Table 3-1.

The groundwater elevations in the bedrock aquifer range from 2.99 feet NGVD and 2.73 feet NGVD (MW-312B and MW-113B in the residential neighborhood southeast of the former Raymark Facility) to a high of 26.84 feet at monitoring well MW-308DB, located north of the former Raymark Facility. This indicates that in general, the groundwater elevations in the bedrock aquifer decrease from the west to east across the OU2 study area and that groundwater is flowing toward the river and creek. A further complicating factor in predicting the actual groundwater flow paths in the bedrock aquifer is caused by the deep bedrock valleys and ridges identified during the site investigation. The groundwater elevation data indicates that groundwater discharges into the northern bedrock valley. Most of this water continues across the valley and flows across OU1 in the overburden aquifer. Refer to Section 3.4.2.2 for overburden groundwater flow details.

Groundwater in the bedrock aquifer under the former Raymark Facility flows to the east and south east into the south bedrock valley, where it also mixes with the groundwater already in the valley. The groundwater elevation data also indicate that some of the overburden groundwater in the valley enters the southern valley wall, where it flows across the residential neighborhood. The actual path that the groundwater takes through the bedrock is difficult to determine, as the groundwater flows through discrete fractures in the bedrock.

3.4.2.5 Overburden - Bedrock Interaction

The interaction between the overburden and bedrock aquifers was evaluated using data collected during the field investigation. These data include geologic mapping of the bedrock outcrops, geophysical investigations, hydraulic conductivity testing of the bedrock, vertical gradients established from the synoptic groundwater monitoring event, the 48-hour aquifer test, and groundwater sampling results.

Several lines of evidence indicate that the bedrock does not act as a separate aquifer and that groundwater in the overburden has the potential to move into and out of the bedrock. First, as shown by surficial geophysical investigations, geologic mapping of bedrock outcrops and the boring logs for the bedrock monitoring wells, the bedrock has numerous fractures. Fractured bedrock was noted in borings and outcrops scattered throughout the OU2 study area. Fractures had two orientations; one with a strike of N35E to N55E, parallel to foliations in the bedrock; and one with a strike of N30W to N35W, nearly perpendicular to the foliations. Borehole geophysics performed on three of the bedrock borings indicates that at least some of these fractures transmit water. See Section 3.2.2.4 for bedrock geology details. Till is a potential confining unit, which could prevent the interaction of bedrock and overburden groundwater. In the OU2 study area, till was located overlying the deepest portions of the bedrock valleys in only a few borings. See Section 3.2.2.1 for details.

Single well pump test and packer tests were conducted to determine the hydraulic conductivity of the bedrock and overburden. In most cases, the hydraulic conductivity of the overburden wells in a cluster were orders of magnitude higher than the hydraulic conductivity of the bedrock wells. However, three bedrock wells had hydraulic conductivities over 50 feet/day, and one well (MW-214B) had a hydraulic conductivity of almost 500 feet/day. The three wells were located in the bedrock ridges east and west of the main bedrock valley, indicating that zones of fractures may exist, which allow groundwater to move into the bedrock from the thick overburden aquifer in the valley. See Table 3-1 and Section 3.4.2.1 for hydraulic conductivity details.

The connection between the bedrock and overburden aquifers was also evaluated by conducting a 48-hour aquifer test. The aquifer test consisted of pumping the overburden aquifer and measuring the response at overburden and bedrock monitoring wells located at

various distances and directions from the pumping well. The aquifer test data is presented in Appendix D. For a detailed discussion of the aquifer test, refer to Section 3.4.2.1.

A review of the monitoring well hydrographs indicates that several of the bedrock monitoring wells had a measurable response during the pumping of the test well. These wells include MW-208B, MW-210B, MW-211B, MW-212B, MW-213DB, MW213B, MW-304B, MW-502B, MW-504B, and MW-505B. The hydrographs for monitoring wells MW-514B and MW-508B indicate that there may be some response from pumping, but the response is not as clear compared to the response observed in the other monitoring wells. Also, monitoring well MW-302B did not indicate any change in groundwater elevation that could be attributed to the pumping of the test well. Monitoring well MW-505B was the closest to the test well at a distance of 34 feet; MW-302B was the furthest away at a distance of 1,320 feet. Drawdown in the bedrock wells affected by the stress caused by the aquifer test ranged from 0.2 feet in MW-210B to 0.8 feet in MW-505B. The bedrock response to the stress originating from the overburden aquifer indicated a hydraulic connection between the overburden and bedrock aquifers at those locations. Groundwater elevation for all the wells monitored as part of the aquifer test is presented in Appendix D.

The vertical gradients between the bedrock and overburden monitoring wells indicate the direction of water movement. Areas with strong positive gradients between the bedrock and deep overburden wells (greater than 0.01) indicate potential groundwater movement from the bedrock up into the overburden (see Section 3.4.2.3 for details). The groundwater is potentially moving up into the overburden from the bedrock as it encounters the more resistant bedrock material, which makes up the ridges. See Section 3.2.2.3 for a discussion of bedrock topography. Areas with strong negative gradients, indicating water moving from the overburden down into the bedrock, are located in the central portion of the site, roughly in line with the saddle in the center of the middle bedrock ridge. The areas with strong negative and positive gradients between the overburden and bedrock are more likely to have groundwater movement between the aquifers and mixing of potentially contaminated groundwater. Refer to Table 3-2 for a summary of the vertical gradients at the site. For a complete discussion of the vertical gradients, refer to Section 3.4.2.3.

In summary, the data gathered during the field investigation indicate that the bedrock aquifer transmits groundwater and the overburden and bedrock aquifers are hydraulically connected in

most areas of the OU2 study area. The bedrock coring, outcrop mapping, and geophysical investigation identified bedrock fractures, and the hydraulic conductivity tests confirmed that the bedrock aquifer has the capability to transmit groundwater. The aquifer test results and the vertical gradients encountered indicated that the overburden and bedrock aquifers are connected. In addition, groundwater sampling results indicate a connection between the aquifers. For the groundwater sampling results, see the discussion in Section 4.0.

3.4.2.6 Groundwater/Surface Water Interactions

Ferry Creek and the Housatonic River are the major surface water features that lie within the OU2 study area. Potentially contaminated groundwater from the OU2 study area flows into both. Seepage studies in Ferry Creek and tidal studies for the river and creek were conducted to study the interaction between the surface water and groundwater in the OU2 study area. The seepage studies were conducted to estimate the amount and rate of groundwater and contamination discharge from the streambed into the creek. The tidal studies were conducted to test whether tide changes in the Housatonic River and Ferry Creek affect the groundwater elevations and flow directions. See Section 3.3 for a discussion of the physical characteristics and classifications of both surface water bodies.

Ferry Creek Seepage Studies

Seepage studies were conducted in Ferry Creek during January to February 2003 and in March 1999 to estimate the amount and rate of groundwater contamination from VOCs and SVOCs discharging into Ferry Creek. The 2003 seepage study was conducted in Ferry Creek on the north side of I-95 behind the 411 Barnum Avenue cut-off. Four seepage meters (two seepage meters and a duplicate upstream, and an experimental seepage meter in a marginal area downstream), were installed in locations identified as potential groundwater discharge areas. However, the Ferry Creek stream bed is almost entirely covered with debris in this area, and the only areas where the seepage meters could be installed had a loose layer of organics up to 2 feet thick. The stream bed material caused the groundwater to move laterally through the loose material instead of filling the sample bags of the seepage meters. Although the seepage meters were left in place for up to 48 hours, the sample bags never filled with water. After removal of the seepage meters, five piezometers were installed in Ferry Creek. The piezometers gave little to no water because of the high percentage of clay and silt and were not sampled. Two

surface water samples were collected from the creek downstream (adjacent to the culvert at I-95) and upstream (south of MW-510M) of the seepage meters. Five piezometers (MW-511M, MW-510M, MW-512M, MW-516M, and MW-509M) were also installed north of Ferry Creek. These piezometers were screened below the organic layers and were sampled with the other groundwater wells.

The March 1999 seepage study consisted of installing 13 seepage meters at 12 locations along the lower reach of Ferry Creek, south of I-95, and collecting four surface water samples from the creek. Seepage meter 2 and 2A were collocated to provide duplicate groundwater samples to meet the analytical data quality control needs. Figure 2-3 presents the location of the seepage meters and the surface water sample locations.

Upon completion of the field study, the groundwater discharge rates for the entire creek were estimated. The first step in estimating the discharge rates was to define the areas within Ferry Creek represented by each seepage meter so that the data from the meters could be integrated over the entire length of Ferry Creek. The seepage meter results, visual evidence of groundwater seepage along the creek banks, and bottom conditions in the creek during low tide allowed for a reasonable assignment of seepage meter results to segments of the creek. The seepage rate measured at each seep meter was applied to that area of the creek that had similar groundwater seep values during periods of low tide.

The highest total VOC concentration was 1,889 µg/l at SM-2. This location also had the highest seepage rate. The data also shows that the highest VOC concentrations were detected in the upper reaches of the creek between I-95 and Ferry Boulevard. These higher concentrations are associated with most, but not all, of the higher seepage rates. The complete chemical data are presented in Appendix D.

The results of the surface water sample analysis indicate that VOCs are present in the surface water in the upper reaches of Ferry Creek. The concentrations of VOCs in the seepage meter samples are one to two orders of magnitude higher than the VOCs in the surface water samples. The order of magnitude difference is probably because of dilution of the contaminants in the groundwater discharging at the base of the creek.

The results of the surface water and groundwater analysis for SVOCs indicated that this class of compounds was not present in the surface water or groundwater with the exception of a few low concentrations, 6 µg/l or lower, at seepage meters SM-2, SM-2A, SM-3, and SM-4. The SVOCs detected at the levels seen may be because of artifacts of the field filtering process. The conclusions from the 1999 Ferry Creek seepage study are as follows:

- Groundwater discharging to Ferry Creek as base flow discharge is contaminated with VOCs.

- SVOCs were only detected in the groundwater samples at isolated locations and at very low concentrations. It is believed that the detection of these SVOCs may be related to field filtering artifacts.

- Groundwater seepage rates vary over five orders of magnitude among the 12 monitored locations. These variations are probably because of changes in the permeability of the different types of sediments that comprise the creek bed.

- Because of tidal influences discussed below, the time that contaminated groundwater discharges to the creek can be roughly estimated as 910 minutes per day, versus 530 minutes per day when groundwater did not discharge to the creek.

- The segment of Ferry Creek from I-95 to the flood control structure receives about 0.68 cubic feet per second, or 437,000 gallons per day, of contaminated ground water.

Tidal Influence Studies

Two short-term studies of the influence of tidal cycles on groundwater discharge and flow directions within the study area were conducted in February 1999 and April 2003. The February 1999 study included monitoring groundwater elevations through two tidal cycles at four well cluster locations; monitoring surface water elevations in Ferry Creek; and monitoring surface water elevations in the Housatonic River. The April 2003 tidal study involved monitoring five well clusters located at the eastern edge of the former Raymark Facility, the Dock property, and a surface water location.

The 1999 study was conducted over a 24-hour period and included two high and low tide cycles. The objective of this study was to evaluate the influence of changes in surface water elevation in Ferry Creek and the Housatonic River on the groundwater, and determine if those changes were sufficient to alter the rate of groundwater discharge and flow direction. The change in water elevations for the monitoring wells, Ferry Creek, and the Housatonic River are presented on graphs in Appendix D.

The test data show that the range of tidal fluctuation measured in Ferry Creek was about 6.2 feet, from a low of –0.6 feet NGVD to a high of 5.6 feet NGVD. These results are consistent with the verified tidal listings for February 1999 (NOAA 2004), which indicated that the low elevation would occur during a spring low tide on February 17th, while the high elevation would occur during the general high tide on February 18th.

The range of tidal fluctuation measured in the Housatonic River was about 8.1 feet, from a low of –2.1 feet NGVD to a high of 6.0 feet NGVD. These results are again consistent with the verified tidal listings for February 1999 that the low elevation would occur during a spring low tide on February 17th, while the high elevation would occur during the general high tide on February 18th (NOAA 2004). The range of fluctuation of the Housatonic River was 2.5 feet greater than the range measured in Ferry Creek because of a flood control structure located at the mouth of Ferry Creek, where it enters the Housatonic River. The flood control structure caused the surface water elevation in Ferry Creek to be about 1.5 feet higher than the Housatonic River surface during low tide, and about 0.4 feet below the Housatonic River during high tide.

The well cluster locations monitored were MW-112, MW-213, MW-302, and MW-304, with a total of 11 wells as shown on Figure 2-3. Each cluster location includes a shallow overburden, deep overburden, and bedrock monitoring well, with the exception of MW-213 where only a shallow overburden and bedrock monitoring well are present.

Well cluster MW-112 is about 20 feet from Ferry Creek, and is the closest cluster to the creek. The data for the MW-112 cluster show that the groundwater response to changes in the tidal fluctuations is measurable, but slight. The range of fluctuation in the cluster was on the order of 0.3 feet in all three wells in the cluster. These changes are not significant enough to reverse or otherwise significantly impact the flow direction of the groundwater through the tidal cycle.

Also, the water levels at the MW-112 cluster experience a delayed response to tidal fluctuations. The highest groundwater levels measured in the wells nearly coincide with the lowest tide levels in Ferry Creek, and the lowest groundwater elevations nearly coincide with the highest tides on Ferry Creek. The lag time in the response of the aquifer at MW-112 with respect to tidal fluctuations in Ferry Creek is a function of the hydraulic conductivity of the aquifer between the wells and the creek, and the distance of the well screens from the creek.

Well cluster MW-302 is located about 350 feet from the Housatonic River, and is the closest well cluster to the river. Very slight, but measurable, tidal influences were detected in groundwater elevations at this location. The fluctuations were on the order of 0.20 feet in the shallow overburden monitoring well (MW-302S), and only 0.05 feet in the deep overburden well (MW-302D) and the bedrock well (MW-302B).

The MW-304 cluster is the furthest cluster from the Housatonic River, at approximately 1,300 feet west of the river and approximately 750 feet north of Ferry Creek. The groundwater elevations at the MW-304 cluster remain relatively constant through the tidal cycle, with a small change of 0.1 feet noted during the test period. The data show that the changes in water level at the MW-304 cluster have a slight sinusoidal shape, possibly because of a slight, dampened tidal influence.

The MW-213 cluster is the furthest location from Ferry Creek, at approximately 1250 feet northeast of the creek, and approximately 650 feet west of the Housatonic River. The monitoring wells at the MW-213 cluster do not show a cyclic response to the high and low tide cycles. The data for MW-213S (shallow overburden monitoring well) do not indicate significant changes in groundwater elevation; however, an increase of 0.05 feet was noted early during the test period, and the test level remained at this elevation for the rest of the period. Similarly, the groundwater elevation in MW-213B (bedrock monitoring well) indicates a constant water elevation at about 3.10 feet with a brief increase to 3.25 feet, resulting in a change of 0.15 feet. Neither of these fluctuations appears to be related to tidal influence since there is no sinusoidal character to the small changes in water level.

The study also showed that surface water elevations were higher than groundwater elevations in the immediate vicinity of the creek and river during the peak of the high tides. The

groundwater levels were lower than the surface water levels about 510 minutes per day, given two tidal cycles per day.

The size, shape and configuration of the monitoring wells, and the permeability of the surrounding soil and rock, can cause a delay or dampening of the response of the wells to changes in groundwater head in an aquifer. This can produce an inaccurate assessment of the differences between the elevation measured in the well, and the true elevation head in the aquifer, unless these differences are taken into consideration. The volume of water in each of the 2-inch wells is minimal, and the hydraulic conductivity of the aquifer in the vicinity of Ferry Creek and the Housatonic River is high enough to fill/remove the volume of water in the wells in response to aquifer changes. Therefore, the water levels in the wells are an accurate representation of changes in the aquifer they are screened in.

The conclusions for the February 1999 tidal influence study are as follows:

- Tides in Ferry Creek and the Housatonic River cause changes in the head of the aquifers within the study area, but these changes are slight, and tend to decrease with distance from the creek and the river.

- The most significant tidal influence is within the immediate area of the creek and river, as shown in the graphic plots for MW-112 adjacent to Ferry Creek, and MW-302 adjacent to the Housatonic River (Appendix D). Tidal influence on groundwater elevations further inland from the surface water bodies is insignificant, as shown by the data for MW-304 and MW-213.

- Tidal influences are not significant enough to alter the general groundwater flow directions within the OU2 study area, nor reverse flow directions at distances beyond the immediate vicinity of the creek or the river.

- When the surface water elevation exceeds the groundwater elevation head at the base of the creek or river, groundwater discharge to the river temporarily ceases, and surface water may actually become influent into the sediment until the tide falls and allows the groundwater to resume discharging to the creek. The depth of penetration of the

surface water into the sediment is not known, but is not expected to be significant - probably less than a foot to several inches.

- Based on the tidal elevation versus groundwater elevation near Ferry Creek, the groundwater ceases to discharge during the high tide. The length of time when discharge ceases was estimated as 510 minutes for each 24-hour day. This is based on surface water elevations exceeding groundwater elevation heads at the creek and river for about 260 minutes per high-tide cycle. Further details are provided in Section 3.4.2.6, Ferry Creek Seepage Study.

The April 2003 tidal study involved the monitoring of five well clusters located at the east edge of the OU1 former Raymark Facility. A surface water monitoring point was established on the pier at the Dock Property adjacent to the boat launching area. The study was conducted over a 24-hour period and included two high tide and two low tide cycles. The objective of this study was to evaluate the influence of changes in surface water elevation in the Housatonic River on the groundwater, and determine if those changes were sufficient to alter the rate and direction of groundwater flow. The change in water elevations for the monitoring wells and the Housatonic River are presented in graphical format in Appendix D.

The test data show that the range of tidal fluctuation measured in the Housatonic River during the 24-hour test was about 9.1 feet, from a low of –2.8 feet NGVD to a high of 6.3 feet NGVD.

The well cluster locations used in the tide study were MW-4, MW-201, MW-309, MW-533, and PC-04, as shown on Figure 2-3. Each cluster location includes a shallow overburden, deep overburden, and bedrock monitoring well, with the exception of MW-201, where only the deep overburden and bedrock monitoring wells were utilized in this study, for a total of 14 wells. The surface water monitoring point for the tidal study was located on the Dock Pier. Refer to Figure 2-3 for the well and surface water monitoring point locations. Refer to Appendix D for the graphs generated by the tidal study. Most of the wells had no measurable response to tide changes.

The closest well cluster to the Housatonic River in the 2003 tidal study was MW-201, located about 750 feet away. When the data recorded by transducers in both wells are plotted on a graph, they indicate a general trend of declining water levels. The graph for MW-201D shows

an unexplained sudden increase (0.025 feet) in water levels late in the day on April 16; however, water levels return to the same slow decline as for MW-201B about 5 hours later. Neither well indicates a synclinal pattern that could be attributed to tidal changes.

The MW-4 cluster was the next closest to the river, 850 feet away. The graph of water levels over time for all three wells indicates a general decrease in water levels. The total change in water levels for all three wells is less than 0.05 feet. None of the wells indicate a pattern related to tidal influence.

The MW-533 cluster is about 1,550 feet away from the Housatonic River. The graph of water levels over time for all three wells in the cluster indicates a similar trend for each well; at the beginning of the test, the levels have a slight increase, but then decline throughout the rest of the tidal cycles. The difference in water levels for all of the wells in the cluster is less than 0.1 foot. None of the fluctuations seen appear to be related to tidal influence, as there is no regular sinusoidal character to the water level changes.

The PC-04 cluster is about 1,850 feet from the Housatonic River. The graph of water levels over time for the three wells in the cluster indicates a similar trend. The water levels in all three wells dropped about 0.08 feet at about 4 PM on April 16, then remained at about the same level for the rest of the test, varying less than 0.03 feet. None of these small fluctuations appear to be related to tidal influence.

The MW-309 cluster is also about 1,850 feet from the Housatonic River. The graph of water levels over time shows that the water levels for each well followed a different trend. Water levels declined over time in MW-309S and increased over time in MW-309D, with the water level elevations converging at about 8 AM on April 17. MW-309B was the only well in the 2003 tidal study that showed a clear response to tidal influence. The water levels in MW-309B followed a sinusoidal pattern with the peaks and low points occurring approximately 360 minutes after tidal peaks and low points. During the period of the study, the total change in water level was 0.15 feet. The time lag noted in MW-309B could be the result of a dampening effect caused by the low hydraulic conductivity of the bedrock in the area. The tidal response in MW-309B indicates it is hydraulically connected to the Housatonic River in spite of its distance from the river.

The data indicate that the surface water elevation exceeds the groundwater elevation in all the wells monitored for an estimated period of between 105 minutes to 210 minutes per high tide cycle. The actual times will vary depending on the elevation and duration of high tide compared to the groundwater elevation at any particular well. The duration when the groundwater level exceeds the surface water is estimated at 315 minutes per 24-hour day.

The conclusions of the April 2003 tidal influence study are as follows:

- Tides in the Housatonic River cause changes in the head in the bedrock aquifer in the vicinity of MW-309. These head changes were not observed in any other bedrock wells monitored as part of this tidal study.

- Tidal influences were not observed in any of the overburden wells monitored as part of this study.

- Based on a comparison of the surface water and groundwater elevations, the surface water elevations are estimated to exceed groundwater elevation heads in the wells monitored during this tidal study for 315 minutes for each 24-hour day. The difference in head indicates the potential for surface water to flow into the aquifer for the 315 minutes each day.

3.4.2.7 Overburden Flow Summary

Groundwater and potential contaminant movement in the OU2 study area is highly complex, because of heterogeneities in the aquifer materials (see Sections 3.2.2.1 and 3.2.2.2) and the resulting differences in hydraulic conductivity (Section 3.4.2.1). The overburden groundwater flow is also affected by changes in underlying bedrock topography (Section 3.2.2.3) and the interaction between the bedrock and deeper overburden groundwater (Section 3.4.2.5), as well as localized effects from surface water bodies (Section 3.4.2.6). This section focuses on groundwater flow in the areas of interest.

Groundwater Flow and Ferry Creek

The vertical gradients and the seepage study results indicate that some groundwater and groundwater contaminants discharge into Ferry Creek. However, the distribution of the groundwater contaminants and the geologic and hydrogeologic conditions indicate that most of the groundwater continues to flow under Ferry Creek and toward the mouth of Ferry Creek and the Housatonic River. See Section 3.4.2.6 for a discussion of the seepage studies and Section 3.4.2.3 for a discussion of vertical gradients.

The geologic conditions encountered at Ferry Creek are presented on Figure 3-3, Geologic Cross Section A-A'. This cross section indicates that Ferry Creek is underlain by approximately 18 feet of organic silt with varying amounts of peat. Thicker organic deposits have been observed in borings not located along the cross-section. However, the thickness of these deposits vary. Other areas of Ferry Creek have sand layers with higher hydraulic conductivities. The organic silts have a much lower hydraulic conductivity in comparison to the sand and gravel that fills the south valley. The silt deposit may act as a semi-confining layer in this portion of the OU2 study area. This interpretation is supported by the results of the seepage study, indicating that the rate of groundwater discharge to Ferry Creek varies greatly at different locations.

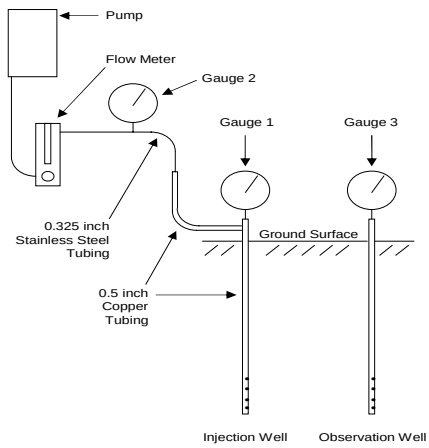
The vertical gradients at monitoring well cluster MW-503 (see Table 3-2) indicate that an upward gradient exists within the overburden. This gradient indicates that groundwater in the overburden has the potential to discharge into the surface water at Ferry Creek. However, the actual rate of groundwater discharge is controlled by the hydraulic conductivity of the organic silt layer that underlies Ferry Creek. Monitoring well MW-503S, which is screened in the organic silt with some sand layers, did not produce enough water to conduct a hydraulic conductivity test, indicating that the hydraulic conductivity is low at this location.

The groundwater contaminants identified in the valley at the MW-212 cluster were also detected in the groundwater in the MW-111 and MW-113 clusters, which are located south of the intersection of the south bedrock valley and Ferry Creek. The observation of contaminants in these downgradient monitoring wells indicates that not all of the groundwater in the south valley discharges into Ferry Creek. Further discussion of the distribution of contaminants is presented in Section 4.

3.5 Soil Gas

The soil gas area of interest is the vadose zone. Material above the water table in the OU2 study area generally consists of fill overlying sand or organic layers. Refer to Section 3.2.2.1 for OU2 study area overburden geology details. Soil gas sampling was conducted using both soil gas profiling and permanent soil gas wells. For sampling details, refer to Section 2. For soil gas results, refer to Section 4.

In order to characterize the potential for soil gas to migrate from the water table, a series of five air permeability tests were conducted at the residential area east of Ferry Boulevard (see Section 2.12.1 and Figure 2-4). Each test employed a couplet of wells made from 10-foot sections of 0.5-inch diameter copper tubing that were perforated along their lowermost 1 foot. The bottoms of the injection well and observation well were placed approximately 9 feet bgs. To isolate the test zone, the annular space between each well and its borehole wall was backfilled with 1.5 to 2 feet of sand followed by hydrated granular bentonite. Both wells were sited more than five feet from buried utility lines or large tree roots. The observation well was placed approximately three feet from the injection well to obtain qualitative information regarding soil permeability and the radius of influence of the injected air. A schematic of the test apparatus is shown below.



Schematic for air injection test to determine soil permeability

During each test, the flow rate was maintained at 60 to 90 ft^3/hr, depending on the soil's ability to accept the injected air. Atmospheric pressure was recorded at the start of each test, and flow rate and pressure readings were recorded periodically until gauge pressures stabilized.

Soil permeability was calculated from measures of atmospheric pressure, gauge pressure, air viscosity, well geometry, air flow rate, head loss, and radius of influence using a steady-state radial flow equation that was modified to account for the compressibility of air (Johnson et al. 1990).

Soil permeability ranged from 2.5 x 10^{-7} cm^2 to 8.3 x 10^{-7} cm^2 (see Table 3-3). This range of values is consistent with field observations of soil texture, because the soils in the test areas are composed almost entirely of sand-size grains, and the calculated permeabilities are characteristic of clean sands (Freeze and Cherry 1979). The permeability results are also consistent with measures of hydraulic conductivity obtained from shallow single well pumping tests. When the soil permeability test results are converted to equivalent hydraulic conductivities, the mean value for the soil permeability tests (150 ft/day) is similar to the mean value for the shallow single well pumping tests listed in Table 3-1 (163 ft/day).

The permeability calculations do not account for the influence of soil moisture on measures of soil permeability. However, the permeability-reducing influence of soil moisture was minimized in these tests, because they took place several days after any significant precipitation, and they were conducted in well-drained sands 8-9 feet below the ground surface and several feet above the water table.

3.6 Climate and Meteorology

The OU2 study area is located in a temperate-humid climate, characterized by highly changeable weather and large daily and annual temperature variations. The most pronounced topographical effect is the land-sea breeze, an occurrence generally associated with the spring through early autumn months. In general, monthly temperatures during the summer months average 3° to 5° lower than nearby inland locations. Temperatures during the fall and winter months are moderated because of the proximity of Long Island Sound. Winter snowfall is generally around 10 inches less than areas a few miles inland, also because of the proximity to Long Island Sound.

Low-lying areas are subject to flooding during periods of exceptionally high tide (from hurricanes and storms). Tides three to five feet higher than normal may be encountered in the presence of slow-moving, deepening low pressure systems.

The local National Oceanic and Atmospheric Administration (NOAA) Climatological Station is located at the Bridgeport-Sikorsky Airport, which is in close proximity to the study area. For the past 30 years, data from this station have been used to describe the general climate in the area.

July is the warmest month with an average temperature of 74.0°F. The coldest month is January with an average temperature of 29.9°F. The maximum temperature observed between 1939-1998 was 103°F. The minimum temperature observed during this period was –7°F. Normal annual precipitation for the region is 44.2 inches, with between three and four inches of rain or water equivalent falling during each month. The area has an average annual snowfall of 25.8 inches, which generally occurs between November and April. However, most snowfall occurs in January and February. Snowfall averages for these two months are 7.4 inches and 7.6 inches, respectively.

Wind speed in the region varies between 9.3 and 13.0 miles per hour (mph) with an average of 11.4 mph. In the warmer months the prevailing wind direction is southwest. In the colder months the prevailing direction is west to northwest (NOAA 1998 and 2002).

4.0 NATURE AND EXTENT OF CONTAMINATION

This section presents summaries of the analytical results developed by several investigations performed during the RI to characterize the nature and extent of chemical contamination in groundwater, soil gas, and indoor air, beneath and downgradient of the former Raymark Facility. Section 4 includes:

- An overview of potential contaminant sources;
- A brief discussion of the upgradient and cross-gradient locations used to evaluate the groundwater within the OU2 study area;
- A discussion of the nature and extent of contamination in the groundwater; and
- A discussion of the nature and extent of contamination in residential soil gas and indoor air.

The primary focus of the nature and extent of groundwater contamination is on samples collected during the 2002/2003 sampling effort. The results, often referred to as “recent”, were used in the evaluation of the spatial distribution of VOC and metals groundwater contamination. Analytical results from all investigations included in this RI Report are in Appendix B. The locations of monitoring wells are shown on Figure 2-1.

4.1 Potential Groundwater Contaminant Sources

This section presents descriptions of the features of the former Raymark Facility, and identifies other features in the study area that were or are potential sources of groundwater contamination downgradient of the Raymark Facility. Refer to Figure 1-2 for the locations of potential contaminant sources.

4.1.1 Former Raymark Facility

The former Raymark Facility operated from 1919 until 1989, when the plant was shut down and permanently closed. As part of the property cleanup activities, the Facility buildings were demolished and a cap was placed over the contaminated areas in 1996 and 1997. Friction materials containing asbestos and non-asbestos components, metals, phenol-formaldehyde resins, and various adhesives were utilized in the manufacturing processes at the Facility. The

primary products were gasket material, sheet packing, and friction materials including clutch facings, transmission plates, and brake linings. As a result of these activities, soils and groundwater at the Facility became contaminated. Additional details on the manufacturing processes that may have contributed to groundwater contamination can be found in Section 1.3.1 as well as in the OU1 Final RI report (HNUS 1995). Table 4-1 presents a list of chemicals used or handled at the former Raymark Facility.

Historically, Raymark waste was used as fill in the low lying areas of the former Raymark Facility. Some of the specific known source areas on the Facility that may have contributed to groundwater contamination include:

- Tanks – Fifty-two underground and 21 above ground tanks were used to store raw materials, solvents, Number 2 and 6 fuel oils, gasoline, waste oils, and possibly process wastes. When the tanks were in use, any spills and leaks that occurred during filling or unloading operations would have seeped into underlying soils and groundwater.

- Toluene release and storage pile – In 1984, an unknown quantity of toluene was released from a 10,000 gallon underground storage tank located in the northern portion of the Facility. Subsequent to the release, 400 cubic yards of contaminated soil were excavated and stored on a concrete slab on-site, and a groundwater extraction well was installed to recover affected groundwater. The recovered groundwater was discharged to the Stratford municipal sewer system. The extraction well operated from 1985 through 1993.

- Lagoons – Lagoons were used to receive and temporarily store process wastewater, manufacturing wastes, and storm water. Lagoons were present at various locations throughout the Facility during its operational history. They were periodically dredged and the sludges/waste were used as fill on-site, and on residential, commercial, and municipal properties throughout Stratford. The sludges left in the lagoons were composed of the same waste constituents found in fill materials present in the remainder of the Facility.

- Acid neutralization pits – Three acid neutralization pits were used during a 30-year period for disposal and neutralization of acids and spent caustic solutions. It appears

that solvents were disposed of in these acid neutralization pits based on the high concentration of solvents in the soils and the presence of NAPL in the groundwater in those areas. The presence of acids could be a significant factor influencing the leaching and migration of metals from soils into groundwater.

- Release of 1,1,1-trichloroethane (1,1,1-TCA) – In 1987, a release of 6,000 gallons of 1,1,1-TCA was discovered north of the acid treatment plant in the center of the property.

- Fill materials – Most of the 33.4 acres that comprised the former Raymark Facility were filled in with manufacturing waste, as Raymark waste was used historically as fill in the low-lying areas. The Raymark fill material contains the four contaminants that meet the definition of Raymark waste (lead, chrysotile asbestos, copper, and Aroclor 1268), as well as, potentially, VOCs, SVOCs, dioxins, and other metals and PCBs.

The scope of the OU1 RI was limited to the boundaries of the former Raymark Facility, and no off-site wells were evaluated, therefore a full assessment of the extent of groundwater contamination present upgradient and downgradient of the former Raymark Facility was not performed. The findings of the OU1 Final RI report indicate groundwater underlying the former Facility is contaminated by a variety of organic and inorganic constituents, with the acid pits and spill areas (toluene and 1,1,1-TCA) being significant sources of organic contamination in groundwater. All contaminants detected in groundwater were also detected in soils on the Facility.

4.1.2 Off-facility Distribution of Raymark Waste

Raymark waste materials were disposed of (dumped) as fill on residential, commercial, and municipal properties throughout Stratford, including many areas within the OU2 study area. Approximately one half of samples collected from those areas of Raymark waste indicate the potential for contaminants to leach into groundwater, based on exceedances of the Connecticut pollutant mobility criteria (PMC) for metals. Groundwater modeling found elevated lead and copper concentrations near the MW-102 well cluster (600 East Broadway), and lead concentrations in Raymark waste in that area also exceeded the CT PMC, indicating the potential to leach (see Appendix E for modeling results). Concentration of metals in groundwater near other areas of Raymark waste do not indicate that groundwater quality has

been impacted by leaching of metals from Raymark waste. However, the PMC exceedances for soils in the area and the groundwater modeling implicate the same contaminants, suggesting that the contaminants may be leaching from the soil into the groundwater.

EPA removed more than 37,000 cubic yards of contaminated waste material from residential properties within Stratford during time-critical removal actions from 1993 to 1996. Contaminants present in those areas were designated a threat to human health (based on EPA-established criteria) and removed. Approximately 30,000 cubic yards of waste material were also excavated and removed from the Wooster Middle School playing fields in 1996 by the CTDEP. All areas were backfilled with clean fill, and excavated materials were transported back to the former Raymark Facility and placed under a RCRA-type cap installed as part of the source control remedial action for OU1. Additional properties located within the OU2 study area that received Raymark waste as fill have previously been investigated as separate operable units. Soil, sediment, surface water, and/or biota samples collected during the OU2 groundwater study have been evaluated under OU3 - Ferry Creek, OU4 - Raybestos Memorial Ballfield, and OU6 - Additional Properties (see Figure 1-2). The potential for contaminants from these operable units to migrate to groundwater is included in this OU2 RI evaluation to determine the nature of the contaminants detected in groundwater.

4.1.3 Other Non-Raymark Related Sources

Activities at additional industrial and commercial properties within and upgradient of the OU2 groundwater study area such as RCRA facilities, manufacturing facilities, gas stations, and dry cleaners, may also have impacted groundwater quality. The focus of this evaluation of the nature and extent of contamination in groundwater is on contaminants that are potentially attributable to operations at the former Raymark Facility, or locations where Raymark waste materials were disposed. Contamination from non-Raymark sources has not been evaluated in this Report.

4.2 Background Locations

The OU2 study area contains many potential sources of groundwater contamination, in addition to the contamination contributed by the former Raymark Facility and the wastes associated with that Facility. In addition, natural subsurface materials can contribute to elevated concentrations

of inorganic constituents that might otherwise be identified as “contamination”. Therefore, an evaluation of the background concentrations of various chemical constituents in the groundwater was conducted to help discriminate between groundwater contamination from Raymark and Raymark waste releases versus releases from other non-Raymark related sources. The background evaluation was conducted by comparing the analytical results for study area samples with those obtained from locations that are unlikely to have been affected, either by past activities at the former Raymark Facility or by disposal of Raymark waste materials. In this way, chemicals detected in the study area can be attributed to naturally occurring sources (such as metals in soils and sediments), contamination that is pervasive in the area (i.e., pesticides use within the community, lead in an urbanized area, or releases from other sources, etc.), or to Raymark-related releases. Discussions of chemicals detected in the background locations are presented in the evaluations of contaminant nature and extent.

Upgradient locations for groundwater were determined by evaluating the groundwater flow direction for the overburden and bedrock aquifers, then selecting monitoring well locations that were situated hydrologically upgradient or cross gradient from the potential contaminant source areas. The average concentrations of contaminants in these upgradient/cross gradient wells were used to compare to concentrations found in other wells, within the same aquifer, at or downgradient of the former Raymark Facility. Groundwater flow directions at the study area were established based on bedrock and topographic elevation information, water level rounds, and contaminant movement from known source areas. The monitoring wells used for establishing upgradient/cross-gradient chemical characteristics are indicated on Figure 2-1. Tables 4-2 and 4-3 present a summary of the results of samples collected from the upgradient/cross-gradient overburden and bedrock monitoring wells, respectively. Table 4-4 presents a comparison of the overburden and bedrock monitoring wells.

Background locations for indoor air were selected to represent indoor air not impacted by contaminants volatilizing from groundwater impacted by activities at the former Raymark Facility. The two locations were selected after consulting with a Stratford Town Official with knowledge of the extent of Raymark-related contamination. Maximum background indoor air concentrations are listed on Table 4-9.

4.3 Nature and Extent of Contamination in Groundwater

Groundwater contamination investigations for the RI have been conducted primarily since 1994. Groundwater data for some wells located on commercial properties in the study area (e.g. 540 Longbrook Avenue and 375 Barnum Avenue) that were sampled prior to 1994 have also been included in this evaluation. Groundwater samples were collected from bedrock and overburden aquifers to evaluate the extent of groundwater contamination that may have resulted from the past practices of Raymark Industries, Inc. Figure 2-1 presents the various monitoring points used to evaluate the nature and extent of contamination within the study area.

The groundwater analytical results are presented in this section according to class of chemical compounds detected (i.e. VOCs, SVOCs, pesticides, PCBs, metals, and dioxins), and by aquifer (overburden, bedrock.) The groundwater analytical results were compared to maximum upgradient/cross-gradient concentrations, and to criteria promulgated by the State of Connecticut Groundwater Remediation Standards, the Surface Water Protection Criteria (SWPC), and residential and industrial/commercial volatilization criteria. Table 4-4 presents the available criteria to which each chemical constituent was compared.

In March 2003, CTDEP released “Proposed Revisions” to Connecticut’s Remediation Standard Regulations Volatilization Criteria (CTDEP 2003). New Groundwater Volatilization Criteria were proposed based on revised indoor air target concentrations (TACs) using new toxicity values, a revised transport model, and improved understanding of the pathway. The TACs are generally based on either residential or industrial exposures and resulting hazard indices of 1.0 or cancer risks of 10^{-6}. For TCE and benzene, the TAC values are based on upgradient/cross-gradient concentrations. In general the proposed Groundwater Volatilization Criteria are lower than (or more stringent than) the existing criteria. One exception is the proposed Groundwater Volatilization Criteria for 1,1-DCE, which is greater than the existing criteria. These proposed revisions have not yet been promulgated, therefore the Connecticut Remediation Standard Regulations (CTDEP 1996) were used to aid in the evaluation of the nature and extent of groundwater contamination.

The primary data used to determine the nature and extent of contamination were collected during the 2002/2003 sampling effort. Different subsets of wells were previously sampled. A

comparison of recent data to historical data is provided for an understanding of contaminant concentrations over time.

Plume Maps – The spatial distribution of groundwater contamination was modeled using Visual Groundwater™ Version 2.2. This software package uses both the spatial coordinates and the chemical analytical data for each monitoring location to: 1) contour contaminant concentrations in three dimensions; and 2) produce two-dimensional images of the distribution of contaminants throughout the study area. Plume maps were created for seven VOCs and eight metal contaminants. The modeling combined data from the two aquifer units, because the aquifer test results (Section 3.4.2.1) and the chemical analytical data indicate hydraulic continuity between the overburden and bedrock.

Figures 4-1 through 4-15 present the plume maps produced by Visual Groundwater™. The patterns of contamination shown in these images are generally compatible with the patterns that would be predicted from both the source information presented in Section 4.1.1 and the groundwater flow directions depicted on Figures 3-7 and 3-8. This compatibility supports the assertion that these images approximate the actual distribution of groundwater contamination within the study area. A further discussion of the plume imaging process, including its strengths and limitations, is included in Appendix E.

Overburden aquifer data points - A total of 193 groundwater samples were collected from monitoring wells screened in the overburden aquifer in the study area (including eight upgradient samples). Of these samples, 192 were analyzed for VOCs, SVOCs, and pesticides and PCBs, 190 were analyzed for metals, and 150 were analyzed for dioxins.

Bedrock aquifer data points - A total of 63 groundwater samples were collected from monitoring wells screened in the bedrock aquifer in the study area (including 11 upgradient samples). Of these samples, 63 were analyzed for VOCs, SVOCs, pesticides and PCBs, and metals, and 55 were analyzed for dioxins.

Tables 4-5 and 4-6 provide data summaries including the frequency of detection and frequency of exceedances of criteria for detected compounds in groundwater samples collected in the overburden and bedrock aquifers, respectively. These tables summarize the analytical results for overburden and bedrock aquifers and show the range of detected concentrations and the

average concentrations for detected compounds. The groundwater analytical results are presented on a disk in Appendix B. The following subsections describe the general nature and extent of contamination by chemical classification.

Groundwater samples were not analyzed for asbestos within the study area outside of the former Raymark Facility (HNUS 1995).

4.3.1 Volatile Organic Compounds in Groundwater

Volatile organic compounds were the most commonly and widely detected chemical class in the groundwater that can be considered entirely the result of man-made sources. Data summaries for individual VOCs are presented in Table 4-5 for the overburden aquifer and Table 4-6 for the bedrock aquifer. The nature and extent of the VOCs exceeding Connecticut's SWPC and/or residential volatilization criteria (RVC) are described below. Figures 4-1 through 4-7 present computer generated images of the spatial distributions of seven of the ten identified VOCs throughout the OU2 study area. Table 4-5a presents data summary and criteria comparisons for VOCs shown in the figures.

The concentrations of ten VOCs: 1,1,1-trichloroethane (1,1,1-TCA), 1,1-dichloroethene (1,1-DCE), 1,2-dichloroethane (1,2-DCA), 1,2-dichloropropane (1,2-DCP), benzene, chlorobenzene, tetrachloroethene (PCE), toluene, trichloroethene (TCE), and vinyl chloride exceeded Connecticut's SWPC and/or RVC in at least one groundwater sample collected during the winter of 2002/2003. Since 1,2-DCP was only detected in two samples, and PCE and 1,2-DCA were only found at high concentrations near the former acid neutralization pits at the Facility, plume maps were not constructed for these three VOCs. Maps of the other seven VOCs at shallow depths are presented in Figures 4-1 through 4-7 along with cross sections positioned through the centers of high concentrations. The shallow plan view illustrates the extent of contamination in the groundwater near the water table. It also enables the reader to identify areas where the upward migration of vapors into basements may be of greatest concern. Cross sectional views help delineate the geometries of high-concentration areas and the variability in contaminant concentrations with depth.

The paragraphs below describe VOC contamination. This evaluation is organized by contaminant group rather than geographic location of contamination. Contaminants are grouped by the similarities in the nature and extent of the contaminant.

1,1-DCP, Benzene, Chlorobenzene, and Toluene

1,1-DCP was only found in the shallow groundwater beneath 375 Barnum Avenue. 1,1-DCP was detected in monitoring wells CRA-6S and CRA-5S at 8 µg/L and 56 µg/L, respectively, and the sample from CRA-5S exceeded the RVC of 14 µg/L. As a result of the limited number of detections, no plume maps were created for 1,1-DCP in this RI Report.

The highest concentrations of benzene (up to 4,600 µg/L) were also found in the shallow groundwater beneath 375 Barnum Avenue (see Figure 2-1 for road names). These exceeded the RVC of 215 µg/l for benzene (see Figure 4-1). Although benzene was observed in the groundwater at 73 locations throughout the OU2 study area, measured concentrations outside the Barnum Avenue area of localized high benzene concentrations were below pertinent regulatory criteria. The absence of high recent or historic benzene concentrations in samples collected from upgradient former Raymark Facility wells suggests the high concentrations at Barnum Avenue did not originate on the Facility.

Chlorobenzene was detected in 133 of the 265 monitoring wells sampled in the OU2 study area. High concentrations of chlorobenzene were observed in two localized areas, and the centroid of the area with the highest concentrations was near the northeast corner of the former Raymark Facility (see Figure 4-2). Here, the highest concentration of chlorobenzene (6,100 µg/L) was measured in the groundwater at PC-03S. The other wells with groundwater concentrations above the RVC of 1,800 µg/L were PC-03D and MW-1M, positioned 190 feet northeast of the PC-03 cluster.

Toluene was found in 33 groundwater samples. As with chlorobenzene, there were two localized areas of high toluene concentrations in the OU2 groundwater system (see Figure 4-3). The highest concentration of toluene (35,000 µg/L) was measured in the shallow groundwater at PC-04S, approximately 300 feet southeast of the 1984 toluene spill at the former Raymark Facility. There was a second localized area of high concentration in the groundwater beneath the former Raymark Facility acid neutralization pits with a concentration of 23,500 µg/L.

1,1,1-TCA and 1,1-DCE

1,1,1-TCA was detected in 128 samples and 1,1-DCE was observed in 131 samples. In general, groundwater concentrations of both contaminants were highest in the overburden and bedrock associated with the southern bedrock valley (see Figure 3-6 and the cross sections on Figures 4-4 and 4-5). The highest concentration points of 1,1,1-TCA (160,000 µg/L) and 1,1-DCE (40,000 µg/L) were detected in samples collected from PC-02D and PC-02B, respectively. These two wells are located beneath former disposal Lagoon 4. Lagoon 4 was located above the southern bedrock valley approximately 550 feet southeast of the 1987 1,1,1-TCA release, and it reportedly received most of the spilled liquid. Unlike benzene and toluene, the highest concentrations of 1,1,1-TCA and 1,1-DCE reside in the deeper portions of the OU2 groundwater system. None of the shallow groundwater samples in the study area exceeded pertinent regulatory criteria for 1,1,1-TCA. However, a large number of both shallow and deep samples exceeded the SWPC and RVC for 1,1-DCE. In fact, 99 percent of the samples in which 1,1-DCE was detected surpassed the RVC of 1 µg/L (the proposed RVC for 1,1 DCE is 190 µg/L). Thirty-seven percent of the samples that contained detectable concentrations of 1,1-DCE surpassed the proposed RVC.

1,2-DCA, PCE, and TCE

Maximum concentrations of 1,2-DCA (50 µg/L), PCE (2,000 µg/L), and TCE (100,000 µg/L) were detected in the groundwater beneath the former acid neutralization pits at the southwest end of the former Raymark Facility. Although 1,2-DCA was detected in 39 of the OU2 groundwater samples and PCE was found in 81 samples, 1,2-DCA and PCE exceedances of the RVC (21 and 1,500 µg/L, respectively) were limited to the groundwater near the former acid pits. Outside the immediate vicinity of the acid pits, the concentrations of 1,2-DCA and PCE were everywhere less than 14 µg/L and 28 µg/L, respectively. As stated earlier in this section, PCE and 1,2-DCE were rarely detected in the groundwater and no plume maps were created in this RI Report. No further discussion of these two contaminants is presented in this section.

TCE exceedances were more widespread (see Figure 4-6). Fifty percent of the 147 samples in which TCE was detected surpassed the RVC of 219 µg/L, and 80 percent surpassed the proposed RVC of 27 µg/L. The primary source of TCE in the OU2 groundwater system appears to be the TCE-rich DNAPL that has been observed in overburden wells installed beneath the

former acid neutralization pits (see Sections 4.1.1 and 5.2). The slow dissolution of this DNAPL source delivers a continuous supply of TCE to the groundwater.

The acid pit TCE source is visible on the plume maps shown in Figure 4-6. Two additional areas of localized high TCE concentrations are evident on the shallow plan view. These additional areas are downgradient from the former acid pits, and the concentration distribution on cross section B-B' suggests that the TCE beneath the acid pits is their primary source. The absence of perfect continuity between the areas of localized high concentrations is likely an artifact of the scarcity of observation points between them. The TCE fate and transport model described in Section 5.3.4.4 further supports the assumption of a single primary TCE source, because it demonstrates that the TCE concentrations in several downgradient wells can be reproduced with a model that relies on a single source located beneath the former acid pits.

Overall, the maps in Figure 4-6 indicate that TCE concentrations are greatest at intermediate depths; and that TCE concentrations can be high in both the overburden and the bedrock.

As shown on Figure 3-7, shallow groundwater flows in a southeasterly direction beneath the former Raymark Facility and the residential area east of I-95 and Ferry Creek. Shallow groundwater flowing beneath the southern end of the former Raymark Facility (which includes the acid pit source area) eventually passes beneath the residential area between Ferry Creek and Minor Avenue. Shallow groundwater flowing beneath the northern end of the former Raymark Facility passes underneath the residential area north of Riverview Place, en route to the Housatonic River.

TCE concentrations exceeded Connecticut's proposed RVC in several groundwater monitoring points in the residential area south of Riverview Place (see Figure 4-6). North of Riverview Place and east of I-95, TCE has only been observed at trace concentrations whenever it has been found. Given the understanding of groundwater flow and the distribution of TCE in this portion of Stratford, it appears the TCE contamination in the groundwater under the residential area originated from the confirmed DNAPL source beneath the former acid neutralization pits. Furthermore, the groundwater flow and TCE source and distribution information strongly suggest the residences north of Riverview Place will not be adversely impacted by this source of TCE.

Based on the available data, the only residences potentially at risk from this source of TCE are those along Willow Avenue, Homestead Avenue, Minor Avenue, Burr Place, Riverview Place, Ferry Boulevard between Willow Avenue and Riverview Place, and Housatonic Avenue from its northern terminus to a short distance south of its intersection with Willow Avenue. Homes further south on Housatonic Avenue and the residential area south of I-95 and west of Ferry Creek are south of the zone of influence of shallow groundwater from the former Raymark Facility. Similarly, the residential area north and west of the former Raymark Facility is not affected by shallow groundwater contamination from the Facility because it is hydraulically upgradient from the Facility (i.e. shallow groundwater flows from this residential area toward the former Raymark Facility). TCE is among the groundwater and indoor air contaminants evaluated in Section 6.0.

Vinyl Chloride

Vinyl chloride exceeded Connecticut's RVC of 2 µg/L in 96 of the 116 samples in which it was detected. Figure 4-7 illustrates its prevalence at high concentrations in the shallow groundwater beneath the former Raymark Facility. Vinyl chloride is a common daughter product of TCE biodegradation (see Section 5.4). The highest concentration of vinyl chloride (530 µg/L) was found at PC-14S, about 160 feet downgradient from the acid pit TCE source. Most of the samples in which vinyl chloride was detected were collected from wells that were sited downgradient from this TCE source. As with TCE, the homes that overlie shallow, vinyl chloride-laden groundwater are found east of Ferry Boulevard and south of the Village Square condominium complex.

Temporal Trends in VOC Concentrations

The concentrations of eight VOCs (1,1,1-TCA, 1,1-DCE, TCE, 1,2-DCE, toluene, vinyl chloride, benzene, and chlorobenzene) detected in the groundwater during the winter of 2002/2003 were compared to the concentrations detected between November 1997 and March 1999 (Table 4-7). Several monitoring wells were sampled more than once between 1997 and 1999, so the most recent sample collected from each location during this 1.3-year time interval was compared to the concentration measured in the winter of 2002/2003. The comparisons were limited to the wells in which the VOCs were detected during both time periods. The Wilcoxon signed-rank test – the nonparametric analog to the paired t-test – was used to determine whether or not the

concentrations of the eight VOCs changed significantly ($\alpha = 0.05$) between the two time periods. According to the test results for the entire OU2 study area, 1,1-DCE, TCE, toluene, and vinyl chloride concentrations did not change significantly between 1997-1999 and 2002/2003, while the concentrations of the other four VOCs decreased. The results for the residential area east of Ferry Boulevard indicate that VOC concentrations, with the exception of toluene, did not change significantly in this portion of the study area (Table 4-8). Changes in toluene concentrations could not be evaluated because the sample number was too small. The groundwater fate and transport modeling results presented in Section 5.3.4 and Appendix E also indicate that: 1) the concentrations of 1,1-DCE and TCE did not change significantly between 1997-1999 and 2002/2003; and 2) their concentrations are not expected to change at any time in the foreseeable future.

Summary and Discussion of Potential Source Areas

The concentrations of ten VOCs (listed in Section 4.3.1) exceeded Connecticut's SWPC and/or RVC in one or more groundwater samples. 1,1-DCP was only detected at two monitoring locations, and both were in the shallow groundwater beneath 375 Barnum Avenue. Benzene concentrations were also highest at shallow depths beneath 375 Barnum Avenue. The absence of elevated levels of benzene in upgradient monitoring well samples and abandoned Facility well samples suggests this localized area of high concentrations did not originate on the Raymark property.

By contrast, the eight remaining VOCs (excluding 1,1-DCP and benzene) were most abundant beneath the former Raymark Facility. 1,1,1-TCA and 1,1-DCE concentrations were high throughout much of the OU2 study area, but they were highest in the deep overburden and bedrock beneath former Lagoon 4. At this location, both contaminants were present at concentrations greater than 1 percent of their respective pure phase solubilities, suggesting the occurrence of a nearby DNAPL source (Pankow and Cherry 1996). Toluene and chlorobenzene exceedances were restricted to the northern end of the former Raymark Facility near the site of the 1984 toluene spill. Concentrations of toluene and chlorobenzene were high enough to infer the presence of NAPL near the PC-03 and PC-04 well clusters. The highest concentrations of TCE, PCE, and 1,2-DCA were found in the DNAPL recovery wells at the southwestern end of the former Raymark Facility, where both TCE and PCE were present at

concentrations greater than 1 percent saturation. The highest concentrations of vinyl chloride were detected in the shallow groundwater downgradient from this confirmed DNAPL source.

NAPLs have not been observed outside the former Raymark Facility. However, dissolved concentrations of TCE, 1,1,1-TCA, and 1,1-DCE exceeded 1 percent saturation in off-site groundwater samples collected in the winter of 2002/2003. TCE was present at slightly less than 2 percent of its pure phase solubility in MW-514D, located about 800 feet southeast of the southern end of the facility. 1,1,1-TCA and 1,1-DCE were present at more than 1 percent saturation in samples collected from MW-211D and MW-211B. The MW-211 well cluster lies 650 feet downgradient from the suspected DNAPL source beneath Lagoon 4, and it is within 200 feet of the facility. It is not clear whether: 1) there is a separate DNAPL source near the MW-211 well cluster; 2) DNAPL extends from beneath Lagoon 4 to the vicinity of these wells; or 3) the DNAPL only occurs in the deep overburden and bedrock near Lagoon 4. The fate and transport modeling described in Section 5 demonstrates that it is possible the 1,1,1-TCA and 1,1-DCE at MW-211 originated from a DNAPL source that only occurs near Lagoon 4. The modeling results suggest that groundwater concentrations of 1,1,1-TCA and 1,1-DCE are still very high 650 feet from the Lagoon 4 source, not because there is a separate source near MW-211, but because the plume has not traveled far enough from Lagoon 4 for dilution and degradation to reduce dissolved concentrations to below 1 percent saturation.

4.3.2 Semi-Volatile Organic Compounds in Groundwater

The types and distribution of SVOCs in the overburden and bedrock aquifers are presented below. Tables 4-5 and 4-6 provide summaries of the analytical results for contaminants detected in the overburden and bedrock aquifers, respectively. A summary of results for the overburden recovery wells is included at the end of Table 4-5. Groundwater modeling was not conducted for SVOCs, so no figures are presented. Results referred to as “recent” reflect the comprehensive groundwater sampling effort performed during the winter of 2002/2003.

<u>Overburden</u>

The overall distribution of SVOCs in the overburden aquifer during the 2002/2003 sampling effort is discussed below. One hundred and eighty-four groundwater samples were collected

from the overburden aquifer and analyzed for SVOCs. Twenty-eight SVOCs were detected in 80 of the 184 samples collected.

Acenaphthylene, benzo(b)fluoranthene, and phenanthrene were the only compounds detected in samples from 2003/2003 at concentrations greater than the SWPC. Concentrations of acenaphthylene ranged from 0.45 µg/L to 1.3 µg/L in three samples, which exceeded the SWPC of 0.3 µg/L. The highest concentration was detected at PC-02S, on the southern boundary of the former Raymark Facility. Benzo(b)fluoranthene was detected in one well directly downgradient of the former Raymark Facility (CRA-6D) at 0.32 µg/L, exceeding the SWPC of 0.3 µg/L. Concentrations of phenanthrene ranged from 0.2 µg/L to 3.1 µg/l in 15 samples. All detected concentrations exceeded the SWPC of 0.077 µg/L. The maximum concentration was detected at PC-02S, where the maximum concentration of acenaphthylene was also detected. Acenaphthylene, benzo(b)fluoranthene, and phenanthrene were not detected in upgradient wells.

2,4-Dimethylphenol, 2-chlorophenol, 2-methylphenol, 4-methylphenol, benzo(b)fluoranthene, caprolactam, naphthalene, and phenol concentrations in study area wells exceeded the maximum concentrations detected in upgradient wells, but did not exceed the SWPC.

Similarly, results from historical sampling also indicate concentrations of acenaphthylene and phenanthrene exceeded the SWPC. Concentrations of acenaphthylene ranged from 1 µg/L to a maximum concentration of 10 µg/L at MW-104M and MW-120S in May 1994. Historical samples from PC-02S contained acenaphthylene at concentrations greater than the SWPC, similar to the most recent sample from the same location. Concentrations of phenanthrene ranged from 0.6 µg/L at PC-09S to a maximum concentration of 36 µg/l at PC-02S in December 1998; all exceeding the SWPC of 0.077 µg/L.

Historically, acenaphthylene was not detected in any upgradient overburden samples, but phenanthrene was detected in one upgradient overburden well, MW-313S (the western-most well), at 2 µg/L.

Overburden Recovery Wells

Groundwater samples were collected during 2002/2003 from each of the five DNAPL recovery system wells located on the former Raymark Facility. Fifteen SVOCs were detected. Benzo(k)fluoranthene and phenanthrene were detected at MW-RW1 at concentrations greater than the SWPC (0.3 µg/L and 0.077 µg/L, respectively). No other SVOCs exceeded the SWPC. 4-Chloro-3-methylphenol and naphthalene were detected in recovery wells at concentrations greater than the maximum concentrations detected in overburden upgradient wells.

Bedrock

The overall distribution of SVOCs in the bedrock aquifer during the 2002/2003 sampling effort is discussed below. Fifty-two samples were collected from the bedrock aquifer and analyzed for SVOCs. Twelve SVOCs were detected in 31 of the 52 samples collected. Phenanthrene was the only compound with a concentration greater than the SWPC of 0.077 µg/L (0.12 µg/L at PC-11B). It was not detected in any other wells. The concentrations of caprolactam ranged from 4 µg/l to a maximum concentration of 810 µg/L at MW-4B, where the maximum upgradient concentration in bedrock of 650 µg/l was exceeded.

In historical bedrock groundwater samples, no compounds were detected at concentrations greater than the SWPC.

Summary

SVOCs were detected infrequently and at relatively low concentrations in groundwater samples from throughout the study area. Only three compounds had concentrations exceeding the SWPC, and the distribution of those exceedances, primarily in the overburden aquifer, appears to be sporadic. Compared to previous sampling rounds, concentrations of SVOCs from the 2002/2003 sampling effort appear to be similarly low, and detections of SVOCs are still sporadic throughout the study area.

4.3.3 Pesticides and PCBs in Groundwater

The type and distribution of pesticides and PCBs in the overburden and bedrock aquifers are presented below. Tables 4-5 and 4-6 provide summaries of the analytical results for contaminants detected in the overburden and bedrock aquifers, respectively. Groundwater modeling was not conducted for pesticides or PCBs, so no figures are presented.

Overburden

The type and distribution of pesticides and PCBs in the overburden aquifer during the 2002/2003 sampling effort are presented below. One hundred and eighty-three samples were collected from the overburden aquifer and analyzed for pesticides and PCBs.

Twelve pesticides were detected in 15 of the 183 overburden groundwater samples collected. Concentrations of dieldrin, heptachlor, and heptachlor epoxide exceeded their respective SWPC (0.1 µg/L, 0.05 µg/L, and 0.05 µg/L). Dieldrin concentrations ranged from 0.072 µg/L to 0.25 µg/L in two samples; the highest concentration was found in MW-302S, located downgradient of the former Raymark Facility, near the Housatonic River. Heptachlor concentrations ranged from 0.026 µg/L to 0.087 µg/L in two samples; the highest concentration was found at MW-506M located at the southwestern boundary of the former Raymark Facility. Heptachlor epoxide concentrations ranged from 0.03 µg/L to 0.66 µg/L in six samples; the highest concentration was found at PC-12D on the former Raymark Facility property near the southern boundary. No pesticides were detected in upgradient overburden wells.

Historically, concentrations of dieldrin and heptachlor also exceeded the SWPC. Dieldrin was detected in 9 of 226 samples analyzed for pesticides with one exceedance of the SWPC at MW-302S, while heptachlor was detected in 4 of 221 samples, with one exceedance of the SWPC at PC-04S. Both exceedances were detected during the same sampling round – the winter of 1998/1999. No other pesticides were ever detected in MW-302S, and low concentrations of a few pesticides were detected at PC-04S during the winter of 1998, but none were detected the previous sampling round (winter 1997). Additional pesticides were detected in various overburden wells that were not detected in upgradient overburden wells. The concentrations and the frequencies of detection were relatively low.

No PCBs were detected in overburden groundwater samples collected during 2002/2003.

Three Aroclors were detected in two historical overburden samples (PC-10M, on the northern boundary of the former Raymark Facility and MW-305S, on the property formerly known as Tilo Industries, southwest of the former Raymark Facility) collected during the winter of 1998/1999. Concentrations of Aroclors 1260 and 1262 at PC-10M did not exceed the SWPC of 0.05 µg/L for PCBs. The concentration of Aroclor 1254 detected at MW-305S (1.2 µg/L) did exceed the SWPC of 0.5 µg/L for PCBs. No other PCBs were detected in overburden groundwater samples.

Bedrock

The types and distribution of pesticides and PCBs for the bedrock aquifer during the 2002/2003 sampling effort are presented below. Fifty-two samples were collected from the bedrock aquifer and analyzed for pesticides and PCBs.

Eight pesticides were detected in eight bedrock groundwater samples. Concentrations of dieldrin and heptachlor epoxide exceeded the SWPC. Dieldrin was detected in only one sample, at MW-215B at a concentration of 0.42 µg/L. Heptachlor epoxide was detected in five samples. Concentrations ranged from 0.02 µg/L to 0.18 µg/L. Two samples had concentrations greater than the SWPC. PCBs were not detected in bedrock groundwater samples during the 2002/2003 sampling effort.

No pesticides or PCBs were detected in upgradient bedrock wells.

Historically, concentrations of beta-BHC and delta-BHC exceeded the maximum upgradient concentrations at that time of 0.005 µg/L and 0.0021 µg/L, respectively. Exceedances of maximum upgradient concentrations were detected in samples collected during the winter 1998/1999 at wells MW-113B, MW-212B, MW-213B, MW-301B, MW-304B, and MW-310B. Neither beta-BHC nor delta-BHC was detected in wells MW-113B, MW-212B, or MW-213B during previous sampling rounds. Other pesticides such as 4,4'-DDT, endosulfan I, endosulfan sulfate, gamma-BHC, and gamma-chlordane were detected very infrequently and at low concentrations in downgradient wells, and not detected at all in upgradient wells.

Aroclor 1221, the only PCB detected in bedrock groundwater samples, was detected at MW-212B during November 1998. The detected concentration, 1.1 µg/L exceeded the SWPC of 0.5 µg/L. No other PCBs were detected in any upgradient or downgradient bedrock groundwater samples.

Summary

Pesticides and PCBs were infrequently detected in overburden and bedrock groundwater samples collected from the OU2 study area. Pesticides and PCBs were detected more frequently in overburden samples, but concentrations were relatively low in samples from both aquifers. No PCBs were detected in overburden or bedrock samples collected during 2002/2003.

4.3.4 Metals in Groundwater

Metals are ubiquitous in groundwater because they occur naturally in the environment. Naturally-occurring heavy metals are usually found in the groundwater at low concentrations. Their occurrence at high concentrations generally indicates an anthropogenic source or an abundance of suspended solids in the groundwater sample. Groundwater samples collected from the OU2 monitoring wells were acquired using methods that minimized the amount of suspended particulates, so the samples should represent the true concentrations of metals in the groundwater.

Data summaries for individual metals are presented in Table 4-5 for the overburden aquifer and Table 4-6 for the bedrock aquifer. A summary of the nature and extent of the metals that exceeded Connecticut's SWPC are described below. In addition, Figures 4-8 through 4-15 present computer-generated images of the spatial distributions of selected metals throughout the study area; Table 4-5a presents data summary and criteria comparisons for metals shown in the figures.

Arsenic, beryllium, cadmium, copper, lead, mercury, nickel, selenium, silver, thallium, and zinc exceeded SWPC in one or more groundwater samples. Computer images were not created for mercury, selenium, or silver because few samples contained detectable concentrations of these

elements, and the small number of samples that exceeded pertinent regulatory criteria did so by a narrow margin.

Groundwater samples from 12 out of 257 OU2 monitoring wells contained detectable levels of mercury. The highest concentration of mercury (2.8 µg/L) was found in samples from MW-101S beneath 600 East Broadway and RW-1 beneath the former Raymark Facility. The only other sample that exceeded the SWPC of 0.4 µg/L was collected from MW-211D.

Selenium was found in 28 groundwater samples, and concentrations in two samples exceeded the SWPC of 50 µg/L. The highest concentration of selenium (90.3 µg/L) was detected in the bedrock beneath the former Raymark Facility at PC-01B. Although silver was detected in 43 samples, only one sample exceeded the silver SWPC of 12 µg/L. A silver concentration of 15 µg/L was measured in the bedrock groundwater on the east side of East Main Street at MW-1B.

Concentrations of beryllium surpassed the SWPC of 4 µg/L in 16 of the 22 samples in which it was detected. Nevertheless, the SWPC was not exceeded in the shallow groundwater. Figure 4-8 shows that beryllium concentrations were much higher in the deeper overburden and bedrock, and the highest concentrations (up to 113 µg/L) were found in the deep overburden and bedrock associated with the southern bedrock valley.

The spatial distribution of thallium was similar to that of beryllium. As with beryllium, thallium concentrations were by far the highest in the deep overburden and bedrock of the southern bedrock valley (see Figure 4-9). The maximum concentration of thallium (210 µg/L) was detected in the bedrock valley groundwater at PC-01B. Thallium was also relatively abundant in the vicinity of the DNAPL recovery wells at the former Raymark Facility and at the eastern edge of the 411 Barnum Avenue cut-off property. However, concentrations were below the SWPC of 63 µg/L at both locations. Overall, thallium concentrations ranged from 2.4 µg/L to 210 µg/L, and the SWPC was surpassed in 7 of the 40 samples in which thallium was detected.

Arsenic and zinc contamination was more pervasive. All but 1 of the 60 samples with measurable levels of arsenic exceeded the SWPC of 4 µg/L, and 43 percent of the 125 samples with measurable levels of zinc exceeded the SWPC of 123 µg/L. Arsenic concentrations reached a maximum of 148 µg/L in the OU2 study area, and areas of high localized concentrations were found in the groundwater beneath the former Raymark Facility, 600 East

Broadway, the Dock property, and the 411 Barnum Avenue cut-off property, as well as the overburden and bedrock in the southern bedrock valley (see Figure 4-10). Zinc exceedances were also observed in these areas (see Figure 4-11). The highest concentrations of zinc (up to 10,800 µg/L) were measured in the groundwater beneath the former acid neutralization pits at the former Raymark Facility. High levels of zinc (up to 8820 µg/L) were also found in the groundwater from the deep overburden and bedrock of the southern bedrock valley. Additional areas of localized high zinc concentrations were detected beneath 540 Longbrook Avenue, the Raybestos Memorial Ball Field, the Housatonic Boat Club, and the south side of I-95 near the Main Street overpass. Figure 4-11 shows elevated levels of zinc rimming the outer margin of the southern half of the shallow groundwater image. The continuity of this v-shaped area is an artifact of the contouring algorithm, and it is driven by the following five shallow data points that lie near the edges of the model domain: MW-4, at the southern end of 540 Longbrook Avenue; MW-313S, west of Sutton Avenue; MW-102S, at 600 East Broadway; MW-532S, at the Housatonic Boat Club; and MW-535S, east of Minor Avenue. Zinc concentrations at these five locations ranged from 96 µg/L to 466 µg/L. Since there were few intervening monitoring points, the computer program connected these five isolated areas.

The spatial distributions of copper and lead in groundwater were similar (see Figures 4-12 and 4-13). At shallow depths the highest concentrations of both elements (797 µg/L copper and 37 µg/L lead) were found at PC-10S on the western edge of the former Raymark Facility. Elsewhere at shallow depths both elements were relatively abundant beneath 600 East Broadway and the former Raymark Facility acid neutralization pits. Copper concentrations remained high a short distance downgradient from the former acid pits, and they were high beneath 540 Longbrook Avenue. Deeper in the aquifer, both elements exceeded their SWPC in the overburden and bedrock of the southern bedrock valley. Overall, concentrations of copper ranged from 0.5 µg/L to 9960 µg/L, with 23 percent exceeding the SWPC of 48 µg/L. Concentrations of lead detected ranged from 1.4 µg/L to 227 µg/L, with 29 percent exceeding the SWPC of 13 µg/L.

Figures 4-14 and 4-15 show the highest concentrations of cadmium (up to 553 µg/L) and nickel (up to 17,900 µg/L) beneath 540 Longbrook Avenue and western portions of the former Raymark Facility. Further to the east in the southern bedrock valley, the distribution of these two metals resembled those for beryllium and copper. Cadmium was detected in 59

groundwater samples, and 47 percent of which exceeded the SWPC of 6 µg/L. Nickel was detected in 185 samples, 8 percent of which exceeded the SWPC of 880 µg/L.

Summary

Concentrations of arsenic, selenium, and nine heavy metals exceeded Connecticut's SWPC in at least one groundwater sample collected from the OU2 study area during the winter of 2002/2003. Metal contamination was most noticeable at the four locations noted in the following discussion. Shallow groundwater beneath portions of 600 East Broadway exceeded the SWPC for arsenic, copper, lead, mercury, and zinc. Groundwater beneath 540 Longbrook Avenue and the abutting portion of the former Raymark Facility often exceeded the SWPC for cadmium, copper, lead, nickel, and zinc. Groundwater in the overburden and bedrock near the former Raymark Facility acid neutralization pits surpassed the SWPC for beryllium, cadmium, copper, lead, nickel, and zinc. Groundwater from the deep overburden and bedrock associated with the southern bedrock valley often exceeded the SWPC for arsenic, beryllium, cadmium, copper, lead, nickel, thallium, and zinc. The groundwater near the former acid pits and in the deeper portions of the southern bedrock valley was strongly acidic with a pH as low as 3.0. Beyond the immediate vicinity of the acid pits and in the shallow overburden of the southern bedrock valley, the pH was typically near neutral and heavy metal concentrations tended to be much lower.

4.3.5 Dioxins in Groundwater

The types and distribution of dioxins in the overburden and bedrock aquifers are presented below. Tables 4-5 and 4-6 provide summaries of the analytical results for contaminants detected in the overburden and bedrock aquifers, respectively. Groundwater modeling was not conducted for dioxins, so no figures are presented. Results discussed below reflect only the comprehensive groundwater sampling effort performed during the winter of 2002/2003. Groundwater samples collected during previous sampling rounds were not analyzed for dioxins.

The term "dioxins" is commonly used to refer to a specific group of polychlorinated dibenzo-p-dioxin chemical compounds. The toxicity of one specific compound, 2,3,7,8-tetrachloro-dibenzo-p-dioxin (2,3,7,8-TCDD), has been studied more than other known dioxins and furans. The toxicities of all other dioxins and furans are expressed in relation to 2,3,7,8-TCDD. Concentrations of each individual dioxin and furan in a sample are multiplied by Toxicity

Equivalent Factors (TEFs) to yield 2,3,7,8-TCDD equivalent concentrations. These values are then totaled to yield total dioxin Toxicity Equivalent (TEQ) concentrations. The TEF used for these calculations are found in “Toxic Equivalency Factor for PCBs, PCDDs, PCDFs for Human and Wildlife” (Van der Berg et. al.1998).

Overburden

The distribution of dioxins in the overburden aquifer during the 2002/2003 sampling effort is presented below. One hundred and forty-four groundwater samples were collected from the overburden aquifer and analyzed for dioxins.

Sixty-three samples contained dioxins. Toxicity Equivalent concentrations ranged from 0.00029 pg/L to 18 pg/L. The highest concentration was detected at MW-211D. Dioxins were also detected in overburden wells upgradient of the former Raymark Facility. The maximum upgradient concentration for overburden wells was 13 pg/L. Samples from three locations exceeded that concentration: MW-211D, MW-504D, and MW-PC05M. There is no SWPC for dioxins to use for comparison.

Bedrock

The distribution of dioxins in the bedrock aquifer during the 2002/2003 sampling effort is presented below. Forty-nine groundwater samples were collected from the bedrock aquifer and analyzed for dioxins.

Nineteen samples contained dioxins. Toxicity Equivalent concentrations ranged from 0.00047 pg/L to 8 pg/L. The highest concentration was detected at MW-215B. Dioxins were also detected in bedrock wells upgradient of the former Raymark Facility. The maximum upgradient concentration for bedrock wells was 1.8 pg/L. Samples from three locations exceeded that concentration: MW-217B, MW-215B, and MW-215DB. There is no SWPC for dioxins to use for comparison.

Summary

Low concentrations of dioxins were infrequently detected in overburden and bedrock groundwater samples collected from the OU2 study area.

4.4 Nature and Extent of Contamination in Indoor Air and Soil Gas

Soil gas and indoor air sampling were conducted between April 2000 and March 2003. Both soil gas samples and indoor air samples were collected to evaluate the extent of groundwater contamination that may have resulted from past manufacturing and waste handling practices of Raymark Industries, Inc. In addition, the sampling effort was conducted to determine whether the contaminated groundwater is volatilizing or migrating into the homes and businesses downgradient of the former Raymark facility.

The soil gas and indoor air sampling was performed at numerous homes and light commercial buildings within the area of concern, downgradient and east of the former Raymark Facility. The indoor air area of concern can be defined as the area of the groundwater plume where shallow groundwater concentrations exceed the Connecticut groundwater volatilization criteria. TCE has been identified as the contaminant of greatest concern through preliminary groundwater comparisons to volatilization criteria and through preliminary examination of indoor air sampling results. The proposed Connecticut groundwater volatilization criteria for residential scenarios for TCE is 27 µg/L. The proposed Connecticut groundwater volatilization criteria for industrial worker scenarios for TCE is 67 µg/L. Figure 4-20 delineates the estimated area of the plume where shallow groundwater concentrations of TCE are expected to exceed these criteria. The indoor air area of interest for residential properties is the zone within the area delineated by the Connecticut groundwater volatilization criteria for residential scenarios of 27 µg/L. The indoor air area of interest for commercial properties is the area within the zone delineated by the Connecticut groundwater volatilization criteria for industrial worker scenarios of 67 µg/L.

Figure 2-4 shows sampling locations for soil gas and shallow groundwater within the area of interest. Sample results for soil gas and indoor air for properties where sampling was performed are not specifically indicated on any figures to avoid presenting individual property results.

4.4.1 Volatile Organic Compounds in Indoor Air

Indoor air samples were collected during April 2000, February and March 2001, February and March 2002, July 2002, February 2003, and March 2003. Indoor air samples were collected to evaluate the impact on homes from volatilizing and migrating volatile organic compounds from contaminated groundwater resulting from past practices of the Raymark Industries, Inc. manufacturing and waste handling.

EPA collected Tedlar Bag grab samples and integrated SUMMA canister samples. Typically grab samples were collected for the seven target VOCs that were identified as contaminants in the groundwater within the OU2 study area that exceeded the CTDEP groundwater volatilization criteria. The seven target VOCs were: 1,1-dichloroethene (1,1-DCE), vinyl chloride, trichloroethene (TCE), benzene, 1,1,1-trichloroethane (1,1,1-TCA), chlorobenzene, and toluene. These were selected after reviewing the available groundwater data for the entire study area. These seven VOCs exceed the CTDEP groundwater volatilization criteria at one or more locations within the OU2 study area. Only three of the seven VOCs, TCE, vinyl chloride, and 1,1-DCE, were actually detected above the groundwater volatilization criteria within the groundwater beneath the indoor air area of interest.

SUMMA canister samples, collected over an 8-hour period, were analyzed for an extensive list of VOCs. The results are summarized on Table 4-9 and indicate the presence of numerous VOCs in the subject locations. Since SUMMA canister results are considered more reliable, and they were available from each location where grab samples were collected, only SUMMA results are shown on the table. In addition, data collected after installation of sub-slab ventilation systems were not included.

Numerous VOCs, including 1,1,2,2-tetrachloroethane, 1,1,2-TCA, 1,1-DCE, 1,2,4-trimethylbenzene, 1,2-dibromoethane, 1,2-DCA, 1,3,5-trimethylbenzene, 1,3-butadiene, 1,4-dichlorobenzene, 4-methyl-2-pentanone, acetone, acrylonitrile, benzene, benzyl chloride, bromodichloromethane, carbon tetrachloride, chloroethane, chloroform, chloromethane, cis-1,2-DCE, dibromochloromethane, dibromofluoromethane, ethylbenzene, hexane, methyl tert-butyl ether, methylene chloride, PCE, tetrahydrofuran, toluene, xylene, TCE, vinyl acetate, and vinyl chloride, were detected at concentrations that exceed human health criteria (EPA Region IX preliminary remediation goals for ambient air) as discussed in Section 6.2. The

contaminants detected at highest concentrations were methylene chloride and xylene. Approximately 30 different VOCs were detected in at least one half of the samples and 25 VOCs were detected in samples from homes being considered as background locations. Benzene, toluene, and xylene were detected in every sample in every home, including the background locations.

A closer review of the indoor air sampling results for the three target contaminants detected at concentrations exceeding the groundwater volatilization criteria, within the groundwater beneath the indoor air area of interest, reveals the following:

- 1,1-DCE was detected in 42 of 69 samples, with a maximum concentration of 48 ug/m^3.
- TCE was detected in 39 of 69 samples, with a maximum concentration of 52.5 ug/m^3.
- Vinyl chloride was detected in 2 of 69 samples, with a maximum concentration of 0.62 ug/m^3.

These three contaminants were not detected in indoor air at background locations.

Summary

The indoor air sampling results indicate some volatilization of the target VOCs into the subject locations. 1,1-DCE, TCE, and vinyl chloride were detected in indoor air; however, the concentrations detected were not as high as modeled indoor air concentrations predicted from the maximum groundwater concentrations reported. 1,1-DCE and TCE were widely detected. Vinyl chloride was not widely detected. In addition, numerous VOCs, which were not detected in groundwater at concentrations expected to result in indoor air concentrations of concern, were detected at concentrations that exceed human health criteria in indoor air samples. Refer to Section 6.0 for an evaluation of the risks to human health from indoor air contaminants.

4.4.2 Volatile Organic Compounds in Soil Gas

This evaluation of the nature and extent of VOCs in soil gas is based on the soil gas data collected by TtNUS during the 2002 groundwater and soil gas profiling effort (see Section 2.12.1). The purpose of this sampling effort was to determine the extent of VOC contamination in residential areas hydraulically downgradient from the former Raymark Facility.

Between February and July 2002, soil gas samples were collected from a total of 84 locations using a geoprobe-based sampling system that is described in Section 2.12.1. Samples were analyzed for seven target VOCs: 1,1,1-TCA, 1,1-DCE, benzene, chlorobenzene, PCE, toluene, and TCE. The analytical results are tabulated in Appendix B.

In general, soil gas VOC concentrations decreased with distance above the water table, and concentrations were higher along roadsides. Property owner and physical access constraints limited sampling to areas close to roadways, so the apparent absence of contamination far from roads is an artifact of the sampling program. Additionally, it is not clear to what extent the isolated areas of high concentrations detected along the roadways are caused by the discontinuous nature of soil gas contamination or the lack of access to properties between these areas. The uneven distribution of soil gas sample locations is evident in Figure 2-4.

1,1-DCE was detected at 59 of the 84 sample locations, and concentrations were the highest in the soils beneath portions of Willow Avenue, Housatonic Avenue, and Barnum Avenue. Figure 4-16 shows the lateral distribution of 1,1-DCE at 8 feet below the ground surface. This depth was chosen because it illustrates the extent of contamination at the assumed depth of the bottom of basements. The highest concentration point (1040 ppb/v) was found at SG42 on Willow Avenue. Figure 4-17 is a cross section through the localized areas of high 1,1-DCE concentrations on Willow Avenue and Housatonic Avenue, which shows the general trend of increasing soil gas contamination with depth.

1,1,1-TCA was found at 86 percent of the sample locations, and its distribution was similar to that for 1,1-DCE. Localized areas of high 1,1,1-TCA concentrations were also detected at locations on Housatonic Avenue and Willow Avenue, and a third localized area of high concentrations was detected beneath Ferry Boulevard north of Minor Avenue and south of Riverview Place The highest concentration point of 1,1,1-TCA (1990 ppb/v) was found at SG16. SG16 was located next to Housatonic Avenue along the stretch between Willow Avenue and Homestead Avenue.

Fifty-eight locations had detectable levels of TCE in the soil gas; the distribution is shown on Figures 4-18 and 4-19. Once again, contamination appears to be higher beneath portions of Willow Avenue, Housatonic Avenue, and Ferry Boulevard, and concentrations tend to increase

with depth. As with 1,1,1-TCA, the highest concentration point of TCE (688 ppb/v) was measured in the deep soil gas at SG16.

Benzene, chlorobenzene, PCE, and toluene were detected much less frequently. Benzene was found at only three locations, and all three were beneath the parking lot at the 411 Barnum Avenue cut-off property. Benzene concentrations ranged from 14 to 150 ppb/v, and the maximum concentration was measured at SG218 near the Barnum Avenue cut-off. Chlorobenzene was also found at three locations beneath the parking lot at the 411 Barnum Avenue cut-off property, with concentrations ranging from 74 to 88 ppb/v. Of the 15 locations where PCE was detected, the highest concentrations (up to 36 ppb/v) were found at SG204, located on White Street near its intersection with East Broadway. The 14 remaining locations were spread throughout the area of interest. Toluene was detected at 15 locations, and the highest concentration by far (2385 ppb/v) was found at SG116 near the intersection of Housatonic Avenue and Minor Avenue. Two additional localized areas of high toluene concentrations were detected beneath the northeast and southwest edges of the 411 Barnum Avenue cut-off property at SG218 (318 ppb/v) and SG221 (352 ppb/v), respectively. As with PCE, the 12 remaining toluene locations were scattered throughout the soil gas study area.

Summary

VOC concentrations in soil gas tended to increase with depth. 1,1,1-TCA, 1,1-DCE, and TCE were widely detected, and concentrations were the highest along portions of Barnum Avenue, Ferry Boulevard, Housatonic Avenue, and Willow Avenue. By contrast, benzene, chlorobenzene, PCE, and toluene were found much less frequently and typically at lower concentrations.

5.0 CONTAMINANT FATE AND TRANSPORT

Contaminant fate and transport within the OU2 study area focuses on the groundwater contamination – its sources, migration pathways, and behavior during transport to potential human and ecological receptors. The potential source areas for groundwater contamination were described in general terms in Section 4.1 and are described in more detail in this section. The groundwater migration pathways were presented in Section 4.3, and the pathway for contaminated soil vapor emanating from the groundwater was described in Section 4.4. From these previous discussions, it is apparent that groundwater contamination from the former Raymark Facility is impacting Ferry Creek, the Housatonic River, and indoor air within structures overlying the contaminated groundwater. As described in Section 3.1.2, groundwater is not being used as a source of potable water within the study area, so there is no direct groundwater use impact. Therefore, the major objective of the fate and transport evaluation is to obtain reasonable estimates of contaminant concentrations that are occurring, or will occur in Ferry Creek, the Housatonic River, and indoor structures above contaminated groundwater.

5.1 Properties and Processes Controlling Contaminant Fate and Transport

Contaminant properties and aquifer processes affecting the fate and transport of groundwater contaminants are described in the sections below. Section 5.1.1 examines the impact of the physical phase of a contaminant on its fate and transport. Section 5.1.2 describes how transport processes move contaminants through the groundwater.

5.1.1 Physical Properties of the Contaminants in Groundwater

Organic and inorganic contaminants introduced at the ground surface through spills, discharges, and disposal practices have contaminated the groundwater within the OU2 study area. There are three principal physical phases in which contaminants occur in groundwater, and the physical phase dictates how contaminants are transported and changed within the aquifer. All three phases have been identified within the OU2 study area and are summarized as follows:

Dissolved Phase - Dissolved aqueous phase organic and inorganic contaminants generally migrate in the direction of groundwater flow. Three processes transport dissolved-phase contaminants: advection, mechanical dispersion, and molecular diffusion. These processes are

presented in Section 5.1.2 in more detail. In addition, dissolved contaminants may interact with both the aquifer matrix and other groundwater constituents during transport. A variety of chemical, physical, and biological interactions can significantly retard or accelerate the rate of dissolved contaminant transport, or transform the contaminants into other chemicals or states.

Non-Aqueous Phase Liquid (NAPL) – Organic liquid contaminants that occur within the groundwater as non-dissolved product are called non-aqueous phase liquids (NAPLs). Organic liquids such as gasoline have densities less than water and are referred to as light non-aqueous phase liquids (LNAPLs). Chlorinated solvents have densities greater than water and are examples of dense non-aqueous phase liquids (DNAPLs). Both forms of NAPL are believed to occur within the OU2 study area.

NAPL migrates differently than dissolved phase contaminants and often does not move in exactly the same direction as the groundwater. DNAPL, which is denser than water, tends to sink within the aquifer and its direction of migration is dictated more by gravity and geology than by groundwater flow. LNAPL, which is lighter than water, tends to float on the water table and follow groundwater flow. NAPL transport is discussed in more detail in Section 5.1.2.

Although NAPLs are not dissolved phase contaminants, their slow dissolution provides a continuing source of dissolved phase organics to the groundwater. A DNAPL may migrate in one direction according to gravity, while the dissolved phase generated at its surface moves in a different direction with the groundwater flow, producing a complex pattern of contamination. Depending on the local hydrogeology, groundwater chemistry, and composition of the NAPL, the rate of dissolution may be so slow that the NAPL will provide a continuous source of dissolved phase contamination for many centuries.

The concentrations and spatial distributions of organic contaminants in the groundwater indicate that: 1) DNAPLs exist at two of the investigated locations; and 2) these DNAPLs are a source of the dissolved phase contamination migrating toward Ferry Creek and the Housatonic River. The first location is in the vicinity of former Lagoon No. 4 on the former Raymark Facility, where 1,1,1-TCA and 1,1-DCE are believed to exist as DNAPL. The other location is beneath the former acid neutralization pits on the former Raymark Facility, where TCE-rich DNAPL has been observed in soil borings and well samples.

The concentrations of organic contaminants in the groundwater at one investigated location indicate the presence of LNAPL. Toluene is believed to exist as LNAPL in the north end of the former Raymark Facility where a spill of toluene was reported in 1984.

Suspended Solid Phase - Metals and some organic compounds can migrate through aquifers as suspended solids. Research has demonstrated that clay- and silt-sized particles in the colloidal range can migrate through an aquifer with the groundwater, and that certain organic compounds and metals can be retained on the colloid surfaces through surface charge attractions. Special groundwater sampling techniques have been used during the sampling of monitoring wells for this RI Report to obtain samples that include the colloids that would be transported with the groundwater, but exclude the particulates that would be introduced from the disturbance in the well caused by sampling. Several of the contaminants identified in the OU2 study area, particularly the metals, most commonly occur in the dissolved phase, but can also occur in the suspended solid phase. Therefore, some of the contamination detected in the OU2 groundwater may be moving as suspended solids.

5.1.2 Fate and Transport Processes for Contaminants in Groundwater

The major contaminant transport processes in groundwater are advection, mechanical dispersion, molecular diffusion, and NAPL transport. Advection, movement with the bulk flow of the groundwater, is the principal process by which dissolved and suspended phase contaminants are transported in aquifers. Dispersion and diffusion are secondary transport processes. NAPL transport differs from dissolved phase transport because it is more strongly influenced by the geologic features in the aquifer. Two additional processes that are important in determining the ultimate fate of contaminants in groundwater are degradation and retardation. Each of these fate and transport processes is briefly described below, and each is considered in the fate and transport modeling presented in Section 5.3.

Advection - The dominant transport mechanism for dissolved and suspended phase contaminants is advection, which is simply the movement of the contaminants with the bulk flow of the groundwater. Advective transport results from the entrainment of contaminants in a flow field, and it is driven by a potential gradient, such as pressure or hydraulic head. Flow directions for advective transport can be determined from the groundwater contours in Figures 3-7 and 3-8.

Mechanical Dispersion - Mechanical dispersion is a mixing process that results from velocity variations within bodies of moving fluids. Variable velocity regimes are caused by irregularities in the media in groundwater environments, and these irregularities exist at a variety of scales. For example, velocity variations at the microscopic scale arise from: 1) fluids moving faster through the centers of pores than along the edges; 2) fluids moving faster through large pore spaces than through narrow ones; and 3) some fluid particles following more tortuous flow paths than others as they travel around individual soil particles. At the macroscopic scale, velocity variations result from the presence of layers or lenses of materials having contrasting hydraulic conductivities.

The mixing caused by dispersion increases as aquifer heterogeneity increases, and it results in the dilution of a solute body as contaminated water mixes with uncontaminated water along the margins of a plume. Dispersion also results in the spreading of a contaminant plume over a larger area (both parallel and perpendicular to the direction of flow) than would be expected by advection alone. The effects of dispersion were incorporated into the contaminant transport models presented later in Section 5.3 via the longitudinal and transverse dispersivity parameters.

Molecular Diffusion - Molecular diffusion is movement in response to a concentration gradient. Dissolved contaminants will move from areas of high concentration to areas of low concentration within an aquifer even if the groundwater is not moving, because the process is driven by the random thermal motion of the contaminant molecules. Diffusive transport is a slow process; therefore, its impact is usually small compared to the more rapid processes of advection and dispersion. Diffusion is the dominant transport mechanism only in low-permeability hydrogeologic systems, and it is not incorporated into the models used for this investigation.

NAPL Migration - The subsurface transport of NAPLs is a complex process that is strongly influenced by the geology of the system. When a NAPL is spilled on the ground surface or released beneath it, the NAPL will migrate downward through the unsaturated zone toward the water table, primarily by gravity-driven flow. The presence of low permeability layers will inhibit downward migration and force the NAPL to move laterally. If the NAPL encounters a low permeability layer and the layer is continuous, downward movement may cease. If the layer is

discontinuous, the NAPL will eventually spill over its edge and continue to migrate downward toward the water table.

As the NAPL moves downward, the quantity of mobile free product decreases and the quantity trapped within soil pores increases. Depending on the volume of NAPL released, it may or may not reach the water table. If the NAPL reaches the water table and its density is less than water, it will remain in pockets at the top of the water column. If the density of the NAPL is greater than water, it will continue to move downward through the water column under the influence of gravity. Once again, downward migration may cease if the NAPL runs into a continuous, low permeability stratum. The direction of DNAPL transport in aquifers is driven primarily by gravity and the occurrence of relatively high and low permeability features; consequently, the direction of DNAPL transport may or may not coincide with the direction of groundwater flow. Since DNAPL migration is gravity-driven, its rate of movement is relatively fast, and it reaches its maximum lateral and vertical extent in the subsurface soon after it is introduced. Additional movement of the DNAPL takes place only if the system is disturbed by pumping, compaction, etc. The effect of the compaction of OU1 (former Raymark Facility) soils during site cleanup is unknown.

NAPLs are believed to exist in at least three locations within the OU2 study area, as described in Section 5.1.1. In the north end of the former Raymark Facility, the toluene-rich LNAPL resides in the upper portion of the water column and appears to be moving in the direction of shallow groundwater flow (see Section 5.3.4.5). Beneath the former acid neutralization pits, soil boring and well sample data indicate the TCE-rich DNAPL is restricted to the upper 40 feet of the water column; its downward migration is halted by the presence of a continuous silt-rich stratum filling the bottom of the northern bedrock valley as shown on Figure 3-5. DNAPL migration beneath former Lagoon 4 is more difficult to interpret. Free product has not been observed in soil borings or monitoring wells at this location, but groundwater concentration profiles for 1,1,1-TCA and 1,1-DCE suggest this DNAPL sank to the bottom of the southern bedrock valley beneath former Lagoon 4. Some of the DNAPL mass appears to be resting on the bedrock-overburden interface near monitoring well PC-02D, while an unknown portion entered adequately-sized and permeable bedrock fractures and migrated through at least the upper 20 feet of the bedrock near PC-02B. Further discussion of NAPL in the study area is presented in Section 5.2.

Degradation - Organic contaminants may be degraded by biological or nonbiological means. Degradation decreases the concentration of a solute in a plume, but it does not necessarily slow the rate of plume movement. Although some compounds degrade relatively quickly via abiotic pathways, biodegradation is typically the more important destructive mechanism. In biodegradation, microorganisms oxidize or reduce contaminants in their quest to obtain energy and nutrients. Depending on the microorganisms and contaminants present, biodegradation can occur under aerobic or anaerobic conditions.

Biodegradation is accomplished by microbially-mediated electron transfer reactions. In these reactions contaminants may be used as electron donors (a source of energy) or electron acceptors, or they may be fortuitously degraded by an enzyme or cofactor produced during the oxidation of other organic carbon sources (a process referred to as cometabolism). Through degradation, chemicals that were not originally present may be produced. In some cases, these daughter products may be more toxic than the original compounds released. Examples include chlorinated ethenes (PCE, TCE, DCE), which can be transformed to the more toxic vinyl chloride by sequential dehalogenation, and 1,1,1-TCA, which can be transformed to 1,1-DCE by abiotic degradation. Depending on environmental conditions these daughter products may resist further degradation and remain in the groundwater for a long time. The effects of biodegradation were incorporated into the organic contaminant fate and transport models presented in Section 5.3 through the decay constant. The value of the decay constant is based on the half-life of the contaminant.

Retardation - Most groundwater contaminants react to some extent with the aquifer's solid surfaces. Consequently, their transport is affected not only by the processes of advection, dispersion, and diffusion, but also by surface reactions. If the contaminants participate in adsorption/desorption reactions with mineral surfaces or the oxyhydroxide or organic coatings on these surfaces, the rate of contaminant transport will be slower than the rate of groundwater flow. The extent to which the movement of a plume is retarded relative to the rate of groundwater flow depends on the solute's propensity to sorb to the aquifer's surfaces. The propensity to sorb is governed by many factors including: the chemical character of the solute, the composition of the aquifer’s solid surfaces, and the groundwater chemistry.

Single parameter distribution coefficients (K_ds) are often used to quantify the tendency for a solute to sorb to media surfaces. K_ds are based on a linear model of adsorption – i.e. it is

assumed that the mass of solute sorbed increases linearly as its dissolved concentration increases. K_d theory works best for nonionic organic solutes such as chlorinated solvents and fuel hydrocarbons (Stumm and Morgan 1996). These compounds sorb primarily to organic coatings on mineral surfaces. A number of studies have shown that nonionic organic solute K_ds can be estimated from the fraction of organic carbon in the soil and the octanol-water partition coefficient (K_{ow}) of the solute. Site-specific K_{ow}-based distribution coefficients were used to describe the adsorption of organic compounds modeled in Section 5.3.

K_d theory also works for ionic solutes that: 1) sorb weakly; 2) participate in few reactions; 3) are present in low concentrations; and 4) move through geochemically homogeneous groundwater systems (Bethke and Brady 2000). The K_d construct is too simple to accurately represent the sorptive behavior of heavy metals, because they tend to sorb strongly and react to form various species and complexes. Nevertheless, the K_d approach is applied almost universally to models of heavy metal transport, because the site-specific data needed for more robust and realistic sorption models (Davis and Kent 1990; Koretsky 2000) are rarely available. For this same reason, the fate and transport of heavy metals in the OU2 groundwater system were evaluated with K_d-based models.

5.2 Contaminant Source Locations

Section 4.1 describes the potential contaminant sources that may be causing groundwater contamination within the OU2 study area. Of these potential sources of contamination, six locations were selected as meriting further evaluation based on their location at, or proximity to, the former Raymark Facility. These locations have been identified by combining several lines of evidence, including:

- reported spills or disposal of liquid and/or solid wastes;
- monitoring well data confirming the presence of contaminants associated with the wastes; and
- localized areas of groundwater contamination on the plume maps presented in Section 4.0, Figures 4-1 through 4-15.

The following is a listing and description of the six source locations (Source or Location) selected for fate and transport evaluation. The location of each of these sources is shown on Figure 5-1.

Source A - The 1984 reported toluene spill location near the northern end of the former Raymark Facility. Toluene-rich LNAPL was recovered from groundwater extraction wells placed near the spill. The LNAPL recovery system was abandoned prior to capping the former Raymark facility property, but high levels of toluene in post-cap groundwater samples collected from nearby shallow monitoring wells suggest the continued presence of toluene as LNAPL. This localized area of high toluene concentrations is shown on Figure 4-3.

Source B - The location of former acid neutralization pits on the former Raymark Facility. TCE-rich DNAPL has been observed in three wells (J-2, J-5, and RW-3) installed in the westernmost pit, and up to 810,000 µg/L of TCE has been measured in the groundwater (ELI 1995). TCE concentrations are shown on Figure 4-6.

Source C - The 1987 reported 1,1,1-TCA spill location. The 1,1,1-TCA reportedly ran into the site drainage that emptied into former Lagoon No. 4, which is Source D. 1,1,1-TCA concentrations are shown on Figure 4-4.

Source D - The location of Lagoon No. 4 on the former Raymark Facility. This lagoon (one of four lagoons on the former Raymark Facility) reportedly received the 1,1,1-TCA spilled at Source C in 1987. 1,1,1-TCA and 1,1-DCE concentrations in monitoring wells beneath the lagoon indicate that these compounds exist both as DNAPL and as dissolved phase contaminants. Figures 4-4 and 4-5 show localized areas of high concentrations at depth as would be expected with a DNAPL. The locations of all four lagoons are shown on Figure 1-2.

Source E – Two waste lagoons on the 540 Longbrook Avenue property immediately upgradient from monitoring well MW-Z. Source E is upgradient and adjacent to the former Raymark Facility property. Shallow groundwater collected from MW-Z has high levels of cadmium, nickel, and zinc. Figures 4-11, 4-14, and 4-15 show localized areas of high concentrations of these metals in the shallow subsurface near the down gradient edge of the lagoon complex.

Source F – Raymark Facility waste surrounding the MW-102 well cluster in the southwestern corner of 600 East Broadway. The 600 East Broadway property is an area where Raymark wastes were disposed. High concentrations of arsenic, copper, lead, and zinc were detected in groundwater samples collected from MW-102S, which is installed through Raymark wastes at the 600 East Broadway property. Copper, lead, and zinc (in addition to barium, cadmium, chromium, and nickel) have been detected in soil samples collected from areas of Raymark waste throughout Stratford at concentrations greater than the CT pollutant mobility criteria (PMC). The PMCs are the soil standards that indicate the potential for contamination to leach to groundwater. At 600 East Broadway, lead was the only contaminant detected at concentrations exceeding the CT PMC.

No apparent upgradient source of these metals exists since the upgradient property is residential. Figures 4-10, 4-11, 4-12, and 4-13 show the localized areas of high concentrations of these contaminants at 600 East Broadway. Cross sections on these figures indicate the contaminants occur beneath the 600 East Broadway property in the shallow subsurface, consistent with a source from leaching waste. Raymark waste at this property is not necessarily representative of all Raymark waste based on the changing conditions of wastes disposed in the lagoons in the former Facility; the nature in which the wastes were dumped onto properties; and the fact that other companies also dumped materials in Stratford.

5.3 Contaminant Fate and Transport Modeling

The fate and transport of several contaminants apparently emanating from the sources identified in Section 5.2 were evaluated using a one-dimensional groundwater modeling approach. The transport model explicitly accounts for many of the transport processes described in Section 5.1.2, such as advection, dispersion, retardation, and degradation. The following sections describe the formulation and limitations of the model, the methodology used to parameterize the model, and an estimate of the contaminant concentrations that are reaching, or will reach, either the Housatonic River or Ferry Creek. Information on model calibration and sensitivity is included in Appendix E.

5.3.1 Transport Model Formulation

Contaminant fate and transport can be modeled using analytical solutions to advection dispersion equations of the form:

$$\frac{\partial C}{\partial t} = D_x \frac{\partial^2 C}{\partial x^2} + D_y \frac{\partial^2 C}{\partial y^2} + D_z \frac{\partial^2 C}{\partial z^2} - v_x \frac{\partial C}{\partial x} - \frac{r}{n} \qquad (1)$$

The term on the left hand side of this equation represents the change in solute concentration with time. The first three terms on the right hand side describe the spreading (i.e. dispersion) of the solute in three dimensions as it moves through the aquifer. The fourth term on the right hand side represents the advective transport of the solute. Advection is mass transport caused by the physical movement of groundwater; therefore, the direction and rate of advective transport are governed by the direction and rate of groundwater flow. The final term in the equation describes a biological or chemical reaction of the solute (e.g. biodegradation or radioactive decay), which results in a loss of contaminant mass from the system.

Domenico (1987) developed an analytical solution to equation (1) that predicts the concentration of a contaminant along the longitudinal axis of a plume emanating from a continuous source.

$$C(x,0,0,t) = \frac{C_0}{2} \exp\left\{ \left(\frac{x}{2\alpha_x}\right)\left[1-\left(1+\frac{4\lambda\alpha_x}{v_x}\right)^{\frac{1}{2}}\right]\right\} erfc\left[\frac{x - v_x t\left(1+\frac{4\lambda\alpha_x}{v_x}\right)^{\frac{1}{2}}}{2(\alpha_x v_x t)^{\frac{1}{2}}}\right] \qquad (2)$$

$$\left\{ erf\left[\frac{Y}{4(\alpha_y x)^{\frac{1}{2}}}\right] erf\left[\frac{Z}{2(\alpha_z x)^{\frac{1}{2}}}\right]\right\}$$

where:

- C = predicted concentration at a given location (x,0,0) and time (t)
- C_0 = initial concentration at source
- x = distance from source

α_x = longitudinal dispersivity

λ = decay constant (λ = 0.693/half life)

v_x = groundwater velocity

α_y = horizontal transverse dispersivity

α_z = vertical transverse dispersivity

Y = source dimension in y direction

Z = source dimension in z direction

If the solute sorbs to the aquifer's solid surfaces as it moves through the system, the migration of the solute will be retarded relative to the flow of groundwater. The process of adsorption can be incorporated into equation (2) by substituting the contaminant velocity for the groundwater velocity according to

$$v_c = \frac{v_x}{1 + \frac{(1-n)}{n}\rho_s K_d} \qquad (3)$$

where:

v_c = contaminant velocity

n = porosity

ρ_s = bulk density of the aquifer solids

K_d = solute-specific distribution coefficient

As shown in equation (3), analytical models of reactive transport describe adsorption in terms of single parameter distribution coefficients. The distribution coefficient describes the solute's propensity to sorb to the aquifer's solid surfaces. The propensity to sorb is affected by many factors including: the chemical character of the solute, the composition of the aquifer's solid surfaces, and the groundwater chemistry. Hence, distribution coefficients are highly site- and solute-specific. K_d-based models work best for nonionic organic contaminants, such as chlorinated solvents. Sections 5.1.2 and 5.3.4.2 describe some of the limitations associated with the K_d approach to modeling ionic solute transport.

5.3.2 Model Assumptions and Limitations

Equations (2) and (3) assume the aquifer is homogeneous and isotropic, the flow field is uniform and one-dimensional, the source is continuous with a finite planar geometry, spreading occurs in two horizontal directions and one vertical direction, biodegradation can be described by a first order decay process, sorption can be described by simple linear models, and there are no additional sources or sinks along the flow path. Furthermore, the use of these equations to model transport from a groundwater source to a surface water receptor assumes the centerline (i.e. the longitudinal axis) of the plume intersects the bed of the stream. To the extent that these model assumptions are valid in the OU2 groundwater system, equations (2) and (3) can provide reasonable estimates of the concentrations of contaminants entering Ferry Creek or the Housatonic River at points in the streams where the centerlines of the plumes intersect the streams.

This modeling exercise was not intended to provide a description of contaminant fate and transport at the level of detail needed to evaluate remedial alternatives. Instead, the primary objective of this modeling exercise was to support the human health and ecological risk assessment component of the OU2 RI by generating reasonable worst-case concentrations for those contaminants predicted to enter Ferry Creek or the Housatonic River. The concentrations generated using equations (2) and (3) are considered worst case because they are the concentrations predicted to occur at the point in the stream bed where the centerline of the plume meets the stream bed. The concentrations are considered reasonable because representative values rather than maximum values were assigned to model parameters for which site-specific values were not available.

Both Ferry Creek and the Housatonic River are tidally-influenced surface water bodies. The cyclic, tidally-influenced fluctuations in surface water elevations cause fluctuations in groundwater discharge velocities to these rivers. Discharge velocities increase as surface water elevations decrease; conversely, velocities decrease as surface water elevations increase. Discharge will temporarily cease when water elevations in the river reach those in the aquifer. When surface water elevations exceed groundwater elevations, surface water will begin to move into the stream bank sediments. The dynamic interplay between surface water elevation, discharge velocity, and flow direction is affected by tidal strength and groundwater elevation, both of which vary over time. In order to model the effects of changing discharge rates and

surface water-groundwater mixing in the hyporheic zone, a detailed understanding of the temporal changes in tides and groundwater elevations is needed along with a mathematical model that does not assume a uniform flow field.

5.3.3 Modeling Methodology

The selection of contaminants to be modeled was based on three criteria: 1) does the solute have the potential to adversely effect human health or the aquatic biological community if it enters one of these surface water bodies; 2) could a source of the contaminant be identified; and 3) does the scientific literature contain sufficient information on the behavior of the contaminant to model its migration using equations (2) and (3). The modeled sources and pathways are shown in Figure 5-1. Each source and pathway delineated on Figure 5-1 is discussed in Section 5.3.4.

One or more sources were identified for each of the modeled contaminants using the three lines of evidence described in Section 5.2. Flow path directions and flow path lengths were determined from potentiometric surface maps and an evaluation of the vertical migration of the plume along its flow path (Figure 3-7 and 3-8). The hydraulic gradient was calculated using information contained in these maps along with additional hydraulic head data where needed. Hydraulic conductivity was obtained from pumping test data. Longitudinal dispersion was calculated using the regression equation presented in Xu and Eckstein (1995), and transverse dispersivities were based on the relationships between longitudinal and transverse dispersivities discussed in Gelhar et al. (1992).

Site-specific measurements of soil bulk density, porosity, and effective porosity were not available and had to be estimated from soil mineralogy and texture (EPA 1998). Boring logs (Appendix A) and geologic cross sections (Figures 3-3 to 3-5) indicate the soils along the flow paths shown on Figure 5-1 are predominantly quartz-rich sands. Peat layers and organic-rich silt layers are present in the OU2 study area, but these units are not continuous – even in the bed of Ferry Creek. For example, organic-rich strata were not observed in the sediments where flow line F meets the creek – this stretch of the creek appears to be underlain by sands and gravelly sands to a depth of at least 20 feet (see boring log for A1-SD-03 in Appendix A). Although Ferry Creek is underlain by organic-rich silt near the projected discharge point for flow path D, it is a relatively short distance from the discharge point for F, and temporal

concentration and head data suggest the locations of these discharge points vary over time. Therefore, to be conservative, flow along both paths was assumed to occur solely through sandy soils.

There were no soil borings where flow paths B and E meet the Housatonic River. At these locations, the contaminants would probably encounter organic-rich sediments beneath the riverbed that would delay their arrival and reduce the concentrations of organic contaminants entering the river. The modeling results discussed in Section 5.3.4 suggest that the inorganic contaminants from Source E would not arrive at the river in the foreseeable future if they traveled only through sand. Consequently, the primary modeling objective was met for the Source E contaminants without attempting to account for the additional retardation imposed by an unknown quantity of organic-rich sediment in the riverbed. The potential impact of the riverbed sediments on the TCE emanating from source B is described in Section 5.3.4.4.

Table 5-1 presents a summary of the parameter values assigned to equations (2) and (3) for each of the modeling scenarios described in Section 5.3.4.

5.3.4 Pathways and Predicted Migration of Groundwater Plumes

The sections below describe the movement of groundwater contaminants from the sources shown on Figure 5-1 to discharge points along Ferry Creek or the Housatonic River. The source areas are not presented in alphabetic order.

5.3.4.1 Source F and Related Pathway Shown on Figure 5-1

The migration of arsenic, copper, lead, and zinc from the 600 East Broadway property to Ferry Creek was modeled with the assumption that the Raymark waste contiguous with monitoring well MW-102S is the source of these four contaminants. [See Figure 3-11 in Appendix E. Figure 3-11 has been reproduced from the OU6 RI (TtNUS 2004) and included in Appendix E for convenience]. There are no known sources of contamination upgradient from MW-102S, and although there are other areas of Raymark waste and non-Raymark waste on the property that could potentially act as sources to the groundwater, high concentrations of arsenic, copper, lead, and zinc were only detected in the shallow groundwater at MW-102S. VOCs were not detected at this property above groundwater standards and therefore were not modeled.

Based on a review of aerial photographs, the Raymark waste appears to have been placed in the southwest corner of the Facility at some time between the years 1972 and 1992, so 1982 was chosen to be the starting date. According to Figure 3-7, shallow groundwater flows radially away from MW-102S toward the northeast, east, and southeast. The flow path with the largest source dimensions and the shortest distance to Ferry Creek trends to the northeast and intersects Ferry Creek near sediment boring A1-SD-03 (see Figure 5-1). This worst case flow path was modeled to support the risk assessments presented in Sections 6 and 7. A second flow path that trends southeast toward the swale between 230 and 250 Ferry Boulevard was also modeled (see Figure 5-1) to evaluate the likelihood that the metals detected in surface water collected from the swale could have originated from 600 East Broadway.

The source dimension in the Y direction (Y) for the northeast flow path (53 m) was defined as the width of the northeast boundary of the Raymark waste source (see Figure 3-11 in Appendix E). Y for the southeast flow path (6.1 m) was defined as the width of that portion of Raymark waste that lies in the path of groundwater flowing southeast from MW-102S. The thickness of the source for the northeast flow path (1.4 m) was the average depth to the bottom of Raymark waste in the six soil borings that define the source. The thickness of the source for the southeast flow path (2.4 m) was the depth to the bottom of Raymark waste at MW-102S.

To be conservative, the historic maximum concentrations of arsenic, copper, lead, and zinc detected in groundwater from MW-102S were the initial source concentrations in the models. According to Figure 3-7, the northeast flow line passes close to the MW-103 well cluster and intersects Ferry Creek near sediment boring A1-SD-03. The southeast flow line passes between MW-310 and MW-101, and intersects the swale near soil borings FBSWL-102, FBSWL-103, SP-SB4, and SP-SB8 (TtNUS 2004). Since the soils at these locations are sandy silts, organic silts, and silty sands to a depth of approximately 15 to 20 feet bgs, and since the discharge area along Ferry Creek near A1-SD-03 is underlain by sand, the models assumed the contaminants move through silty sand, and bulk density and porosity values representative of silty sand were assigned to equations (2) and (3). The hydraulic conductivity along both flow paths was set at the mean value for single well pumping tests conducted in silty sands and peat, and distribution coefficients were derived from published adsorption data. Davis et al. (1993) measured the adsorption of lead and zinc onto carbon-poor, quartz-rich sands under pH conditions similar to those found in the groundwater beneath 600 East Broadway. Distribution coefficients for arsenic and copper were estimated from the pH and Eh of the groundwater and

the adsorption behavior of these two contaminants relative to those for which published data were available.

The modeling results are shown in Table 5-2 and Figure 5-2. According to the model, arsenic originating in the groundwater near MW-102S is predicted to reach Ferry Creek at the point where the centerline (i.e. the longitudinal axis) of the plume intersects the stream with a maximum value of 84 µg/L by the year 2110. The leading edge of the zinc plume is predicted to have already reached Ferry Creek, and a maximum sustained point concentration of 330 µg/L may discharge to the creek by the year 2030. Concentrations of copper and lead above Connecticut's surface water protection criteria are not forecast to arrive at Ferry Creek until the year 2200. Although the K_d-based models predict that the copper and lead plumes will eventually extend to the creek, the copper and lead in the groundwater at MW-102S may never actually reach the creek because of their strong propensity to sorb to the aquifer matrix.

Maximum concentrations of arsenic (24 µg/L) and zinc (97 µg/L) are forecast to reach the Ferry Boulevard swale around the years 2130 and 2040, respectively. Copper and lead emanating from the MW-102S source are not predicted to reach the swale in the present century (2200). Consequently, it is highly unlikely that any arsenic, copper, or lead currently detected in surface water samples collected from the swale originated from the groundwater beneath 600 East Broadway. Additionally, the model assumed that the contaminants pass exclusively through silty sand en route to the swale. In actuality, the swale is underlain by several feet of organic-rich soil. The thick layer of organic-rich soil beneath the swale will strongly sorb metals and severely restrict groundwater discharge; as a result, it appears that the groundwater beneath 600 East Broadway will never be a significant source of contamination to the surface water in the swale.

5.3.4.2 Source E and Related Pathway

In this modeling scenario (Appendix E presents further details on the modeling efforts) the two lagoons on the 540 Longbrook Avenue property immediately upgradient from monitoring well MW-Z are assumed to be the source of the high concentrations of cadmium, nickel, and zinc detected in the shallow groundwater at MW-Z. According to Figure 3-1, the land immediately down gradient from this well is covered by asphalt and/or concrete for more than 1000 feet. Since Source E is shallow and there is little infiltration to drive the contaminants deeper in the

aquifer along the first 50 percent of the projected flow path, it was assumed that the metals migrating from this source would remain in the shallow groundwater system delineated in Figure 3-7, pass over a swale in the bedrock surface near MW-301 (Figure 3-6), and discharge to the Housatonic River.

The model assumed the historic maximum concentrations of cadmium, nickel, and zinc detected in MW-Z represented the steady state concentrations of these elements in the groundwater at the source. Source geometry (Y = 65 m, Z = 9 m) coincided with the geometry of the lagoons, and the starting date for the model was chosen to be the year in which the lagoons were closed (1970). The average hydraulic conductivity of the overburden in the OU2 study area (109 feet per day) was used to calculate groundwater velocity, and distribution coefficients were taken from Davis et al. (1993) and Christensen et al. (1996). Christensen et al. (1996) measured cadmium and nickel distribution coefficients for 18 aquifer sands from 12 locations in Denmark. The physical and geochemical properties of the Finderup 1 sands were most similar to those at the Raymark Facility, so the Finderup 1 distribution coefficients were used in this modeling scenario.

According to the analytical model, the leading edges of the cadmium and nickel plumes are not predicted to arrive at the Housatonic River before the year 2200. By contrast, the leading edge of the zinc plume is forecast to arrive within the next 50 years and a maximum sustained concentration of 1100 µg/L is expected 150 years later. The model also predicts that zinc concentrations in 2003 should be greater than 300 µg/L in the closest down gradient well (PC-09S); however, the zinc content of the groundwater in both PC-09S and PC-09D was less than 2 µg/L in 2003. Even if the center line of the plume passed 150 feet north of the PC-09 wells, the 3-D version of equation (2) predicts zinc concentrations at PC-09S to be 100 µg/L in 2003. Perhaps the zinc concentrations in these two wells are so low because the plume has not yet reached them. It may be moving through soils with a below average hydraulic conductivity, and/or the actual zinc distribution coefficient (K_d) may be larger than 3.5 ml/g. Alternatively, the reliance on a simple K_d value to describe sorption may be overestimating the extent of the plume and its rate of advance. K_d-based models do not impose a mass balance on the system, and they assume a plume migrates in the direction of groundwater flow at a constant rate determined by the retardation factor. In actuality, the rate of heavy metal plume migration decreases with time (Bethke and Brady 2000).

The maximum historic concentrations of cadmium (2140 µg/l), nickel (71,500 µg/l), and zinc (44,800 µg/l) were found in groundwater samples collected from PC-16M near the westernmost acid neutralization pit on the former Raymark Facility. However, groundwater chemistry changes radically with distance from this source, so a more complex and data intensive modeling tool (e.g. PHREEQC) would be required to simulate metal migration. Since groundwater sample results show that cadmium, nickel, and zinc concentrations are orders of magnitude lower in nearby down gradient wells, it appears the metals rapidly leave the liquid phase when they leave the local acidic environment beneath the pit, and there is no need to develop complex models of metal migration from this source.

5.3.4.3 Sources C and D and Related Pathways

The highest concentrations of 1,1,1-TCA (up to 185,000 µg/L) and 1,1-DCE (up to 42,000 µg/L) were detected in groundwater samples collected from monitoring wells PC-02D and PC-02B. These monitoring wells were installed near Lagoon 4 on the former Raymark Facility. PC-02D was placed immediately above the bedrock-overburden interface near the longitudinal axis of the southern bedrock valley shown in Figures 3-6 and 3-8. PC-02B was sited slightly below the top of rock. According to Figure 3-8, the groundwater in this portion of the valley flows south toward Ferry Creek. The spatial distributions of 1,1,1-TCA and 1,1-DCE within the valley support this conceptual model of groundwater flow (see Figures 4-4 and 4-5). The concentrations of both contaminants tend to decrease as distance from the PC-02 well cluster increases and distance from Ferry Creek decreases. Furthermore, the highest concentrations within individual well clusters tend to occur at progressively shallower depths as the distance from PC-02 increases and the distance to Ferry Creek decreases. These observations suggest the contamination near PC-02D and PC-02B may be moving southward and upward through the valley toward Ferry Creek as shown on pathway D on Figure 5-1.

In 1987, approximately 6000 gallons of 1,1,1-TCA were released at Source C on Figure 5-1, and the contaminant reportedly entered the Facility drainage system, which emptied into Lagoon 4. 1,1,1-TCA is a potential source of 1,1-DCE; 1,1-DCE may form by hydrolysis of 1,1,1-TCA or by thermal degradation of 1,1,1-TCA during use. 1,1-DCE is also used in the manufacture of plastics and adhesives (Howard 1989). 1,1,1-TCA and 1,1-DCE were present in groundwater samples collected in 1997 from PC-02B at 15 percent and 10 percent of their respective pure phase solubilities. Maximum concentrations in groundwater samples collected

from PC-02D indicate 13 percent 1,1,1-TCA saturation and 8 percent 1,1-DCE saturation. Since monitoring wells rarely encounter DNAPLs and screen lengths tend to be longer than the vertical extent of DNAPL zones, dilution usually produces less than saturated conditions in samples collected from wells near DNAPLs. Because saturated conditions are usually not found in wells near DNAPL, the presence of at least 1 percent saturation is reasonable for inferring the presence of DNAPL (Pankow and Cherry 1996).

1,1,1-TCA and 1,1-DCE were present at concentrations far above 1 percent saturation in both PC-02D and PC-02B; therefore, these two wells are probably very close to a DNAPL zone resulting from the 1987 1,1,1-TCA spill and perhaps from additional releases of these contaminants into Lagoon 4. Since the source is deep and the centerline of the plume rises en route to Ferry Creek, equation (2) was modified to account for transverse dispersion in two z-directions. Furthermore, the spatial distribution of 1,1-DCE in the southern bedrock valley strongly suggests the centerline of the 1,1-DCE plume does not intersect Ferry Creek, but instead passes beneath it (Figure 4-5). For that reason, the 3-D version of equation (2) was used to forecast the maximum concentration of 1,1-DCE entering Ferry Creek from the upper margin of the plume. The model was calibrated to concentration data from monitoring wells positioned near flow path D. Flow path D is shown on Figure 5-1.

The historic maximum groundwater concentration of 1,1-DCE was selected as the source concentration in the model (see Table 5-1). The source dimensions were assumed to be 30 meters in the Y direction and 10 meters in the Z direction. The Y dimension coincides with the width of Lagoon 4 in the down gradient direction and the estimated width of the bedrock valley floor at this location. The Z dimension encompasses the PC-02B well screen, the overlying bedrock, and the bottommost 8 feet of the overburden. The hydraulic conductivity from the large-scale pumping test conducted in the bedrock valley near the centerline of the 1,1-DCE plume was used to calculate velocity. A site-specific distribution coefficient was obtained from the octanol-water partition coefficient (K_{ow}) for 1,1-DCE and the fraction of organic carbon (f_{oc}) in the soil (Karikoff 1984). The f_{oc} in the OU2 soils was based on the total organic carbon in the most transmissive sands sampled during the drilling of upgradient well boreholes (EPA 1998). A site-specific decay constant was calculated from monitoring well data following the method of Zhang and Heathcote (2003).

As shown in Table 5-3, the model matches down gradient concentrations quite well, particularly at the monitoring locations closest to the source. The model results indicate the plume reached a steady state from the source to Ferry Creek prior to 1997, supporting the statistical results presented in Section 4.3.1. The model results also suggest a sustained 1,1-DCE concentration of 500 µg/L enters Ferry Creek at the point where the plume's vertical plane of symmetry intersects the creek bed. There are no monitoring points in the creek or near the north shore of the creek at this exact location. However, in 2003, 870 µg/L of 1,1-DCE was measured at -3 feet elevation mean sea level (MSL) in MW-520S, positioned about 35 meters upstream near the north shore of the creek. The 3-D version of equation (2) predicts that the concentration should be 200 µg/L at this location. The model also slightly under-predicts 1,1-DCE levels in the groundwater beneath Ferry Creek at MW-111D and MW-111M (see Table 5-3). The higher concentrations observed at these three locations may reflect contributions from additional sources of 1,1-DCE. For example, it is possible that 1,1-DCE is being created by the hydrolysis of 1,1,1-TCA along the upper margin of the plume where the pH is near neutral. Moreover, groundwater flow and contaminant concentration information strongly suggest that some of the 1,1-DCE present beneath the acid neutralization pits on the former Raymark Facility property commingles with the Lagoon 4 plume near MW-514 and follows flow path D toward Ferry Creek.

Deep contamination passes beneath Ferry Creek and apparently follows the bedrock valley to discharge into the Housatonic River (see Figure 3-6). This is verified by the absence of detectable concentrations of target VOCs, including 1,1-DCE, in all of the shallow wells south of the creek, suggesting: 1) the creek is effectively capturing the shallow contamination that originates beneath the former Raymark Facility; and 2) the contamination passing beneath Ferry Creek does not get into the shallow groundwater before it reaches the edge of the Housatonic River.

5.3.4.4 Source B and Related Pathway

In 1993, two 1-meter thick zones of TCE-rich DNAPL were observed in the former J cluster wells, which were drilled into the soils of the westernmost acid neutralization pit (HNUS 1995). TCE-rich DNAPL was also observed in the soils of the easternmost acid neutralization pit in soil boring SB-10, approximately 200 feet northeast of the J cluster. In 2003, TCE-rich DNAPL was found in recovery well RW-3, near the site of the now-abandoned J cluster wells. All of the acid neutralization pits were reportedly closed by 1975, so the DNAPL was apparently emplaced

before this time. Previously, the flow path of the dissolved TCE plume emanating from the acid pit DNAPL source was not known (TtNUS 2000c). Based on the data available at that time and the aim to provide a worst case estimate of the TCE concentration discharging to local surface waters, it was assumed that the plume moved in a southerly direction from the source and discharged to Ferry Creek. However, additional data collected in 2002 and 2003, including 59 new soil borings and more extensive bedrock topography, water level, and water quality information, strongly suggest: 1) the dissolved TCE plume extends southeast from the source; and 2) the core of the plume remains in the overburden and discharges to the Housatonic River through a swale in the bedrock ridge between the MW-302 and MW-214 well clusters (see Figure 3-6 and flow path B on Figure 5-1).

Recent and historic soil boring observations and groundwater sample data suggest the TCE source lies beneath two, and perhaps all three, acid neutralization pits to a depth of 12 meters below the water table, where its vertical extent is restricted by silt-rich strata (see Figure 3-5). The silt-rich strata fill the bottom of the northern bedrock valley that trends northeast-southwest beneath the western end of the former Raymark Facility property (see Figures 3-6 and 3-8). The 2003 water level data shown on Figures 3-7 and 3-8 indicate the driving force for groundwater flow in the valley is toward the southeast. Since the top of the down gradient wall of the valley lies only a meter or so above the bottom of the source (i.e. the top of the silt-rich strata), the majority of the contamination flows over the southeastern wall of the bedrock valley and moves through the overburden beneath the former Raymark Facility until it reaches the southern bedrock valley along the eastern margin of the property. Only the lowermost portion of the dissolved plume enters bedrock fractures. The new water quality data support this revised conceptual model of groundwater flow and TCE transport, because groundwater samples from the old and new wells south of the source contain little TCE, while those from the bedrock and overburden wells southeast of the source contain up to 5700 µg/L and 21,000 µg/L of TCE, respectively. Upon reaching the southern bedrock valley, cross section A-A' on Figure 4-6 shows the lower portion of the plume is captured by the deep flow system depicted in Figure 3-8. The upper portion of the plume remains in the shallow flow system shown in Figure 3-7, crosses over the southern bedrock valley, moves beneath the residential area east of Ferry Boulevard, and discharges into the Housatonic River.

Since the analytical model assumes the media is homogeneous and the flow field is one-dimensional, only the upper portion of the TCE plume – the portion that moves through the

overburden and does not get captured by the deeper flow system upon reaching the southern bedrock valley – was modeled. The source was assumed to be a continuous zone of TCE contamination beneath the three acid neutralization pits, extending 120 meters in the Y direction and 8 meters in the Z direction. The Y dimension needed to be as large as 120 meters for the modeled concentrations to match the measured concentrations in MW-302D and MW-214D with reasonable estimates of dispersion. Y was chosen to be 8 meters because the bottom of the source zone was 12 meters below the water table, and groundwater from the upper 4 meters of the water column immediately down gradient from the source contained less than 200 µg/L of TCE. In fact, more than 80 percent of the down gradient groundwater samples collected from the upper 4 meters of the water column in 2003 contained less than 100 µg/L of TCE. Since the upper 4 meters of the water column generally contained little TCE and the bedrock surface was roughly 12 meters below the water table along most of flow path B, the model ignored Z spreading in both vertical directions.

The source concentration was assumed to be the average concentration of TCE in groundwater samples collected from recovery wells 1, 3, and 6. These three wells were sited beneath the easternmost and westernmost acid neutralization pits. The average overburden hydraulic conductivity calculated from single well pumping tests was used to represent the hydraulic conductivity along the flow path. The TCE distribution coefficient and decay constant were computed from site-specific data.

Modeled TCE concentrations provide a close match to measured concentrations in down gradient wells (see Table 5-3). The model indicates a sustained concentration of 5700 µg/L currently reaches the base of the Housatonic riverbed sediments at the point where the centerline of the plume intersects the bed. Everywhere else, the concentrations reaching the riverbed (from this source) are less than 5700 µg/L.

After the TCE reaches the base of the Housatonic riverbed sediments, its fate is less clear. Since the thickness and composition of the riverbed sediments is unknown, the model assumes 5700 µg/L discharge directly to the river. If the discharge zone is underlain by organic-poor sands (as in parts of Ferry Creek), 5700 µg/L is a reasonable estimate of the maximum sustained TCE concentration reaching the Housatonic River from the acid pit source. On the other hand, if the riverbed sediments within the plume discharge zone are thick, fine-textured,

and organic-rich, the actual maximum concentration entering the river should be much less than 5700 µg/L.

The most tenable conclusion from the TCE modeling is that the concentration of TCE entering the Housatonic River from the acid pit source has reached a maximum value, and it should remain at this value until the acid pit source is destroyed, depleted, or cut-off. As discussed in Section 5.3.4.3, the model corroborates the results of the statistical analyses presented in Section 4.3.1, which indicate the 1,1-DCE and TCE plumes have reached a steady state throughout the OU2 study area.

5.3.4.5 Source A and Related Pathway

In 1981, an underground toluene storage tank near location A on Figure 5-1 was replaced when a leak was found (Pedrosa 1983). In 1984, the Connecticut Department of Environmental Protection ordered Raymark to remove an underground storage tank that had released an unknown quantity of toluene at location A (ELI 1995). Figure 4-3 shows that toluene contamination is restricted to the shallow groundwater, and Figure 3-7 shows that shallow groundwater flows in an easterly to southeasterly direction away from the spill site. The water table is too flat and the monitoring well density is too sparse to model toluene migration with equations (2) and (3); nevertheless, groundwater sampling results shed some light on its fate and transport. In 1997, more than 12 years after the toluene was spilled, the PC-06 monitoring well cluster was installed at the spill site and toluene was not detected in the groundwater. At the same time, 170,000 µg/L of toluene were detected in the shallow groundwater at PC-04S, located 350 feet to the east. This concentration of toluene represents approximately 30 percent of its pure phase solubility, so it appears the toluene LNAPL has been migrating eastward from the spill site and was close to PC-04S in 1997.

PC-04S is sited on the former Raymark Facility property about 20 feet west of East Main Street. In 1997, toluene was also present at high concentrations (up to 94,000 µg/L) in one monitoring well (MW-1S) and two direct push borings (DPA1-2 and DPA1-5) on the opposite side of East Main Street. Toluene has not been found in the groundwater 800 feet further to the east in MW-4S. However, the MW-4 cluster is the only monitoring location east of MW-3S, DPA1-2, and DPA1-5, so the lateral extent of the toluene plume is poorly understood.

Groundwater quality data collected between 1997 and 2003 indicate the dissolved-phase toluene is biodegrading. The rate of biodegradation is unknown, but dissolved oxygen concentrations are low and carbon dioxide concentrations are very high in the shallow groundwater at PC-04S and PC-03S, and toluene concentrations have decreased dramatically. Between 1997 and 2003, toluene concentrations dropped from 170,000 to 35,000 µg/L in PC-04S, and from 12,000 to 310 µg/L in PC-03S.

5.3.5 Model Uncertainty

There are two types of uncertainties in groundwater models: geologic uncertainties, and parameter uncertainties. Geologic uncertainties are those related to the locations of geologic boundaries: aquifer boundaries, boundaries between stratigraphic units, etc. Parameter uncertainties refer to uncertainties in the values of hydrogeologic parameters such as hydraulic conductivity. Fortunately, both sources of uncertainty can be reduced by collecting site-specific data. The large number of borings at the site has reduced the geologic uncertainties associated with the models presented in Section 5.3.4.

One of the greatest sources of parameter uncertainty in these contaminant transport models is associated with the source geometry. The sensitivity analysis in Appendix E shows that source dimensions exert a strong influence on concentration predictions. Another significant source of parameter uncertainty in the inorganic transport models lies in the distribution coefficients used to describe the adsorption and desorption behavior of the contaminants. The assigned K_d values were based on adsorption studies conducted at other sites because site-specific adsorption data were not available. Although the K_d values affect the modeled arrival times for the inorganic contaminants, they do not affect the predicted concentrations.

5.4 <u>Contaminant Persistence</u>

The organic contaminants of greatest concern in the OU2 groundwater system are the chlorinated aliphatic hydrocarbons because of their relatively high toxicities and their widespread occurrence at concentrations above surface water protection and volatilization criteria. Chlorinated aliphatic hydrocarbon plumes can persist for centuries or longer in some natural groundwater environments, depending on the mass of the source(s) and nature of the attenuation processes active in the system (Pankow and Cherry 1996). Attenuation processes

include: sorption, dispersion, dilution from recharge, volatilization, biodegradation, and abiotic degradation. Sorption retards the migration of contaminant plumes, and dispersion and dilution from recharge reduce contaminant concentrations in groundwater, but these processes do not destroy contaminants. Volatilization is a destructive process, but its impact on reducing contaminant mass is limited when the plume is below the water table. Although some compounds degrade relatively quickly via abiotic pathways, biodegradation is typically the most important destructive mechanism.

In natural systems chlorinated aliphatic hydrocarbons may be biodegraded by electron acceptor reactions (reductive dechlorination), electron donor reactions, or cometabolism. The most important biodegradation process for TCE and other highly chlorinated solvents is reductive dechlorination. In reductive dechlorination, chlorine atoms are sequentially removed and replaced by hydrogen atoms. The chlorinated hydrocarbon acts as the electron acceptor and additional carbon sources, such as fuel hydrocarbons or naturally occurring organic carbon, are the electron donors. If the source of electron donors is depleted before the chlorinated hydrocarbon is degraded, the reductive dechlorination process will cease, regardless of the ideality of other environmental conditions such as Eh or pH. Evidence for the occurrence of reductive dechlorination in an aquifer includes the accumulation of intermediate daughter products and an increase in chloride ion concentrations. Highly chlorinated aliphatics are the most susceptible to reductive dechlorination because they are the most highly oxidized. Reductive dechlorination proceeds more slowly with less oxidized compounds such as DCE and vinyl chloride.

Chlorinated aliphatic hydrocarbons resist degradation and tend to persist in aerobic environments because their oxidation yields so little energy. Microorganisms can not obtain sufficient energy for growth from the oxidation of highly chlorinated solvents. However, the less chlorinated and less oxidized aliphatics can be used as electron donors under aerobic and some anaerobic conditions.

In the presence of other electron acceptors and donors, chlorinated hydrocarbons may also be degraded by cometabolism – a process in which the chlorinated hydrocarbon is fortuitously degraded by an enzyme or cofactor produced during the oxidation of other organic carbon sources. Cometabolism can occur in aerobic and anaerobic environments, and the rate of degradation increases as the degree of dechlorination decreases.

The EPA has developed a protocol for evaluating the natural attenuation of chlorinated aliphatic hydrocarbons in aquifers (EPA 1998). The initial screening process begins with an evaluation of groundwater chemistry data to determine if and where the potential for natural biodegradation by reductive dechlorination exists. The screening process focuses on reductive dechlorination because this pathway is responsible for the initial biotransformation of most chlorinated solvents.

For the first step in the screening process, groundwater sample data from upgradient and cross-gradient wells, source area wells, down gradient wells within the plume, and down gradient wells beyond the plume are compared with the parameter values listed in the second column of Tables 5-5 through 5-23. Points are awarded if an individual parameter value in a groundwater sample favors reductive dechlorination. No points are awarded, or points are subtracted if a parameter value does not favor reductive dechlorination. For example, a low redox potential indicates a reductive pathway is possible, so samples having oxygen-reduction potential (ORP) readings below the cutoffs listed in the screening tables are awarded points. By contrast, high sulfate concentrations may interfere with reductive dechlorination because sulfate will compete with the chlorinated hydrocarbon as an electron acceptor. If sulfate concentrations exceed the value listed in the screening tables, no points are awarded for sulfate. Reductive dechlorination is likely when the analytical data for a groundwater sample yield an overall score of 15 points or more.

Sample data from 28 monitoring wells were analyzed to determine if and where reductive dechlorination is potentially occurring within the OU2 study area. Six locations (MW-303B, MW-307B, MW-308B, MW-501B, MW-501D, and MW-501S) provided information on upgradient concentrations of electron acceptors, electron donors, carbon dioxide, alkalinity, and chloride ions. Seven monitoring wells represented the TCE-rich (GW-J2, GW-J5, RW-1, RW-3, RW-6) and 1,1-DCE-rich (PC-02B, PC-02D) DNAPL source areas. These seven locations were used to identify the dominant electron accepting process at each source. Nine wells (MW-211B, MW-211D, MW-211M, MW-212D, MW-212M, MW-212S, MW-503S, MW-111D, and MW-111M) are situated within the dissolved 1,1-DCE plume. Five wells (PC-14D, PC-14S, MW-302B, MW-302D, and MW-302S) are situated within the dissolved TCE plume. Information from these 14 locations was used to determine if and where the plumes are degrading with distance from their respective sources. One additional well (MW-113M) provided information on the metabolic byproducts in the remediated groundwater in front of the leading edges of the plumes.

The data from upgradient and cross-gradient wells presented in Tables 5-5 to 5-9 show the uncontaminated groundwater is oxygen-rich. Dissolved oxygen contents ranged from 2.9 to 8 mg/L and the observed electrode potential (Eh) ranged from 230 to 446 millivolts (mV). The upgradient groundwater is also rich in nitrate and sulfate – compounds that interfere with reductive dechlorination because they compete with the chlorinated hydrocarbons as electron acceptors. Mean upgradient concentrations for nitrate and sulfate were 3.1 and 37.1 mg/L, respectively. Furthermore, the uncontaminated groundwater is poor in naturally-occurring organic carbon – the food source needed for reductive dechlorination when anthropogenic sources of reduced carbon are absent. Total organic carbon contents in the upgradient samples ranged from less than 2 mg/L to 5 mg/L.

The TCE source area evaluation relied on 1993 groundwater data from the former "J" cluster wells on the former Raymark Facility, in addition to the 2003 data from RW-1, RW-2, and RW-3, because several key parameters including chloride, sulfate, sulfide, nitrate, and total organic carbon were not measured in the groundwater collected from the DNAPL recovery wells. Groundwater from J-2 and J-5 had extremely high concentrations of nitrate, sulfate, and chloride, presumably from the acids that were routinely dumped in the overlying soils. As expected, the pH was very low from the disposal of acids, and the total organic carbon content was high because of the TCE (Table 5-10). Although detection limits for the J cluster samples were too high to measure most other organic compounds in the source area groundwater, the 2003 recovery well samples showed that the source area groundwater also contains high concentrations of 1,1,1-TCA, 1,1-DCA, PCE, 1,2-DCE, and toluene (Tables 5-11 and 5-12). 1,1,1-TCA, PCE, and toluene are present in the TCE-rich DNAPL (see Appendix B); however, the presence/absence of 1,2-DCE and 1,1-DCA in the DNAPL could not be confirmed because their detection limits were 40,000,000 µg/L. It may be that some or all of the 1,2-DCE and 1,1-DCA in the source area groundwater are parent compounds rather than daughter products, since alternative electron acceptors are plentiful, and the low pH and extremely high concentrations of toxic contaminants create an environment that is hostile to microbial activity.

Downgradient from the TCE source area, but within the TCE plume, the potential for natural biodegradation varied widely. Groundwater in the core of the plume collected from PC-14D, MW-302B, and MW-302D showed limited to adequate evidence of reductive dechlorination (Tables 5-13 and 5-14). Close to the source in PC-14D the pH was still quite low, the groundwater was well-oxygenated, and nitrate and sulfate were plentiful. Nevertheless, the

presence of elevated levels of ferrous iron, methane, chloride, and fuel hydrocarbons, together with low concentrations of potential daughter products, suggested that some biodegradation was taking place. Much further from the source in MW-302B and MW-302D, alternative electron acceptors were still plentiful and reduced carbon sources were limited, but there were indications of biodegradation including increased alkalinity, chloride, 1,2-DCE, 1,1-DCA, and chloroethane.

The greatest potential for reductive dechlorination was observed in the shallow groundwater about 65 meters from the former acid pits at PC-14S. At this location, the environment was reducing, the pH was near neutral, and metabolic byproducts and daughter products were abundant (Table 5-13). By contrast, the lowest potential for reductive dechlorination was found in the shallow, oxygen-rich groundwater much farther from the source at MW-302S (Table 5-15). Across the TCE plume, the shallow groundwater chemistry was highly variable, perhaps resulting from geologic heterogeneities in the shallow overburden.

Groundwater near the 1,1,1-TCA and 1,1-DCE source at PC-02D and PC-02B was oxygen-poor, strongly acidic, and loaded with ferrous iron and sulfate (Table 5-16). The ferrous iron and sulfate apparently originated upgradient from the 1,1,1-TCA and 1,1-DCE source, since the iron and sulfate concentrations were roughly the same in the groundwater at MW-533D and MW-533B. Down gradient from the source, but within the dissolved-phase plume, the potential for reductive dechlorination tended to increase as the distance from the PC-02 wells increased (Tables 5-17 to 5-22). Once again, groundwater from the shallow overburden varied in its potential for anaerobic biodegradation (Tables 5-20 and 5-21), but the potential was consistently high in the moderate to deep overburden more than 470 meters from the source.

Upgradient concentrations of carbon dioxide, alkalinity, and chloride were used as benchmarks for assessing the likelihood of increased microbial activity in the contaminant-laden groundwater. Beyond the leading edges of the TCE and 1,1-DCE plumes in the overburden filling the southern bedrock valley, parent compounds were no longer present, but concentrations of these three metabolic byproducts were several times greater than the values from upgradient wells (Table 5-23). Sulfide and methane, additional indicators of upgradient biodegradation, were also present.

The previous evaluation of contaminant persistence (TtNUS 2000c) was based on a smaller number of groundwater samples and a more limited suite of screening parameters. It indicated that some reductive dechlorination was taking place, but the degradation rates were unknown and the process appeared to be restricted to the edges of the chlorinated solvent plumes. This evaluation benefited not only from a greater number of samples and a larger suite of analytes, but also from lower chemical detection limits and an understanding that the chlorinated solvent plumes had reached a steady state. The more sensitive laboratory methods allowed daughter products to be detected in highly contaminated groundwater, and steady state conditions enabled the calculation of degradation rates (Zhang and Heathcote 2003).

The evaluation of contaminant persistence indicates the chlorinated aliphatic hydrocarbons in the OU2 groundwater system are undergoing biodegradation, and the process is not restricted to the outer margins of the plumes. Contaminant concentrations observed in samples collected from wells located down gradient from known or suspected sources often decrease with increasing distance from the source. Daughter products, including 1,2-DCE, 1,1-DCA, and vinyl chloride, are present in many of the groundwater samples. After accounting for the effects of sorption and dispersion, the fate and transport modeling results suggest the half-lives for 1,1-DCE and TCE in the OU2 groundwater system are four years and three years, respectively. The evaluation supports the earlier finding that significant biodegradation is not taking place near the sources. However, there is ample evidence for its occurrence in the cores of the plumes further from the sources where the groundwater environment is more hospitable. Nevertheless, the chlorinated hydrocarbon plumes will persist in the OU2 groundwater for decades, or even centuries, until the DNAPL sources that continually replenish the plumes are contained, destroyed, or depleted.

5.5 Summary of Fate and Transport of Contaminants in Groundwater

The migration of contaminants emanating from four sources shown on Figure 5-1 was modeled using the set of equations presented in Section 5.3.1. Other potential sources of groundwater contamination exist within the study area (e.g. metals in the soils and groundwater beneath the former acid neutralization pits on the Raymark property). These additional sources may be contributing contamination to Ferry Creek or the Housatonic River; however, they were not modeled because the sources were poorly defined or the chemistry of the groundwater environment varied dramatically along the projected plume flow path. Although the models

developed in Section 5.3.1 do not account for all of the contamination entering the surface water in the study area, they do provide specific information regarding the fate and transport of contaminants from four sources. They also provide general information regarding the behavior of these contaminants in the OU2 groundwater system.

Cadmium, copper, lead, and nickel are migrating very slowly in the OU2 groundwater system. The cadmium and nickel observed in the groundwater near MW-Z, and the copper and lead observed on the groundwater near MW-102S are not expected to adversely impact local surface waters in the near future. Zinc is more mobile. Zinc leaching from the fill near MW-102S is predicted to begin entering Ferry Creek at a maximum sustained point concentration of approximately 330 µg/L in roughly 25 years. Arsenic is relatively immobile in oxidizing and strongly reducing environments, but its mobility is enhanced under weakly reducing conditions. Since the groundwater near MW-102S has a moderately low redox potential, arsenic is predicted to enter Ferry Creek at a maximum sustained point concentration of 84 µg/L in approximately 100 years.

1,1-DCE and TCE are more mobile than arsenic and the five heavy metals. The modeling results suggest that the 1,1-DCE emanating from the DNAPL source near PC-02B and PC-02D should already be discharging to Ferry Creek at a maximum sustained point concentration of about 500 µg/L. The TCE originating from the DNAPL source beneath the former acid neutralization pits should have already reached the base of the Housatonic riverbed sediments at a maximum sustained point concentration of 5700 µg/L. The TCE concentration discharging to the surface water could not be predicted, because the character of the riverbed sediments in the vicinity of the discharge zone is not known.

Equations (2) and (3) constitute a relatively simple mathematical model that was used to describe fate and transport in a complex system. Therefore, it should be viewed simply as a tool to provide approximate arrival times and contaminant concentrations, help constrain hypotheses regarding the fate and transport of contaminants in the OU2 groundwater system, and guide future data collection efforts. Despite the limitations inherent in the modeling approach taken, the results strongly suggest that: 1) the concentrations of 1,1-DCE and TCE entering Ferry Creek and the Housatonic River will remain at their present levels until their sources are contained, destroyed, or depleted; and 2) the concentrations of the modeled inorganic contaminants discharging to Ferry Creek are expected to increase over time.

5.6 Fate and Transport of Contaminants in Soil Gas

Chemicals in the shallow groundwater that have high Henry's Law coefficients or vapor pressures tend to partition (volatilize) to the ambient air in the soil. Once in the soil gas, the chemicals may undergo physical transport by advection or diffusion, or they may be transformed through chemical processes such as hydrolysis or photolysis. Volatile organic compounds include chlorinated aliphatic hydrocarbons such as chlorinated solvents and their degradation products, and aromatic hydrocarbons such as benzene and toluene.

5.6.1 Contaminant Transport from Groundwater to Soil Gas

In February 2002 and March 2002, co-located groundwater and soil gas samples were collected at 34 locations in the OU2 study area (see Section 2.12.1) and relationships between groundwater and soil gas VOC concentrations were quantified. The VOCs analyzed in both groundwater and soil gas were: 1,1,1-TCA, 1,1-DCE, benzene, chlorobenzene, PCE, toluene, and TCE. Relationships were only evaluated for 1,1,1-TCA, 1,1-DCE, and TCE, because the other four compounds were detected in both groundwater and soil gas at fewer than five locations.

The frequency distributions for 1,1,1-TCA, 1,1-DCE, and TCE exhibited strong positive skewness. In other words, the histograms for these three compounds had long tails that extended to the right. Since the distributions were not bell-shaped, the nonparametric Spearman-Rho correlation test was applied to the data. The test results presented in Table 5-4 indicate that:

1. VOC concentrations in groundwater samples collected just below the water table were most strongly correlated with VOC concentrations in the overlying soil gas.

2. VOC concentrations in groundwater samples collected from depths greater than 15 feet below the water table were not correlated ($\alpha = 0.05$) with measures of VOCs in the overlying soil gas.

3. Relationships between VOC concentrations in groundwater and VOC concentrations in soil gas are not strong enough to predict the concentrations in one media from those in the other.

Apparently, differences in physical and chemical properties of VOCs, and heterogeneities in geologic, hydrologic, and geochemical conditions cloud the relationships between groundwater and soil gas and render simple statistical methods inappropriate for predictive purposes.

5.6.2 Contaminant Transport from Groundwater and Soil Gas to Indoor Air

As mentioned in Section 2.12.6, indoor air samples were also collected in 2002 from the basements of a few homes sited above contaminated groundwater. Based on the 2002 groundwater, soil gas, and indoor air sampling results, there are no statistically significant relationships ($\alpha = 0.05$) between the concentrations of target VOCs in groundwater and their concentrations in indoor air, nor are there any significant relationships between the concentrations of target VOCs in soil gas and indoor air. As a result of the small number of homes sampled in 2002 and the common occurrence of sample nondetects, sample numbers for relationships between measured VOC concentrations in groundwater and indoor air and measured VOC concentrations in soil gas and indoor air ranged from zero to five. Statistical evaluations were limited to data sets having sample numbers greater than three.

Johnson and Ettinger (1991) published a model for estimating the transport of contaminant vapors from groundwater or soil to the air inside buildings sited directly above the contamination. The model accounts for the partitioning of chemicals from the groundwater to the soil gas, the advective and diffusive transport of the vapor-phase chemicals through the soil, and the subsequent transport of the vapors through cracks in the basement slab floor. The Johnson and Ettinger (JE) model was applied to the 2002 groundwater, soil gas, and indoor air data, and the measured and predicted concentrations of VOCs in homes were compared using the Spearman-Rho correlation test and the Wilcoxon signed rank test. The results from these two statistical tests showed that:

1. When the JE model was used to forecast indoor air concentrations from measured groundwater concentrations, it tended to over-predict actual indoor air results.

2. When the JE model was used to forecast indoor air concentrations from measured soil gas concentrations, it under-predicted actual indoor air results.

3. The JE model did not over-predict or under-predict actual indoor air values in a systematic fashion (i.e. there were no quantifiable relationships between measured and predicted indoor air values).

It is not clear to what extent the inability to detect statistically significant relationships was caused by small sample numbers, unidentified contaminant sources, or variability in soil properties and/or basement integrity. However, a simpler variant of the JE model, summarized in the 1995 American Society of Testing and Materials (ASTM) Risk-Based Corrective Action (RBCA) Standard, was able to predict measured indoor air concentrations from soil gas data ($r = 0.88$) when the values for the one soil variable and the one basement variable were varied within reasonable limits and one residential property (364 Housatonic Avenue) was excluded from the analysis. These modeling results suggest that variability in soil properties and basement integrity may be responsible for the poor match between the JE model predictions (which assumed uniform soil and basement characteristics) and actual indoor air results. Nevertheless, the conclusions drawn from any of these analyses are tenuous in light of the small number of homes sampled.

6.0 BASELINE HUMAN HEALTH RISK ASSESSMENT

Section 6.0 presents the methodology and the results of a baseline human health risk assessment (HHRA) conducted for the Raymark OU2 groundwater study area described in Section 1.3. The objective of this assessment is to estimate potential current or future risks to the public from the chemicals detected in the groundwater within the study area. Section 6.1 provides an overview of the risk assessment process. Sections 6.2 through 6.5 outline the methodology and results of the baseline HHRA. Appendix F-1, Table 1 presents an overview of the various media, exposure points, potential receptors, and exposure pathways evaluated in this risk assessment. A detailed discussion of the potential receptors, exposure locations, and exposure pathways listed in Appendix F-1, Table 1 is presented in Section 6.3. Section 6.6 presents results of modeled indoor air exposures based on measured groundwater concentrations. An analysis of the uncertainties is presented in Section 6.7. The risk assessment conducted for this report follows the most recent guidance from the EPA (EPA 1989b and 1991), including regional EPA guidance (EPA 1989a, 1994, 1995b, 1996b, and 1999a). Tables were prepared following the standard format in accordance with Risk Assessment Guidance for Superfund, Human Health Risk Evaluation Manual (RAGS HHEM) Part D (EPA 1997e). These tables are presented in Appendix F-1.

This HHRA evaluates potential exposures to groundwater in the study area for four groups of receptors:

- Residents,
- Industrial/Commercial workers,
- Recreational users of Ferry Creek, and
- Fishermen consuming oysters from Ferry Creek or the mouth of the Housatonic River.

Residents This HHRA estimates the potential for human health risk from exposures to groundwater in the study area for residents within the indoor air area of interest described in Section 4.4 and shown in Figure 4-20. Residents were evaluated for exposures to indoor air contaminants that may be present as a result of volatilization of groundwater contaminants through building foundations into indoor air using two methods:

- Evaluation of indoor air sampling data, and

- Evaluation of indoor air concentrations modeled from shallow groundwater data.

Several limited indoor air sampling efforts were conducted in the residential area of OU2 between April 2000 and March 2003. Volatile contaminants have been detected in groundwater beneath this area. This HHRA estimates the potential for human health risk from exposures to indoor air contaminants detected in indoor air samples collected prior to installation of sub-slab ventilation systems. Sub-slab ventilation systems have been installed at several homes in the area. This HHRA evaluates risks assuming the absence of these systems or any other abatement measure.

An evaluation of potential risks from exposures to indoor air concentrations modeled from measured shallow groundwater concentrations in this area is included in the HHRA for comparison to measured indoor air concentrations and risks estimated based on those measured indoor air concentrations.

As described in Section 2, soil gas samples have also been collected within the indoor area of interest; however, this HHRA did not evaluate soil gas data.

Residents are not evaluated for direct exposures to groundwater through domestic water use, since groundwater in the OU2 study area and surrounding areas is not used as a drinking water source because of brackish conditions and productivity constraints.

Industrial/Commercial Workers This HHRA estimates the potential for human health risk from exposures to groundwater in the study area for industrial/commercial workers within the indoor air area of interest (described in Section 4.4 and shown in Figure 4-20). Industrial/commercial workers were evaluated for exposures to indoor air contaminants that may be present as a result of volatilization of groundwater contaminants through building foundations into indoor air using two methods:

- Evaluation of indoor air sampling data, and
- Evaluation of indoor air concentrations modeled from shallow groundwater data.

Several limited indoor air sampling efforts were conducted in the indoor air area of interest between April 2000 and March 2003. The majority of these samples were collected in homes;

however, a few were obtained from non-residential properties. Volatile contaminants have been detected in groundwater beneath this area. This HHRA estimates the potential for human health risk from industrial/commercial exposures to indoor air contaminants detected in indoor air samples collected from both residential and non-residential properties.

An evaluation of potential risks from exposures to indoor air concentrations modeled from measured shallow groundwater concentrations in this area is included in the HHRA for comparison to measured indoor air concentrations and risks estimated based on those measured indoor air concentrations.

Recreational users This HHRA estimates the potential for human health risk from exposures to groundwater in the study area for recreational users of Ferry Creek. Recreational users of Ferry Creek were evaluated for exposures to future surface water concentrations resulting from migration of groundwater to Ferry Creek. This evaluation used surface water concentrations modeled from measured groundwater concentrations.

Current surface water exposures at Ferry Creek have been evaluated in the OU3 Area I RI (TtNUS 1999a) and are summarized in section 6.5.2 of this report.

Current and future surface water concentrations in the Housatonic River may also be impacted by groundwater contamination. The groundwater transport modeling, described in Section 5, indicates the movement of trichloroethene (TCE) toward the Housatonic River. Current and future surface water concentrations in the Housatonic River have not been evaluated in this report.

Fishermen This HHRA provides a qualitative evaluation of fishermen consuming oysters from Ferry Creek or the Housatonic River in the future. Human consumption of oysters is not evaluated quantitatively in this risk evaluation. However, a qualitative comparison of predicted future concentrations of contaminants in oysters to current oyster status as described in the OU3 Area II RI evaluation of risks from human consumption of biota is included. Predicted future oyster tissue concentrations were modeled from predicted future surface water concentrations. This multiple step modeling is highly uncertain, therefore the qualitative discussion is included in the uncertainty section (Section 6.7.4). Risks from exposures to

current concentrations of contaminants in oysters have been evaluated in the OU3 Area II RI and are summarized in Section 6.5.2 of this report.

6.1 Introduction - Overview of Risk Assessment Process

A risk assessment provides the framework for developing risk information necessary to assist in determining the need for remediation and developing potential remedial alternatives for a site. A baseline HHRA consists of five major components, as follows:

- Data evaluation and identification of chemicals of potential concern (COPCs);
- Exposure assessment;
- Toxicity assessment;
- Risk characterization; and
- Characterization of uncertainty in the risk estimates.

To assess potential public health risks, four major aspects of chemical contamination and exposure must be considered: contaminants with toxic characteristics must be found in environmental media; the contaminants must be released by either natural processes or by human action; potential exposure points must exist; and human receptors must be present at the point of exposure. Risk is a function of both toxicity and exposure. If any one of the requirements listed above is absent for a specific site, the exposure route is regarded as incomplete and no potential risks will be considered for human receptors.

The data evaluation component of the HHRA is primarily concerned with selecting COPCs that are representative of the type and magnitude of potential human health effects. Both current and historical data are considered in developing a list of COPCs. In turn, these COPCs are used to evaluate potential risks. A discussion of the process and site-specific issues is contained in Section 6.2, as is a discussion of the models used to develop exposure point concentrations.

The exposure assessment identifies potential human exposure pathways at the study area under consideration. Exposure routes are identified based on information on study area chemical concentrations, chemical release mechanisms, human activity patterns, and other pertinent information to develop a conceptual site model. A discussion of the exposure

assessment is contained in Section 6.3. Section 6.3.1 presents the conceptual site model. Section 6.3.2 presents the potential routes of exposure. Section 6.3.3 presents potential human receptors and the relevant exposure assumptions. Section 6.3.4 presents exposure pathways and the equations for estimating chemical intake.

The toxicity assessment presents the available human health criteria for all the selected COPCs. This assessment is contained in Section 6.4. Quantitative toxicity indices are presented where they are available. A discussion of health effects and dose-response parameters such as Reference Doses (RfDs) and Cancer Slope Factors (CSFs) is presented for each COPC.

The risk characterization (Section 6.5) describes how the estimated intakes are combined with the toxicity information to estimate risks. Section 6.6 presents an alternate evaluation of indoor air exposures based on indoor air concentrations modeled from measured shallow groundwater concentrations. Supporting tables for this evaluation are presented in Appendix F-2. Uncertainties associated with the risk assessment process are discussed qualitatively in Section 6.7.

6.2 Data Evaluation

Data evaluation is a study area-specific task that uses a variety of information to determine which of the detected chemicals at a study area are most likely to present a risk to potential receptors. The end result of this qualitative selection process is a list of COPCs and representative exposure point concentrations for each medium. Exposure Point Concentrations (EPCs) are defined as the contaminant concentrations at the point of exposure. The methodology used to identify COPCs for the OU2 RI Report is provided in Section 6.2.1. The rationale for the selection and/or exclusion of each detected chemical for the OU2 groundwater study area is presented in Section 6.2.2. The methodologies used to determine EPCs for the selected COPCs are presented in Section 6.2.3.

6.2.1 Selection of Chemicals of Potential Concern

COPCs for the baseline human health risk assessments were limited to those chemicals that exceed a selection criterion. For this risk assessment, state risk-based and health-based

criteria and EPA Region IX risk-based criteria were used to reduce the number of chemicals considered in a risk assessment. These risk-based criteria are chemical concentrations based on a fixed level of risk from medium-specific exposures. The premise of this screening step is that risk is typically dominated by a few chemicals and that, although dozens may actually be detected, many chemicals may contribute minimally to the total risk.

Maximum detected concentrations (in a single sample at any depth) in the indoor air and groundwater of the OU2 study area were compared to the risk-based and health-based screening criteria. If the maximum concentration exceeded the screening criteria, that chemical was retained as a COPC for the appropriate exposure route. For example, benzene was retained based on a comparison of study area data to ambient air criteria and was therefore evaluated as a COPC for the inhalation of indoor air.

All available Contract Laboratory Program (CLP) data were used to identify indoor air COPCs for the study area, except indoor air data collected from homes after installation of sub-slab ventilation systems. For indoor air exposure COPCs, measured CLP indoor air data from all rounds of indoor air sampling within the indoor air area of interest were used in the COPC selection process. Indoor air field-screening data were not used in the risk assessment.

All available laboratory data were used to identify groundwater COPCs for the study area. For surface water exposures, all groundwater laboratory data from the 2002/2003 sampling efforts within the study area were used in the COPC selection process. For modeled indoor air exposures, only shallow groundwater laboratory data from within the indoor air area of interest were used in the COPC selection process. Both historical data and more recent data were included in this dataset. The COPC selection tables are discussed in Section 6.2.2. Analytical results qualified as rejected (R) during the data validation process were not considered because of their potential unreliability. Groundwater field-screening data were not used in the risk assessment.

Three groups of COPCs were identified and presented separately in this baseline HHRA, as follows: indoor air COPCs based on ambient air criteria; groundwater COPCs based on surface water protection criteria; and groundwater COPCs based on volatilization criteria. Only chemicals selected as COPCs based on comparisons to selection criteria were evaluated quantitatively in the baseline risk assessment. Contaminants lacking selection criteria were not

selected as COPCs for the pathway of concern. Indoor air COPCs based on ambient air criteria were used to estimate risks from indoor air exposures. Groundwater COPCs based on surface water protection criteria were used to identify groundwater contaminants to be evaluated for their potential migration to surface water. As described in Section 5.2, sufficient information to model migration of these contaminants was available for a limited number of these groundwater contaminants. The groundwater volatilization criteria were used to identify potential indoor air contaminants detected in shallow groundwater. These groundwater COPCs are presented in Appendix F-2 for purposes of discussion and evaluation of modeled indoor air concentrations.

Previously, the groundwater volatilization criteria were used to identify potential indoor air contaminants in groundwater throughout the OU2 study area prior to the initiation of indoor air sampling efforts. These early efforts led to the identification of seven potential indoor air contaminants as discussed in Section 2.11. Subsequent indoor air sampling focused mainly on these contaminants.

Selection criteria were chosen based on applicability to the exposure scenarios identified in Appendix F-1, Table 1. A discussion of the criteria used for COPC selection is provided in the remainder of this section.

Criteria for the Selection of Indoor Air COPCs EPA Region IX Preliminary Remediation Goals (PRGs) for Ambient Air (EPA 2002a) were used to identify COPCs for potential indoor air exposures within the indoor air area of interest of the OU2 study area. (See Figure 4-20 to locate the indoor air area of interest). The EPA Region IX PRGs for Ambient Air were developed for the protection of human health associated with indoor air exposures. These criteria are based on target cancer risks of 1E-06 and hazard indices of 1.0 under residential indoor air exposure scenarios. In developing the PRGs for Ambient Air, EPA developed target indoor air concentrations assuming residents are exposed to indoor air concentrations of contaminants 24 hours/day, 350 days/year, for 30 years. The EPA Region IX PRGs for Ambient Air used to identify COPCs are presented in Appendix F-1, Table 2.1. Indoor air COPCs are those chemicals detected at maximum concentrations in excess of the EPA Region IX PRGs for Ambient Air. The “indoor air exposures” category is used to evaluate residents who may be potentially exposed through inhalation of indoor air in their homes and industrial/commercial workers who may be potentially exposed through inhalation of indoor air at commercial properties.

Criteria for the Selection of Surface Water Protection COPCs Connecticut Surface Water Protection Criteria (SWPC) (CTDEP 1996) were used to identify COPCs for potential future surface water exposures to contaminants present in groundwater up-gradient of Ferry Creek. The SWPC are groundwater concentrations developed for the protection of human health and ecological concerns associated with potential surface water contamination. The SWPC were developed by application of a dilution factor to Connecticut's water quality criteria. The water quality criteria are surface water concentrations developed for the protection of human health and aquatic life. The human health exposures considered are those associated with use of surface water as a drinking water source and ingestion of fish. The exposure (human health or aquatic life) resulting in the most restrictive value was selected for the basis of the SWPC. The SWPC used to identify COPCs are presented in Appendix F-1, Table 2.2. Surface water protection COPCs are those chemicals detected at concentrations in excess of the SWPC. The "surface water protection" category is used to evaluate potential future human surface water exposures resulting from migration of contaminants presently located in groundwater.

Criteria for the Selection of Groundwater Volatilization COPCs Connecticut Groundwater Volatilization Criteria for residential exposures (CTDEP 1996) were used to identify COPCs for potential indoor air exposures to contaminants present in groundwater within the indoor air area of interest of the OU2 study area. (See Figure 4-20 to locate the indoor air area of interest). In March 2003, CTDEP released "Proposed Revisions" to Connecticut's Remediation Standard Regulations Volatilization Criteria (CTDEP 2003). These proposed revisions have not yet been finalized; however, the proposed Groundwater Volatilization Criteria for TCE and 1,1-DCE were used to identify COPCs. The Connecticut Groundwater Volatilization Criteria were developed for the protection of human health associated with indoor air exposures. These criteria are based on target cancer risks of 1E-06 and hazard indices of 1.0 under residential indoor air exposure scenarios. In developing the Groundwater Volatilization Criteria, CTDEP developed target indoor air concentrations assuming residents are exposed to indoor air concentrations of contaminants 24 hours/day, 350 days/year, for 30 years. The concentrations of groundwater associated with the target indoor air concentrations were calculated using the transport model and default parameters presented in the Risk Based Corrective Action Process (RBCA 1994). The Connecticut Groundwater Volatilization Criteria used to identify COPCs are presented in Appendix F-2, Table 2. Groundwater volatilization COPCs are those chemicals detected at maximum concentrations in excess of the Connecticut Groundwater Volatilization Criteria. The "modeled indoor air exposures" category is used to evaluate groundwater contamination to

which residents and industrial/commercial workers may be potentially exposed through volatilization of contaminants into indoor air in their homes or businesses and for comparison to measured indoor air sample evaluations.

6.2.2 Identification of COPCs

Appendix F-1, Tables 2.1 and 2.2 present a summary of the COPCs for quantitative risk assessment for indoor air exposures and future surface water exposures, respectively. Appendix F-2, Table 2 presents a summary of the groundwater COPCs for an alternate quantitative risk assessment for indoor air exposures. COPCs were identified based on a comparison of indoor air and groundwater data to the COPC screening criteria defined in Section 6.2.1. For indoor air exposure COPCs, measured CLP indoor air data from all rounds of indoor air sampling within the indoor air area of interest, except data collected from homes after installation of ventilation systems, were used in the COPC selection process. For surface water exposures, all groundwater laboratory data from the 2002/2003 sampling efforts within the study area were used in the COPC selection process. For groundwater COPCs for indoor air exposures, only shallow groundwater laboratory data from within the indoor air area of investigation were used in the COPC selection process. Both historical data and more recent data were included in this dataset. A discussion of COPCs is presented in the following paragraphs.

Indoor Air Exposure COPCs The COPC selection process for indoor air exposure is summarized on Appendix F-1, Table 2.1. The following chemicals were identified as indoor air exposure COPCs based on a comparison of maximum indoor air concentrations within the indoor air area of investigation to EPA Region IX PRGs for Ambient Air.

- VOCs (1,1,2,2-tetrachloroethane, 1,1,2-trichloroethane, 1,1-dichloroethene, 1,2,4-trimethylbenzene, 1,2-dibromoethane, 1,2-dichloroethane, 1,3,5-trimethylbenzene, 1,3-butadiene, 1,4-dichlorobenzene, 4-methyl-2-pentanone, acetone, acrylonitrile, benzene, benzyl chloride, bromodichloromethane, carbon tetrachloride, chloroethane, chloroform, chloromethane, cis-1,2-dichloroethene, dibromochloromethane, dibromofluoromethane, ethylbenzene, hexane, methyl tert-butyl ether, methylene chloride, tetrachloroethene, tetrahydrofuran, toluene, xylenes, trichloroethene, vinyl acetate, and vinyl chloride)

Surface Water Exposure COPCs The COPC selection process for surface water exposure to groundwater contaminants is summarized on Appendix F-1, Table 2.2. The following chemicals were identified as surface water exposure COPCs based on a comparison of maximum groundwater concentrations to the SWPC.

- VOCs (1,1-dichloroethene, 1,1,1-trichloroethane, benzene, tetrachloroethene, and trichloroethene)
- PAHs (acenaphthylene, benzo(b)fluoranthene, benzo(k)fluoranthene, and phenanthrene)
- Metals (arsenic, beryllium, cadmium, lead, mercury, nickel, selenium, silver, thallium, and zinc)
- Pesticides (gamma-chlordane, dieldrin, heptachlor, and heptachlor epoxide)

Groundwater COPCs for Indoor Air Exposure The COPC selection process for indoor air exposure to groundwater contaminants is summarized on Appendix F-2, Table 2. The following chemicals were identified as groundwater COPCs for indoor air exposure based on a comparison of maximum shallow groundwater concentrations within the indoor air area of interest of the study area to Connecticut groundwater volatilization criteria for residential exposures.

- 1,1-dichloroethene
- Trichloroethene
- Vinyl chloride

The maximum shallow groundwater concentrations within the indoor air area of interest of these same three contaminants also exceed Connecticut groundwater volatilization criteria for industrial/commercial exposures.

6.2.3 Exposure Point Concentrations

According to the regional guidance, risk assessments are conducted using an exposure point concentration for each COPC. The exposure point concentration (EPC) is generally defined as the 95 percent upper confidence limit (UCL) on the mean (EPA 1992a, 1992b, and 1994). However, because of the high degree of variability between homes and the absence of indoor

air data from every home and commercial property within the area of interest, maximum indoor air concentrations were used for reasonable maximum exposure EPCs, and average indoor air concentrations were used for central tendency exposure EPCs. A value of one-half the detection limit was substituted for nondetected values in the calculation of average indoor air concentrations.

For modeling future surface water concentrations, measured groundwater concentrations at a presumed source were used. Since surface water exposures are based on modeled surface water concentrations, the modeled concentrations were used as the EPCs. Modeled future surface water concentrations were also used to estimate future contaminant concentrations in oyster tissue for qualitative discussion.

For purposes of modeling indoor air concentrations, maximum shallow groundwater concentrations within the indoor air area of interest were used to estimate worst-case indoor air concentrations. These concentrations were then used as the indoor air EPCs. The shallow groundwater dataset included the shallowest samples at each permanent groundwater monitoring well location. The average depth below ground surface (bgs) to the top of the water table within the indoor air area of interest is 11.85 feet bgs. Shallow groundwater samples were collected at depths of 5.8 to 24.0 feet bgs. A list of groundwater samples used to evaluate indoor air exposures to residents and industrial/commercial workers is presented in Appendix F-2, Table 1. The maximum detected concentration reported for field duplicate pair samples was used in the calculation of maximum groundwater concentrations.

Exposure point concentrations used in the risk assessment are presented in Appendix F-1, Tables 3.1 and 3.2, and Appendix F-2, Table 3. A discussion of the models used to estimate future surface water, oyster tissue, and indoor air concentrations is provided in the remainder of this section.

Surface Water Concentration Model The migration of groundwater contaminants in the OU2 study area to local surface water bodies was modeled. A discussion of the modeling of the migration of several VOCs and inorganics to generate reasonable worst case concentrations for those contaminants predicted to enter Ferry Creek or the Housatonic River at a single point in the river bed is presented in detail in Section 5. Contaminants to be modeled were selected based on three criteria: 1) does the solute have the potential to adversely effect human health

or the aquatic biological community if it enters one of these surface water bodies; 2) could a source of the contaminant be identified; and 3) does the scientific literature contain sufficient information on the behavior of the contaminant to model its migration. The modeled sources and pathways are shown in Figure 5.1, and each source and pathway delineated on Figure 5.1 is discussed in Section 5.1.3.

Several groundwater COPCs selected in Appendix F-1, Table 2.2 on the basis of concentrations greater than SWPC were not modeled because of infrequent detections or exceedances of selection criteria, because of their absence in shallow groundwater, because a source could not be identified, because they do not appear to be derived from Raymark waste, or because the scientific literature does not contain sufficient information on the behavior of the contaminant to model its migration. Acenaphthylene, benzo(b)fluoranthene, benzo(k)fluoranthene, phenanthrene, gamma-chlordane, dieldrin, heptachlor, and heptachlor epoxide were detected infrequently and have low mobility. Mercury, selenium, silver, and thallium were detected infrequently above the selection criteria and most of the detected concentrations exceeding the selection criterion were just slightly greater than the selection criteria. 1,1,1-Trichloroethane and beryllium were only detected in high concentrations in deep groundwater. Tetrachloroethene exceeds the selection criterion only at the source. Benzene does not appear to be derived from Raymark waste. Refer to Section 4.3.1 for additional discussions about possible benzene sources.

Among the COPCs modeled, cadmium, lead, and nickel were not evaluated in the HHRA because they are not expected to arrive at either Ferry Creek or the Housatonic River within the foreseeable future.

The model indicates that trichloroethene is migrating toward the Housatonic River, rather than toward Ferry Creek. As indicated on Table 5-2, it is expected that maximum predicted concentrations of trichloroethene have already reached the river. Further sampling data in the Housatonic River are needed to verify this prediction.

Surface water concentrations in Ferry Creek of the remaining COPCs for surface water protection, 1,1-dichloroethene, arsenic, and zinc, were estimated by applying a dilution factor to modeled seep concentrations. The dilution factor was used to account for mixing of groundwater seep with creek waters. A dilution factor of 0.1 was selected, based on a review of

Ferry Creek surface water concentrations of VOCs and groundwater concentrations. The resulting modeled surface water concentrations are presented in Appendix F-1, Table 3.2 along with the detected groundwater source concentrations on which they are based.

Oyster Tissue Concentration Model Oyster tissue concentrations were estimated from biological concentration factors (BCFs) and worst-case surface water concentrations estimated for the Housatonic River from the groundwater modeling presented in Section 5.0. The surface water concentrations in the river were estimated by applying an additional dilution factor of 10 to the Ferry Creek estimate (Appendix F-3). This estimate is likely to be very conservative, at least for the volatile organic compounds, given that five- to ten-fold attenuation appears to be occurring between the upper and lower parts of Ferry Creek. There are no data available for Ferry Creek to make a more accurate estimate.

The BCFs used to estimate oyster tissue concentrations for arsenic and zinc (Appendix F-3, Table 1) were derived from data in EPA's ECOTOX database (EPA, 2004b). Only BCFs for marine invertebrates were considered. The zinc BCFs were based on marine bivalves. Polychaete worms were the only marine organisms for which arsenic BCFs were available, so arsenic BCFs were based on these organisms. The BCFs for each metal were downloaded from the database and listed together with their associated surface water concentrations. Using the water concentration estimates for both Ferry Creek and the Housatonic River, the BCFs were placed into two groups according to their water concentrations. From each group the median value was selected as the BCF to be used in the calculations.

The results of this multiple-step modeling effort are very uncertain worst-case estimates of future oyster tissue concentrations. A discussion of predicted oyster tissue concentrations in comparison to measured oyster tissue concentrations reported in the OU3 RI is presented in Section 6.7.4.

Indoor Air Concentration Model The Johnson and Ettinger (1991) Model for Subsurface Vapor Intrusion into Buildings is a one-dimensional analytical solution for estimating the transport of contaminant vapors from either subsurface soils or groundwater into an indoor space (a building) located directly above or close to the source of contamination. The model assumes the volatilized contaminant moves by diffusion through the subsurface until it reaches the zone of influence of the building where convection moves the vapors through the cracks

between the foundation and the floor. Results can be calculated with an infinite source (steady state) or a finite source (quasi-steady-state).

The EPA has developed a user's guide and a series of spreadsheets that allow for site-specific application of the Johnson and Ettinger model (EPA 1997d). Inputs to the model include chemical properties of the contaminant, saturated and unsaturated zone soil properties, and structural properties of the building. The model will:

- reverse-calculate a site-specific target concentration for soil or groundwater and an associated indoor air concentration given a target risk or target hazard quotient; or

- forward-calculate an incremental cancer risk or hazard quotient and an associated indoor air concentration given an initial soil or groundwater concentration.

The model may be run as a first-tier screening tool or a more involved second-tier estimate. The first-tier screening models for soil contamination and groundwater contamination should be run as steady state. In the screening models, only the most sensitive model parameters may be user-specific. Examples are chemical abstract service (CAS) number of the contaminant, initial soil or groundwater concentration, depth to enclosed space floor, depth to water (groundwater model only), average soil/groundwater temperature, soil type above water (groundwater model only), and for the vadose zone soil – soil type or permeability, dry bulk density, total porosity, water-filled porosity, and (for soil model only) fraction of organic carbon. Other parameters for the screening models input by the user are averaging times, exposure duration and frequency, and target risk or target hazard quotient if reverse-calculating a site-specific target concentration. Second-tier estimates may be obtained using additional site-specific values for up to three soil strata and site-specific values for the structural properties of the building.

Because the source of groundwater contamination may be located up-gradient of the enclosed structure for which the indoor inhalation pathway is to be assessed, the second-tier model for contaminated groundwater is based on an infinite source of contamination; however, site-specific values for all other model parameters may be user defined.

The Johnson and Ettinger (1991) Model for Subsurface Vapor Intrusion into Buildings was used in this risk assessment for estimating the transport of contaminant vapors from groundwater into

indoor spaces (homes and commercial properties) located directly above or close to the source of contamination. As mentioned previously, the evaluation was limited to the indoor air area of interest, which represents the area of greatest concern. Since this area represents a relatively large area with many homes and a few non-residential properties, no single home or commercial property was evaluated. Rather, the model was used to estimate risks for a hypothetical home or commercial property within the indoor air area of interest based on maximum shallow groundwater concentrations. Because risks were not determined for any particular building, specific building structural properties were not available to enter into the model. Soils in this area were characterized as sands with little stratification. For these reasons, the first-tier screening model for groundwater contamination was selected.

Soil parameters selected for use in the screening model were chosen based on a combination of measured values and values consistent with the soil type. Groundwater temperatures were measured during groundwater sampling efforts. Average groundwater temperatures were 14.5 degrees Celsius. Depths to the water table were also measured during groundwater sampling. Depths to the water table at the locations of maximum concentrations were entered into the model. Soil water-filled porosity was set at 0.1 cm^3/cm^3, soil total porosity was set at 0.43 cm^3/cm^3, and dry bulk density was set at 1.5 g/cm^3, based on the presence of sand as the soil type. It was assumed that the hypothetical home or commercial property has a basement extending 200 cm below ground surface.

Residential and industrial/commercial exposure parameters, such as exposure frequency, exposure duration, etc., are discussed in Section 6.3. Appendix F-2 presents the spreadsheets for the Johnson & Ettinger indoor air model.

Indoor air concentrations predicted from the model are presented in Appendix F-2, Table 3. Appendix F-2, Table 3 presents average and maximum detected groundwater concentrations and the indoor air concentrations developed from the maximum groundwater concentrations. Actual indoor air concentrations have been measured in homes and a few non-residential properties within the indoor air area of interest. A discussion of the results of the actual indoor air sampling in comparison to the indoor air concentrations predicted by the Johnson & Ettinger model based on groundwater sampling is included in Section 6.6.

6.3 Exposure Assessment

The exposure assessment defines and evaluates the exposures experienced by a receptor population. To have an exposure, several factors must be present: there must be a source of contamination, there must be a mechanism through which a receptor can come into contact with the contaminants in that medium, and there must actually (or potentially) be a receptor present at the point of contact.

The exposure assessment presented consists of several sections that characterize the physical site setting and the receptors of concern, identifies the potential contaminant migration and exposure pathways, and presents the equations used to quantify exposure in terms of contaminant intake (dose). Appendix F-4 of this report contains sample calculations for the exposure assessment. Exposure assumptions are presented in Appendix F-1, Tables 4.1 and 4.2. Intakes are presented in Appendix F-1, Tables 7.1, 7.2, 8.1, and 8.2.

6.3.1 Conceptual Site Model

This section discusses the general conceptual site model for the OU2 study area. A conceptual site model facilitates a consistent and comprehensive evaluation of the risks to human and ecological health by creating a framework for identifying the paths by which human health may be impacted by contaminants predicted to exist at the source areas. A conceptual site model depicts the relationships between the following elements necessary to construct a complete exposure pathway:

- Sources of contamination and potential COPCs;
- Contaminant release mechanisms and transport pathways:
- Exposure mechanisms and exposure routes; and
- Receptors.

The conceptual site model was developed to provide the basis for identifying the potential risks to human health and the environment. The model considers the current and future conditions within the study area and the actual or potential receptors who could come into contact with the COPCs.

The conceptual site model first considers the contaminant sources assumed to be available, either currently or in the future. For this model, Raymark Facility waste migrating from the former Raymark Facility or disposed within the study area, or the contaminated soils and groundwater within the study area are considered the sources. Contaminants may be released from these sources by mechanisms such as wind, water erosion, leaching to the subsurface, or excavation within areas of contamination. Once released from the source, contaminants are transported in media such as air, surface water, soils, or groundwater. Contaminants may also be released from groundwater through volatilization into indoor air spaces or by mixing with surface water bodies. Receptors may be exposed either directly or indirectly to contaminants in environmental media through a variety of mechanisms. The exposure mechanisms considered include residential activities, recreational activities, etc. These exposure mechanisms generally act along one or more exposure routes such as ingestion, inhalation, or direct dermal contact.

The conceptual site model also indicates those exposure routes that are carried through the quantitative risk assessment for each receptor. An objective of the development of the conceptual site model was to focus attention on those pathways that contribute the most to the potential impacts on human health and the environment and to provide the rationale for screening out other exposure pathways that are minor components of the overall risk.

Aspects of the conceptual site model are described below:

Sources of Contamination and Potential COPCs. As a result of the actions at Raymark Industries, Inc. described in Section 1.3.1, groundwater at the former Raymark Facility has been contaminated primarily with VOCs and metals.

Contaminant Release and Migration Mechanisms. Chemicals may be released from the study area by a variety of mechanisms. These mechanisms include stormwater runoff and subsequent surface soil erosion, soluble chemical infiltration and subsequent migration through the subsurface soil to the water table where the chemicals may migrate downgradient, wind erosion of surface soil from unpaved areas, disturbance of contaminants in soil through human excavation or animal burrowing activities, and through cracks in asphalt pavement, if present. Contaminant fate and transport are discussed in Section 5.

Soluble chemicals released to the ground surface may also migrate downward through the soil column with infiltrating precipitation. The migration of these chemicals may be somewhat impeded by the chemical's tendency to bind to soil organic material. Eventually, these soluble chemicals may reach the water table. Once in the groundwater, chemicals may continue to migrate downgradient via dispersion and advection. Eventually, these chemicals may discharge with the groundwater to surface bodies such as Ferry Creek. Once in the surface water, contaminants may enter the food chain of aquatic biota. Volatile contaminants may be released from groundwater and move upwards into indoor air.

Exposure Mechanisms and Exposure Routes. The potential for groundwater exposure is based on factors such as current and future land uses, human activity patterns, site access controls, and chemical behavior in the environment. Based on these variables, exposure scenarios were developed to characterize the potential for human exposure under current and future site conditions. The future scenario accounts for possible or anticipated changes in land use and site characteristics that may alter exposure and/or concentrations of COPCs in a given medium.

The exposure assessment is based on the assumptions that, in general, chemical concentrations in shallow groundwater are able to migrate into indoor air and/or surface water in Ferry Creek.

The exposure routes through which receptors may be exposed are inhalation of indoor air, dermal contact with contaminated surface water, and ingestion of contaminated biota.

A summary of the potentially significant exposures identified for quantitative evaluation is provided in Appendix F-1, Table 1.

The OU2 study area is presented on Figure 1-1 and described in detail in Section 1.3.2. The study area includes commercial/industrial properties, highways, residential properties, Ferry Creek and other ecological areas (the delineated wetland boundaries along Ferry Creek). Property within the study area has been developed for commercial/industrial, residential, or recreational purposes, or is undeveloped (wetlands). The primary land-use within the study area is industrial; however, a portion of the area is residential and areas of Ferry Creek are used for recreational purposes.

The lower reaches of Ferry Creek and the Housatonic River are used for recreational fishing and boating. The mouth of the Housatonic River is considered to be a recreational fishery and there may be the potential for human consumption of biota taken from the river. Coastal waterways are assumed to support various recreational activities, as well as recreational and commercial fishing. Currently recreational shellfishing is prohibited in the Town of Stratford. The lower Housatonic River, near the mouth of Ferry Creek, contains important commercial seed beds for oyster cultivation. Oysters remain in these seed beds for one year or less before being moved to a different location for the remainder of their growing period. Oysters originating from these oyster beds are commercially shipped through the region. EPA representatives have observed people crabbing from the Ferry creek flood control gates located on Broad Street.

Exposed Populations The study area is located in Stratford, Fairfield County, Connecticut. The principal industries within the community of Stratford include manufacturing aircraft, air conditioners, chemicals, plastic, paper, rubber goods, electrical and machine parts, and toys. There were 49,389 people reported to live in the town in 2003. Potentially exposed populations within the Raymark OU2 groundwater study area are discussed in detail in Section 6.3.4.

Receptors Several potential receptor populations were initially considered for inclusion in the exposure assessment. However, the majority of these receptors were eliminated from further evaluation based on the current land use, site access, COPCs, and the likelihood of exposure. Of the receptors initially considered (residents, recreational users, industrial/commercial workers, construction workers, and trespassers), the receptors retained for quantitative evaluation are current/future residents, current/future industrial/commercial workers, and future recreational users. These receptors are included in the baseline risk assessment.

Under current conditions, potential human receptors (residents and industrial/commercial workers) are assumed to be exposed to indoor air. Similar indoor air exposure is likely for receptors under current and future conditions.

Current surface water conditions in Ferry Creek were detailed in the OU3 Ferry Creek Area I RI and showed cancer risk estimates ranging from 5.5E-06 to 1.9E-05 for recreational receptors exposed to surface water while wading. In the future, contaminated groundwater currently located at some distance upgradient of Ferry Creek may migrate to Ferry Creek and increase contaminant concentrations in the surface water. Under future conditions, recreational

receptors were evaluated for exposure to surface water modeled from upgradient groundwater concentrations.

Current surface water conditions in the Housatonic River have not been evaluated. Current and future surface water concentrations in the Housatonic River may be impacted by groundwater contamination. The groundwater transport modeling, described in Section 5, indicates the migration of trichloroethene toward the Housatonic River. As indicated on Table 5-2, it is expected that maximum predicted concentrations of trichloroethene have already reached the river. Current and future recreational receptors may be exposed to surface water in the Housatonic River while swimming or boating. Further sampling is needed in order to estimate current or future surface water concentrations in the Housatonic River and potential human health risks from exposures to surface water in the Housatonic River.

Human consumption of oysters obtained from the Housatonic River near the mouth of Ferry Creek was evaluated in the OU3 Ferry Creek RI and showed cancer risk estimates ranging from 8.9E-07 to 6.2E-06. In the future, contaminated shallow groundwater contributing to increased Ferry Creek surface water concentrations, may increase contaminant concentrations in oysters. Human consumption of oysters is not evaluated quantitatively in this risk evaluation. However, a qualitative comparison of predicted future concentrations of contaminants in oysters to current oyster concentration data is included in Section 6.7.4.

Groundwater in the study area is not used, or expected to be used, in the future as a potable water supply because of brackish conditions and productivity constraints.

Direct contact with groundwater may occur during excavation within the study area. However, such contact represents a minor pathway and therefore is not evaluated in this report. Trespassers are not expected to contact groundwater.

6.3.2 Potential Routes of Exposure

A receptor can come into contact with contaminants in a variety of ways, which are generally the result of interactions between a receptor's behavior or lifestyle and an exposure medium. This HHRA defines an exposure route as a stylized description of the behavior that brings a receptor

into contact with a contaminated medium. The exposure routes considered in this HHRA are discussed below.

Indoor Air This pathway is based on the scenario that a receptor is immersed in air that contains volatile organic vapors originating from the source areas as part of daily living. Subsequent exposure of the receptor occurs upon inhalation of the ambient air.

Direct Contact with Surface Water Receptors may come into direct contact with surface water affected by the release of chemicals from groundwater. During the receptor's period of contact, the individual may be exposed by dermal absorption of certain contaminants from the surface water. This exposure would be of short duration. The “attractiveness” of Ferry Creek was considered when evaluating the potential for exposure to surface water. Therefore, recreational use of Ferry Creek is limited to wading. For this reason, ingestion of surface water is not evaluated.

Direct Contact with Groundwater As discussed previously, direct contact with groundwater in the study area during excavation activities is considered a minor pathway and is not evaluated in this baseline risk assessment. It is possible that an excavation (for construction, utility maintenance, etc.) could be deep enough to come into contact with the shallow groundwater. In such an instance, workers could be exposed to the groundwater through dermal contact.

Potable use of groundwater is also not evaluated. Groundwater in the OU2 study area and surrounding areas is not used as a drinking water source because of brackish conditions and productivity constraints.

Ingestion of Biota As discussed previously, current human consumption of oysters was evaluated in the OU3 Area II RI. Future human consumption of oysters is evaluated qualitatively in this report through a comparison of predicted oyster tissue concentrations and concentrations evaluated in the OU3 RI. Receptors may contact contaminants through ingestion of contaminated oysters.

6.3.3 Potential Receptors

Potential receptors have been identified under both current and future land use conditions. These receptors were identified by analyzing the interaction of current and anticipated future land use practices and the identified sources of contamination. Three receptor groups have been defined for this risk assessment. These receptors are as follows:

- Residents at properties located within the OU2 study area;
- Industrial/commercial workers at properties located within the OU2 study area; and
- Future recreational users - Individuals (adults and children) who may periodically visit (recreate) along Ferry Creek.

Each of these receptor groups is evaluated quantitatively. Appendix F-1, Table 1 presents the receptors and exposure pathways identified for the OU2 study area and provides the rationale for the quantitative evaluation of the selected exposure pathways.

Two bounding estimates of the indoor air exposure scenarios are considered, as presented in EPA Region I guidance. The first is identified as a central tendency exposure (CTE) receptor, which was developed using both regional guidance (EPA 1994) and professional judgment regarding site-specific conditions. The second class of receptor is called the reasonable maximum exposure (RME) and was developed according to EPA guidance (EPA 1989b and 1994). The RME is conceptually the “high end” exposure, above the 90th percentile of the population distribution, but not higher than the individual in the population with the highest exposure. Therefore, the RME scenario represents a “reasonable worst case” exposure scenario.

For the future recreational surface water exposure scenario, only the RME scenario is evaluated. Current risks from recreational surface water exposures at Ferry Creek have been previously evaluated in the OU3 Area I RI and are summarized later in Section 6.5.2. This evaluation seeks to predict worst case future surface water risks as contaminated groundwater migrates downgradient and enters Ferry Creek. Maximum source concentrations have been combined with reasonable groundwater flow and solute transport parameters to obtain reasonable maximum surface water concentrations within the foreseeable future.

Residents Residents were evaluated for exposures to indoor air. Residents are assumed to inhale an average of 20 m^3/day of indoor air for 9 years for the CTE and for 30 years for the RME. The proposed exposure duration values are based on EPA guidance for RME and CTE evaluation. Residents are exposed to indoor air in their homes 350 days per year, which is consistent with EPA and CTDEP default values. The average body weight for residents, aged 0 to 30 years, is 70 kg.

Industrial/Commercial Workers Industrial/commercial workers were evaluated for exposures to indoor air. Industrial/commercial workers are assumed to inhale an average of 20 m^3/24 hours of indoor air for 8 hours/day for 9 years for the CTE and for 25 years for the RME. The proposed exposure duration values are based on EPA guidance for RME and CTE evaluation. industrial/commercial workers are exposed to indoor air in their workplace 250 days per year, which is consistent with EPA and CTDEP default values. The average body weight for industrial/commercial workers is 70 kg.

Recreational Users Recreational users were evaluated for exposures to surface waters while wading. The "attractiveness" of Ferry Creek was considered when evaluating the potential for exposure to surface water. Receptors may come into direct contact with surface water containing chemicals in a suspended or dissolved phase. This exposure would be of short duration and individuals may be exposed through dermal contact. Ingestion of surface water is not considered because this is a non-swimming scenario.

Recreational users of the OU2 study area were assumed to be adults and young children. The proposed exposure duration values are based on EPA guidance for RME evaluation. Values for small children for the RME reflect the entire age span for the receptor evaluated. RME exposure durations for child and adult receptors under the recreational scenario are 6 years and 24 years, respectively. For all receptors, the event frequency was set at one event/day. The exposure times were set at 1 hour/day.

Site-specific considerations were used to determine exposure frequencies for recreational users. Recreational users were assumed to be exposed 90 days/year, corresponding to 3 days/week April through October. Only a receptor's hands, feet, and lower legs were assumed to be exposed for wading scenarios. These body parts represent 25 percent of total body surface area. The skin surface areas for the wading scenario were set at 4,500 cm^2 for adults

and 1,900 cm^2 for small children. These values represent 25 percent of total body surface area for the appropriate ages of receptors as presented in the Exposure Factor Handbook (EPA, 1997c). The average body weight for adult recreational users is 70 kg. The average body weight for child recreational users, aged 0 to 6 years, is 15 kg.

6.3.4 Exposure Pathways

An exposure pathway consists of four elements: a source and mechanism of release, a route of contaminant transport through an environmental medium, a contact point for a human receptor, and an exposure route at the point of contact. All four components must be present for the exposure pathway to be considered complete. This section summarizes the potentially complete exposure pathways that are quantitatively evaluated in the risk assessment and provides the rationale for those pathways that are not evaluated. Appendix F-1, Table 1 presents a summary of the potentially complete and incomplete exposure pathways and receptors.

The primary routes of exposure for potential human receptors in the OU2 study area are inhalation of indoor air and dermal contact with surface water. Other potential exposure routes such as use of groundwater as a drinking water source and direct contact with groundwater during excavation work, were not considered for the following reasons:

- The shallow aquifer within the study area is not used as a potable water supply. Shallow groundwater at the site discharges to Ferry Creek and its tributaries. Thus, domestic groundwater exposures by residents are eliminated. In addition, as previously mentioned, groundwater at the site is not used or expected to be used in the future as a potable water supply because of brackish conditions and productivity constraints.

- Direct contact with groundwater during excavation work is expected to represent a minor exposure pathway. Therefore, this pathway has not been evaluated in this report.

- Ingestion of surface water is unlikely during wading. This pathway is typically evaluated for recreational users only in swimming scenarios.

Quantification of Exposure

Estimates of exposure are based on the contaminant concentrations at the exposure points and on scenario-specific assumptions and intake parameters. The models and equations used to quantify intakes are described below and have been obtained from a variety of EPA guidance documents, which are cited in the specific intake estimation discussions that follow.

Exposures depend on the predicted concentrations of chemicals in environmental media and local land use practices, and both are subject to change over time. This results in a large number of possible combinations of receptors, media, exposure pathways, and concentrations. As mentioned previously, Appendix F-1, Table 1 presents a summary of the exposure pathways evaluated in the quantitative risk assessment. Some of these scenarios (such as residential, industrial/commercial, and recreational scenarios) may be applicable under both current and future land use conditions. Recreational scenarios are applicable under both current and future land-use conditions; however, current surface water conditions in Ferry Creek have been evaluated under the OU3 RIs using measured surface water concentrations.

Exposure model parameters are presented in Appendix F-1, Tables 4.1A, 4.1B, 4.2A, and 4.2B. The values reflect current EPA guidance and comments received from EPA Region I. All parameters are referenced in footnotes on each table. These parameters are used in the equations presented in this section, along with the exposure point concentrations presented in Appendix F-1, Tables 3.1 and 3.2, to calculate intakes, which are used to determine risks. Individual chemical intakes for each receptor/exposure route combination are presented in Appendix F-1, Tables 7.1A, 7.1B, 7.2A, 7.2B, 8.1A, 8.1B, 8.2A, and 8.2B. The equations used to quantify intakes are presented below.

Inhalation of Indoor Air

The estimation of intake of contaminants in indoor air is determined using measured indoor air concentrations of contaminants in the indoor air area of interest within the OU2 study area. This pathway is evaluated for current/future residents and current/future industrial/commercial workers. In general, intakes associated with indoor air inhalation are calculated using the following equation:

$$Intake_{ai} = \frac{(C_{ai})(IR)(EF)(ED)(CF)}{(BW)(AT)}$$

where: $Intake_{ai}$ = intake of contaminant "i" from air ($^{mg/Kg}$/day)

C_{ai} = concentration of contaminant "i" in air (µg/m^3)

IR = inhalation rate (m^3/day)

EF = exposure frequency (days/yr)

ED = exposure duration (yr)

CF = conversion factor (10^{-3} mg/ug)

BW = body weight (kg)

AT = averaging time (days);
for non-carcinogens, AT=ED*365 days/yr;
for carcinogens, AT=70 yr*365 days/yr

Appendix F-1, Table 4.1A contains a summary of the input parameters for inhalation of indoor air by residents. Appendix F-1, Table 4.1B contains a summary of the input parameters for inhalation of indoor air by industrial/commercial workers.

Dermal Contact with Surface Water

Recreational users of Ferry Creek were evaluated for dermal exposure to surface water while wading. The following equation was used to estimate exposures resulting from dermal contact with water (EPA 2001b):

$$DAD_{wi} = \frac{(DA_{event})(EV)(ED)(EF)(SA)}{(BW)(AT)}$$

where: DAD_{wi} = dermally absorbed dose of chemical "i" from water (mg/kg/day)

DA_{event} = absorbed dose per event (mg/cm^2-event)

EV = event frequency (events/day)

ED = exposure duration (yr)

EF = exposure frequency (days/yr)

SA = skin surface area available for contact (cm^2)

BW = body weight (kg)

AT = averaging time (days);
for non-carcinogens, AT=ED*365 days/yr;
for carcinogens, AT=70yr*365 days/yr

The absorbed dose per event (DA_{event}) is estimated using a nonsteady-state approach for organic compounds and a more traditional steady-state approach for inorganics. For organics, the following equations apply:

$$If\ t_{event} > t^{*},\ then:\ DA_{event} = (FA)(K_p)(C_{wi})(CF)\left(\frac{t_{event}}{1+B} + 2\,\tau_{event}\left(\frac{1+3B+3B^2}{(1+B)^2}\right)\right)$$

$$If\ t_{event} \le t^{*},\ then:\ DA_{event} = 2\,(FA)(K_p)(C_{wi})(CF)\left(\frac{\sqrt{6\,\tau_{event}\,t_{event}}}{\pi}\right)$$

where: FA = fraction absorbed (dimensionlesss)
t_{event} = duration of event (hr/event)
t^* = time it takes to reach steady-state conditions (hr)
K_p = permeability coefficient from water through skin (cm/hr)
C_{wi} = concentration of chemical "i" in water (mg/L)
τ_{event} = lag time per event (hr/event)
π = constant (unitless; equal to 3.141592654)
CF = conversion factor (10^{-3} L/cm^3)
B = partitioning constant derived by Bunge Model (dimensionless)

Values for the chemical-specific parameters (t_{event}, t^*, K_p, τ_{event}, and B) were obtained from the EPA's "Risk Assessment Guidance for Superfund, Volume I: Human Health Evaluation Manual, (Part E, Supplemental Guidance for Dermal Risk Assessment)" (EPA 2001b). If no published values were available for a particular organic compound, they were calculated using equations provided in the cited guidance. Details regarding the procedures used to derive the constants, as well as sample calculations, are provided in Appendix F-4.

The following nonsteady-state equation is used to estimate DA_{event} for inorganics:

$$DA_{event} = (K_p)(C_{wi})(t_{event})_{(CF)}$$

In general, the recommended default K_p value of 0.001 was used for inorganic constituents (EPA, 2001b).

Input parameters for dermal contact with surface water are summarized in Appendix F-1, Tables 4.2A and 4.2B.

6.4 Toxicity Assessment

The toxicity assessment for the COPCs examines information concerning the potential human health effects of exposure to COPCs. The goal of the toxicity assessment is to provide, for each COPC, a quantitative estimate of the relationship between the magnitude and type of exposure and the severity or probability of human health effects. The toxicity values presented in this section are integrated with the exposure assessment (Section 6.3) to characterize the potential for the occurrence of adverse health effects (Section 6.5).

The toxicological evaluation involves a critical review and interpretation of toxicity data from epidemiological, clinical, animal, and in vitro studies. This review of the data ideally determines both the nature of the health effects associated with a particular chemical and the probability that a given quantity of a chemical could result in the referenced effect. This analysis defines the relationship between the dose received and the incidence of an adverse effect for the chemicals of potential concern.

The entire toxicological database is used to guide the derivation of cancer slope factors (CSFs) for carcinogenic effects and reference doses (RfDs) for non-carcinogenic effects. These data may include epidemiological studies, long-term animal bioassays, short-term tests, and evaluations of molecular structure. Data from these sources are reviewed to determine if a chemical is likely to be toxic to humans. Because of the lack of available human studies, the majority of toxicity data used to derive CSFs and RfDs comes from animal studies.

For non-carcinogenic effects, the most appropriate animal model (the species most biologically similar to the human) is identified. Pharmacokinetic data often enter into this determination. In the absence of sufficient data to identify the most appropriate animal model, the most sensitive species is chosen. The RfD is generally derived from the most comprehensive toxicological study that characterizes the dose-response relationship for the critical effect of the chemical. Preference is given to studies using the exposure route of concern; in the absence of such data, an RfD for one route of exposure may be extrapolated from data from a study that evaluated a different route of exposure. Such extrapolation must take into account pharmacokinetic and toxicological differences between the routes of exposure. Uncertainty factors are applied to the highest no-observed-adverse-effect-level (NOAEL) to adjust for inter- and intraspecies variation, deficiencies in the toxicological database, and use of subchronic rather than chronic animal studies. Additional uncertainty factors may be applied to estimate a NOAEL from a lowest-observed-adverse-effect-level (LOAEL) if the key study failed to determine a NOAEL. When chemical-specific data are not sufficient, an RfD may be derived from data for a chemical with structural and toxicologic similarity.

CSFs for weight-of-evidence Group A or B chemicals are generally derived from positive cancer studies that adequately identify the target organ in the test animal data and characterize the dose-response relationship. CSFs are derived for Group C compounds for which the data are sufficient, but are not derived for Group D or E chemicals. (An explanation/definition of these weight-of-evidence classes is provided in Section 6.4.2). No consideration is given to similarity in the animal and human target organ(s), because a chemical capable of inducing cancer in any animal tissue is considered potentially carcinogenic to humans. Preference is given to studies using the route of exposure of concern, in which normal physiologic function was not impaired, and in which exposure occurred during most of the animal's lifetime. Exposure and pharmacokinetic considerations are used to estimate equivalent human doses for computation of the CSF. When a number of studies of similar quality are available, the data may be combined in the derivation of the CSF.

Toxicological profiles for each of the major COPCs are presented in Appendix F-5. These profiles present a summary of the available literature on carcinogenic and non-carcinogenic effects associated with human exposure to the chemical. For more in depth information see www.epa.gov/iris or www.atsdr.cdc.gov. Brief summaries of the toxicity profiles for the major COPCs are presented in Section 6.4.3

6.4.1 Non-carcinogenic Effects

For non-carcinogens, it is assumed that there exists a dose below which no adverse health effects will be seen. Below this "threshold" dose, exposure to a chemical can be tolerated without adverse effects. For non-carcinogens, a range of exposure exists that can be tolerated. Toxic effects are manifested only when physiologic protective mechanisms are overcome by exposures to a chemical above its threshold level. Maternal and developmental endpoints are considered systemic toxicity.

The potential for non-carcinogenic health effects resulting from exposure to chemicals is assessed by comparing an exposure estimate (intake or dose) to an RfD. The RfD is expressed in units of mg/kg/day and represents a daily intake of contaminant per kilogram of body weight that is not sufficient to cause the threshold effect of concern. An RfD is specific to the chemical, the route of exposure, and the duration over which the exposure occurs. Separate RfDs are presented for ingestion and inhalation pathways. In particular, reference concentrations (RfCs) in units of mg/m^3 are typically presented for the inhalation pathway. Because characterization of non-carcinogenic effects requires an estimate of dose in units of mg/kg/day, the inhalation RfC must be converted to an inhalation RfD. The conversion is performed by assuming that humans weigh 70 kg and inhale 20 m^3 of air per day [i.e., the inhalation RfC (mg/m^3) is multiplied by 20 m^3/day and divided by 70 kg to yield an inhalation RfD (mg/kg/day)]. EPA does not support the route-to-route extrapolation of inhalation RfDs from oral RfDs.

To derive an RfD, EPA reviews all relevant human and animal studies for each compound and selects the study (studies) pertinent to the derivation of the specific RfD. Each study is evaluated to determine the NOAEL or, if the data are inadequate for such a determination, the LOAEL. The NOAEL corresponds to the dose (in mg/kg/day) that can be administered over a lifetime without inducing observable adverse effects. The LOAEL corresponds to the lowest daily dose that induces an observable adverse effect. The toxic effect characterized by the LOAEL is referred to as the "critical effect." To derive an RfD, the NOAEL (or LOAEL) is divided by uncertainty factors to ensure that the RfD will be protective of human health. Uncertainty factors are applied to account for extrapolation of data from laboratory animals to humans (interspecies extrapolation), variation in human sensitivity to the toxic effects of a compound (intraspecies differences), derivation of a chronic RfD based on a subchronic rather than a chronic study, or derivation of an RfD from the LOAEL rather than the NOAEL. In addition to

these uncertainty factors, modifying factors between 1 and 10 may be applied to reflect additional qualitative considerations in evaluating the data. For most compounds, the modifying factor is one.

A dermal RfD is developed by multiplying an oral RfD (based on an administered dose) by the gastrointestinal tract absorption factor. The resulting dermal RfD, based on an absorbed dose, is used to evaluate the dermal (absorbed) dose calculated by the dermal exposure algorithms.

The primary sources of information for RfD values are the EPA in Washington, D.C. (EPA 1997b, 2002a, and 2004a). EPA's database (IRIS - the Integrated Risk Information System) (EPA, 2004a) was consulted as the primary source for RfD values, as well as for CSFs. EPA intends that IRIS supersede all other sources of toxicity information for risk assessment. If values are not available in IRIS, the annual Health Effects Assessment Summary Tables (HEAST) (EPA, 1997b) were consulted, as well as the current Region IX EPA Preliminary Remediation Goals (PRGs) Table (EPA 2002a). If no RfD is available from any of these sources, non-carcinogenic risks are not quantified and potential exposures are addressed in the uncertainty section, see Section 6.7.

Ingestion and dermal RfDs for the COPCs at the Raymark OU2 groundwater study area are presented in Appendix F-1, Table 5.1. Inhalation RfDs for the COPCs at the OU2 study area are presented in Appendix F-1, Table 5.2. Appendix F-1, Tables 5.1 and 5.2 also include the primary target organs affected by a particular chemical. This information may be used in the risk characterization section (Section 6.5) to segregate risks by target organ effects, unless the total hazard index (HI) is below unity.

6.4.2 Carcinogenic Effects

The toxicity information considered in the assessment of potential carcinogenic risks includes a slope factor and a weight-of-evidence classification consistent with EPA's 1986 *Guidelines for Carcinogenic Risk Assessment* (EPA 1986). A revised weight-of-evidence classification system has been developed and presented in the *Draft Revised Guidelines for Carcinogenic Risk Assessment* (EPA 1999b); however, none of the COPCs for OU2 are impacted at this time. The 1986 weight-of-evidence classification qualitatively describes the likelihood that a chemical is a human carcinogen and is based on an evaluation of the available data from human and animal

studies. A chemical may be placed in one of the following five groups in EPA's 1986 classification system to denote its potential for carcinogenic effects:

- Group A - known human carcinogen
- Group B1 or B2 - probable human carcinogen
- Group C - possible human carcinogen
- Group D – cannot be classified as a human carcinogen because of a lack of data
- Group E - evidence of non-carcinogenicity in humans

The CSF is the toxicity value used to quantitatively express the carcinogenic hazard of cancer-causing chemicals. It is defined in the IRIS glossary as: "An upper-bound, approximately a 95 percent confidence limit, on the increased cancer risk from a lifetime exposure to an agent. This estimate, usually expressed in units of proportion (of a population) affected per mg/kg/day, is generally reserved for use in the low-dose region of the dose-response relationship, that is, for exposures corresponding to risks less than 1 in 100." (EPA 2004a). CSFs are derived from studies of carcinogenicity in humans and/or laboratory animals and are typically calculated for compounds in Groups A, B1, and B2; however, some Group C carcinogens also have CSFs and some B2 carcinogens, such as lead, have none. CSFs are specific to a chemical and route of exposure and are expressed in units of $(mg/kg/day)^{-1}$ for both oral and inhalation routes. Inhalation cancer toxicity values are usually expressed as inhalation unit risks in units of reciprocal $\mu g/m^3$ $[1/(\mu g/m^3)]$. Because cancer risk characterization requires an estimate of reciprocal dose in units of 1/(mg/kg/day), the inhalation unit risk must be converted to the mathematical equivalent of an inhalation CSF, or risk per unit dose (mg/kg/day). This is done by assuming that humans weigh 70 kg and inhale 20 m^3 of air per day [i.e., the inhalation unit risk ($1/\mu g/m^3$) is divided by 20 m^3, multiplied by 70 kg, and multiplied by 1,000 $\mu g/mg$ to yield the mathematical equivalent of an inhalation CSF (1/mg/kg/day)]. Oral and dermal CSFs for COPCs at the OU2 study area are presented in Appendix F-1, Table 6.1. Inhalation CSFs for the COPCs at the OU2 study area are presented in Appendix F-1, Table 6.2. The primary source of information for these values is the EPA IRIS database, followed by other EPA sources described for non-carcinogens.

CSFs exist for several (but not all) Group C compounds, which are identified as "possible" human carcinogens. These compounds typically exhibit inadequate evidence of carcinogenicity in humans and limited evidence in animals. In this human health risk assessment, Group C

compounds are evaluated quantitatively as Group A/B1/B2 compounds, but the risks associated with exposure to Group C compounds are also discussed separately if these chemicals are major risk drivers, underscoring the uncertainty associated with these estimations.

Dermal CSFs are derived from the corresponding oral values. In the derivation of a dermal CSF, the oral CSF is divided by the gastrointestinal absorption efficiency to determine a CSF based on an absorbed dose rather than an administered dose. The oral CSF is divided by the absorption efficiency because CSFs are expressed as reciprocal doses. Dermal CSFs and the absorption efficiencies used in their determination are also included in Appendix F-1, Table 6.1. Adjustments were made to the oral CSFs according to EPA guidance following Table 4.1, "Summary of Gastrointestinal Absorption Efficiencies and Recommendations for Adjustment of Oral Slope Factors for Specific Compounds" (EPA 2001b).

6.4.3 Toxicity Summaries for Major Chemicals of Potential Concern

This section contains brief summaries of the toxicological profiles for the major COPCs. The detailed profiles are contained in Appendix F-5.

1,1-Dichloroethene 1,1-Dichloroethene (1,1-DCE), also known as 1,1-dichloroethylene and vinylidine chloride, is a colorless liquid that is used primarily in the production of polyvinyl chloride (PVC) copolymers and as an intermediate for synthesis of organic chemicals. The major application for PVC copolymers is the production of flexible films for food packaging such as Saran® wrap.

The primary effect of acute exposure to high concentrations (approximately 4000 ppm) of 1,1-DCE vapor in humans is central nervous system (CNS) depression. Occupational exposure has been reported to cause liver dysfunction in workers. 1,1-DCE is irritating when applied to the skin and prolonged contact can cause first degree burns. Direct contact with the eyes may cause conjunctivitis and transient corneal injury. In experimental animals, the liver and kidneys are target organs for the toxic effects of 1,1-DCE. An RfD was derived for chronic exposure and subchronic exposure to 1,1-dichloroethene based on liver lesions seen in rats in a 2-year drinking water study. The oral RfD is currently under review and may be subject to change. An inhalation RfC for 1,1-DCE is under review and currently unavailable.

1,1-DCE was assigned to weight-of-evidence group C, possible human carcinogen, on the basis of one inhalation study. A CSF formerly available has been withdrawn.

Trichloroethene Trichloroethene (TCE), also known as trichloroethylene, is an industrial solvent used primarily in metal degreasing and cleaning operations.

Human and animal data indicate that exposure to TCE can result in toxic effects on a number of organs and systems, including the liver, kidney, blood, skin, immune system, reproductive system, nervous system, and cardiovascular system. TCE can be absorbed through the lungs, mucous membranes, gastrointestinal tract, and the skin. In humans, acute inhalation exposure to TCE causes central nervous system symptoms such as headache, dizziness, nausea, and unconsciousness. RfDs and RfCs for subchronic and chronic oral and inhalation exposure to TCE are presently under review by EPA.

Epidemiologic studies have been inadequate to determine if a correlation exists between exposure to TCE and increased cancer risk. Although EPA's Science Advisory Board recommended a weight-of-evidence classification of C-B2 continuum (C = possible human carcinogen; B2 = probable human carcinogen), the agency has not adopted a current position on the weight-of-evidence classification (EPA 2001c). In an earlier evaluation, TCE was assigned to weight-of-evidence Group B2, probable human carcinogen, based on tumorigenic responses in rats and mice for both oral and inhalation exposure and on inadequate data in humans.

The toxicity of TCE is presently under review by EPA. EPA has recently prepared an External Review Draft entitled *Trichloroethylene Health Risk Assessment: Synthesis and Characterization* (EPA 2001c), which recommends an inhalation CSF for TCE of 0.4 $(mg/kg/day)^{-1}$ and an inhalation RfD of 0.02 mg/kg/day. This risk assessment uses these recently proposed TCE toxicity values in calculations of risk in Appendix F-1 and Appendix F-2; however, for purposes of discussion, risk estimates using the CSF for TCE previously available are included in Appendix F-6. Prior to the publication of the proposed TCE toxicity values, RfDs for TCE were not available and, therefore, no non-cancer health hazards are estimated for TCE in Appendix F-6. The proposed CSFs for TCE are approximately two orders of magnitude greater than those previously available. Thus risks calculated using the proposed CSF are approximately two orders of magnitude greater than those presented in Appendix F-6.

Vinyl Chloride In animal studies, lifetime dietary ingestion of vinyl chloride increased mortality and induced mild histopathologic effects in the liver. Long-term inhalation studies in animals identified liver effects. Several early occupational studies associated vinyl chloride exposure with a syndrome known as vinyl chloride disease, which includes acroosteolysis (dissolution of the ends of the distal phalanges of the hands), circulatory disturbances in the extremities, Raynaud's Syndrome (sudden, recurrent bilateral cyanosis of the digits), scleroderma, hematologic effects, effects on the lungs, and impaired liver function and liver damage. Mild neurologic effects were also associated with occupational exposure. The principal target organs for vinyl chloride appear to be the central nervous system and the liver.

The EPA (2004a) lists vinyl chloride as an EPA cancer weight-of-evidence Group A compound (human carcinogen) and presents a verified oral CSF of 1.9 per mg/kg/day, based on the increased incidence of liver and lung tumors in a lifetime dietary study in rats. An inhalation unit risk of 8.8E-06 per :g/m^3, equivalent to 0.03 per mg/kg/day, assuming humans inhale 20 m^3 of air/day and weigh 70 kg, is based on liver tumors in rats intermittently exposed by inhalation for 12 months.

Arsenic Arsenic is found in nature at low levels. Inorganic arsenic compounds are mainly used to preserve wood. They are also used to make insecticides and weed killers.

Inorganic arsenic is a human poison; organic arsenic is less harmful. High levels of inorganic arsenic in food or water can be fatal. Arsenic damages many tissues including nerves, stomach and intestines, and skin. Breathing high levels can result in a sore throat and irritated lungs. Lower levels of exposure to inorganic arsenic may cause nausea, vomiting, diarrhea, decreased production of red and white blood cells, abnormal heart rhythm, blood vessel damage, and a "pins and needles" sensation in hands and feet. Long term exposure to inorganic arsenic may lead to a darkening of the skin and the appearance of small "corns" or "warts" on the palms, soles, and torso. Direct skin contact may cause redness and swelling.

EPA has classified arsenic as a human carcinogen of Group A. Breathing inorganic arsenic increases the risk of lung cancer. Ingesting inorganic arsenic increases the risk of skin cancer and tumors of the bladder, kidney, liver, and lung.

Zinc Zinc is a nutritionally required trace element. Absorbed zinc is distributed primarily to the liver, with subsequent redistribution to bone, muscle, and kidney. The highest tissue concentrations are found in the prostate. The half-life of zinc absorbed from the gastrointestinal (GI) tracts of humans in normal zinc homeostasis is approximately 162 to 500 days.

Orally, zinc exhibits a low order of acute toxicity. Animals dosed with 100 times the dietary requirement showed no evidence of toxicity. In humans, acute poisoning from foods or beverages prepared in galvanized containers is characterized by GI upset. Chronic oral toxicity in animals is associated with poor growth, GI inflammation, arthritis, lameness, and anemia, possibly secondary to copper deficiency.

6.5 Risk Characterization

This section provides a characterization of the potential human health risks associated with the potential exposure to all COPCs in indoor air and groundwater in the OU2 study area. Section 6.5.1 outlines the methods used to estimate the type and magnitude of health risks, and Section 6.5.2 presents the results for the current and potential future land use conditions for the OU2 study area. Section 6.5.3 presents a qualitative evaluation of potential future oyster tissue concentrations resulting from migration of contaminants in groundwater to surface water at Ferry Creek and the Housatonic River.

6.5.1 Risk Characterization Methodology

Potential human health risks resulting from exposure to all COPCs are estimated using algorithms established by EPA. The methods described by EPA are protective of human health and are likely to overestimate (rather than underestimate) risk. The methodology uses specific algorithms to calculate risk as a function of chemical concentration, human exposure parameters, and toxicity.

Risks from hazardous chemicals are calculated for either carcinogenic or non-carcinogenic effects. Some carcinogenic chemicals may also exhibit non-carcinogenic effects. Potential impacts are then characterized for both types of health effects.

Non-carcinogens The hazards associated with the effects of non-carcinogenic chemicals are evaluated by comparing an exposure level or intake to a reference dose (RfD). The ratio of the intake to the RfD is called the hazard quotient (HQ) and is defined as follows (EPA, 1989b):

$$HQ_i = \frac{Intake_i}{RfD_i}$$

where: HQ_i = Hazard Quotient for chemical "i" (unitless)

$Intake_i$ = Intake of chemical "i" (mg/kg/day), a function of exposure and chemical concentration

RfD_i = Reference Dose of chemical "i" (mg/kg/day)

If the ratio of the intake to the RfD exceeds unity, there exists a potential for non-carcinogenic (toxic) effects to occur. A hazard index (HI) is generated by summing the individual HQs for all the COPCs. If the value of the HI exceeds unity, there is a potential for non-carcinogenic health effects associated with that particular chemical mixture, and therefore it is necessary to segregate the HQs by target organ effects. The HQ should not be construed as a probability, but rather as a numerical indicator of the extent to which a predicted intake exceeds or is less than an RfD.

Chemical Carcinogens Risks attributable to exposure to chemical carcinogens are estimated as the probability of an individual developing cancer over a lifetime as a result of exposure to a potential carcinogen. At low doses, the incremental lifetime cancer risk (ILCR) is determined as follows (EPA 1989b):

$$ILCR_i = (Intake_i)(CSF_i)$$

where: $ILCR_i$ = Incremental Lifetime Cancer Risk for chemical "i", expressed as a unitless probability

$Intake_i$ = Intake of chemical "i" (mg/kg/day)

CSF_i = Cancer Slope Factor of chemical "i" $(mg/kg/day)^{-1}$

Risks below 1E-6 (or a risk less than 1 in 1 million) are generally considered to be acceptable by EPA, and risks greater than 1E-4 (1 in 10,000) are generally considered to be unacceptable.

The CTDEP regulations use 1E-5 (1 in 100,000) as a break point between acceptable and unacceptable risk for cumulative cancer risk from multiple contaminants and a break point of 1E-6 (one-in-a-million) cancer risk for individual contaminants (CTDEP 1996).

Risks are estimated for all carcinogenic compounds regardless of the class designation (See Section 6.4.2).

6.5.2 Risk Characterization Results

A summary of the quantitative risk assessment for the OU2 study area is provided in this section. Sample calculations are provided in Appendix F-4. Appendix F-1, Tables 7.1A, 7.1B, 7.2A, and 7.2B and Appendix F-1, Tables 8.1A, 8.1B, 8.2A, and 8.2B present non-cancer and cancer risk estimates for each receptor and media. Appendix F-1, Tables 9.1A, 9.1B, 9.2A, and 9.2B present summaries of cancer risks and health hazard indices from all applicable media and pathways for each exposure scenario. Appendix F-1, Tables 10.1A and 10.1B reduce the information developed in Appendix F-1, Tables 9.1A and 9.1B to the major risk drivers for the residential and industrial/commercial indoor air exposure scenarios, the only media and scenarios with cancer risks greater than 1E-04 or hazard indices greater than 1.0. Table 6-1 summarizes the results and provides the target organ hazard indices and primary contributors to unacceptable risks for all scenarios. Indoor air risks have been mitigated at some residences, as a result of the installation of sub-slab ventilation systems. However, the risks would return in the future if the systems stop working.

Non-carcinogenic Risks Hazard indices (HIs) developed for current/future residents, current/future industrial/commercial workers, and future recreational users are summarized in the table below:

Exposure Scenario	RME Case	CTE Case
Current/Future Resident - Indoor air (Adult/Child)	37	6.4
Current/Future Industrial/Commercial Workers – Indoor Air (Adult)	8.9	1.5
Future Recreational Users - Surface Water (Adult) (Child)	 0.0008 0.0016	 Not evaluated Not evaluated

HIs are in excess of unity for current and future residential exposures to indoor air and current and future industrial/commercial worker exposures to indoor air. 1,2,4-Trimethylbenzene, 1,2-dibromoethane, 1,3,5-trimethyl benzene, 1,4-dichlorobenzene, benzene, chloroform, xylenes, and TCE were the main contributors to the HI for residents. 1,2,4-Trimethylbenzene, 1,2-dibromoethane, 1,4-dichlorobenzene, and xylenes were the main contributors to the HI for industrial/commercial workers. The chemical-specific (and target-organ specific) HIs for each of these contaminants alone are in excess of unity under the RME scenario. Adverse non-carcinogenic health effects are possible from exposure to these contaminants in indoor air. See Tables 7.1A RME, 7.1B RME, 7.1A CTE, and 7.1B CTE in Appendix F-1 for details on the non-cancer HI calculations.

HIs are less than unity for future adult and child recreational exposures at Ferry Creek.

Current frequent recreational user exposures to surface water in Ferry Creek were evaluated in the OU3 Area I RI. HIs were below one indicating that adverse non-carcinogenic health effects are not anticipated under the conditions established in the OU3 Area I RI exposure assessment.

Carcinogenic Risks Incremental cancer risk estimates developed for current/future residents, current/future industrial/commercial workers, and future recreational users are summarized in the table below:

Exposure Scenario	RME Case	CTE Case
Current/Future Resident -Indoor air (Adult/Child)	4.8E-03	1.3E-04
Current/Future Industrial/Commercial Worker – Indoor Air (Adult)	9.7E-4	3.0E-5
Future Recreational Users - Surface Water		
(Adult) (1)	6.8E-08	Not evaluated
(Child) (1)	3.4E-08	Not evaluated

1) Summation of total risk for recreational user (adult plus child): 1.0E-07

The EPA cancer risk range is 10^{-4} to 10^{-6}. The CTDEP target cancer risk level is 10^{-6} for single contaminants and 10^{-5} for total risk from multiple contaminants. The risk estimates for residential exposures to indoor air exceed the EPA cancer risk range (10^{-4} to 10^{-6}), when either the RME or CTE case is evaluated. The major contributors to residential cancer risk at the site are 1,1,2,2-tetrachloroethane, 1,3-butadiene, benzyl chloride, methylene chloride, and TCE.

See Tables 8.1A RME and 8.1A CTE in Appendix F-1 for details on the residential cancer risk calculations. As detailed on Appendix F-1, Table 9.1A, residential cancer risks for each individual volatile organic contaminant with available inhalation cancer slope factors are greater than the CTDEP target risk level of 10^{-6} for single contaminants under the RME scenario. The risk estimates for industrial/commercial exposures to indoor air exceed the EPA cancer risk range (10^{-4} to 10^{-6}), when the RME case is evaluated. The major contributors to industrial/commercial cancer risk at the site are 1,1,2,2-tetrachloroethane, 1,3-butadiene, and TCE. See Tables 8.1B RME and 8.1B CTE in Appendix F-1 for details on the industrial/commercial cancer risk calculations. As detailed on Appendix F-1, Table 9.1B, industrial/commercial cancer risks for the major contributors to risk and several other volatile organic contaminants are greater than the CTDEP target risk level of 10^{-6} for single contaminants under the RME scenario.

Cancer risks for the adult and child recreational users are added together for a lifetime exposure. The risk estimates for future recreational users exposed to surface water in Ferry Creek do not exceed the EPA cancer risk range (10^{-4} to 10^{-6}) or the CTDEP target total risk level of 10^{-5} for multiple contaminants, indicating that increased carcinogenic health effects are not anticipated under the conditions established for these receptors in the exposure assessment. As detailed on Appendix F-1, Tables 9.2A and 9.2B, cancer risks for each individual contaminant are less than the CTDEP target risk level of 10^{-6} for single contaminants.

Cancer risk estimates evaluated in the OU3 Area I RI for current frequent recreational user exposures to surface water in Ferry Creek ranged from 5.5E-06 to 1.9E-05 for lifetime exposures. These cancer risk estimates fall within the EPA cancer risk range (10^{-4} to 10^{-6}), but RME risk estimates exceed the CTDEP target total risk level of 10^{-5} for multiple contaminants. The primary contributors to risk were Aroclors, 1,1-DCE, and vinyl chloride. Cancer risks for each of these individual contaminants are greater than the CTDEP target risk level of 10^{-6} for single contaminants.

6.6 Modeled Indoor Air Evaluation

Indoor air exposures were also evaluated through modeling indoor air concentrations from maximum shallow groundwater concentrations within the indoor air area of interest. Appendix F-2, Table 1 presents the list of groundwater samples used for the selection of COPCs for this

evaluation. Appendix F-2, Table 2 presents the COPC selection table. As discussed in Section 6.2, only three contaminants were detected in shallow groundwater within the indoor air area of interest at concentrations exceeding Connecticut groundwater volatilization criteria. This limited number of COPCs contrasts with the numerous COPCs identified from the measured indoor air data. This implies the likelihood of indoor sources of contamination. The Johnson and Ettinger (1991) Model for Subsurface Vapor Intrusion into Buildings was used in estimating indoor air concentrations resulting from volatilization of contaminants in groundwater into indoor spaces (homes and commercial properties) and associated inhalation risks from residential and industrial/commercial exposures.

Indoor air concentrations predicted from the model are presented in Appendix F-2, Table 3. Appendix F-2, Table 3 presents average and maximum detected groundwater concentrations for the three COPCs and the indoor air concentrations developed from the maximum groundwater concentrations. Predicted indoor air concentrations for these COPCs are significantly higher than actual detected indoor air concentrations. This may be because of variability of construction material between homes, structural integrity of basement walls, and other property-specific factors. It is also possible that in the limited sampling performed to date, buildings with the highest indoor air concentrations resulting from volatilization of groundwater contaminants have not yet been sampled. Johnson and Ettinger model spreadsheets for each COPC are included in Appendix F-2. These spreadsheets predict indoor air cancer risks from residential exposures to the three groundwater COPCs of 1.2E-02 and hazard indices of 12.2. This hazard index falls between the RME and CTE hazard indices estimated for residents based on actual indoor air concentrations. Hazard quotients for both TCE and 1,1-DCE exceed unity. Cancer risks from exposures to these modeled indoor air concentrations are significantly greater than RME cancer risks estimated based on actual indoor air concentrations. These risks exceed EPA's cancer risk range (10^{-4} to 10^{-6}). TCE was the main contributor to risk estimates.

The spreadsheets predict indoor air cancer risks from industrial/commercial exposures to the three groundwater COPCs of 2.5E-03 and hazard indices of 2.9. This hazard index falls between the RME and CTE hazard indices estimated based on actual indoor air concentrations. Hazard quotients for both TCE and 1,1-DCE exceed unity. Cancer risks from exposures to these modeled indoor air concentrations are greater than RME cancer risks estimated based on actual indoor air concentrations. These risks exceed EPA's cancer risk range (10^{-4} to 10^{-6}). TCE was the main contributor to risk estimates.

6.7 Uncertainties Analysis

There is uncertainty associated with all aspects of the baseline human health risk assessment presented in the preceding sections. This section presents a summary of these uncertainties, with a discussion of how they may affect the final risk numbers discussed in Section 6.5.

There is uncertainty associated with all steps of the risk assessment process. The selection of contaminants of potential concern is based on exposure assumptions and toxicity information, which in turn have associated uncertainties. Uncertainty in the selection of COPCs and exposure point concentrations is associated with the current status of the predictive databases and the procedures used to include or exclude constituents as chemicals of potential concern, and the methods and models used and the assumptions made to determine exposure point concentrations. Uncertainty associated with the exposure assessment includes the values used as input variables for a given intake route, and the predictions regarding future land use and population characteristics. Uncertainty in the toxicity assessment includes the quality of the existing data to support dose-response relationships and the weight-of-evidence used for determining the carcinogenicity of chemicals of concern. Uncertainty in risk characterization includes that associated with exposure to multiple chemicals and the cumulative uncertainty from combining conservative assumptions made in the exposure and toxicity assessments.

While there are various sources of uncertainty as described above throughout the entire risk assessment, assumptions were made so that the final calculated risks would be conservative estimates that are protective of public health. Thus, the resultant uncertainty in the numerical risk assessments is in how much lower the actual risks are.

Generally, risk assessments carry two types of uncertainty: measurement and informational uncertainty. Measurement uncertainty refers to the variance that can be attributed to sampling techniques and laboratory analysis of contaminants. For example, this type of uncertainty is associated with analytical data collected for each site. The risk assessment reflects the accumulated variances of the individual values used.

Informational uncertainty refers to estimates of toxicity and exposure. Often, this gap is significant, such as the absence of information on the effects of human exposure to low doses

of a chemical, the biological mechanism of action of a chemical, or the behavior of a chemical in soil.

Once the risk assessment is complete, the results must be reviewed and evaluated to identify the type and magnitude of uncertainty involved. Reliance on the results from a risk assessment without a consideration of uncertainties, limitations, and assumptions inherent in the process can be misleading. For example, to account for uncertainties in the development of exposure assumptions, conservative estimates must be made to ensure that the particular assumptions made are protective of sensitive subpopulations or the maximum exposed individuals. If a number of conservative assumptions are combined in an exposure model, the resulting calculations can propagate the uncertainties associated with those assumptions, thereby producing a much larger uncertainty for the final results. This uncertainty is biased toward over-predicting both carcinogenic and non-carcinogenic risks. Thus, both the results of the risk assessment and the uncertainties associated with those results must be considered when making risk management decisions.

This interpretation is especially relevant when the risks exceed the point-of-departure for defining "acceptable" risk. For example, when risks calculated using a high degree of uncertainty are below an "acceptable" risk level (1E-6), the interpretation of no significant risk is straightforward. However, when risks calculated using a high degree of uncertainty are above an "acceptable" risk level (1E-4), a conclusion can be difficult unless uncertainty is considered.

EPA guidance on risk assessment (EPA 1992a and 1994) requires risk assessors to use exposure and toxicity assumptions from the "high end" and the "central tendency" of their distributions. These values correspond to the RME and CTE scenarios. The RME is conceptually the “high end” exposure above the 90th percentile of the population distribution, but not higher than the individual in the population with the highest exposure. The CTE reflects the central (average) estimates of exposure.

6.7.1 Uncertainty in Data Evaluation

There is uncertainty associated with the selection of COPCs and the calculation of exposure point concentrations on the final risk values in the quantitative risk assessment.

The objective of this assessment is to estimate potential current or future risks to the public from the chemicals detected in the groundwater within the study area. Indoor air samples were collected to establish evidence of volatilization of groundwater contaminants into indoor air spaces and to calculate risks from exposures to these contaminants. However, indoor air sampling does not discriminate between contaminants present because of volatilization from groundwater and contaminants originating from indoor sources. The sources of the indoor air contaminants are not fully understood. Based on the identification of three contaminants in groundwater within the indoor air area of investigation above CT groundwater volatilization criteria, the groundwater is a possible source for these three contaminants in indoor air. For all other indoor air COPCs, the source may be indoor sources or groundwater (where indoor COPCs are in the groundwater). Based on the comparison of measured groundwater data to Connecticut groundwater volatilization criteria, these chemicals were not present in groundwater at concentrations that would lead to expectations of concern to indoor air. Indoor air COPCs identified through comparison to EPA Region IX PRGs for ambient air may include contaminants not present in groundwater. This serves to overestimate risks from volatilization of contaminants from groundwater into indoor air spaces.

EPA has not published groundwater screening values based on potential volatilization to indoor air or migration to surface water. In lieu of EPA screening values, Connecticut groundwater volatilization criteria and Connecticut surface water protection criteria were used for groundwater COPC selection. These criteria are conservatively based on target cancer risk levels of 10^{-6} and hazard quotients of 1.0. The CTDEP has not developed these criteria for all the groundwater contaminants detected in the OU2 study area. Contaminants without Connecticut criteria were eliminated from the quantitative risk assessment. Because of the elimination of contaminants that lack Connecticut groundwater protection criteria for the two exposure pathways of concern, total risks may be underestimated. However, it is assumed that the State of Connecticut, in the development of its groundwater protection criteria, has selected the contaminants of greatest concern for each potential exposure pathway. Therefore, it is unlikely that any contaminants that may pose a significant risk were eliminated from the risk assessment.

Uncertainty may arise in the selection of groundwater COPCs for use in the evaluation of modeled indoor air concentrations because of the use of CTDEP Groundwater Volatilization Criteria (CTDEP 1996). In March 2003, CTDEP released “Proposed Revisions” to

Connecticut's Remediation Standards Regulations Volatilization Criteria (CTDEP 2003). New Groundwater Volatilization Criteria were proposed based on revised indoor target air concentrations (TACs) using new toxicity values, a revised transport model, and improved understanding of the pathway. The TACs are generally based on either residential or industrial exposures and resulting hazard indices of 1.0 or cancer risks of 10^{-6}. For TCE and benzene, the TAC values are based on background concentrations. These proposed revisions have not yet been accepted.

In general the proposed Groundwater Volatilization Criteria are less than (or more stringent than) the existing criteria. One exception is the proposed Groundwater Volatilization Criteria for 1,1-DCE, which is greater than the existing criteria. It is possible that by using the proposed Groundwater Volatilization Criteria for COPC selection an increased number of contaminants may be identified as groundwater COPCs for use in the evaluation of modeled indoor air concentrations. However, no additions or deletions from the COPC list for the groundwater in the indoor air area of interest would occur among the seven target contaminants originally identified within groundwater throughout the OU2 study area.

Background concentrations were not used to eliminate COPCs. Maximum indoor air concentrations from two homes located outside the indoor air area of investigation are presented in Appendix F-1, Table 2.1 for comparison purposes. Comparison of this background data to concentrations of indoor air detected within the indoor air area of investigation indicates that the presence of some indoor air COPCs may be attributable to background conditions.

Indoor air and groundwater COPCs were selected if the maximum detected concentration in indoor air or groundwater exceeded its respective risk-based screening criterion. Even if the compound was detected at a very low frequency, i.e., less than five percent, if the maximum detected concentration exceeded the screening criterion, the compound was still identified as a COPC. Several groundwater COPCs selected on the basis of concentrations greater than SWPC were not modeled because of infrequent detections or exceedances of selection criteria, because of their absence in shallow groundwater, because a source could not be identified, because they do not appear to be derived from Raymark waste, or because the scientific literature does not contain sufficient information on the behavior of the contaminant to model its migration. Among the COPCs modeled, four metals were not evaluated in the HHRA because they are not expected to arrive at either Ferry Creek or the Housatonic River within the

foreseeable future; therefore, these contaminants were effectively eliminated from the risk evaluation. This contributes to the uncertainty of the risk assessment. The model indicates that TCE is migrating toward the Housatonic River, rather than toward Ferry Creek. It is expected that maximum predicted concentrations of TCE have already reached the river. Further sampling data in the Housatonic River are needed to estimate current or future surface water concentrations. The absence of an evaluation of future TCE concentrations in the Housatonic River may result in an underestimation of risks.

Calculation of Exposure Point Concentrations

Variability between homes and potential indoor sources of contaminants introduces uncertainty in calculations of indoor air exposure point concentrations. Indoor air data from throughout the indoor air area of interest were combined to determine a single set of EPCs. No distinctions in data sampling locations based on first floor versus basement levels were made. Maximum detected concentrations of indoor air contaminants from all homes and non-residential properties sampled within the indoor air area of interest were used to evaluate RME risks. Average concentrations were used to evaluated CTE risks. The use of maximum concentrations is intended to result in a high end estimate of risk; however, since sampling was not performed in every home and commercial property within the area of concern, it is possible that higher concentrations of contaminants may be present.

Section 5.3.2 discusses uncertainty in surface water concentration modeling. The maximum detected chemical concentrations in groundwater were used to model future worst-case surface water concentrations. The model used to predict concentrations of contaminants entering Ferry Creek is a relatively simple mathematical model that is being used to describe fate and transport in a complex system. As a result, it is not intended to provide highly accurate predictions of contaminant migration. Despite the limitations inherent in the simple modeling approach taken, the results strongly suggest that concentrations of 1,1-DCE and TCE have reached maximum, sustained concentrations and concentrations of inorganic contaminants entering Ferry Creek are expected to increase over time and remain at higher concentrations until the sources are depleted. As a result, future risks associated with surface water exposures at Ferry Creek are expected to be greater than current risks. Surface water concentrations at Ferry Creek may present unacceptable risks in the future.

The use of a dilution factor based on a comparison of groundwater seep data and surface water concentrations in the absence of specific stream flow information for Ferry Creek, introduces uncertainty in the exposure point concentrations. However, the selection of a dilution factor of 0.1 is likely to underestimate dilution; therefore risks based on these concentrations are likely overestimated.

Exposure point concentrations for indoor air are modeled concentrations based on measured groundwater concentrations. The model assumes the groundwater concentrations entered into the model represent concentrations directly below the building being evaluated. Since groundwater samples were not collected from locations beneath buildings, but were from a relatively few irregularly spaced locations within the indoor air area of interest, uncertainty is introduced. The maximum shallow groundwater concentrations from the indoor air area of interest were used to assess indoor air RME risks. The model then estimates indoor air concentrations for a hypothetical home or building located directly above the maximum groundwater concentrations. In addition to uncertainty related to groundwater sampling, uncertainty is associated with the soil and building parameters used in the model. The model used estimated reasonable soil parameters based on the identified soil type and default building structural parameters. Soil parameters were selected based on reported values for sand. Default building structural parameters are used because the risk evaluation is for a hypothetical building. As discussed in Section 6.6, indoor air sampling within the indoor air area of interest detected the presence of the identified COPCs at concentrations below the maximum concentrations predicted by the model. It is difficult to determine the accuracy of the model.

6.7.2 Uncertainty in the Exposure Assessment

Uncertainty in the exposure assessment arises from the determination of land use conditions, selection of receptors, and selection of exposure parameters. Each of these factors is discussed below.

Land-Use Conditions Land-use conditions were based on current uses. If future use changes, this HHRA may under or over-estimate risks.

Exposure Routes and Receptor Identification Exposure routes and receptor groups were based on discussions with the EPA and site visits. This may either under- or over-estimate the risks, with the final result dependent on how well the receptors were defined.

One scenario of potential concern, but not evaluated quantitatively in this RI, is future human consumption of oysters. Current human consumption of oysters was evaluated in the OU3 Area II RI (TtNUS 2000b). The results indicate that there are currently no significant risks associated with human consumption of oysters in the study area. At the time, arsenic and zinc were not detected in oyster tissue. In the future, contaminated shallow groundwater contributing to increasing surface water concentrations may increase contaminant concentrations in oysters. See Section 6.7.4.

Selection of Exposure Parameters Each exposure factor selected for use in this risk assessment has some associated uncertainty. Generally, exposure factors are based on surveys of physiological and lifestyle profiles across the United States. The attributes and activities studied in these surveys generally have a broad distribution. To avoid underestimation of exposure, EPA guidelines on the RME receptor were used; they generally consist of the 95th percentile for most parameters. Therefore, the selected values for the RME receptor represent the upper bound of the observed or expected habits of the majority of the population.

Many of the exposure parameters were determined from statistical analyses on human population characteristics. Often the database used to summarize a particular exposure parameter (body weight) is quite large. Consequently, the values chosen for such variables in the RME scenario have low uncertainty. For many parameters for which limited information exists (dermal absorption of organic chemicals from surface water), there is greater uncertainty.

Many of the quantities used to calculate exposures and risks in this report were selected from a distribution of possible values. For the RME scenario, the value representing the 90-95th percentile was selected for some, but not all, parameters to ensure that the assessment bounds the actual risks from a postulated exposure. In order to estimate a central tendency estimate of exposure, EPA has suggested the use of the CTE receptor, whose intake variables are set at approximately the 50th percentile of the distribution. The risks for this receptor seek to incorporate the range of uncertainty associated with various intake assumptions. EPA Region I

provides some default parameters (EPA 1994); however, many of the parameters were estimated using professional judgment.

6.7.3 Uncertainty in the Toxicological Evaluation

A toxicity evaluation is the hazard identification and dose-response assessment of a chemical. The hazard identification deals with characterizing the nature and strength of the evidence of causation, or the likelihood that a chemical that induces adverse effects in animals will also induce adverse effects in humans. Hazard identification of carcinogenicity is an evaluation of the weight-of-evidence that a chemical causes cancer. Positive animal cancer test data suggest that humans contain tissue(s) that may also manifest a carcinogenic response; however, the animal data cannot necessarily be used to predict the target tissue in humans. In contrast, in the hazard assessment of non-cancer effects, positive animal data suggest the nature of the effects (the target tissues and type of effects) anticipated in humans.

Uncertainty in hazard assessment arises from the nature and quality of the animal and human data. Uncertainty is reduced when similar effects are observed across species, strain, sex, and exposure route; when the magnitude of the response is clearly dose-related; when pharmacokinetic data indicate a similar fate in humans and animals; when postulated mechanisms of toxicity are similar for humans and animals; and when the chemical of concern is structurally similar to other chemicals for which the toxicity is more completely characterized.

Uncertainty in the dose-response evaluation includes the determination of a CSF for the carcinogenic assessment and derivation of an RfD for the non-carcinogenic assessment. The slope factor is an upper bound estimate of the human cancer risk per milligram of contaminant per milligram of body weight per day. The RfD is an estimate, with uncertainty spanning perhaps an order of magnitude, of daily exposure to humans below which there is likely to be no appreciable risk of adverse effect over a lifetime. Uncertainty is introduced from interspecies (animal to human) extrapolation, which, in the absence of quantitative pharmacokinetic or mechanistic data, is usually based on consideration of interspecies differences in basal metabolic rate. Uncertainty also results from intraspecies variation. Most toxicity experiments are performed with animals that are very similar in age and genotype so that intragroup biological variation is minimal; however, the human population of concern may reflect a great deal of heterogeneity, including unusual sensitivity or tolerance to the COPC. Even toxicity data

from human occupational exposure reflect a bias because only those individuals sufficiently healthy to attend work regularly (the "healthy worker effect") and those not unusually sensitive to the chemical are likely to be occupationally exposed.

Finally, uncertainty arises from the quality of the key study from which the quantitative estimate is derived and from the database. For cancer effects, the uncertainty associated with dose-response factors is mitigated by assuming the 95 percent upper bound for the slope factor. Another source of uncertainty in carcinogenic assessment is the method by which data from high doses in animal studies are extrapolated to the dose range expected for environmentally exposed humans. The linearized multistage model, which is used in nearly all quantitative estimations of human risk from animal data, is based on a nonthreshold assumption of carcinogenesis. There is evidence to suggest, however, that epigenetic carcinogens, as well as many genotoxic carcinogens, have a threshold below which they are non-carcinogenic (William and Weisburger, 1991). Therefore, the use of the linearized multistage model is conservative for chemicals that exhibit a threshold for carcinogenicity.

For non-cancer effects, additional uncertainty factors may be applied in the derivation of the RfD to mitigate poor quality of the key study or gaps in the database. Additional uncertainty for non-cancer effects arises from the use of an effect level in the estimation of an RfD, because this estimation is predicated on the assumption of a threshold below which adverse effects are not expected. Therefore, an uncertainty factor is usually applied to estimate a no-effect level. Additional uncertainty arises in estimation of an RfD for chronic exposure from less-than-chronic data. Unless empirical data indicate that effects do not worsen with increasing duration of exposure, an additional uncertainty factor is applied to the no-effect level in the less-than-chronic study. Uncertainty in the derivation of RfDs is mitigated by the use of uncertainty and modifying factors that normally range between 3 and 10. The resulting combination of uncertainty and modifying factors may reach 1,000 or more.

Group C carcinogens are classified as possible human carcinogens because the evidence for their carcinogenicity in animals is limited. The inclusion of these compounds in the estimation of total carcinogenic risk adds to the uncertainty of the final risk numbers by potentially overestimating the human health effects.

The derivation of dermal RfDs and CSFs from oral values may cause uncertainty. This is particularly the case when no gastrointestinal absorption rates are available in the literature or when only qualitative statements regarding absorption are available.

Uncertainty in the dose-response evaluation is associated with a lack of toxicity values for some contaminants. In this risk assessment no EPA verified RfCs were available for some of the COPCs being evaluated for indoor air exposures. Use of route-to-route extrapolation of inhalation RfDs or RfCs from oral RfDs is not supported by EPA because of the high level of uncertainty this process introduces. The lack of verified toxicity values for potential inhalation exposures results in an under estimation of non-cancer risks.

Use of the proposed toxicity factors for TCE from EPA's External Review Draft entitled *Trichloroethylene Health Risk Assessment: Synthesis and Characterization* (EPA 2001c) may overestimate risks from exposure to TCE. EPA and the scientific community are still reviewing this document. Appendix F-6 presents the risk calculations for site exposures to TCE using the TCE toxicity factors available prior to the preparation of this document. Cancer risks estimated using this approach are approximately two orders of magnitude smaller than risks calculated using the proposed CSF for TCE. Since RfDs have not previously been available for evaluation of non-cancer health hazards from exposures to TCE, the total HIs presented in Appendix F-6 reflect zero contribution from TCE.

Uncertainty in the final calculations of risk results from assumptions made regarding additivity of effects from exposure to multiple compounds from various exposure routes. High uncertainty exists when cancer risks for several substances are summed across different exposure pathways. This assumes that each substance has a similar effect and/or mode of action. Often compounds affect different organs, have different mechanisms of action, and differ in their fate in the body, so additivity may not be an appropriate assumption. However, the assumption of additivity is made to provide a conservative estimate of risk.

Finally, the risk characterization does not consider antagonistic or synergistic effects. Little or no information is available to determine the potential for antagonism or synergism for the COPCs. Therefore, this uncertainty cannot be discussed for its impact on the risk assessment, since it may either underestimate or overestimate potential human health risk.

6.7.4 Qualitative Evaluation of Potential Risks from Oyster Consumption

The mouth of the Housatonic River is considered to be a recreational fishery and there may be the potential for human consumption of biota taken from the river. Coastal waterways are assumed to support recreational and commercial fishing. Currently recreational shellfishing is prohibited in the Town of Stratford. The lower Housatonic River, near the mouth of Ferry Creek, contains important commercial seed beds for oyster cultivation. Oysters remain in these seed beds for one year or less before being moved to a different location for the remainder of their growing period.

Current human consumption of oysters was evaluated in the OU3 Area II RI (TtNUS 2000b) without regard to the current recreational shellfishing prohibition. The results indicate that there are currently no significant risks associated with human consumption of oysters in the study area. Cancer risk estimates ranged from 8.9E-07 to 6.2E-06. Hazard indices were below one. At the time, arsenic and zinc were not detected in oyster tissue.

In the future, contaminated shallow groundwater contributing to increasing surface water concentrations, may increase contaminant concentrations in oysters. As described in Section 6.2.3, future oyster tissue concentrations were estimated from modeled future surface water concentrations in Ferry Creek and the Housatonic River. The modeled future surface water concentrations were estimated from unfiltered groundwater concentrations. Thus predicted future oyster concentrations are the result of multiple stage modeling and must be considered very crude estimates. The predicted future oyster tissue arsenic concentrations are 0.076 mg/kg in Ferry Creek and 0.0076 mg/kg in the Housatonic River (see Appendix F-3). These concentrations are greater than the EPA Region III risk-based concentrations (RBCs) for fish ingestion (EPA 2004c) of 0.0021 mg/kg. The predicted future oyster tissue zinc concentrations are 5.05 mg/kg in Ferry Creek and 1.6 mg/kg in the Housatonic River. These concentrations are less than the EPA Region III RBCs for fish ingestion. It is possible based on increased arsenic concentrations that future risks from human consumption of oysters may be higher than 10^{-6}; however, because oysters remain in the commercial seed beds for a small portion of their growing period, it is unlikely that the commercial oyster cultivation will be impacted.

6.8 Conclusions

The Human Health Risk Assessment (HHRA) identified VOCs (1,1,2,2-tetrachloroethane, 1,1,2-trichloroethane, 1,1-dichloroethene (1,1-DCE), 1,2,4-trimethylbenzene, 1,2-dibromoethane, 1,2-dichloroethane, 1,3,5-trimethylbenzene, 1,3-butadiene, 1,4-dichlorobenzene, 4-methyl-2-pentanone, acetone, acrylonitrile, benzene, benzyl chloride, bromodichloromethane, carbon tetrachloride, chloroethane, chloroform, chloromethane, cis-1,2-dichloroethene, dibromochloromethane, dibromofluoromethane, ethylbenzene, hexane, methyl tert-butyl ether, methylene chloride, tetrachloroethene (PCE), tetrahydofuran, toluene, xylenes, trichloroethene (TCE), vinyl acetate, and vinyl chloride) as the contaminants of potential concern for exposures to indoor air within the study area. These contaminants were selected based on comparison of indoor air data to EPA Region IX Preliminary Remediation Goals for ambient air.

VOCs (1,1-DCE, 1,1,1-trichloroethane, benzene, PCE, and TCE); PAHs (acenaphthylene, benzo(b)fluoranthene, benzo(k)fluoranthene, and phenanthrene); pesticides (gamma-chlordane, dieldrin, heptachlor, and heptachlor epoxide); and metals (arsenic, beryllium, cadmium, lead, mercury, nickel, selenium, silver, thallium, and zinc) were identified as the contaminants of potential concern for future exposures resulting from migration of groundwater contaminants into surface water within the study area. These contaminants were selected based on comparison of groundwater data to Connecticut surface water protection criteria. Future surface water concentrations for many of the contaminants selected on the basis of exceedances of surface water protection criteria were not modeled because of infrequent detections or exceedances of selection criteria, because of their absence in shallow groundwater, because a source could not be identified, because they do not appear to be derived from Raymark waste, or because the scientific literature does not contain sufficient information on the behavior of the contaminant to model its migration. For surface water exposures in Ferry Creek, modeled concentrations of 1,1-DCE, arsenic, and zinc were evaluated.

The HHRA also provided a comparison of groundwater data to Connecticut groundwater volatilization criteria. This comparison identified VOCs (1,1-DCE, TCE, and vinyl chloride) as the contaminants of potential concern for exposures resulting from volatilization of groundwater contaminants into indoor air spaces within the study area.

Two routes of exposure were evaluated quantitatively in the human health risk assessment: indoor air and surface water in Ferry Creek. Potential future human exposures through consumption of oysters were evaluated qualitatively in this HHRA. Current human exposures were evaluated quantitatively in the OU3 Area II RI (TtNUS 2000b). See Table 6-1 for a summary of the potential risks that could result from exposure to indoor air or future surface water exposures based on indoor air data and modeled surface water concentrations.

Indoor Air

Residential. Based on indoor air data evaluated for residential inhalation exposures, non-carcinogenic and carcinogenic risks have been identified for the combined adult and child frequent resident. Non-cancer hazard indices (HI) that were calculated are in excess of unity for residential exposures to indoor air. 1,2,4-trimethylbenzene, 1,2-dibromoethane, 1,3,5-trimethyl benzene, 1,4-dichlorobenzene, benzene, chloroform, xylenes, and TCE were the main contributors to the hazard index. Adverse non-carcinogenic health effects are possible from exposure to each of these contaminants in indoor air. The cancer risk estimates for lifetime residential exposures to indoor air exceed the EPA cancer risk range (10^{-4} to 10^{-6}). The major contributors to cancer risk at the site are 1,1,2,2-tetrachloroethane, 1,3-butadiene, benzyl chloride, methylene chloride, and TCE. Cancer risks for these and several other volatile organic contaminants are greater than the CTDEP target risk level of 10^{-6} for single contaminants.

Residential inhalation exposures resulting from volatilization of groundwater contaminants into indoor air spaces were also evaluated through an alternative method. Indoor air exposures to the three groundwater contaminants with exceedances of Connecticut groundwater volatilization criteria (1,1-DCE, TCE, and vinyl chloride) were evaluated using the Johnson and Ettinger Model to estimate indoor air concentrations from groundwater data. The predicted indoor air hazard index from exposures to these modeled indoor air concentrations falls between the RME and CTE hazard indices estimated based on actual indoor air concentrations. Adverse non-carcinogenic health effects are possible from exposure to TCE and 1,1-DCE. Cancer risks from exposures to these modeled indoor air concentrations are greater than cancer risks estimated based on actual indoor air concentrations. TCE was the main contributor to cancer risk estimates exceeding the EPA cancer risk range (10^{-4} to 10^{-6}). Both vinyl chloride and TCE exceed the CTDEP target risk level of 10^{-6} for single contaminants.

Industrial/Commercial. Based on indoor air data evaluated for industrial/commercial inhalation exposures, non-carcinogenic and carcinogenic risks have been identified for the industrial/commercial worker. Non-cancer hazard indices (HI) that were calculated are in excess of unity for industrial/commercial exposures to indoor air. 1,2,4-Trimethylbenzene, 1,2-dibromoethane, 1,4-dichlorobenzene, and xylenes were the main contributors to the hazard index. Adverse non-carcinogenic health effects are possible from exposure to each of these contaminants in indoor air. The cancer risk estimates for industrial/commercial exposures to indoor air exceed the EPA cancer risk range (10^{-4} to 10^{-6}). The major contributors to cancer risk at the site are 1,1,2,2-tetrachloroethane, 1,3-butadiene, and TCE. Cancer risks for these and several other volatile organic contaminants are greater than the CTDEP target risk level of 10^{-6} for single contaminants.

Industrial/commercial inhalation exposures resulting from volatilization of groundwater contaminants into indoor air spaces were also evaluated through an alternative method. Indoor air exposures to the three groundwater contaminants with exceedances of Connecticut groundwater volatilization criteria (1,1-DCE, TCE, and vinyl chloride) were evaluated using the Johnson and Ettinger Model to estimate indoor air concentrations from groundwater data. The predicted indoor air hazard index from exposures to these modeled indoor air concentrations falls between the RME and CTE hazard indices estimated based on actual indoor air concentrations. Adverse non-carcinogenic health effects are possible from exposure to TCE and 1,1-DCE. Cancer risks from exposures to these modeled indoor air concentrations are greater than cancer risks estimated based on actual indoor air concentrations. TCE was the main contributor to cancer risk estimates exceeding the EPA cancer risk range (10^{-4} to 10^{-6}). Both vinyl chloride and TCE exceed the CTDEP target risk level of 10^{-6} for single contaminants.

Summary. The indoor air concentration model and indoor air sampling were used to estimate risks from indoor air exposures of residents and industrial/commercial workers to groundwater contaminants volatilizing into indoor air spaces of homes and commercial properties. Because of the variability of sampling and the large number of buildings within the indoor air area of interest, no single home or commercial property was evaluated. Rather, the model and sampling were used to estimate risks for a hypothetical home or commercial property within the indoor air area of interest. The area of interest at which these risk estimates may be applicable can be defined as the area of the groundwater plume where shallow groundwater concentrations of the greatest risk driver exceed the Connecticut groundwater volatilization criteria. TCE has been

identified as the greatest risk driver through the modeled indoor air evaluation. The proposed Connecticut groundwater volatilization criteria for residential scenarios is 27 µg/L. The proposed Connecticut groundwater volatilization criteria for industrial worker scenarios is 67 µg/L. Figure 4-20 delineates the estimated area of the plume where shallow groundwater concentrations of TCE are expected to exceed these criteria. Residential risk estimates may be applicable to residential properties within the area delineated by the Connecticut groundwater volatilization criteria for residential scenarios of 27 µg/L. These residential properties have had sub-slab ventilation systems installed during 2003 and 2004. Industrial/commercial worker risk estimates may be applicable to commercial properties within the area delineated by the Connecticut groundwater volatilization criteria for industrial worker scenarios of 67 µg/L.

Surface Water

Based on predicted models of groundwater contamination reaching Ferry Creek, surface water exposures are not expected to present a risk for future adult and child recreational users of Ferry Creek. Hazard indices (HI) are less than unity for future adult and child recreational exposures (wading) at Ferry Creek indicating that adverse non-carcinogenic health effects are not anticipated. The cancer risk estimates for future recreational users exposed to surface water in Ferry Creek do not exceed the EPA cancer risk range (10^{-4} to 10^{-6}) or the CTDEP target total risk level of 10^{-5} for multiple contaminants, indicating that increased carcinogenic health effects are not anticipated.

Oyster Tissue

Future oyster tissue concentrations of arsenic and zinc were estimated from modeled future surface water concentrations in Ferry Creek and the Housatonic River. The predicted future oyster tissue concentrations were compared to the EPA Region III Risk-Based Concentrations for fish ingestion. Based on this comparison, it is possible that future risks from human consumption of oysters may be higher than 10^{-6}.

7.0 ECOLOGICAL RISK ASSESSMENT

A screening-level ecological risk assessment was conducted to evaluate potential risks to ecological receptors in Ferry Creek from site-related contaminants in groundwater. The risk assessment followed Steps 1 and 2 of EPA's *Ecological Risk Assessment Guidance for Superfund* (EPA 1997a). Sections 7.1 through 7.5 below comprise Step 1 (Screening-Level Problem Formulation and Ecological Effects Evaluation) of EPA's risk assessment process, while Sections 7.6 and 7.7 comprise Step 2 (Screening-Level Exposure Estimate and Risk Calculation).

7.1 Environmental Setting

This section presents the ecological characteristics of the study area, describes water and sediment conditions, and identifies the ecological receptors that are potentially exposed to site-related groundwater contaminants. A comprehensive discussion of the nature and extent of groundwater contamination is presented in Section 4.0.

7.1.1 Study Area Description

The main focus of the OU2 ecological risk assessment is the Ferry Creek area between I-95 and the control structure at the Broad Street Bridge (Figure 2-1). This area was the subject of the *OU3 Area I – Ferry Creek RI* (TtNUS1999a). The portion of Ferry Creek downstream of the former Raymark Facility passes through what was once an extensive salt marsh bordering the Housatonic River (B&RE 1998). Much of the marsh has been filled for commercial, industrial, and residential development. The eastern bend of Ferry Creek was displaced by filling, and downstream of this bend, the banks are steep and topped by a berm of fill material. Marsh remnants near the creek are dominated by *Phragmites australis* (a reed that is typically associated with physical or hydrological disturbances in tidal marshes). Upland vegetation along Ferry Creek consists of small trees, shrubs, and coarse herbs.

7.1.2 Water and Sediment Conditions

All of Ferry Creek within the study area is tidally influenced. At low tide the creek becomes very shallow, fed only by freshwater flow from Long Brook and from groundwater seeps. Salinity

measurements in Ferry Creek range up to approximately 18 parts per thousand, similar to measurements in the Housatonic (NOAA 1998). The average salinity of seawater is 35 parts per thousand.

Dissolved oxygen measurements in Ferry Creek ranged from 4.2 to 8.2 mg/L in August 1995, at temperatures of 24.1°C and 26.7°C, respectively (NOAA 1998). The pH ranged from 5.5 to 7.93. These values are typical for salt marshes receiving fresh water input.

7.1.3 Habitats and Potentially Exposed Receptor Groups

The majority of the study area has been disturbed by commercial and residential development, including paving, building, and dredging activities. This has impacted the wetland areas and associated habitats. *Phragmites* is dominant in most remaining wetlands, and biotic diversity is low.

At least 53 species of fish and 11 invertebrate species are expected to use the Housatonic River for spawning, adult forage, or as a nursery ground for juveniles (NOAA 1998). Recreational fish and crustacean species include Atlantic menhaden, black sea bass, bluefish, four species of flounder, American eel, striped bass, white perch, and the blue crab. The American eel and the eastern oyster are commercially harvested in this area. A commercial larval bed for eastern oyster cultivation is located in the Housatonic River near the mouth of Ferry Creek.

Little information is available on wildlife use of the area around Ferry Creek. Black-crowned night herons and red-winged blackbirds have been observed near Ferry Creek and geese, swans, and shore birds are common on the lower Housatonic River. The Atlantic sturgeon might occur in the vicinity of Ferry Creek, and bald eagles and peregrine falcons may use the area while in transit. No other threatened or endangered species are known from the study area (NOAA 1998).

7.2 Exposure Pathways

Ecological receptors are often exposed to contaminants through several routes. This evaluation is focused on the groundwater pathway, so the main concern is exposure of aquatic

organisms to contaminants from direct contact with sediment pore water and overlying surface water. Because contaminants in water and sediment may become entrained in the food web, another exposure route is ingestion of food, water, and sediment. This exposure route has been evaluated in previous studies (NOAA 1998), and was not evaluated in this report.

7.3 Chemicals of Potential Concern

As explained in Section 7.6, ecological risk was evaluated for two scenarios: recent exposure and future exposure. Ecological chemicals of potential concern (COPCs) in the recent exposure scenario are shown in Tables 7-1 and 7-2. The COPCs in Table 7-1 are those previously identified in *OU3 Ferry Creek – Area I Remedial Investigation Report* (TtNUS 1999a). Table 7-2 presents analytes in the seepage study (see Sections 2.10.6 and 2.12.7). Chemicals shown in Table 7-2 with concentrations that exceeded ecological screening values are COPCs.

The future exposure scenario was evaluated through modeled surface water concentrations. The fate and transport of contaminants in the OU2 groundwater system was modeled using analytical solutions to advection dispersion equations. A discussion of the modeling of the migration of several VOCs and inorganics to generate reasonable worst case concentrations for those contaminants predicted to enter Ferry Creek or the Housatonic River at a single point in the river bed is presented in detail in Section 5.0. Contaminants to be modeled were selected based on three criteria: 1) does the solute have the potential to adversely effect human health or the aquatic biological community if it enters one of these surface water bodies, 2) could a source of the contaminant be identified, and 3) does the scientific literature contain sufficient information on the behavior of the contaminant to model its migration. Ecological COPCs in the future exposure scenario consisted of arsenic, cadmium, copper, lead, nickel, zinc, 1,1-DCE, and TCE.

Several groundwater COPCs selected on the basis of concentrations greater than CT Surface Water Protection Criteria (SWPC) or Ambient Water Quality Criteria (AWQC) were not modeled because of infrequent detections or exceedances of selection criteria, because of their absence in shallow groundwater, because a source could not be identified, because they do not appear to be derived from Raymark waste, or because the scientific literature does not contain sufficient information on the behavior of the contaminant to model its migration.

Acenaphthylene, benzo(b)fluoranthene, benzo(k)fluoranthene, phenanthrene, 4,4'-DDT, dieldrin, endosulfan I, endosulfan II, endrin, gamma-BHC, gamma-chlordane, heptachlor, and heptachlor epoxide were detected infrequently. Mercury, selenium, silver, and thallium were detected infrequently above selection criteria and most of the detected concentrations exceeding criteria were just slightly greater than the selection criteria. 1,1,1-Trichloroethane and beryllium were only detected at high concentrations in deep groundwater. Tetrachloroethene exceeds criteria only at the source. Benzene does not appear to be derived from Raymark waste. Scientific literature does not contain sufficient information on the behavior of cyanide. The model cannot account for the radically different behaviors of chromium III and chromium VI in the absence of speciation data.

Among the ecological COPCs modeled, cadmium, copper, lead, and nickel were not evaluated in the ecological risk assessment because they are not expected to arrive at either Ferry Creek or the Housatonic River within the foreseeable future.

The model indicates that trichloroethene is migrating toward the Housatonic River, rather than toward Ferry Creek. It is expected that maximum predicted concentrations of trichloroethene have already reached the river. Further sampling data in the Housatonic River, are needed to estimate current or future surface water concentrations.

Surface water concentrations in Ferry Creek of the remaining ecological COPCs for surface water protection, 1,1-dichloroethene, arsenic, and zinc, were estimated by applying a dilution factor to account for mixing with creek waters to the modeled concentrations. A dilution factor of 0.1 was selected, based on a review of Ferry Creek surface water concentrations of VOCs and groundwater seep concentrations. The resulting modeled surface water concentrations are presented in Table 5-2. A detailed discussion of contaminant fate and transport was presented in Section 5.0.

7.4 Selection of Ecological Endpoints

"Ecological endpoints" refers to setting goals within the risk assessment and addressing how the goals will be met. Goals for the assessment, or assessment endpoints, are the protection of aquatic life (the indigenous benthic community, oysters, swimming invertebrates, and fish) that may be exposed to contaminants in the study area. The goals are met by taking

measurements that relate to the assessment endpoints. This investigation assessed the risk to aquatic life by comparing estimated and measured chemical concentrations in water to surface water toxicity threshold values. By assessing risk in this way, it is assumed that sediment-dwellers are exposed through direct contact with pore water. In essence, the discharging groundwater becomes sediment pore water as it moves through the sediment and discharges into surface water.

7.5 Ecological Effects Evaluation

Potential adverse effects of contamination were evaluated by comparing measured and modeled surface water concentrations to toxicity threshold values for aquatic receptors. These surface water toxicity threshold values are referred to as ecological screening values. Saltwater ecological screening values were preferentially used since the salinity in Ferry Creek ranges up to approximately 18 parts per thousand. Ecological screening values were selected using the following hierarchy;

- Federal chronic AWQC (EPA 2002b) were preferentially used as ecological screening values. CT water quality criteria (WQC) (CTDEP 2002) were identical to the federal criteria for the chemicals of interest in this report.

- If federal chronic AWQC were not available, *ECO Update: Ecotox Thresholds* (EPA 1996a) was used as a source of ecological screening values.

- Tier II secondary chronic values calculated by Oak Ridge National Laboratory (ORNL) (ORNL 1996) were used as the third choice in the selection hierarchy. The ORNL Tier II secondary chronic values were calculated using EPA's guidance for the Great Lakes Water Quality Initiative, and are designed to be conservatively low estimates of chronic water quality criteria (ORNL 1996). Other sources were used as necessary when ecological screening values were not available in the above three sources. All ecological screening values and their sources are noted in Tables 7-1, 7-2, and 7-3. These three tables also present a hazard quotient for each chemical evaluated, which is defined as the ratio of the maximum chemical concentration to the ecological screening value.

7.6 Exposure Estimate

Exposure to site-related COPCs was evaluated for two scenarios: recent exposure and future exposure. Data from surface water samples collected in Ferry Creek (TtNUS 1999a) and data from the seepage study (see Sections 2.10.6 and 2.12.7) were used to estimate recent exposure. Future contaminant concentrations in groundwater as it discharges into Ferry Creek or the Housatonic River were estimated by groundwater modeling (Section 5.0). For purposes of this evaluation, this future discharge is expected to be from the groundwater seepage points described in Section 3.4.2.6. Future concentrations in Ferry Creek were estimated using dilution factors derived by comparing chemical concentrations in seepage samples to nearby surface water concentrations sampled during the seepage study. Comparisons of mean organic contaminant concentrations in measured seep samples (1,1-DCE: 58 µg/L; TCE: 107 µg/L) to mean concentrations in co-located surface water samples collected in Ferry Creek upstream of Broad Street (1,1-DCE: 4.7 µg/L; TCE: 9.7 µg/L) indicate a dilution factor of about 10. Neither 1,1-DCE nor TCE were detected in surface water collected immediately downstream of the Broad Street Bridge.

7.7 Risk Calculation

Risk was characterized at the point of groundwater seepage and in the overlying surface water of Ferry Creek. This risk is evaluated for aquatic life for recent and future exposure.

7.7.1 Recent Exposure

Data were collected during surface water sampling in Ferry Creek during the mid-1990s for the full suite of analytes and the 1999 seep and surface water sampling for VOCs and SVOCs.

Previously Evaluated Data

The samples taken in the mid-1990s were evaluated in the *OU3 Ferry Creek - Area I RI Report* (TtNUS 1999a). As described in that Report, arsenic, chromium, copper, lead, mercury, nickel, zinc, Aroclor-1262, and 4,4'-DDT had values that exceeded Federal Ambient Water Quality Criteria (FAWQC). The results are summarized in Table 7-1. Note that the maximum detected concentration of chromium (12.3 µg/L) is less than the ecological screening value (50 µg/L).

The previous report (TtNUS 1999a) included chromium as posing potential risk because concentrations in three samples (11.1 to12.3 µg/L) slightly exceeded the ecological screening value for freshwater (11 µg/L). As stated in Section 7.5, however, saltwater ecological screening values were used in this Report since the salinity in Ferry Creek ranges up to approximately 18 parts per thousand.

With the highest hazard quotient (44.5) and frequency with which FAWQC (7 out of 14) are exceeded, the potential risk was greatest for copper in surface water. Maximum hazard quotients for DDT (26.0) and lead (14.6) were also elevated (Table 7-1). Concentrations of DDT exceeded its ecological screening value in one sample, while lead concentrations exceeded the ecological screening value in two samples. Concentrations of other COPCs in Table 7-1 were only slightly greater than their respective ecological screening values, or infrequently exceeded their ecological screening values.

Seepage Study

Data from the seepage study are summarized in Table 7-2. The seepage study data were obtained from 14 seep samples in Ferry Creek and four surface water samples in Ferry Creek. Concentrations of all analytes in the four surface water samples were less than ecological screening values (Table 7-2). Analytes with concentrations greater than ecological screening values in seep samples consisted of 1,1,1-trichloroethane, 1,1-dichloroethane, 1,1-DCE, 1,2-dichloroethene, carbon disulfide, and TCE (Table 7-2).

Concentrations of 1,1,1-trichloroethane, 1,1-DCE, 1,2-dichloroethene, and TCE exceeded their ecological screening values only in seep sample GW-02. Thus, the data indicate the potential risk from these four compounds is limited to the vicinity of sample GW-02. Similarly, the potential risk from 1,1-dichloroethane is limited to the vicinity of seep samples GW-02 and GW-03, the only samples where concentrations of this compound exceeded its ecological screening value. Since the maximum hazard quotients for these five compounds were relatively low, concentrations exceeded ecological screening values in only one or two samples, and concentrations were well below the ecological screening value in nearby surface water samples, the potential risk from these five compounds appears to be negligible or minor at worst.

Carbon disulfide is a natural product of anaerobic biodegradation and naturally occurs in marsh sediments. Carbon disulfide was detected in only one seep sample, and the concentration in this sample (1 µg/L) only slightly exceeded the ecological screening value (0.92 µg/L), with a hazard quotient of 1.1. The singe detected concentration was less than the United States Geological Survey's freshwater chronic guideline for carbon disulfide (2 µg/L) (USGS 1997). However, the detection limit was 10 µg/L in 12 of 13 seep samples in which carbon disulfide was not detected, as well as in the four non-detect surface water samples. The detection limit was 50 µg/L in one seep sample (GW-02). Thus, there is some uncertainty associated with the evaluation of carbon disulfide, since the detection limit exceeded both available ecological guidelines. Overall, however, the detection limits are not excessively high (with the exception of sample GW-02), and further evaluation of carbon disulfide is probably not warranted.

With the exception of carbon disulfide, the detection limits for all analytes in Table 7-2 were less than their respective ecological screening values.

7.7.2 Future Exposure

As discussed in Section 5.0, arsenic, zinc, and 1,1-DCE in groundwater are expected to eventually discharge into Ferry Creek, while TCE is expected to eventually discharge directly into the Housatonic River. Future risk from these chemicals is based on groundwater modeling and rough approximations of subsequent dilution in surface water. Maximum hazard quotients based on predicted future seep concentrations were highest for 1,1-DCE (20), but hazard quotients for arsenic, zinc, and TCE also exceeded 1.0 (Table 7-3). After applying a dilution factor of 10 based on Ferry Creek data, estimated concentrations of arsenic and zinc in Ferry Creek are less than their respective ecological screening values, while the estimated concentration of 1,1,-DCE exceeds its most conservative ecological screening value, with a hazard quotient of 2.0 (Table 7-3). The estimated concentration of 1,1,-DCE is well below EPA Region IV's chronic ecological screening value for saltwater (2240 µg/L) (EPA 2001a). Ecological screening values have not been established by EPA Region I.

Seepage meter data and paired surface water data indicate a dilution factor of approximately 1:10 in Ferry Creek. Data are not available with which dilution factors can be estimated for the Housatonic River. Thus, the expected concentration of TCE after discharge into the river is very uncertain. However, the Housatonic River is much larger than Ferry Creek, so its ability to

dilute concentrations of contaminants seeping into it would probably be orders of magnitude greater than for Ferry Creek. Conservatively applying a dilution factor of 1:10 based on the Ferry Creek data would result in an expected TCE concentration of 570 µg/L after dilution, with a resulting hazard quotient of 1.6. This suggests minimal risk in the Housatonic River, since the actual dilution factor would probably be much greater than 1:10.

Expected future concentrations of arsenic (19 µg/L) and zinc (75 µg/L) in Ferry Creek are considerably less than recent actual measured maximum concentrations of arsenic (75.1 µg/L) and zinc (126 µg/L) in the creek (Table 7-1). Thus, risk from these two metals is not expected to substantially increase as groundwater discharges into Ferry Creek. Future concentrations of copper and lead, the two metals with the highest hazard quotients in Table 7-1, could not be modeled, but maximum groundwater concentrations of these metals are not expected to reach Ferry Creek for approximately 200 years (see Table 5-2).

7.7.3 Uncertainty Analysis

Uncertainties are associated with most steps of an ecological risk assessment. This section describes some of the sources of uncertainty that are associated with this assessment.

Concentrations of chemicals recently detected in Ferry Creek (Section 7.7.1) might result from multiple sources, rather than solely from the former Raymark Facility. In addition, surface water grab samples taken at one point in time introduce some uncertainty regarding sample variability.

The assumptions made in the groundwater modeling may not reflect actual conditions in the field, and thus, uncertainty is introduced into the exposure characterization. Some of these assumptions, such as sustained contaminant sources, may be unrealistically conservative. The lack of information on flow rates in Ferry Creek and the Housatonic River makes dilution estimates very uncertain. Also, the size of the presumed groundwater seepage areas in Ferry Creek and the Housatonic River is not known, and thus, the size of the areas in the creek and river that would be exposed to groundwater seepage at the predicted concentrations is uncertain. The groundwater modeling predicted that maximum groundwater concentrations of TCE would reach the Housatonic River and 1,1-DCE would reach Ferry Creek by 1997, and will remain at those concentrations until the source is depleted. However, actual surface water data

from the expected seepage location are not available, so the accuracy of the predicted concentrations of these two organic compounds is uncertain.

There is uncertainty in the values used to characterize ecological effects, especially for TCE and 1,1-DCE. No federal or Connecticut water quality criteria were available for 1,1-DCE or TCE. The highest hazard quotients for TCE and 1,1-DCE in Table 7-3 (future exposure) are based on Tier II values for freshwater, and are designed to be conservatively low estimates of the chronic water quality criteria. Tier II values for saltwater are not available, but current EPA guidance is to use freshwater Tier II thresholds in a saltwater environment (EPA 1996a). ORNL (1996) reviewed toxicity data from numerous sources and reported the following lowest chronic values for 1,1-DCE: 2,800 µg/L (fish), 4,720 µg/L (daphnids) and 798,000 µg/L (aquatic plants). The lowest chronic values for TCE were 11,100 µg/L (fish) and 7,257 µg/L (daphnids) (ORNL 1996). The predicted future surface water concentrations of 1,1-DCE and TCE at the modeled groundwater seep location and in Ferry Creek or the Housatonic River are well below these values. Similarly, measured concentrations of 1,1-DCE and TCE in Ferry Creek surface water samples collected during the seepage study (Recent Exposure) are well below these values.

There is uncertainty associated with the evaluation of carbon disulfide in the seepage study, since the detection limits were greater than ecological guidelines. However, the detection limits are not excessively high (with the exception of one sample).

Another uncertainty regarding ecological effects is that the toxicity of chemical mixtures is not well understood. All the toxicity information used in this Report for evaluating risk to ecological receptors is for individual chemicals. Chemical mixtures can affect the organisms very differently than the individual chemicals because of synergistic or antagonistic effects.

Only aquatic receptors exposed directly to contaminants in water were evaluated. Previous assessments done for Ferry Creek have evaluated risk to terrestrial wildlife (e.g., birds and mammals) via the food web. Although some terrestrial wildlife exposure to contaminants introduced by groundwater is likely, much of the exposure in the area is from contaminated fill material. Therefore, it is not known with certainty how increasing contaminant loading from groundwater would affect doses to terrestrial wildlife receptors. Risk assessments have indicated that sediment-dwelling organisms are at more risk than terrestrial wildlife (TtNUS 1999a).

7.8 Conclusions

This investigation assessed the risk to aquatic life (the indigenous benthic community, oysters, swimming invertebrates, and fish) by comparing estimated measured chemical concentrations in water to surface water toxicity threshold values, or ecological screening values. By assessing risk in this way, it is assumed that sediment-dwellers are exposed through direct contact with pore water. Sampling performed in the mid 1990's in Ferry Creek indicates a potential risk to aquatic organisms from copper, lead, and 4,4'-DDT. Maximum concentrations of six volatile organic compounds exceeded ecological screening values in some 1999 seep samples, but hazard quotients were relatively low.

The maximum predicted future concentration of 1,1-DCE at the point of discharge exceeds the most conservative ecological screening value for this compound, but was less than alternate screening values. The maximum predicted future concentration of TCE at the point of discharge exceeds its ecological screening value. With only minor dilution, predicted future concentrations of arsenic, zinc, 1,1-DCE and TCE pose negligible risk to aquatic receptors. Site-related groundwater concentrations of copper and lead are not expected to reach Ferry Creek for approximately 200 years.

Surface water sampling at seepage points and in Ferry Creek would reduce the uncertainty regarding current contamination; the "Recent Exposure" scenario evaluated in this report was based on samples collected in the 1990s.

8.0 SUMMARY AND CONCLUSIONS

The OU2 study area is an urban area consisting of 500 acres where groundwater could potentially have been impacted by wastes from the former Raymark Facility. This study area, most of which is located within the tidally influenced 100-year floodplain of the Housatonic River Basin, is composed of approximately 240 acres under cover (buildings, pavement), 10 acres of open water, and 250 acres of undeveloped properties (mainly grassed or trees, gravel and wetlands).

This OU2 RI report summarizes the activities performed under various investigation programs by federal, state, and private contractors. Groundwater, soil gas, and indoor air sampling, and geologic investigations have been performed by numerous entities over a 9-year period (1994-2003). The environmental media evaluated for this OU2 study area are groundwater, soil gas, and indoor air. Sediments, soils, and surface water have been investigated and reported under other operable units and are not discussed in detail within this RI Report.

The objectives of this OU2 RI Report are to:

- Serve as the mechanism for compiling and evaluating the available data needed to characterize the OU2 study area groundwater conditions;

- Characterize the nature and extent of contamination in the groundwater from both the former Raymark Facility and properties impacted by waste from the former Raymark Facility located within the OU2 study area as shown on Figure 1-2;

- Assess the risks to human health and evaluate the ecological receptors within the study area; and

- Serve as the data resource for developing, screening, and evaluating a potential range of alternatives that address the groundwater contamination within the OU2 study area.

As detailed in Section 1.0, the majority of the OU2 study area is located beneath and downgradient of the former Raymark Facility. This study area was targeted for investigation because groundwater beneath and downgradient of the Facility has been impacted by the

Raymark Facility. Raymark manufacturing waste was disposed of on the former Raymark Facility property and on other properties within the study area. These properties were impacted by the Raymark waste through either direct disposal of waste or deposition of Raymark-related contaminants via surface water flow, storm runoff, or other means. Spills at the Facility have contaminated groundwater beneath and downgradient of the Facility.

8.1 Nature and Extent of Contamination Summary

This section summarizes the known nature and extent of groundwater contamination, detailed in Section 4.0.

The Raymark waste contains volatile and semi-volatile organic compounds (VOCs and SVOCs), pesticides, polychlorinated biphenyls (PCBs), dioxins and furans, metals (lead, copper, and barium), and asbestos. This waste was disposed of as fill material on properties located both within and outside the OU2 study area. Additionally, process water and runoff from the former Raymark Facility containing these contaminants were directly discharged to Ferry Creek. Based on these practices and spills that occurred on the former Raymark Facility, groundwater throughout the OU2 study area has been contaminated from the former Raymark Facility discharges, and from the properties that received Raymark waste as fill. This RI Report encompasses the groundwater beneath the majority of properties or areas that are part of several RIs (OU1, OU3, OU4, OU5, and OU6) that have been investigated to date. Areas that may have received waste outside of the OU2 study area are not included in this groundwater RI.

Contaminated areas may have impacted the groundwater either by leaching from the soils, or direct discharging of waste from activities at the former Raymark Facility. This RI Report included samples from wells installed for this investigation, previously existing wells on commercial properties in the study area, and direct-push locations. The focus of this RI Report has been on identifying contaminants that are attributable to the former Raymark Facility, however, the study area is highly industrialized with many other potential sources of hazardous waste contamination. The investigation of Ferry Creek was limited to surface water, groundwater, and sediment sampling. No evaluation of surface water flow rates or other characteristics were performed for either Ferry Creek or the Housatonic River. Samples in the Housatonic River were limited to sediments, surface water, and biota collected at the

confluence of the river and Ferry Creek. Sediment samples were also collected from wetlands in the study area under OU3 and OU5.

8.1.1 Nature and Extent of Groundwater Contamination

The groundwater contamination has been investigated in numerous field events. These investigations have revealed that contaminants of Raymark waste are present in the groundwater within both the overburden and bedrock aquifers at various concentrations, within the entire OU2 study area.

8.1.1.1 Volatile Organic Compounds (VOCs)

VOCs are an identified groundwater contaminant within the OU2 study area. VOCs are commonly used in industrial processes; they have been identified as contaminants or by-products of contaminants used in the former Raymark Facility manufacturing processes, and may have been used in other businesses within the study area.

VOCs are the most common and widely detected chemical class of contaminants in the groundwater that can be attributed to man-made activities. Refer to Figure 5-1 for potential groundwater contamination sources. Two hundred and thirty-five groundwater samples were collected and analyzed for VOCs during the 2002/2003 sampling effort. Ten VOCs exceeded Connecticut's criteria for surface water protection or residential volatilization in at least one groundwater sample collected during the winter of 2002/2003. Eight of these VOCs were most abundant beneath the former Raymark Facility (OU1). 1,1,1-TCA and 1,1-DCE were most abundant near the eastern edge of OU1 in the deep overburden and bedrock beneath former Lagoon 4. At this location, both contaminants were present at concentrations greater than 1 percent of their respective pure phase solubilities, suggesting the occurrence of a nearby DNAPL source. Localized areas of high toluene and chlorobenzene concentrations were restricted to the northern end of OU1 near a 1984 toluene spill. Concentrations of toluene and chlorobenzene were high enough to infer the presence of NAPL. The highest concentrations of TCE, PCE, and 1,2-DCA were found beneath the former acid neutralization pits at the southwestern end of OU1. Maximum concentrations of vinyl chloride were detected a short distance downgradient from this confirmed DNAPL source.

The known extent of VOC contamination for the ten identified contaminants that exceed Connecticut surface water protection criteria (SWPC) and/or residential volatilization criteria (RVC) in groundwater are:

Benzene – The highest concentrations of benzene were found in shallow off-site wells just south of the former Raymark Facility. Benzene has not been detected in the groundwater directly beneath the former Raymark Facility at levels that exceed the SWPC or RVC. Contamination appears to be primarily located in the middle of the study area just south of the Facility. See Figure 4-1 for details. Although benzene was observed in the groundwater at 73 locations throughout the OU2 study area, measured concentrations outside the localized area of high concentrations at Barnum Avenue were below pertinent regulatory criteria. The absence of high recent or historic benzene concentrations in samples collected from upgradient former Raymark Facility wells suggests the high concentrations at Barnum Avenue did not originate on the Facility.

Chlorobenzene – Chlorobenzene was detected in 133 of the 265 OU2 study area monitoring wells. High concentrations were located along the northeast border of the former Raymark Facility and in wells positioned 190 feet further northeast. See Figure 4-2 for details.

Toluene – Toluene exceeded the Connecticut RVC in two localized areas. One is located in the northeast portion of the study area approximately 300 feet southeast of the 1984 toluene spill at the former Raymark Facility. The second localized area is beneath the acid neutralization pits. See Figure 4-3 for details.

1,1,1-TCA – 1,1,1-TCA was detected in 128 samples. Elevated levels of this contaminant are located both in the overburden and bedrock. Most of the contamination appears to be moving through the deep overburden and bedrock associated with the southern bedrock valley. 1,1,1-TCA contamination resides in the deeper portions of the OU2 groundwater study area. The highest detected levels of 1,1,1-TCA are located beneath Lagoon 4 just southeast of the 1987 1,1,1-TCA spill on the Raymark Facility. See Figure 4-4 for details.

1,1-DCE - Elevated levels of this contaminant are located both in the overburden and bedrock, usually in association with 1,1,1-TCA. It also resides deeper in the OU2 groundwater system. Most of the contamination appears to be moving through the deep overburden and bedrock

associated with the southern bedrock valley. The highest detected level of 1,1-DCE is located beneath Lagoon 4 just southeast of the 1987 1,1,1-TCA spill on the former Raymark Facility. See Figure 4-5 for details.

TCE - Although TCE is a widespread contaminant within the study area, nearly all of the localized areas of high TCE concentrations are located on the Raymark Facility property or down gradient from it. TCE concentrations are greatest at intermediate depths and can be high in both the overburden and the bedrock. The maximum concentration of TCE was detected in the groundwater beneath the acid neutralization pits at the southwest end of the former Raymark Facility. See Figure 4-6 for details. The primary source of TCE in the OU2 groundwater system appears to be the TCE-rich DNAPL that has been observed in the overburden wells installed beneath the former acid neutralization pits.

Vinyl Chloride - Vinyl chloride exceeded Connecticut RVC in 96 of the 116 groundwater samples in which it was detected. The highest concentrations were detected at shallow depths beneath the former Raymark Facility, approximately 160 feet downgradient of the acid pits. See Figure 4-7 for details.

1,1-DCP - 1,1-DCP was detected at two monitoring locations, and both were in the shallow groundwater beneath 375 Barnum Avenue. Because of the limited number of detections, no plume maps were created.

1,2-DCA - The maximum concentration of 1,2-DCA was detected in the groundwater beneath the former acid neutralization pits at the southwest end of the Raymark Facility. Although 1,2-DCA was detected in 39 of the OU2 groundwater samples, concentrations only exceeded the Connecticut RVC in the groundwater near the former acid neutralization pits. Since 1,2-DCA was not widely detected, no plume maps are included in this Report.

PCE – PCE was detected in 81 samples. As with 1,2-DCA, the highest groundwater concentration of PCE was found beneath the former acid neutralization pits. PCE concentrations only exceeded Connecticut RVC standards in the immediate vicinity of these pits (21 µg/L). Since PCE was not widely detected, no plume maps are included in this Report.

NAPLs have not been observed outside the former Raymark Facility. However, dissolved concentrations of TCE, 1,1,1-TCA, and 1,1-DCE exceeded 1 percent saturation in off-site groundwater samples collected in the winter of 2002/2003. TCE was present at slightly less than 2 percent of its pure phase solubility in MW-514D, located about 800 feet southeast of the southern end of the facility. 1,1,1-TCA and 1,1-DCE were present at more than 1 percent saturation in samples collected from MW-211D and MW-211B. The MW-211 well cluster lies 650 feet downgradient from the suspected DNAPL source beneath Lagoon 4, and it is within 200 feet of the facility. It is not clear whether: 1) there is a separate DNAPL source near the MW-211 well cluster; 2) DNAPL extends from beneath Lagoon 4 to the vicinity of these wells; or 3) the DNAPL only occurs in the deep overburden and bedrock near Lagoon 4. The fate and transport modeling described in Section 5 demonstrates that it is possible the 1,1,1-TCA and 1,1-DCE at MW-211 originated from a DNAPL source that only occurs near Lagoon 4. The modeling results suggest that groundwater concentrations of 1,1,1-TCA and 1,1-DCE are still very high 650 feet from the Lagoon 4 source, not because there is a separate source near MW-211, but because the plume has not traveled far enough from Lagoon 4 for dilution and degradation to reduce dissolved concentrations to below 1 percent saturation.

8.1.1.2 Semi-Volatile Organic Compounds (SVOCs)

SVOCs are identified as a contaminant within the OU2 study area. Three primary groups of SVOCs were detected in Raymark waste soils within the study area: phenolic compounds, polynuclear aromatic hydrocarbons (PAHs), and phthalates. Many of these compounds are common constituents of various industrial products, used in the manufacture of friction materials (such as those made at Raymark), and are associated with fuels, coal, and petroleum products. Phthalates were used as plasticizers in the manufacture of synthetic products (such as the synthetic resins made at Raymark). Some SVOCs have leached from soils into groundwater.

Two hundred and thirty-six groundwater samples were collected and analyzed for SVOCs during the winter of 2002/2003. SVOCs were detected infrequently and at low concentrations throughout the study area. Only three SVOCs (benzo(b)fluoranthene, acenaphthylene, and phenanthrene) exceeded the Connecticut SWPC, and their distribution within the overburden aquifer is sporadic. Phenanthrene was the only compound detected in the bedrock aquifer at concentrations exceeding the Connecticut SWPC.

8.1.1.3 Pesticides

Pesticides are an identified contaminant in the source areas within the OU2 study area.

Pesticides are assumed to have been used at the Raymark Facility, as indicated by pest control practices common in manufacturing plants. Pesticides were detected in waste that was excavated from residential properties and stored at the Raymark Facility during the emergency removal actions.

There were 235 samples collected and analyzed for pesticides in groundwater. Pesticides were infrequently detected in both the overburden and bedrock groundwater samples, with detection slightly higher in the overburden. Concentrations of dieldrin and heptachlor epoxide exceeded the Connecticut SWPC in both overburden and bedrock aquifers. Concentrations of heptachlor also exceeded the Connecticut SWPC in the overburden aquifer. No pesticides were detected in upgradient overburden or bedrock wells.

8.1.1.4 Polychlorinated Biphenyls (PCBs)

PCBs are an identified contaminant in the OU2 source areas within the study area. PCBs identified within the source areas of the OU2 study area consisted primarily of Aroclor 1262 and Aroclor 1268. PCBs are typically used as plasticizers in the manufacture of brake linings, rubber gaskets, and synthetic resins (which were products used or produced at Raymark).

There were 235 samples collected and analyzed for PCBs in groundwater during the 2002/2003 sampling effort. No PCBs were detected in overburden or bedrock groundwater samples during that sampling. Historically, PCBs were infrequently detected in both the overburden and bedrock groundwater samples, with detection slightly higher in the overburden. Three Aroclors were detected in two historic overburden samples. Aroclor 1254 exceeded the SWPC in one sample from the overburden. For bedrock, one historic sample exceeded the SWPC for Aroclor 1221. No other PCBs were detected in bedrock groundwater samples.

8.1.1.5 Metals

Metals are identified contaminants in the source areas within the OU2 study area. Metals are ubiquitous in groundwater because they occur naturally in the environment. Copper and lead were the most prevalent Raymark-related metals found at concentrations greater than background samples in groundwater. Other metals that were found at concentrations exceeding Connecticut SWPC were arsenic, beryllium, cadmium, mercury, nickel, selenium, silver, thallium, and zinc. These metals are used in fabricating brake and friction products (such as were used at Raymark). All of these metals within the study area can originate from Raymark waste, but other sources of metals exist, particularly for cadmium, nickel, and zinc. Metal contamination was most noticeable at four locations: 1) the shallow groundwater beneath portions of 600 East Broadway exceeded the SWPC for arsenic, copper, lead, mercury, and zinc indicating possible leaching from Raymark waste; 2) the shallow groundwater beneath 540 Longbrook Avenue, which is upgradient from the former Raymark Facility, and the abutting portion of the former Raymark Facility often exceeded the SWPC for cadmium, copper, lead, nickel, and zinc; 3) the overburden and bedrock groundwater surrounding the former Raymark Facility acid neutralization pits surpassed the SWPC for beryllium, cadmium, copper, lead, nickel, and zinc; and 4) the deep overburden and bedrock groundwater associated with a northeast-southwest trending bedrock valley that lies beneath the eastern margin of the former Raymark Facility often exceeded the SWPC for arsenic, beryllium, cadmium, copper, lead, nickel, thallium, and zinc.

Figures 4-8 to 4-15 present the extent of the metals contamination within the study area.

8.1.1.6 Dioxins

Dioxin are an identified contaminant in the source areas within the OU2 study area. Dioxins are not used in manufacturing processes; they are formed during the production of chlorinated compounds (such as pesticides or PCBs) or from incomplete combustion of chlorinated compounds. There is no SWPC for dioxins.

Dioxins were detected in 82 of the 193 samples collected for dioxin. Dioxins are at low concentrations in both overburden and bedrock groundwater samples. Concentrations in overburden ranged from 0.00029 pg/L to 18 pg/L, while concentrations in bedrock ranged from

0.00047 pg/L to 8 pg/L. Concentrations of some samples were higher than the respective upgradient concentrations.

8.1.1.7 Asbestos

Asbestos is an identified contaminant on the source areas within the OU2 study area. Asbestos-containing materials were a primary component of products manufactured at the former Raymark Facility. Asbestos fibers were mixed with phenolic resins to manufacture brake pads, linings, clutches, transmission plates, and gaskets.

Groundwater samples were not analyzed for asbestos within the study area outside of the former Raymark Facility. Previous sample results from the Raymark Facility's consultants are evaluated in the Final RI Report for OU1 (HNUS 1995).

8.1.2 Nature and Extent of Indoor Air and Soil Gas Contamination

Indoor air samples were collected during April 2000, February and March 2001, February and March 2002, July 2002, February 2003, and March 2003. Indoor air samples were collected to evaluate the impact on homes from volatilizing and migrating volatile organic compounds from contaminated groundwater resulting from past practices of the Raymark Industries, Inc. manufacturing and waste handling.

Numerous VOCs were detected in homes at concentrations that exceed human health criteria (EPA Region IX preliminary remediation goals for ambient air). Approximately 30 different VOCs were detected in at least one half of the samples.

Only three VOCs, TCE, vinyl chloride, and 1,1-DCE, were detected above the Connecticut RVC within the groundwater beneath the indoor air area of interest. A closer review of the indoor air sampling results for these three target contaminants reveals the following:

- 1,1 DCE was detected in 42 of 69 samples, with a maximum concentration of 48 ug/m^3.
- TCE was detected in 39 of 69 samples, with a maximum concentration of 52.5 ug/m^3.
- Vinyl chloride was detected in 2 of 69 samples, with a maximum concentration of 0.62 ug/m^3.

These three contaminants were not detected in indoor air at background locations.

Soil gas samples were collected between February and July 2002 from a total of 84 locations, using a geoprobe-based sampling system described in Section 2.12.1. Samples were analyzed for seven target VOCs: 1,1,1-TCA, 1,1-DCE, benzene, chlorobenzene, PCE, toluene, and TCE. VOC concentrations in soil gas tended to increase with depth. 1,1,1-TCA, 1,1-DCE, and TCE were widely detected, and concentrations were highest along portions of Barnum Avenue, Ferry Boulevard, Housatonic Avenue, and Willow Avenue. By contrast, benzene, chlorobenzene, PCE, and toluene were found much less frequently and typically at lower concentrations.

8.2 Contaminant Fate and Transport Summary

Contaminant fate and transport in the environment are controlled by a number of factors including chemical and physical properties of the contaminants, geologic formations, hydrologic conditions, topography, precipitation, and tidal flow.

The groundwater contaminants identified in the nature and extent discussion are associated with the former Raymark Facility. Major pathways of migration within the OU2 study area were wastewater and drainage discharge, leaching of contaminants from waste/fill areas into the groundwater, erosion from the former Raymark Facility to Ferry Creek, and runoff from the Raymark waste/fill areas into Ferry Creek, and then to the Housatonic River. Surface water flowing through the study area (both naturally and tidally) receives contaminated groundwater. This surface water has also eroded the Ferry Creek bank where Raymark waste/fill had been disposed of on properties bordering the creek.

Spills of contaminants and wastewater and drainage discharges from the former Raymark Facility have principally contributed to contamination in the groundwater. While the discharges ceased when the Raymark Facility closed in 1989, spill and disposal areas have remained as continuing sources of contaminants. These areas have continued to leach into the groundwater and will continue to do so for many years. The contaminated waste/fill placed on properties within the OU2 study area also continues to be a potential source of groundwater contamination. The Raymark contaminated waste has resulted in the direct and indirect release of contamination into the groundwater within the study area.

Cadmium, copper, lead, and nickel are migrating very slowly in the OU2 groundwater system. While cadmium, nickel, copper, and lead are not expected to adversely impact local surface waters in the near future, zinc is more mobile. Zinc leaching from the Raymark waste/fill at the 600 East Broadway property is predicted to have already reached Ferry Creek, and maximum concentrations from this source should reach the creek in roughly 25 years. Maximum concentrations of arsenic originating from the 600 East Broadway property are predicted to enter Ferry Creek in approximately 100 years.

The 1,1-DCE emanating from the probable DNAPL source beneath the former Lagoon 4, located at the former Raymark Facility, should already be discharging to Ferry Creek at a maximum sustained concentration of about 500 µg/L. The TCE originating from the confirmed DNAPL source beneath the former acid neutralization pits, located at the former Raymark Facility, should be discharging to the Housatonic River at a sustained concentration as high as 5700 µg/L, depending on the character of the riverbed sediments. Groundwater modeling results indicate the concentrations of 1,1-DCE and TCE entering Ferry Creek and the Housatonic River should remain at their present levels until their sources are contained, destroyed, or depleted. Groundwater concentration data collected between 1997 and 2003 also suggest the concentrations of 1,1-DCE and TCE have reached a steady state throughout the OU2 groundwater system.

The relatively simple mathematical model used in this report to describe fate and transport in a complex system should be viewed as a tool to: 1) provide approximate arrival times and contaminant concentrations; 2) help constrain hypotheses regarding the fate and transport of contaminants in the OU2 groundwater system; and 3) guide future data collection efforts. Despite the limitations inherent in the simple modeling approach taken, the results strongly suggest that: 1) the concentrations of 1,1-DCE and TCE entering Ferry Creek and the Housatonic River will remain at their present levels until their sources are contained, destroyed, or depleted; and 2) the concentrations of all the modeled inorganic contaminants discharging to Ferry Creek are expected to increase over time.

8.3 Risk Assessment Summary

The risk assessment for this RI focused on both human health and ecological risks.

8.3.1 Human Health Risk Assessment

The Human Health Risk Assessment (HHRA) identified VOCs (1,1,2,2-tetrachloroethane, 1,1,2-trichloroethane, 1,1-dichloroethene (1,1-DCE), 1,2,4-trimethylbenzene, 1,2-dibromoethane, 1,2-dichloroethane, 1,3,5-trimethylbenzene, 1,3-butadiene, 1,4-dichlorobenzene, 4-methyl-2pentanone, acetone, acrylonitrile, benzene, benzyl chloride, bromodichloromethane, carbon tetrachloride, chloroethane, chloroform, chloromethane, cis-1,2-dichloroethene, dibromochloromethane, dibromofluoromethane, ethylbenzene, hexane, methyl tert-butyl ether, methylene chloride, tetrachloroethene (PCE), tetrahydofuran, toluene, xylenes, trichloroethene (TCE), vinyl acetate, and vinyl chloride) as the contaminants of potential concern for exposures to indoor air within the study area. These contaminants were selected based on comparison of indoor air data to EPA Region IX Preliminary Remediation Goals for ambient air.

VOCs (1,1-DCE, 1,1,1-trichloroethane, benzene, PCE, and TCE); PAHs (acenaphthylene, benzo(b)fluoranthene, benzo(k)fluoranthene, and phenanthrene); pesticides (gamma-chlordane, dieldrin, heptachlor, and heptachlor epoxide); and metals (arsenic, beryllium, cadmium, lead, mercury, nickel, selenium, silver, thallium, and zinc) were identified as the contaminants of potential concern for future exposures resulting from migration of groundwater contaminants into surface water within the study area. These contaminants were selected based on comparison of groundwater data to Connecticut surface water protection criteria. Future surface water concentrations for many of the contaminants selected on the basis of exceedances of surface water protection criteria were not modeled because of infrequent detections or exceedances of selection criteria, because of their absence in shallow groundwater, because a source could not be identified, because they do not appear to be derived from Raymark waste, or because the scientific literature does not contain sufficient information on the behavior of the contaminant to model its migration. For surface water exposures in Ferry Creek, modeled concentrations of 1,1-DCE, arsenic, and zinc were evaluated.

The HHRA also provided a comparison of groundwater data to Connecticut groundwater volatilization criteria. This comparison identified VOCs (1,1-DCE, TCE, and vinyl chloride) as the contaminants of potential concern for exposures resulting from volatilization of groundwater contaminants into indoor air spaces within the study area.

Two routes of exposure were evaluated quantitatively in the human health risk assessment: indoor area and surface water in Ferry Creek. Potential future human exposures through consumption of oysters were evaluated qualitatively in this HHRA. Current human exposures were evaluated quantitatively in the OU3 Area II RI (TtNUS, 2000b). See Table 6-1 for a summary of the potential risks that could result from exposure to indoor air or future surface water exposures based on indoor air data and modeled surface water concentrations.

Indoor Air. The following summarizes the residential and commercial indoor air evaluations.

Residential. Based on indoor air data evaluated for residential inhalation exposures, non-carcinogenic and carcinogenic risks have been identified for the combined adult and child resident. Non-cancer hazard indices (HI) that were calculated are in excess of unity for residential exposures to indoor air. 1,2,4-Trimethylbenzene, 1,2-dibromoethane, 1,3,5-trimethyl benzene, 1,4-dichlorobenzene, benzene, chloroform, xylenes, and TCE were the main contributors to the hazard index. Adverse non-carcinogenic health effects are possible from exposure to each of these contaminants in indoor air. The cancer risk estimates for lifetime residential exposures to indoor air exceed the EPA target cancer risk range (10^{-4} to 10^{-6}). The major contributors to cancer risk at the site are 1,1,2,2-tetrachloroethane, 1,3-butadiene, benzyl chloride, methylene chloride, and TCE. Cancer risks for these and several other volatile organic contaminants are greater than the CTDEP target risk level of 10^{-6} for single contaminants.

Residential inhalation exposures resulting from volatilization of groundwater contaminants into indoor air spaces were also evaluated through an alternative method. Indoor air exposures to the three groundwater contaminants with exceedances of Connecticut groundwater volatilization criteria (1,1-DCE, TCE, and vinyl chloride) were evaluated using the Johnson and Ettinger Model to estimate indoor air concentrations from groundwater data. The predicted indoor air hazard index from exposures to these modeled indoor air concentrations falls between the RME and CTE hazard indices estimated based on actual indoor air concentrations. Adverse non-carcinogenic health effects are possible from exposure to TCE and 1,1-DCE. Cancer risks from exposures to these modeled indoor air concentrations are greater than cancer risks estimated based on actual indoor air concentrations. TCE was the main contributor to cancer risk estimates exceeding the EPA target cancer risk range (10^{-4} to 10^{-6}). Both vinyl chloride and TCE exceed the CTDEP target risk level of 10^{-6} for single contaminants.

Industrial/Commercial. Based on indoor air data evaluated for industrial/commercial inhalation exposures, non-carcinogenic and carcinogenic risks have been identified for the industrial/commercial worker. Non-cancer hazard indices (HI) that were calculated are in excess of unity for industrial/commercial exposures to indoor air. 1,2,4-Trimethylbenzene, 1,2-dibromoethane, 1,4-dichlorobenzene, and xylenes were the main contributors to the hazard index. Adverse non-carcinogenic health effects are possible from exposure to each of these contaminants in indoor air. The cancer risk estimates for industrial/commercial exposures to indoor air exceed the EPA cancer risk range (10^{-4} to 10^{-6}). The major contributors to cancer risk at the site are 1,1,2,2-tetrachloroethane, 1,3-butadiene, and TCE. Cancer risks for these and several other volatile organic contaminants are greater than the CTDEP target risk level of 10^{-6} for single contaminants.

Industrial/commercial inhalation exposures resulting from volatilization of groundwater contaminants into indoor air spaces were also evaluated through an alternative method. Indoor air exposures to the three groundwater contaminants with exceedances of Connecticut groundwater volatilization criteria (1,1-DCE, TCE, and vinyl chloride) were evaluated using the Johnson and Ettinger Model to estimate indoor air concentrations from groundwater data. The predicted indoor air hazard index from exposures to these modeled indoor air concentrations falls between the RME and CTE hazard indices estimated based on actual indoor air concentrations. Adverse non-carcinogenic health effects are possible from exposure to TCE and 1,1-DCE. Cancer risks from exposures to these modeled indoor air concentrations are greater than cancer risks estimated based on actual indoor air concentrations. TCE was the main contributor to cancer risk estimates exceeding the EPA cancer risk range (10^{-4} to 10^{-6}). Both vinyl chloride and TCE exceed the CTDEP target risk level of 10^{-6} for single contaminants.

Summary. The indoor air concentration model and indoor air sampling were used to estimate risks from indoor air exposures of residents and industrial/commercial workers to groundwater contaminants volatilizing into indoor air spaces of homes and commercial properties. Because of the variability of sampling and the large number of buildings within the indoor air area of interest, no single home or commercial property was evaluated. Rather, the model and sampling were used to estimate risks for a hypothetical home or commercial property within the indoor air area of interest. The area of interest at which these risk estimates may be applicable can be defined as the area of the groundwater plume where shallow groundwater concentrations of the greatest risk driver exceed the Connecticut groundwater volatilization

criteria. TCE has been identified as the greatest risk driver through the modeled indoor air evaluation. The proposed Connecticut groundwater volatilization criteria for residential scenarios is 27 µg/L. The proposed Connecticut groundwater volatilization criteria for industrial worker scenarios is 67 µg/L. Figure 4-20 delineates the estimated area of the plume where shallow groundwater concentrations of TCE are expected to exceed these criteria. Residential risk estimates may be applicable to residential properties within the area delineated by the Connecticut groundwater volatilization criteria for residential scenarios of 27 µg/L. These residential properties have had sub-slab ventilation systems installed during 2003 and 2004. Industrial/commercial worker risk estimates may be applicable to commercial properties within the area delineated by the Connecticut groundwater volatilization criteria for industrial worker scenarios of 67 µg/L.

Surface Water

Based on predicted models of groundwater contamination reaching Ferry Creek, surface water exposures are not expected to present a risk for future adult and child recreational users of Ferry Creek. Hazard indices (HI) are less than unity for future adult and child recreational exposures (wading) at Ferry Creek indicating that adverse non-carcinogenic health effects are not anticipated. The cancer risk estimates for future recreational users exposed to surface water in Ferry Creek do not exceed the EPA target cancer risk range (10^{-4} to 10^{-6}) or the CTDEP target total risk level of 10^{-5} for multiple contaminants, indicating that increased carcinogenic health effects are not anticipated.

Oyster Tissue

Future oyster tissue concentrations of arsenic and zinc were estimated from modeled future surface water concentrations in Ferry Creek and the Housatonic River. The predicted future oyster tissue concentrations were compared to the EPA Region III Risk-Based Concentrations for fish ingestion. Based on this comparison, it is possible that future risks from human consumption of oysters may be higher than 10^{-6}.

8.3.2 Ecological Risk Evaluation

Surface water sampling performed in the mid-1990s in Ferry Creek indicates a potential risk to aquatic organisms from copper, lead, and 4,4'-DDT. Maximum concentrations of six volatile organic compounds exceeded ecological screening values in some 1999 groundwater seep samples, but hazard quotients were relatively low.

The maximum predicted future concentration of 1,1-DCE at the point of discharge in the upper portion of Ferry Creek exceeds the most conservative ecological screening value for this compound, but was less than alternate screening values. The maximum predicted future concentrations of TCE, arsenic, and zinc at the point of discharge exceed their ecological screening values. With only minor dilution, predicted future concentrations of arsenic, zinc, 1,1-DCE, and TCE pose negligible risk to aquatic receptors. Site-related groundwater concentrations of copper and lead are not expected to reach Ferry Creek for approximately 200 years.

8.4 Conclusions

The interpretation of the data and information compiled for this RI indicates that:

- Raymark Facility waste was disposed of on the former Raymark Facility and throughout properties located within and in close proximity to the OU2 study area. This fill material has the potential to leach metals into the shallow groundwater, as evidenced by the synthetic precipitation leaching procedure (SPLP) data. The SPLP analysis indicates the potential for contamination to leach from soils into the groundwater; however, most of the properties with Raymark waste do not have corresponding groundwater data that conclusively indicate that Raymark waste has impacted groundwater quality. Refer to Appendix B for SPLP analytical results within the OU2 study area.

- Volatile organic compounds (VOCs) are present in both shallow and deep fill materials at the former Raymark Facility (HNUS 1995). VOCs were identified within the fill materials, soils, and groundwater at the former Raymark Facility on a frequent basis. However, given the irregular filling pattern, there is not always a direct correlation between VOCs in soil and groundwater samples. VOCs were used at the former

Raymark Facility and, based on filling on-site, waste management practices, and groundwater sampling results, VOCs were released into the groundwater.

- Analysis of groundwater samples reveals there is contamination in Ferry Creek from discharge of contaminated groundwater from the former Raymark Facility and other sources. Although groundwater contamination is ubiquitous through most of the study area, the groundwater contaminants and concentrations are not distributed evenly. Six discrete source areas for groundwater contamination were identified and analyzed for fate and transport. One area (Source E) was associated with 540 Longbrook Avenue, which is upgradient from the former Raymark Facility; one area (Source F) was associated with Raymark waste disposed of at the 600 East Broadway property; and the remaining four areas (Sources A, B, C, and D) were on the former Raymark Facility itself. This report documents that the ultimate fate of the contaminant plumes from these sources is to enter Ferry Creek or the Housatonic River. See Figure 5-1 for the location of these source areas.

- Modeling results suggest that 1) the concentration of 1,1-dichloroethene (1,1-DCE) and trichloroethene (TCE) entering Ferry Creek and the Housatonic River will remain at their present level until their sources are contained, destroyed, or depleted, and 2) the concentration of all modeled inorganic contamination discharging to Ferry Creek is expected to increase over time.

- The Human Health Risk Assessment (HHRA) estimates the potential for human health risk from exposures to groundwater in the study area. Groundwater in the study area and surrounding areas is not used as a drinking water source, therefore the primary pathways of potential concern for human health exposures are inhalation of volatiles present in indoor air due to volatilization of groundwater contaminants through building foundations, direct contact with surface water contamination resulting from migration of groundwater to Ferry Creek, and ingestion of shellfish (oysters), which may be contaminated as a result of migration of groundwater to Ferry Creek and subsequent uptake of surface water contaminants by oysters.

- Risks to human health from inhalation of indoor air were evaluated through both direct measurements of indoor air and indoor air concentrations modeled from groundwater

contamination data. The conclusion of both methods is that potential risk to human health from inhalation of indoor air is of concern for both residents and industrial/commercial workers. The results of the risk evaluation of indoor air data indicate risk at levels of concern from some contaminants present in groundwater, but also risks at levels of concern from other contaminants not detected in groundwater at levels expected to impact indoor air. It is presumed that groundwater is the source of some indoor air contaminants; however, the source for indoor air contaminants that were not detected in groundwater is unclear. The risk from volatilization of contaminants present in groundwater has decreased with the installation of sub-slab ventilation systems at over 100 homes located within the study area.

- Based on predicted models of groundwater contamination reaching Ferry Creek, surface water exposures are not expected to present a risk for future recreational users of Ferry Creek.

- Potential future human exposures through consumption of oysters were evaluated qualitatively. Predicted future oyster contamination indicates that potential future risks from human consumption of oysters may be higher than 10^{-6}.

- Recent surface water sampling indicates a potential risk for sediment and surface water-dwelling organisms that live in the study area.

- Predicted future surface water concentrations of arsenic, zinc, and 1,1-DCE in Ferry Creek and TCE in the Housatonic River may pose negligible risks to aquatic receptors.

TABLES

TABLE 2-1
HISTORY OF ACTIVITIES ASSOCIATED WITH RAYMARK FACILITY AND ENVIRONS
REMEDIAL INVESTIGATION
RAYMARK – OU2 – GROUNDWATER
STRATFORD, CONNECTICUT

DATE	ACTIVITY	COMPANY CONDUCTING ACTIVITY*	GENERAL FINDINGS
1992	Sediment Sampling	Weston	Sediment samples collected as part of an EPA Site Inspection for Raymark Industries. See Section 2.1.
1992-1994	CERCLA Removal Action at the Raymark Facility and on residential properties to abate imminent health risks	EPA	Mitigated imminent health risks posed by site conditions. See Section 1.3.3.
1993	Soil Sampling	Metcalf & Eddy - CTDEP	Soil samples collected from residential properties in or around the Housatonic Boat Club.
1993	Surface Water Sampling at the Raymark Facility	ELI	Surface water samples were collected by Raymark Industries to characterize the quantity and quality of drainage discharge into and out of Lagoon No. 4. See Section 2.2.
1993	Final Site Inspection Report for Raymark Industries issued	Weston	Included collection of sediment samples along Ferry Creek and the Housatonic River to monitor contaminant migration from the Raymark Facility. Numerous site-related organic and inorganic contaminants detected at elevated levels. Soil sampling detected site-related contaminants at the facility and nearby residential properties. Report also summarized previous sampling results (soil, sediment, groundwater). See Section 2.1.
1993	Fish and Shellfish Sampling	EPA and CTDEP	Fish/shellfish analyses from samples collected from various Stratford water bodies, including Housatonic River, Ferry Creek, Selby Pond, and other ponds. Health advisory issued to limit consumption of eels from Selby Pond.
1993-1995	Expanded Site Inspections (ESIs) and Vertical Sampling Program (VSP) – Soil Sampling	Weston	Surficial and subsurface soil and groundwater sampling conducted at various locations throughout Stratford identified contamination. Commercial and residential properties within the study area were investigated by Weston under TAT and ARCS, respectively. See Section 2.3.
1993-1995	Phase I Remedial Investigation	HNUS	Field investigation work included soil boring and sampling program, monitoring well installation and sampling program (100-series wells), surface water and sediment sampling, GPR survey, topographic survey, and wetlands delineation. See Section 2.4.
1993-1994	Comprehensive Site Investigation (CSI) sampling program and reports issued, Stratford Superfund Sites	HNUS	Surficial and subsurface soil investigations and sampling for lead, PCBs, and asbestos; conducted at Stratford residential properties to provide data necessary to proceed with the Stratford Superfund Sites Remediation Program. Reports were prepared by multiple EPA contractors using data developed by others. See Section 2.6.

TABLE 2-1 (cont.)
HISTORY OF ACTIVITIES ASSOCIATED WITH RAYMARK FACILITY AND ENVIRONS
REMEDIAL INVESTIGATION
RAYMARK – OU2 – GROUNDWATER
STRATFORD, CONNECTICUT

DATE	ACTIVITY	COMPANY CONDUCTING ACTIVITY*	GENERAL FINDINGS
1994-1995	Comprehensive Site Investigations (CSI), Stratford Superfund Sites, Final CSI Report issued 1995	Foster Wheeler	Surficial and subsurface soil investigations conducted at Stratford residential properties, using a grid sampling system, to provide data necessary to proceed with the Stratford Superfund Sites Remediation Program. Contamination identified. See Section 2.4.
1994	Hydrologic Runoff Analysis Report issued	ELI	Investigated surface water samples associated with drainage system network and diversion bypass around Lagoon No. 4. Contaminant discharge identified as result of drainage network, not a specific source or spill. See Section 2.2.
1994	Ground Penetrating Radar (GPR) Survey Report issued	Hager-Richter	Data obtained on depth of fill and presence of buried metal objects at three properties within the study area (600 East Broadway, Vacant Lot on Housatonic Blvd., and the Housatonic Boat Club). See Section 2.4.3
1994-1996	Removal Action and Post-Excavation Programs	Foster Wheeler	Site-specific removal actions were performed at 46 properties based on the results of the CSI program (Section 2.5). Post-excavation records for the removal actions document the remediation activities and indicate that the established clean-up criteria were achieved. See Section 2.6.
1995	Final RCRA Facility Investigation Report, Raymark Industries, issued	ELI	Reported results from monitoring wells and soil borings, Phase IIA and IIB groundwater sampling rounds, nature and extent of Raymark Facility contamination. Continued to exceed drinking water standards.
1995	Final Remedial Investigation Report, Raymark Facility	HNUS	Compiled results reported by ELI and other contractors as part of RCRA Facility Investigation and CERCLA time-critical removal actions at Raymark Facility. Widespread groundwater and soil contamination at facility. Recommended additional investigations of surface water, sediment, and groundwater off site.
1996-1997	Phase I OU2 Groundwater Remedial Investigation	B&RE	Field investigation included monitoring well installation (200-series and completion of some 100-series clusters), groundwater profiling, piezometer installation, and groundwater sampling. See Section 2.7.
1997	Ecological Risk Assessment	NOAA	Addressed risks to ecological receptors posed by hazardous Raymark Facility waste materials present in Ferry Creek, portions of the Housatonic River, and associated wetlands. See Section 2.8.
1997	Phase II Commercial Properties Site Investigation	B&RE	Field investigation included soil and sediment sampling; 35 soil borings were advanced to 16 feet. Work was conducted to finalize the commercial properties RI and support the FS. See Section 2.9.

TABLE 2-1 (cont.)
HISTORY OF ACTIVITIES ASSOCIATED WITH RAYMARK FACILITY AND ENVIRONS
REMEDIAL INVESTIGATION
RAYMARK – OU2 – GROUNDWATER
STRATFORD, CONNECTICUT

DATE	ACTIVITY	COMPANY CONDUCTING ACTIVITY*	GENERAL FINDINGS
1997	Draft Phase II and Draft Phase III Tech Memos, Selby Pond issued	B&RE	Reported nature and extent of surface water and sediment contamination in and around Selby Pond. Identified hydrologic connection between Ferry Creek and pond. Recommended consideration of remedial action to be combined with that of Ferry Creek. See OU3 – Ferry Creek RI.
1997	Final Tech Memo, Compilation of Existing Data, Raymark - Ferry Creek issued	B&RE	Compiled existing data. Identified data gaps to be filled during Raymark - Ferry Creek RI. See OU3 – Ferry Creek RI.
1998	Hydrologic and Hydraulic Analysis, Remedial Investigation and Feasibility Study, Raymark – Ferry Creek OU3	ACOE	Evaluated the flooding potential of Ferry Creek and surrounding drainage areas. See OU3 – Ferry Creek RI.
1998	Draft Evaluation of Raymark Superfund Data for PRG Development	SAIC	Evaluated historical and recently collected chemistry and toxicity data for development of preliminary remediation goals for Raymark-related contaminants of concern.
1998-1999	Phase II OU2 Groundwater Remedial Investigation	TtNUS	Field investigation included monitoring well installation (300- and 400-series) and sampling, GPR and EM survey, hydraulic conductivity tests, tidal study, and Ferry Creek seepage study. See Section 2.10.
1999	Evaluation of Ecological Risk to Avian and Mammalian Receptors in the Vicinity of Upper and Middle Ferry Creek	SAIC	Evaluated potential risk to avian and mammalian receptor species utilizing habitat in upper and middle Ferry Creek
1999	Phase III Ecological Risk Assessment; characterization of Areas C-F	SAIC	Conducted Site-Specific Marine Ecological Investigation to assess potential ecological risks to the aquatic environments of Areas C-F
1999	Draft Technical Memorandum Preliminary Screening of Alternatives - OU-3	TtNUS	Basis for Remedial Alternatives Screening Reports
1999	Final Remedial Investigation Report - OU-4	TtNUS	To be incorporated into the FS Report.

TABLE 2-1 (cont.)
HISTORY OF ACTIVITIES ASSOCIATED WITH RAYMARK FACILITY AND ENVIRONS
REMEDIAL INVESTIGATION
RAYMARK – OU2 – GROUNDWATER
STRATFORD, CONNECTICUT

DATE	ACTIVITY	COMPANY CONDUCTING ACTIVITY*	GENERAL FINDINGS
1999	Final Remedial Investigation Report - Area I - OU-3	TtNUS	Remedial Investigation Report issued in 1999.
1999	Seismic Refraction Survey Report	Hager-Richter	Data obtained to determine depth and configuration of top of bedrock. See Section 2.10.
1999-2002	GPR Survey	TtNUS (Hager-Richter)	Determined presence of underground utilities prior to drilling activities. See Sections 2.10, 2.12, and 2.13.
2000	Draft Technical Memorandums, OU3, OU4, OU7, OU8, - Remedial Alternatives Screening Report (4 Reports)	TtNUS	To be incorporated into the FS Report.
2000	Draft Final Remedial Investigation Report -OU2	TtNUS	Revision is this document
2000	Draft Final Remedial Investigation Report - Area II - OU3	TtNUS	For purposes of the FS, this will become OU7
2000	Draft Final Remedial Investigation Report - Area III - OU3	TtNUS	For purposes of the FS, this will become OU8
2000-2003	Residential Indoor Air Study	TtNUS/EPA	Conducted soil gas and indoor air sampling (5 events) to determine if VOC contaminants detected in groundwater beneath residential properties are volatilizing and migrating into homes. See Section 2.12.
2000	Draft Technical Memorandum - OU6 – Additional Properties	TtNUS	Initial presentation of additional property information, revised in 2004.
2002	Geophysical Survey Report	Geophysical Applications	Data obtained to locate suspected bedrock fracture zones. See Section 2.12.4.
2003	Borehole Geophysics Logging Report issued	Geophysical Applications	Data obtained to help identify and characterize hydraulically active bedrock fractures in boring SB-213DB, SB-215DB, and SB-524B. See Section 2.12.5.
2002 -2003	Phase III Groundwater Remedial Investigation	TtNUS	Field investigation included groundwater and soil gas profiling, well repair and redevelopment, well installation, geophysical surveys, groundwater and soil gas sampling, Ferry Creek seepage study, tidal study, and aquifer test. See Section 2.12.

TABLE 2-1 (cont.)
HISTORY OF ACTIVITIES ASSOCIATED WITH RAYMARK FACILITY AND ENVIRONS
REMEDIAL INVESTIGATION
RAYMARK – OU2 – GROUNDWATER
STRATFORD, CONNECTICUT

DATE	ACTIVITY	COMPANY CONDUCTING ACTIVITY*	GENERAL FINDINGS
2002-2003	Draft Final Remedial Investigation-OU6	TtNUS	Soil borings were advanced at 95 properties to identify Raymark waste. Identified 25 properties with Raymark waste. Some of these properties are outside the OU2 study area. See Section 2.13.
2004	Final Remedial Investigation Report –OU6	TtNUS	To be incorporated into the FS Report.

Notes:

* - ACOE = Army Corp of Engineers. ELI was hired by Raymark Industries, Inc. to perform environmental investigations at the Raymark Facility. Metcalf & Eddy performed environmental sampling under contract to CTDEP. Foster Wheeler was contracted by U.S. ACOE to perform environmental investigations to support the Stratford Superfund Sites Removal Action Program**.** Weston was contracted by EPA to perform environmental investigations at the Raymark Facility and environs, including residential and commercial properties in Stratford, under TAT and ARCS. NOAA and their contractor performed ecological risk assessment work for EPA. HNUS/B&RE (now TtNUS) was contracted by EPA to perform environmental investigations at the Raymark Facility and environs to complete associated RI/FS activities under ARCS and RAC. Hager-Richter Geoscience, Inc. was subcontracted by HNUS to perform a GPR survey to support the RI/FS activities. TtNUS was contracted by EPA to perform investigations for the Raymark Superfund site.

CSI - Comprehensive Site Investigation
ESI - Expanded Site Inspection
GPR - Ground Penetrating Radar
VSP - Vertical Sampling Program

TABLE 2-2
GROUNDWATER SAMPLES COLLECTED DURING 2002/2003 AND ANALYSES PERFORMED
REMEDIAL INVESTIGATION
RAYMARK - OU2 - GROUNDWATER
STRATFORD, CONNECTICUT

ZONE	BORING	SAMPLE NUMBER	DIOXINS	METALS	MEE	OS	OV	PAHSI	PESTP	WET
BEDROCK	MW-104B	OU2-MW-104B-07	+	+	+	+	+	+	+	+
BEDROCK	MW-112B	OU2-MW-112B-07	+	+		+	+	+	+	
BEDROCK	MW-113B	OU2-MW-113B-07	+	+	+	+	+	+	+	+
BEDROCK	MW-1B	OU2-MW-1B-07	+	+		+	+	+	+	+
BEDROCK	MW-201B	OU2-MW-201B-07		+	+	+	+	+	+	+
BEDROCK	MW-207B	OU2-MW-207B-07	+	+	+	+	+	+	+	+
BEDROCK	MW-208B	OU2-MW-208B-07	+	+	+	+	+	+	+	+
BEDROCK	MW-209B	OU2-MW-209B-07	+	+	+	+	+	+	+	+
BEDROCK	MW-210B	OU2-MW-210B-07	+	+	+	+	+	+	+	+
BEDROCK	MW-211B	OU2-MW-211B-07	+	+	+	+	+	+	+	+
BEDROCK	MW-212B	OU2-MW-212B-07	+	+	+	+	+	+	+	+
BEDROCK	MW-213B	OU2-MW-213B-07	+	+	+	+	+	+	+	+
BEDROCK	MW-213DB	OU2-MW-213DB-07	+	+	+	+	+	+	+	+
BEDROCK	MW-214B	OU2-MW-214B-07	+	+	+	+	+	+	+	+
BEDROCK	MW-215B	OU2-MW-215B-07	+	+	+	+	+	+	+	+
BEDROCK	MW-215DB	OU2-MW-215DB-07-AVG	+	+	+	+	+	+	+	+
BEDROCK	MW-217B	OU2-MW-217B-07	+	+	+	+	+	+	+	+
BEDROCK	MW-301B	OU2-MW-301B-07	+	+	+	+	+	+	+	+
BEDROCK	MW-302B	OU2-MW-302B-07	+	+	+	+	+	+	+	+
BEDROCK	MW-304B	OU2-MW-304B-07	+	+	+	+	+	+	+	+
BEDROCK	MW-305BR	OU2-MW-305BR-07-AVG	+	+	+	+	+	+	+	+
BEDROCK	MW-306B	OU2-MW-306B-07	+	+	+	+	+	+	+	+
BEDROCK	MW-306DB	OU2-MW-306DB-07	+	+	+	+	+	+	+	+
BEDROCK	MW-310B	OU2-MW-310B-07	+	+	+	+	+	+	+	+
BEDROCK	MW-312B	OU2-MW-312B-07	+	+		+	+	+	+	
BEDROCK	MW-401B	OU2-MW-401B-07	+	+	+	+	+	+	+	+
BEDROCK	MW-402B	OU2-MW-402B-07-AVG	+	+	+	+	+	+	+	+
BEDROCK	MW-4B	OU2-MW-4B-07		+		+	+	+	+	+
BEDROCK	MW-502B	OU2-MW-502B-07	+	+	+	+	+	+	+	+
BEDROCK	MW-503B	OU2-MW-503B-07	+	+	+	+	+	+	+	+
BEDROCK	MW-504B	OU2-MW-504B-07		+		+	+	+	+	
BEDROCK	MW-505B	OU2-MW-505B-07	+	+		+	+	+	+	
BEDROCK	MW-506B	OU2-MW-506B-07	+	+	+	+	+	+	+	+
BEDROCK	MW-507B	OU2-MW-507B-07	+	+		+	+	+	+	
BEDROCK	MW-508B	OU2-MW-508B-07	+	+	+	+	+	+	+	+
BEDROCK	MW-514B	OU2-MW-514B-07	+	+		+	+	+	+	
BEDROCK	MW-524B	OU2-MW-524B-07	+	+	+	+	+	+	+	+
BEDROCK	MW-533B	OU2-MW-533B-07	+	+	+	+	+	+	+	+
BEDROCK	MW-PC01B	OU1-MW-PC01B-07	+	+	+	+	+	+	+	+
BEDROCK	MW-PC02B	OU1-MW-PC02B-07	+	+	+	+	+	+	+	+
BEDROCK	MW-PC03B	OU1-MW-PC03B-07	+	+	+	+	+	+	+	+
BEDROCK	MW-PC04B	OU1-MW-PC04B-07	+	+	+	+	+	+	+	+
BEDROCK	MW-PC05B	OU1-MW-PC05B-07	+	+	+	+	+	+	+	+
BEDROCK	MW-PC06B	OU1-MW-PC06B-07	+	+		+	+	+	+	
BEDROCK	MW-PC08B	OU1-MW-PC08B-07	+	+	+	+	+	+	+	+
BEDROCK	MW-PC10B	OU1-MW-PC10B-07	+	+	+	+	+	+	+	+
BEDROCK	MW-PC11B	OU1-MW-PC11B-07	+	+	+	+	+	+	+	+
BEDROCK	MW-PC11BF	OU1-MW-PC11BF-07		+						
BEDROCK	MW-PC12B	OU1-MW-PC12B-07	+	+		+	+	+	+	+
BEDROCK	MW-PC13B	OU1-MW-PC13B-07	+	+	+	+	+	+	+	+
BEDROCK	MW-PC14B	OU1-MW-PC14B-07	+	+	+	+	+	+	+	+
BEDROCK	MW-PC15B	OU1-MW-PC15B-07	+	+	+	+	+	+	+	+

TABLE 2-2 (cont.)
GROUNDWATER SAMPLES COLLECTED DURING 2002/2003 AND ANALYSES PERFORMED
REMEDIAL INVESTIGATION
RAYMARK - OU2 - GROUNDWATER
STRATFORD, CONNECTICUT

ZONE	BORING	SAMPLE NUMBER	DIOXINS	METALS	MEE	OS	OV	PAHSI	PESTP	WET
BEDROCK	MW-PC16B	OU1-MW-PC16B-07	+	+	+	+	+	+	+	+
OVERBURDEN	CRA-2D	OU2-MW-CRA2D-07	+	+	+	+	+	+	+	+
OVERBURDEN	CRA-2S	OU2-MW-CRA2S-07		+		+	+	+	+	
OVERBURDEN	CRA-5D	OU2-MW-CRA5D-07		+		+	+	+	+	
OVERBURDEN	CRA-5S	OU2-MW-CRA5S-07		+		+	+	+	+	
OVERBURDEN	CRA-6D	OU2-MW-CRA6D-07	+	+	+	+	+	+	+	+
OVERBURDEN	CRA-6S	OU2-MW-CRA6S-07-AVG	+	+	+	+	+	+	+	+
OVERBURDEN	CRA-7	OU2-MW-CRA7-07		+		+	+	+	+	
OVERBURDEN	CRA-8	OU2-MW-CRA8-07		+		+	+	+	+	
OVERBURDEN	CRA-8	OU2-MW-CRA8F-07		+						
OVERBURDEN	MW-1	OU2-MW-1-07		+		+	+	+	+	
OVERBURDEN	MW-10	OU2-MW-10-07		+		+	+	+	+	
OVERBURDEN	MW-101D	OU2-MW-101D-07		+		+	+	+	+	
OVERBURDEN	MW-101M	OU2-MW-101M-07		+		+	+	+	+	
OVERBURDEN	MW-101S	OU2-MW-101S-07		+		+	+	+	+	
OVERBURDEN	MW-102D	OU2-MW-102D-07		+		+	+	+	+	
OVERBURDEN	MW-102M	OU2-MW-102M-07		+		+	+	+	+	
OVERBURDEN	MW-102S	OU2-MW-102S-07		+		+	+	+	+	
OVERBURDEN	MW-103D	OU2-MW-103D-07		+		+	+	+	+	
OVERBURDEN	MW-103M	OU2-MW-103M-07		+		+	+	+	+	
OVERBURDEN	MW-104D	OU2-MW-104D-07	+	+	+	+	+	+	+	+
OVERBURDEN	MW-104M	OU2-MW-104M-07	+	+	+	+	+	+	+	+
OVERBURDEN	MW-104S	OU2-MW-104S-07-AVG	+	+	+	+	+	+	+	+
OVERBURDEN	MW-110D	OU2-MW-110D-07-AVG		+		+	+	+	+	
OVERBURDEN	MW-110M	OU2-MW-110M-07	+	+		+	+	+	+	
OVERBURDEN	MW-110S	OU2-MW-110S-07	+	+		+	+	+	+	
OVERBURDEN	MW-111D	OU2-MW-111D-07	+	+	+	+	+	+	+	+
OVERBURDEN	MW-111M	OU2-MW-111M-07	+	+	+	+	+	+	+	+
OVERBURDEN	MW-111S	OU2-MW-111S-07	+	+	+	+	+	+	+	+
OVERBURDEN	MW-112D	OU2-MW-112D-07	+	+	+	+	+	+	+	+
OVERBURDEN	MW-112M	OU2-MW-112M-07-AVG	+			+	+	+	+	
OVERBURDEN	MW-113M	OU2-MW-113M-07	+	+	+	+	+	+	+	+
OVERBURDEN	MW-1M	OU2-MW-1M-07-AVG	+	+		+	+	+	+	+
OVERBURDEN	MW-1S	OU2-MW-1S-07	+	+		+	+	+	+	+
OVERBURDEN	MW-201D	OU2-MW-201D-07		+	+	+	+	+	+	+
OVERBURDEN	MW-202D	OU2-MW-202D-07-AVG	+	+	+	+	+	+	+	+
OVERBURDEN	MW-202S	OU2-MW-202S-07	+	+	+	+	+	+	+	+
OVERBURDEN	MW-203D	OU2-MW-203D-07		+		+	+	+	+	
OVERBURDEN	MW-203S	OU2-MW-203S-07		+		+	+	+	+	
OVERBURDEN	MW-204D	OU2-MW-204D-07		+		+	+	+	+	
OVERBURDEN	MW-205D	OU2-MW-205D-07	+	+		+	+	+	+	
OVERBURDEN	MW-206D	OU2-MW-206D-07-AVG		+		+	+	+	+	
OVERBURDEN	MW-206MR	OU2-MW-206MR-07		+		+	+	+	+	
OVERBURDEN	MW-206SR	OU2-MW-206SR-07	+	+		+	+	+	+	
OVERBURDEN	MW-207DR	OU2-MW-207DR-07	+	+	+	+	+	+	+	+
OVERBURDEN	MW-207MR	OU2-MW-207MR-07-AVG	+	+	+	+	+	+	+	+
OVERBURDEN	MW-207SR	OU2-MW-207SR-07	+	+	+	+	+	+	+	+
OVERBURDEN	MW-208D	OU2-MW-208D-07	+	+	+	+	+	+	+	+
OVERBURDEN	MW-208M	OU2-MW-208M-07	+	+	+	+	+	+	+	+
OVERBURDEN	MW-208S	OU2-MW-208S-07	+	+	+	+	+	+	+	+
OVERBURDEN	MW-209D	OU2-MW-209D-07	+	+	+	+	+	+	+	+
OVERBURDEN	MW-209S	OU2-MW-209S-07	+	+	+	+	+	+	+	+

TABLE 2-2 (cont.)
GROUNDWATER SAMPLES COLLECTED DURING 2002/2003 AND ANALYSES PERFORMED
REMEDIAL INVESTIGATION
RAYMARK - OU2 - GROUNDWATER
STRATFORD, CONNECTICUT

ZONE	BORING	SAMPLE NUMBER	DIOXINS	METALS	MEE	OS	OV	PAHSI	PESTP	WET
OVERBURDEN	MW-210D	OU2-MW-210D-07	+	+	+	+	+	+	+	+
OVERBURDEN	MW-210S	OU2-MW-210S-07	+	+	+	+	+	+	+	+
OVERBURDEN	MW-211D	OU2-MW-211D-07	+	+	+	+	+	+	+	+
OVERBURDEN	MW-211M	OU2-MW-211M-07	+	+	+	+	+	+	+	+
OVERBURDEN	MW-211S	OU2-MW-211S-07	+	+	+	+	+	+	+	+
OVERBURDEN	MW-212D	OU2-MW-212D-07-AVG	+	+	+	+	+	+	+	+
OVERBURDEN	MW-212M	OU2-MW-212M-07	+	+	+	+	+	+	+	+
OVERBURDEN	MW-212S	OU2-MW-212S-07	+	+	+	+	+	+	+	+
OVERBURDEN	MW-213S	OU2-MW-213S-07	+	+	+	+	+	+	+	+
OVERBURDEN	MW-214D	OU2-MW-214D-07	+	+	+	+	+	+	+	+
OVERBURDEN	MW-214M	OU2-MW-214M-07	+	+	+	+	+	+	+	+
OVERBURDEN	MW-214S	OU2-MW-214S-07	+	+	+	+	+	+	+	+
OVERBURDEN	MW-215S	OU2-MW-215S-07	+	+	+	+	+	+	+	+
OVERBURDEN	MW-217D	OU2-MW-217D-07	+	+	+	+	+	+	+	+
OVERBURDEN	MW-217S	OU2-MW-217S-07	+	+	+	+	+	+	+	+
OVERBURDEN	MW-2A	OU2-MW-2A-07		+		+	+	+	+	
OVERBURDEN	MW-2S	OU2-MW-2S-07		+		+	+	+	+	
OVERBURDEN	MW-301D	OU2-MW-301D-07-AVG	+	+	+	+	+	+	+	+
OVERBURDEN	MW-301S	OU2-MW-301S-07	+	+	+	+	+	+	+	+
OVERBURDEN	MW-302D	OU2-MW-302D-07-AVG	+		+	+	+	+		+
OVERBURDEN	MW-302S	OU2-MW-302S-07	+	+	+	+	+	+	+	+
OVERBURDEN	MW-304D	OU2-MW-304D-07	+	+	+	+	+	+	+	+
OVERBURDEN	MW-304S	OU2-MW-304S-07-AVG	+	+	+	+	+	+	+	+
OVERBURDEN	MW-305DR	OU2-MW-305DR-07	+	+	+	+	+	+	+	+
OVERBURDEN	MW-305SR	OU2-MW-305SR-07	+	+	+	+	+	+	+	+
OVERBURDEN	MW-310D	OU2-MW-310D-07-AVG	+	+	+	+	+	+	+	+
OVERBURDEN	MW-310S	OU2-MW-310S-07	+	+	+	+	+	+	+	+
OVERBURDEN	MW-312D	OU2-MW-312D-07	+	+		+	+	+	+	
OVERBURDEN	MW-312S	OU2-MW-312S-07	+	+		+	+	+	+	
OVERBURDEN	MW-3A	OU2-MW-3A-07		+		+	+	+	+	
OVERBURDEN	MW-3S	OU2-MW-3S-07		+		+	+	+	+	
OVERBURDEN	MW-4	OU2-MW-4-07		+		+	+	+	+	
OVERBURDEN	MW-401S	OU2-MW-401S-07	+	+	+	+	+	+	+	+
OVERBURDEN	MW-402S	OU2-MW-402S-07	+	+	+	+	+	+	+	+
OVERBURDEN	MW-4M(MW-4D)	OU2-MW-4M(MW-4D)-07	+	+		+	+	+	+	+
OVERBURDEN	MW-4S	OU2-MW-4S-07	+	+		+	+	+	+	+
OVERBURDEN	MW-502D	OU2-MW-502D-07	+	+	+	+	+	+	+	+
OVERBURDEN	MW-502S	OU2-MW-502S-07	+	+	+	+	+	+	+	+
OVERBURDEN	MW-503D	OU2-MW-503D-07	+	+	+	+	+	+	+	+
OVERBURDEN	MW-503M	OU2-MW-503M-07	+	+	+	+	+	+	+	+
OVERBURDEN	MW-503S	OU2-MW-503S-07	+	+	+	+	+	+	+	+
OVERBURDEN	MW-504D	OU2-MW-504D-07	+	+		+	+	+	+	
OVERBURDEN	MW-504S	OU2-MW-504S-07	+	+		+	+	+	+	
OVERBURDEN	MW-505D	OU2-MW-505D-07	+	+		+	+	+	+	
OVERBURDEN	MW-505M	OU2-MW-505M-07-AVG	+	+		+	+	+	+	
OVERBURDEN	MW-505S	OU2-MW-505S-07	+	+		+	+	+	+	
OVERBURDEN	MW-506D	OU2-MW-506D-07	+	+	+	+	+	+	+	+
OVERBURDEN	MW-506M	OU2-MW-506M-07	+	+	+	+	+	+	+	+
OVERBURDEN	MW-506S	OU2-MW-506S-07	+	+	+	+	+	+	+	+
OVERBURDEN	MW-507D	OU2-MW-507D-07	+	+		+	+	+	+	
OVERBURDEN	MW-507S	OU2-MW-507S-07	+	+		+	+	+	+	
OVERBURDEN	MW-508D	OU2-MW-508D-07	+	+	+	+	+	+	+	+

TABLE 2-2 (cont.)
GROUNDWATER SAMPLES COLLECTED DURING 2002/2003 AND ANALYSES PERFORMED
REMEDIAL INVESTIGATION
RAYMARK - OU2 - GROUNDWATER
STRATFORD, CONNECTICUT

ZONE	BORING	SAMPLE NUMBER	DIOXINS	METALS	MEE	OS	OV	PAHSI	PESTP	WET
OVERBURDEN	MW-508S	OU2-MW-508S-07	+	+	+	+	+	+	+	+
OVERBURDEN	MW-509M	OU2-MW-509M-07	+	+		+	+	+	+	
OVERBURDEN	MW-510M	OU2-MW-510M-07	+	+		+	+	+	+	
OVERBURDEN	MW-511M	OU2-MW-511M-07	+	+		+	+	+	+	
OVERBURDEN	MW-512M	OU2-MW-512M-07	+	+		+	+	+	+	
OVERBURDEN	MW-513S	OU2-MW-513S-07	+	+		+	+	+	+	
OVERBURDEN	MW-514D	OU2-MW-514D-07	+	+		+	+	+	+	
OVERBURDEN	MW-514S	OU2-MW-514S-07	+	+		+	+	+	+	
OVERBURDEN	MW-515S	OU2-MW-515S-07-AVG	+	+		+	+	+	+	
OVERBURDEN	MW-516M	OU2-MW-516M-07-AVG	+	+	+	+	+	+	+	+
OVERBURDEN	MW-517S	OU2-MW-517S-07	+	+		+	+	+	+	
OVERBURDEN	MW-518S	OU2-MW-518S-07	+	+		+	+	+	+	
OVERBURDEN	MW-519S	OU2-MW-519S-07	+	+		+	+	+	+	
OVERBURDEN	MW-520S	OU2-MW-520S-07	+	+		+	+	+	+	
OVERBURDEN	MW-521S	OU2-MW-521S-07	+	+		+	+	+	+	
OVERBURDEN	MW-522S	OU2-MW-522S-07	+	+		+	+	+	+	
OVERBURDEN	MW-523S	OU2-MW-523S-07	+	+		+	+	+	+	
OVERBURDEN	MW-525S	OU2-MW-525S-07	+	+		+	+	+	+	
OVERBURDEN	MW-526S	OU2-MW-526S-07	+	+	+	+	+	+	+	+
OVERBURDEN	MW-527S	OU2-MW-527S-07	+	+	+	+	+	+	+	+
OVERBURDEN	MW-528S	OU2-MW-528S-07	+	+		+	+	+	+	
OVERBURDEN	MW-529S	OU2-MW-529S-07-AVG	+	+	+	+	+	+	+	+
OVERBURDEN	MW-530S	OU2-MW-530S-07	+	+		+	+	+	+	
OVERBURDEN	MW-531S	OU2-MW-531S-07	+	+		+	+	+	+	
OVERBURDEN	MW-532S	OU2-MW-532S-07	+	+		+	+	+	+	
OVERBURDEN	MW-533D	OU2-MW-533D-07	+	+	+	+	+	+	+	+
OVERBURDEN	MW-533S	OU2-MW-533S-07	+	+	+	+	+	+	+	+
OVERBURDEN	MW-534S	OU2-MW-534S-07-AVG	+	+		+	+	+	+	
OVERBURDEN	MW-535S	OU2-MW-535S-07	+	+		+	+	+	+	
OVERBURDEN	MW-536S	OU2-MW-536S-07	+	+	+	+	+	+	+	+
OVERBURDEN	MW-6	OU2-MW-6-07		+		+	+	+	+	
OVERBURDEN	MW-6	OU4-MW-6-07		+		+	+	+	+	
OVERBURDEN	MW-7	OU2-MW-7-07		+		+	+	+	+	
OVERBURDEN	MW-8	OU2-MW-8-07-AVG		+		+	+	+	+	
OVERBURDEN	MW-9	OU2-MW-9-07-AVG		+		+	+	+	+	
OVERBURDEN	MW-BR-1	OU2-MW-BR-1-07	+	+		+	+	+	+	
OVERBURDEN	MW-BR-2	OU2-MW-BR-2-07	+	+		+	+	+	+	
OVERBURDEN	MW-PC01D	OU1-MW-PC01D-07	+	+	+	+	+	+	+	+
OVERBURDEN	MW-PC01M	OU1-MW-PC01M-07	+	+	+	+	+	+	+	+
OVERBURDEN	MW-PC01S	OU1-MW-PC01S-07	+	+	+	+	+	+	+	+
OVERBURDEN	MW-PC02D	OU1-MW-PC02D-07	+	+	+	+	+	+	+	+
OVERBURDEN	MW-PC02M	OU1-MW-PC02M-07	+	+	+	+	+	+	+	+
OVERBURDEN	MW-PC02S	OU1-MW-PC02S-07	+	+	+	+	+	+	+	+
OVERBURDEN	MW-PC03D	OU1-MW-PC03D-07-AVG	+	+	+	+	+	+	+	+
OVERBURDEN	MW-PC03S	OU1-MW-PC03S-07	+	+	+	+	+	+	+	+
OVERBURDEN	MW-PC04D	OU1-MW-PC04D-07-AVG	+	+	+	+	+	+	+	+
OVERBURDEN	MW-PC04S	OU1-MW-PC04S-07	+	+	+	+	+	+	+	+
OVERBURDEN	MW-PC05D	OU1-MW-PC05D-07	+	+	+	+	+	+	+	+
OVERBURDEN	MW-PC05M	OU1-MW-PC05M-07	+	+	+	+	+	+	+	+
OVERBURDEN	MW-PC05S	OU1-MW-PC05S-07	+	+	+	+	+	+	+	+
OVERBURDEN	MW-PC06D	OU1-MW-PC06D-07		+		+	+	+	+	
OVERBURDEN	MW-PC06M	OU1-MW-PC06M-07				+	+	+	+	

TABLE 2-2 (cont.)
GROUNDWATER SAMPLES COLLECTED DURING 2002/2003 AND ANALYSES PERFORMED
REMEDIAL INVESTIGATION
RAYMARK - OU2 - GROUNDWATER
STRATFORD, CONNECTICUT

ZONE	BORING	SAMPLE NUMBER	DIOXINS	METALS	MEE	OS	OV	PAHSI	PESTP	WET
OVERBURDEN	MW-PC06S	OU1-MW-PC06S-07	+	+		+	+	+	+	
OVERBURDEN	MW-PC07S	OU1-MW-PC07S-07	+	+		+	+	+	+	
OVERBURDEN	MW-PC08D	OU1-MW-PC08D-07	+	+	+	+	+	+	+	+
OVERBURDEN	MW-PC08S	OU1-MW-PC08S-07	+	+	+	+	+	+	+	+
OVERBURDEN	MW-PC09D	OU1-MW-PC09D-07	+	+	+	+	+	+	+	+
OVERBURDEN	MW-PC09S	OU1-MW-PC09S-07	+	+	+	+	+	+	+	+
OVERBURDEN	MW-PC10D	OU1-MW-PC10D-07	+	+	+	+	+	+	+	+
OVERBURDEN	MW-PC10M	OU1-MW-PC10M-07-AVG	+	+	+	+	+	+	+	+
OVERBURDEN	MW-PC10S	OU1-MW-PC10S-07-AVG	+	+	+	+	+	+	+	+
OVERBURDEN	MW-PC11D	OU1-MW-PC11D-07	+	+	+	+	+	+	+	+
OVERBURDEN	MW-PC11M	OU1-MW-PC11M-07	+	+	+	+	+	+	+	+
OVERBURDEN	MW-PC11S	OU1-MW-PC11S-07	+	+	+	+	+	+	+	+
OVERBURDEN	MW-PC12D	OU1-MW-PC12D-07	+	+	+	+	+	+	+	+
OVERBURDEN	MW-PC12S	OU1-MW-PC12S-07	+	+	+	+	+	+	+	+
OVERBURDEN	MW-PC13D	OU1-MW-PC13D-07	+	+	+	+	+	+	+	+
OVERBURDEN	MW-PC13M	OU1-MW-PC13M-07	+	+	+	+	+	+	+	+
OVERBURDEN	MW-PC13S	OU1-MW-PC13S-07	+	+	+	+	+	+	+	+
OVERBURDEN	MW-PC14D	OU1-MW-PC14D-07	+	+	+	+	+	+	+	+
OVERBURDEN	MW-PC14S	OU1-MW-PC14S-07	+	+	+	+	+	+	+	+
OVERBURDEN	MW-PC15D	OU1-MW-PC15D-07	+	+	+	+	+	+	+	+
OVERBURDEN	MW-PC15S	OU1-MW-PC15S-07	+	+	+	+	+	+	+	+
OVERBURDEN	MW-PC16D	OU1-MW-PC16D-07-AVG	+	+	+	+	+	+	+	+
OVERBURDEN	MW-PC16M	OU1-MW-PC16M-07	+	+	+	+	+	+	+	+
OVERBURDEN	MW-PC16S	OU1-MW-PC16S-07	+	+	+	+	+	+	+	+
OVERBURDEN	MW-PZ13	OU2-MW-PZ13-07		+		+	+	+	+	
OVERBURDEN	MW-PZ14	OU2-MW-PZ14-07		+		+	+	+	+	
OVERBURDEN	MW-Z	OU2-MW-Z-07		+		+	+	+	+	
OVERBURDEN	OU4-MW-3	OU4-MW-3-07		+		+	+	+	+	
OVERBURDEN	SPDA-E222,S132	OU2-MW-SPDA-E222,S132-07		+		+	+	+	+	
OVERBURDEN	WA120AG	OU2-MW-WA120AG-E512-541-07		+		+	+	+	+	
RECOVERY WELL	RW1	OU1-MW-RW1-07		+		+	+	+	+	
RECOVERY WELL	RW2	OU1-MW-RW2-07		+		+	+	+	+	
RECOVERY WELL	RW2F	OU1-MW-RW2F-07		+						
RECOVERY WELL	RW3	OU1-MW-RW3-07		+		+	+	+	+	
RECOVERY WELL	RW5	OU1-MW-RW5-07		+		+	+	+	+	
RECOVERY WELL	RW6	OU1-MW-RW6-07		+		+		+	+	
RECOVERY WELL	RW6A	OU1-MW-RW6A-07					+			
UPGRADIENT_BEDROCK	MW-216B	OU2-MW-216B-07		+	+	+	+	+	+	+
UPGRADIENT_BEDROCK	MW-216DB	OU2-MW-216DB-07-AVG		+	+	+	+	+	+	+
UPGRADIENT_BEDROCK	MW-303B	OU2-MW-303B-07	+	+	+	+	+	+	+	+
UPGRADIENT_BEDROCK	MW-307B	OU2-MW-307B-07		+	+	+	+	+	+	+
UPGRADIENT_BEDROCK	MW-307DB	OU2-MW-307DB-07		+	+	+	+	+	+	+
UPGRADIENT_BEDROCK	MW-308B	OU2-MW-308B-07	+	+	+	+	+	+	+	+
UPGRADIENT_BEDROCK	MW-308DB	OU2-MW-308DB-07	+	+	+	+	+	+	+	+
UPGRADIENT_BEDROCK	MW-309B	OU2-MW-309B-07	+	+	+	+	+	+	+	+
UPGRADIENT_BEDROCK	MW-311B	OU2-MW-311B-07	+	+	+	+	+	+	+	+
UPGRADIENT_BEDROCK	MW-313B	OU2-MW-313B-07		+		+	+	+	+	
UPGRADIENT_BEDROCK	MW-501B	OU2-MW-501B-07	+	+	+	+	+	+	+	+
UPGRADIENT_OVERBURDEN	MW-309D	OU2-MW-309D-07-AVG	+	+	+	+	+	+	+	+
UPGRADIENT_OVERBURDEN	MW-309S	OU2-MW-309S-07	+	+	+	+	+	+	+	+
UPGRADIENT_OVERBURDEN	MW-311D	OU2-MW-311D-07	+	+	+	+	+	+	+	+
UPGRADIENT_OVERBURDEN	MW-311S	OU2-MW-311S-07	+	+	+	+	+	+	+	+

TABLE 2-2 (cont.)
GROUNDWATER SAMPLES COLLECTED DURING 2002/2003 AND ANALYSES PERFORMED
REMEDIAL INVESTIGATION
RAYMARK - OU2 - GROUNDWATER
STRATFORD, CONNECTICUT

ZONE	BORING	SAMPLE NUMBER	DIOXINS	METALS	MEE	OS	OV	PAHSI	PESTP	WET
UPGRADIENT_OVERBURDEN	MW-313D	OU2-MW-313D-07		+		+	+	+	+	
UPGRADIENT_OVERBURDEN	MW-313S	OU2-MW-313S-07		+		+	+	+	+	
UPGRADIENT_OVERBURDEN	MW-501D	OU2-MW-501D-07	+	+	+	+	+	+	+	+
UPGRADIENT_OVERBURDEN	MW-501S	OU2-MW-501S-07	+	+	+	+	+	+	+	+

MEE - Methane, Ethane, Ethene
OV - Volatile organic compounds
OS - Semivolatile organic compounds
PAHSI - Low concentration semivolatile organic compound analysis
PESTP - pesticides and/or PCBs
WET - wet chemistry parameters such as Carbon Dioxide, chloride, ferrous iron, nitrate, nitrite, sulfate.

TABLE 3-1
HYDRAULIC CONDUCTIVITY VALUES FOR SINGLE WELL PUMP TESTS
REMEDIAL INVESTIGATION
RAYMARK - OU2- GROUNDWATER
STRATFORD, CONNECTICUT

Monitoring Well Identifier	Aquifer	K (ft/day)	Aquifer materials	Notes
MW-102M	INTERMEDIATE	102.14	silty sand, sandy gravel	2
MW-104D	DEEP	43.65	sand and gravel	3
MW-104M	INTERMEDIATE	19.38	fine sand	3
MW-110S	SHALLOW	1.31	silty sand and peat	1
MW-112B	BEDROCK	1.79	bedrock	3
MW-112D	DEEP	12.76	sand and gravel	3
MW-112M	INTERMEDIATE	13.51	sand and silty sand	3
MW-113B	BEDROCK	2.80	bedrock	3
MW-113M	INTERMEDIATE	49.15	sand and gravel	3
MW-201B	BEDROCK	0.06	bedrock	3
MW-201D	DEEP	3.17	gravelly sand	3
MW-202D	DEEP	199.15	gravelly sand	2
MW-207B	BEDROCK	2.63	bedrock	1
MW-207DR	DEEP	22.33	sand	2
MW-208D	DEEP	29.58	sand and gravel	3
MW-208M	INTERMEDIATE	49.76	sand and gravel	3
MW-209B	BEDROCK	0.18	bedrock	3
MW-209D	DEEP	21.02	Sand	3
MW-211B	BEDROCK	0.02	bedrock	3
MW-211D	DEEP	67.47	gravelly sand	3
MW-211M	INTERMEDIATE	3.49	sand	3
MW-212D	DEEP	25.53	sand	3
MW-212M	INTERMEDIATE	62.89	sand	3
MW-212S	SHALLOW	188.98	gravelly sand	2
MW-213DB	DEEP BEDROCK	9.07	bedrock	1
MW-214B	BEDROCK	475.90	bedrock	2
MW-214D	DEEP	25.10	medium sand	3
MW-214D	DEEP	27.66	medium sand	2
MW-214M	INTERMEDIATE	35.72	medium sand	3
MW-215DB	DEEP BEDROCK	<0.07	bedrock	2
MW-217S	SHALLOW	518.33	gravelly sand	2
MW-301B	BEDROCK	0.31	bedrock	1
MW-301D	DEEP	244.74	gravelly sand	3
MW-302D	DEEP	12.07	fine sand	3
MW-302S	SHALLOW	331.97	sand and gravel	2
MW-304B	BEDROCK	1.55	bedrock	3
MW-304D	DEEP	114.20	medium to fine sand	3
MW-305BR	BEDROCK	22.54	bedrock	2
MW-305DR	DEEP	11.98	silty sand	2
MW-306DB	DEEP BEDROCK	0.09	bedrock	1
MW-310B	BEDROCK	0.08	bedrock	1
MW-310D	DEEP	51.95	sand	3
MW-312B	BEDROCK	1.24	bedrock	1

TABLE 3-1 (cont.)
HYDRAULIC CONDUCTIVITY VALUES FOR SINGLE WELL PUMP TESTS
REMEDIAL INVESTIGATION
RAYMARK - OU2- GROUNDWATER
STRATFORD, CONNECTICUT

Monitoring Well Identifier	Aquifer	K (ft/day)	Aquifer materials	Notes
MW-401B	BEDROCK	73.22	bedrock	3
MW-402B	BEDROCK	0.62	bedrock	3
MW-501D	DEEP	154.61	sand and gravel	2
MW-501B	BEDROCK	<4.21	bedrock	2
MW-502B	BEDROCK	1.14	bedrock	1
MW-503M	INTERMEDIATE	298.96	gravelly sand	2
MW-503D	DEEP	80.34	silty sand	2
MW-505M	INTERMEDIATE	27.90	sand	2
MW-505D	DEEP	326.01	sand and gravel	2
MW-505B	BEDROCK	0.05	bedrock	1
MW-506S	SHALLOW	115.03	silty sand	2
MW-506D	DEEP	70.98	sand and gravel	2
MW-507B	BEDROCK	51.73	bedrock	2
MW-508S	SHALLOW	56.70	gravelly sand	2
MW-508B	BEDROCK	64.46	bedrock	2
MW-509M	INTERMEDIATE	269.72	gravelly sand	2
MW-510M	INTERMEDIATE	68.23	sand	2
MW-514D	DEEP	144.24	gravelly sand	2
MW-515S	SHALLOW	210.55	gravelly sand	2
MW-520S	SHALLOW	50.44	silty sand	2
MW-523S	SHALLOW	227.95	gravelly sand	2
MW-524B	BEDROCK	0.53	bedrock	1
MW-525S	SHALLOW	246.43	gravelly sand	2
MW-528S	SHALLOW	243.34	gravelly sand	2
MW-530S	SHALLOW	3.26	gravelly sand, some silt	1
MW-532S	SHALLOW	67.32	silty sand	2
MW-534S	SHALLOW	317.49	sand and gravel	2
MW-535S	SHALLOW	20.68	fill	2
FOSTER-WHEELER OU1 DATA				
PC-01B	BEDROCK	0.52	bedrock	1
PC-01D	DEEP	222.34	fine gravel	3
PC-04B	BEDROCK	0.36	bedrock	1
PC-06B	BEDROCK	0.05	bedrock	1
PC-08B	BEDROCK	0.06	bedrock	1
PC-09D	DEEP	329.47	gravelly sand	3
PC-10B	BEDROCK	0.27	bedrock	1
PC-10M	INTERMEDIATE	11.02	sand	3
PC-13B	BEDROCK	0.39	bedrock	1
PC-13M	INTERMEDIATE	2.47	bedrock	3
PC-15B	BEDROCK	0.35	bedrock	1
PC-15D	DEEP	7.45	sand	3
PC-16M	INTERMEDIATE	1.50	silty sand	1
PC-16S	SHALLOW	2.84	silty sand	1

1. Hydraulic conductivity determined from low-flow purging
2. Hydraulic conductivity determined from well development or redevelopmer
3. Hydraulic conductivity calculated in 1999 R
Note: Hydraulic conductivity calculations are shown in Appendix D

TABLE 3-2
SUMMARY OF VERTICAL GRADIENTS
REMEDIAL INVESTIGATION
RAYMARK - OU2 - GROUNDWATER
STRATFORD, CONNECTICUT

Monitoring Well Identifier	Aquifer	Elevation of Midpoint of Well Screen	Water Level Elevations		Vertical Gradients [1,2]	
			3/24/1999	4/15/2003	1999	2003
MW-101D	DEEP	-55.4	3.55	3.93		
MW-101M	INTERMEDIATE	-14.4	2.46	3.82	0.02659	0.00268
MW-101S	SHALLOW	-3.3	4.75	4.82	-0.20631	-0.09009
MW-102D	DEEP	-97.0	4.00	4.40		
MW-102M	INTERMEDIATE	-15.4	4.95	4.31	-0.01164	0.00110
MW-102S	SHALLOW	1.6	6.00	6.07	-0.06176	-0.10353
MW-103D	INTERMEDIATE	-65.0	3.67	4.01		
MW-103M	SHALLOW	-20.4	3.62	3.97	0.00112	0.00090
MW-104B	BEDROCK	-94.3	3.09	3.32		
MW-104D	DEEP	-50.2	3.04	3.32	0.00113	0.00000
MW-104M	INTERMEDIATE	-16.1	3.16	3.41	-0.00352	-0.00264
MW-104S	SHALLOW	0.2	4.82	4.91	-0.10184	-0.09202
MW-110D	DEEP	-37.8	3.27	3.55		
MW-110M	INTERMEDIATE	-13.3	2.98	3.43	0.01184	0.00490
MW-110S	SHALLOW	-1.2	9.10	9.30	-0.50579	-0.48512
MW-111D	DEEP	-73.8	3.05	3.34		
MW-111M	INTERMEDIATE	-18.2	3.25	3.09	-0.00360	0.00450
MW-111S	SHALLOW	2.3	3.40	3.42	-0.00732	-0.01610
MW-112B	BEDROCK	-45.0	3.08	3.33		
MW-112D	DEEP	-33.0	3.08	3.31	0.00000	0.00167
MW-112M	INTERMEDIATE	-21.9	3.00	3.17	0.00721	0.01261
MW-113B	BEDROCK	-94.4	2.33	2.73		
MW-113M	INTERMEDIATE	-25.9	2.75	3.06	-0.00613	-0.00482
MW-201B	BEDROCK	-50.5	4.63	5.13		
MW-201D	DEEP	-20.3	4.57	5.06	0.00199	0.00232
MW-202D	DEEP	-19.8	2.79	4.83		
MW-202S	SHALLOW	1.6	NA	4.86	NA	-0.00140
MW-203D	DEEP	-13.3	4.53	4.99		
MW-203S	SHALLOW	1.2	NA	5.04	NA	-0.00345
MW-206D	DEEP	-51.5	2.28	5.07		

TABLE 3-2 (cont.)
SUMMARY OF VERTICAL GRADIENTS
REMEDIAL INVESTIGATION
RAYMARK - OU2 - GROUNDWATER
STRATFORD, CONNECTICUT

Monitoring Well Identifier	Aquifer	Elevation of Midpoint of Well Screen	Water Level Elevations		Vertical Gradients [1,2]	
			3/24/1999	4/15/2003	1999	2003
MW-206MR	INTERMEDIATE	-16.9	4.54	5.01	-0.06532	0.00173
MW-206SR	SHALLOW	-1.5	4.44	5.04	0.00649	-0.00195
MW-207B	BEDROCK	-70.3		4.84		
MW-207DR	DEEP	-57.6	4.35	5.12	-0.34252	-0.02205
MW-207MR	INTERMEDIATE	-26.1	4.35	4.84	0.00000	0.00889
MW-207SR	SHALLOW	1.3	4.28	4.81	0.00255	0.00109
MW-208B	BEDROCK	-74.2	6.02	6.85		
MW-208D	DEEP	-45.0	5.99	6.84	0.00103	0.00034
MW-208M	INTERMEDIATE	-27.1	6.01	6.85	-0.00111	-0.00056
MW-208S	SHALLOW	3.5	5.96	6.91	0.00164	-0.00196
MW-209B	BEDROCK	-80.2	5.47	6.09		
MW-209D	DEEP	-51.0	5.43	6.21	0.00137	-0.00411
MW-209S	SHALLOW	3.6	6.52	6.73	-0.01997	-0.00953
MW-210D (new)	DEEP	-19.1	3.39	3.75		
MW-210S	SHALLOW	-4.1	3.32	3.67	0.00467	0.00533
MW-211B	BEDROCK	-142.9	4.09	4.62		
MW-211D	DEEP	-115.9	4.01	4.52	0.00297	0.00371
MW-211M	INTERMEDIATE	-81.2	4.14	4.59	-0.00374	-0.00201
MW-211S	SHALLOW	0.1	3.91	4.56	0.00283	0.00037
MW-212B	BEDROCK	-64.5	3.46	3.97		
MW-212D	DEEP	-40.5	3.51	3.94	-0.00208	0.00125
MW-212M	INTERMEDIATE	-28.5	3.48	3.92	0.00250	0.00167
MW-212S	SHALLOW	-2.5	3.48	3.89	0.00000	0.00115
MW-213B	BEDROCK	-21.9	3.44	3.76		
MW-213S	SHALLOW	2.6	3.46	3.76	-0.00082	0.00000
MW-214B	BEDROCK	-40.8		3.51		
MW-214D	DEEP	-25.3	3.31	3.52	-0.21355	-0.00065
MW-214M	INTERMEDIATE	-15.6	3.34	3.55	-0.00309	-0.00309
MW-214S	SHALLOW	0.3	3.30	3.57	0.00252	-0.00126
MW-215B	BEDROCK	-12.3	3.36	3.69		
MW-215S	SHALLOW	6.7	3.36	3.68	0.00000	0.00053

TABLE 3-2 (cont.)
SUMMARY OF VERTICAL GRADIENTS
REMEDIAL INVESTIGATION
RAYMARK - OU2 - GROUNDWATER
STRATFORD, CONNECTICUT

Monitoring Well Identifier	Aquifer	Elevation of Midpoint of Well Screen	Water Level Elevations		Vertical Gradients [1,2]	
			3/24/1999	4/15/2003	1999	2003
MW-216DB	DEEP BEDROCK	-52.3	5.97	6.31		
MW-216B	BEDROCK	-22.0	5.98	6.28	-0.00033	0.00099
MW-217B	BEDROCK	-41.3	3.11	3.72		
MW-217D	DEEP	-13.8	3.12	3.83	-0.00036	-0.00400
MW-217S	SHALLOW			3.30	Not Installed	0.03849
MW-301B	BEDROCK	-55.7	3.41	3.70		
MW-301D	DEEP	-26.6	3.56	3.74	-0.00515	-0.00137
MW-301S	SHALLOW	1.6	3.64	3.90	-0.00284	-0.00567
MW-302B	BEDROCK	-53.0	4.06	4.13		
MW-302D	DEEP	-17.4	3.03	3.85	0.02893	0.00787
MW-302S	SHALLOW	2.7	3.01	3.80	0.00100	0.00249
MW-304B	BEDROCK	-67.7	3.75	4.54		
MW-304D	DEEP	-44.1	3.73	4.22	0.00085	0.01356
MW-304S	SHALLOW	-1.4	3.77	4.24	-0.00094	-0.00047
MW-305BR	BEDROCK	-109.0	4.19	4.53		
MW-305DR	DEEP	-43.5	4.02	4.20	0.00260	0.00504
MW-305SR	SHALLOW	-0.7	3.07	2.73	0.02220	0.03435
MW-306DB	DEEP BEDROCK	-50.2	7.99	8.83		
MW-306B	BEDROCK	-19.2	7.66	8.61	0.01065	0.00710
MW-307DB	DEEP BEDROCK	-21.0	25.22	25.80		
MW-307B	BEDROCK	12.3	26.98	26.52	-0.05285	-0.02162
MW-308DB	DEEP BEDROCK	-16.7	26.28	26.84		
MW-308B	BEDROCK	17.9	26.17	26.68	0.00318	0.00462
MW-309B	BEDROCK	-102.9	4.69	5.14		
MW-309D	DEEP	-77.4	4.66	5.06	0.00118	0.00314
MW-309S	SHALLOW	-0.8	4.65	5.03	0.00013	0.00039
MW-310B	BEDROCK	-71.8	3.05	3.34		
MW-310D	DEEP	-36.1	3.16	3.49	-0.00309	-0.00421
MW-310S	SHALLOW	-2.4	3.23	3.59	-0.00208	-0.00297

TABLE 3-2 (cont.)
SUMMARY OF VERTICAL GRADIENTS
REMEDIAL INVESTIGATION
RAYMARK - OU2 - GROUNDWATER
STRATFORD, CONNECTICUT

Monitoring Well Identifier	Aquifer	Elevation of Midpoint of Well Screen	Water Level Elevations		Vertical Gradients [1,2]	
			3/24/1999	4/15/2003	1999	2003
MW-311B	BEDROCK	-67.5	5.01	5.57		
MW-311D	DEEP	-40.7	4.91	5.44	0.00373	0.00485
MW-311S	SHALLOW	2.2	6.80	7.29	-0.04406	-0.04312
MW-312B	BEDROCK	-100.7	2.67	2.99		
MW-312D	DEEP	-76.5	2.75	2.92	-0.00331	0.00289
MW-312S	SHALLOW	1.8	4.58	4.74	-0.02337	-0.02324
MW-313B	BEDROCK	-37.1	5.38	5.88		
MW-313D	DEEP	-13.4	5.42	5.88	-0.00168	0.00000
MW-313S	SHALLOW	2.4	5.25	5.70	0.01079	0.01143
MW-401B	BEDROCK	-23.9	6.26	6.94		
MW-401S	SHALLOW	3.0	6.16	6.80	0.00372	0.00521
MW-402B	BEDROCK	-78.1	4.81	5.24		
MW-402S	SHALLOW	0.5	4.99	5.51	-0.00229	-0.00344
MW-501B	BEDROCK	-36.5		6.90	not installed	
MW-501D	DEEP	-24.8		5.75	not installed	0.09829
MW-501S	SHALLOW	4.7		8.01	not installed	-0.07661
MW-502B	BEDROCK	-25.9		4.30	not installed	
MW-502D	DEEP	-13.5		4.23	not installed	0.00565
MW-502S	SHALLOW	0.2		4.26	not installed	-0.00219
MW-503B	BEDROCK	-89.3		3.61	not installed	
MW-503D	DEEP	-74.5		3.66	not installed	-0.00338
MW-503M	INTERMEDIATE	-33.0		3.64	not installed	0.00048
MW-503S	SHALLOW	0.8		2.76	not installed	0.02604
MW-504B	BEDROCK	-63.3		4.16	not installed	
MW-504D	DEEP	-47.0		4.14	not installed	0.00123
MW-504S	SHALLOW	0.0		4.10	not installed	0.00085
MW-505B	BEDROCK	-72.8		4.14	not installed	
MW-505D	DEEP	-58.6		4.21	not installed	-0.00493
MW-505M	INTERMEDIATE	-10.0		4.10	not installed	0.00226
MW-505S	SHALLOW	-0.3		3.90	not installed	0.02062

TABLE 3-2 (cont.)
SUMMARY OF VERTICAL GRADIENTS
REMEDIAL INVESTIGATION
RAYMARK - OU2 - GROUNDWATER
STRATFORD, CONNECTICUT

Monitoring Well Identifier	Aquifer	Elevation of Midpoint of Well Screen	Water Level Elevations		Vertical Gradients [1,2]	
			3/24/1999	4/15/2003	1999	2003
MW-506B	BEDROCK	-79.9		10.47	not installed	
MW-506D	DEEP	-62.8		6.66	not installed	0.22281
MW-506M	INTERMEDIATE	10.5		6.59	not installed	0.00096
MW-506S	SHALLOW	6.6		6.55	not installed	0.01020
MW-507B	BEDROCK	-38.4		3.52	not installed	
MW-507D	DEEP	-24.6		3.67	not installed	-0.01087
MW-507S	SHALLOW	-0.1		3.74	not installed	-0.00286
MW-508B	BEDROCK	-52.1		3.57	not installed	
MW-508D	DEEP	-39.2		3.61	not installed	-0.00310
MW-508S	SHALLOW	1.2		3.63	not installed	-0.00050
MW-514B	BEDROCK	-69.6		3.74	not installed	
MW-514D	DEEP	-49.3		4.07	not installed	-0.01626
MW-514S	SHALLOW	3.1		4.15	not installed	-0.00153
MW-533B	BEDROCK	-98.0		4.82	not installed	
MW-533D	DEEP	-89.2		4.78	not installed	0.00455
MW-533S	SHALLOW	2.9		5.11	not installed	-0.00358
PC-01B	BEDROCK	-80.5	4.58	5.18		
PC-01D	DEEP	-60.2	4.59	5.15	-0.00049	0.00148
PC-01M	INTERMEDIATE	-33.6	4.62	5.15	-0.00113	0.00000
PC-01S	SHALLOW	-3.0	4.70	5.19	-0.00261	-0.00131
PC-02B	BEDROCK	-107.6	4.20	4.61		
PC-02D	DEEP	-84.6	4.41	4.61	-0.00913	0.00000
PC-02M	INTERMEDIATE	-58.1	4.38	4.84	0.00113	-0.00868
PC-02S	SHALLOW	-0.3	4.44	4.61	-0.00104	0.00398
PC-03B	BEDROCK	-26.1	4.61	4.97		
PC-03D	DEEP	-7.2	4.62	5.06	-0.00053	-0.00477
PC-03S	SHALLOW	1.2	4.63	5.10	-0.00119	-0.00477
PC-04B	BEDROCK	-38.4	4.64	5.02		
PC-04D	DEEP	-17.5	4.64	5.09	0.00000	-0.00334
PC-04S	SHALLOW	-1.5	4.66	5.07	-0.00125	0.00125

TABLE 3-2 (cont.)
SUMMARY OF VERTICAL GRADIENTS
REMEDIAL INVESTIGATION
RAYMARK - OU2 - GROUNDWATER
STRATFORD, CONNECTICUT

Monitoring Well Identifier	Aquifer	Elevation of Midpoint of Well Screen	Water Level Elevations		Vertical Gradients [1,2]	
			3/24/1999	4/15/2003	1999	2003
PC-05B	BEDROCK	-75.5	4.64	5.11		
PC-05D	DEEP	-51.0	4.66	5.11	-0.00082	0.00000
PC-05M	INTERMEDIATE	-39.0	4.68	4.87	-0.00167	0.02000
PC-05S	SHALLOW	1.5	4.49	5.18	0.00469	-0.00766
PC-06B	BEDROCK	-98.8	4.79	5.20		
PC-06D	DEEP	-57.7	4.72	5.09	0.00170	0.00268
PC-06M	INTERMEDIATE	-37.5	4.77	5.11	-0.00247	-0.00099
PC-06S	SHALLOW	1.1	4.51	5.21	0.00673	-0.00259
PC-08B	BEDROCK	-35.8	4.44	4.83		
PC-08D	DEEP	-15.1	3.44	4.91	0.04824	-0.00386
PC-08S	SHALLOW	-1.0	4.46	4.92	-0.07234	-0.00071
PC-09D	DEEP	-20.7	4.27	4.71		
PC-09S	SHALLOW	-3.0	4.34	4.30	-0.00395	0.02316
PC-10B	BEDROCK	-60.8	5.48	6.17		
PC-10D	DEEP	-42.3	5.37	6.04	0.00595	0.00703
PC-10M	INTERMEDIATE	-22.8	5.33	6.01	0.00205	0.00154
PC-10S	SHALLOW	0.3	5.36	6.04	-0.00130	-0.00130
PC-11B	BEDROCK	-54.4	22.30	22.30		
PC-11D	DEEP	-33.3	4.74	5.20	0.83381	0.81197
PC-11M	INTERMEDIATE	-18.3	4.80	5.36	-0.00400	-0.01067
PC-11S	SHALLOW	-1.6	4.67	5.05	0.00780	0.01861
PC-12B	BEDROCK	-47.0	4.83	4.51		
PC-12D	DEEP	-25.1	4.35	4.79	0.02192	-0.01279
PC-12S	SHALLOW	-2.5	4.29	4.87	0.00265	-0.00353
PC-13B	BEDROCK	-74.9	5.85	6.48		
PC-13D	DEEP	-56.0	5.79	6.53	0.00317	-0.00264
PC-13M	INTERMEDIATE	-35.1	5.85	6.55	-0.00287	-0.00096
PC-13S	SHALLOW	1.5	5.85	6.56	0.00000	-0.00027
PC-14B	BEDROCK	-47.8	5.49	5.83		
PC-14D	DEEP	-25.0	4.99	5.95	0.02193	-0.00526
PC-14S	SHALLOW	-0.1	5.33	6.23	-0.01365	-0.01124

TABLE 3-2 (cont.)
SUMMARY OF VERTICAL GRADIENTS
REMEDIAL INVESTIGATION
RAYMARK - OU2 - GROUNDWATER
STRATFORD, CONNECTICUT

Monitoring Well Identifier	Aquifer	Elevation of Midpoint of Well Screen	Water Level Elevations		Vertical Gradients [1,2]	
			3/24/1999	4/15/2003	1999	2003
PC-15B	BEDROCK	-28.6	5.46	5.80		
PC-15D	DEEP	-9.0	5.51	6.32	-0.00255	-0.02657
PC-15S	SHALLOW	0.4	5.39	6.34	0.01277	-0.00213
PC-16B	BEDROCK	-85.3	6.04	6.91		
PC-16D	DEEP	-55.8	6.08	6.89	-0.00136	0.00068
PC-16M	INTERMEDIATE	-23.3	6.06	7.03	0.00062	-0.00431
PC-16S	SHALLOW	2.3	5.94	6.97	0.00469	0.00234
MW-1B	BEDROCK	-50.4	4.48	5.07		
MW-1M	INTERMEDIATE	-17.2	4.60	5.05	-0.00361	0.00060
MW-1S	SHALLOW	3.6	4.60	5.07	0.00000	-0.00097
MW-4B	BEDROCK	-29.1	4.51	5.38		
MW-4M (MW-4D)	INTERMEDIATE	-7.2	4.57	4.97	-0.00274	0.01872
MW-4S	SHALLOW	2.7	4.54	5.02	0.00303	-0.00505

Notes:
1) The vertical gradients were calculated from adjacent monitoring wells in a cluster (MW-104B vs MW104D, MW-104D vs MW-104M, etc.).
2) Positive value indicates upward vertical gradient; - indicates downward vertical gradient.
3) All measurements reported in feet. Elevations are based on feet mean sea level (NGVD 1929).

TABLE 3-3
SOIL AIR PERMEABILITY TEST RESULTS
REMEDIAL INVESTIGATION
RAYMARK – OU2 – GROUNDWATER
STRATFORD, CONNECTICUT

Test Location		**Flow Rate** (ft^3/hr)	**Atmospheric Pressure** (in. Hg)	**Gauge 2 Pressure** (in. H_20)	**Permeability** (cm^2)
P-1	Housatonic Ave.	83	30.06	4.1	$6.4x10^{-7}$
P-2	Minor Ave.	87	30.05	6.0	$3.4x10^{-7}$
P-3	Willow Ave.	60	30.07	5.0	$2.5x10^{-7}$
P-4	Willow Ave.	90	30.47	4.5	$6.3x10^{-7}$
P-5	Housatonic Ave.	90	30.36	4.0	$8.3x10^{-7}$

TABLE 4-1
CHEMICALS OR MATERIALS USED OR HANDLED AT THE RAYMARK FACILITY
REMEDIAL INVESTIGATION
RAYMARK – OU2 – GROUNDWATER
STRATFORD, CONNECTICUT

CHEMICAL COMPOUND/MATERIAL	DESCRIPTION	INFORMATION SOURCES			
		NO. 1	NO. 2	NO. 3	NO. 4
Acetone	2-Propanone	X		X	
Adhesive CR04					X
Alcohol			X		X
Aluminum	Alumina		X		X
Ammonia Aqua		X			X
Arco 4545					X
Asbestos			X	X	X
Boiler Feed Water		X			
1-Butanol	N-Butyl Alcohol			X	
2-Butanone	MEK			X	
N-Butyl Alcohol				X	
Carbon Tetrachloride	Perchloromethane			X	
Caustic	Sodium Hydroxide	X			X
Caustic Liquid/Sludge	Sodium Hydroxide				X
China Oil					X
Chinawood Oil	Meta Para Cresol; Phenolic Mixture	X			
Ching Oil					X
Chlorinated Fluorocarbons				X	
Coal	Natural Solid		X		
Coal Tar Resin	Petroleum-Like Fuel				X
Copper			X		
Cotton			X		
Cresolic Acid	Cresol; Methylphenol			X	
Cresylic Acid	Cresol; Methylphenol	X		X	X
Denatured Alcohol		X			
Denatured Ethanol					X
Dust (Dry)					X
Dust (Wet)					X
Fiberglass Fibers			X		
Fire Water		X			
Formaldehyde Resin					X
Formaldehyde (37%)		X		X	
#2 Fuel Oil	Diesel Oil	X			
#6 Fuel Oil		X	X		
Gilsonite	Asphaltic Material				X
Graphite	Black Lead		X		
Hexamethylene Tetramine	Methanamine				X
Hycar Rubber			X		
Hydraulic Oil			X		
Iron Hydroxide Sludge			X		

TABLE 4-1 (cont.)
CHEMICALS OR MATERIALS USED OR HANDLED AT THE RAYMARK FACILITY
REMEDIAL INVESTIGATION
RAYMARK – OU2 – GROUNDWATER
STRATFORD, CONNECTICUT

CHEMICAL COMPOUND/MATERIAL	DESCRIPTION	INFORMATION SOURCES			
		NO. 1	NO. 2	NO. 3	NO. 4
Latex	Hydrocarbon Polymer	X	X		X
Lead			X	X	X
Linseed Oil	Flaxseed Oil	X			
Liquid Phenolic Resin	Condensation of Phenol with Aldehydes		X		
Meta Para Cresol	Phenolic Mixture	X			
Methanol	Methyl Alcohol				X
Methylbenzene	Toluene			X	
Methyl Chloride	Dichloromethane			X	
Methyl Chloroform	1,1,1-Trichloroethane		X	X	
Methylethyl Ketone	2-Butanone	X		X	X
Methylphenol	Cresol			X	
Mineral Spirits					X
Monochlorobenzene	Phenyl Chloride	X			X
Muriatic Acid	Hydrochloric Acid		X		
Naptha	Petroleum Product	X	X		
Nitric Acid		X	X		
Nylon					X
Phenol	Tung Oil	X	X	X	X
Phenol Formaldehyde Copolymer	Synthetic Thermosetting Polymer				X
Phenolic Resin	Condensation of Phenol with Aldehydes				X
Phenolic Resin 424					X
Phenolic Resin 439					X
Phenolic Resin 478					X
Pickle Liquor	Waste Acid Containing Dissolved Metals			X	
Polybutadiene Resin	Synthetic Thermoplastic Polymer				X
Powdered Metals					X
2-Propanone	Acetone	X		X	
Process CNSL		X			X
Raw Cashew Nut Oil		X			X
RC 439	477 Saturant	X			
RC 845					X
Reclaimed City Water		X			
Red Oxide	Iron Oxide		X		
Resin Solution CR04					X
Rinsate Water					X
Rubber	Polyisoprene		X		
Rubber Cement			X		

TABLE 4-1 (cont.)
CHEMICALS OR MATERIALS USED OR HANDLED AT THE RAYMARK FACILITY
REMEDIAL INVESTIGATION
RAYMARK – OU2 – GROUNDWATER
STRATFORD, CONNECTICUT

CHEMICAL COMPOUND/MATERIAL	DESCRIPTION	INFORMATION SOURCES			
		NO. 1	NO. 2	NO. 3	NO. 4
Sartomer 845					X
Saturant 295E	90% Anacardic Acid; Sulfur Blistering Compound	X			
Saturant 439		X			
Saturant 451		X			X
Saturant 500-3					X
Saturant 500-F					X
Saturant 8240		X			
Saturant 850F					X
Saturant 851					X
Saturant RC 581					X
Scrap Resin	Petroleum and Mineral Spirits	X			
Scrap Saturant					X
#3 Sludge					X
Soap Saturant 850F					X
Solvent 204		X			
Steel			X		X
Steel Wool			X		
Sulfuric Acid	Battery Acid		X		
Tetrachloroethylene	Perchloroethylene (PCE)			X	
Textile Spirits					X
Toluene				X	X
Toluol	Cresol	X	X		
1,1,1-Trichloroethane (TCA)		X	X	X	
Trichloroethylene (TCE)				X	
Tung Oil		X			X
Unleaded Gasoline		X			
Varsol	Petroleum Aliphatic Solvents				X
Varsol #18		X			X
Vegetable Oil					X
VMP Naptha	Varnish; Petroleum Spirits	X			
Waste Oil		X			
White Water		X	X		X

Information Sources:
No. 1 - Overall Site Plan, Sheet No. S1 (ELI, 1995).
No. 2 - RCRA Facility Investigation Report, Section 2.0 (ELI, 1995).
No. 3 - RCRA Application, Part A, 8/15/80.
No. 4 - RCRA Application, Part B, 8/15/80.

TABLE 4-2
DATA SUMMARY AND CRITERIA COMPARISONS - UPGRADIENT OVERBURDEN
REMEDIAL INVESTIGATION
RAYMARK - OU2 - GROUNDWATER
STRATFORD, CONNECTICUT

PARAMETER	Detects	Count	Average	Average Detected Conc.	Minimum Conc.	Maximum Conc.	MaxLocation	ICVol	> ICVol	RVC	> RVC	SWPC	>SWPC
DIOXINS (PG/L)													
1,2,3,4,6,7,8-HpCDD	2	6	2.8	4.1	2.9	5.3 J	OU2-MW-501S-07						
1,2,3,4,6,7,8-HpCDF	1	6	1.6	3.2	3.2	3.2	OU2-MW-309D-07-AVG						
1,2,3,4,7,8,9-HpCDF	2	6	2.4	4.5	4	5 EMPC	OU2-MW-501S-07						
1,2,3,4,7,8-HxCDD	3	6	2.4	3.4	2.6	4.7 J	OU2-MW-501S-07						
1,2,3,4,7,8-HxCDF	2	6	2.1	3.7	3.1 JB	4.2	OU2-MW-309D-07-AVG						
1,2,3,6,7,8-HxCDD	2	6	1.9	3.3	2.2	4.4 EMPC	OU2-MW-501S-07						
1,2,3,6,7,8-HxCDF	4	6	2.7	3.6	1.6 J	5.6 J	OU2-MW-501S-07						
1,2,3,7,8,9-HxCDD	3	6	2.7	4.1	3.9 JB	4.3 EMPC	OU2-MW-501S-07						
1,2,3,7,8,9-HxCDF	3	6	3.2	5.3	3.6 EMPC	7 J	OU2-MW-501S-07						
1,2,3,7,8-PeCDD	3	6	3.4	5.5	4.7	6.6 J	OU2-MW-501S-07						
1,2,3,7,8-PeCDF	2	6	2.9	7	5.4	8.6 EMPC	OU2-MW-501S-07						
2,3,4,6,7,8-HxCDF	3	6	2.2	3.6	2.6 JB	4.7 J	OU2-MW-501S-07						
2,3,4,7,8-PeCDF	3	6	2.4	4	2.8 EMPC	4.7 J	OU2-MW-501S-07						
2,3,7,8-TCDD	1	6	1.7	3.3	3.3	3.3	OU2-MW-309D-07-AVG						
2,3,7,8-TCDF	2	6	1.5	2.4	2.3 EMPC	2.5	OU2-MW-309D-07-AVG						
OCDD	5	6	14.6	17.3	9.8 EMPC	27.6 J	OU2-MW-501D-07						
OCDF	1	6	7.7	4.9	4.9	4.9	OU2-MW-309D-07-AVG						
Total HpCDD	2	6	2.8	4.1	2.9	5.3 J	OU2-MW-501S-07						
Total HpCDF	1	6	2.7	6.8	6.8	6.8	OU2-MW-309D-07-AVG						
Total HxCDD	2	6	7	15.8	8.1	23.5 EMPC	OU2-MW-309S-07						
Total HxCDF	2	6	6.8	14.4	12.5 EMPC	16.3	OU2-MW-309D-07-AVG						
Total PeCDD	2	6	2.8	4.9	4.7	5.1 EMPC	OU2-MW-309S-07						
Total PeCDF	3	6	4.7	8.6	2.8 EMPC	13.3 EMPC	OU2-MW-501S-07						
Total TCDD	1	6	2	3.3	3.3	3.3	OU2-MW-309D-07-AVG						
Total TCDF	1	6	1.3	2.5	2.5	2.5	OU2-MW-309D-07-AVG						
Toxicity Equivalency	4	4	3.3	3.3	0.0012 J	13 J	OU2-MW-501S-07						
METALS (UG/L)													
Barium	8	8	46.4	46.4	8.7	140	OU2-MW-309D-07-AVG						
Calcium	8	8	61600	61600	7920	222000	OU2-MW-309D-07-AVG						
Chromium	1	8	0.57	1.1	1.1 J	1.1 J	OU2-MW-313S-07					1200	0
Cobalt	2	8	0.88	2.1	2	2.2	OU2-MW-309D-07-AVG						
Copper	3	8	4.5	10.8	1.4	29	OU2-MW-313S-07					48	0
Iron	2	8	547	2160	502	3820	OU2-MW-313D-07						
Magnesium	8	8	25100	25100	1610	146000	OU2-MW-309D-07-AVG						
Manganese	7	8	611	698	17.5	2030	OU2-MW-309D-07-AVG						
Nickel	2	8	2.4	4.4	1.9 J	7	OU2-MW-309D-07-AVG					880	0
Potassium	8	8	5720	5720	1200 J	12000	OU2-MW-309D-07-AVG						

TABLE 4-2 (cont.)
DATA SUMMARY AND CRITERIA COMPARISONS - UPGRADIENT OVERBURDEN
REMEDIAL INVESTIGATION
RAYMARK - OU2 - GROUNDWATER
STRATFORD, CONNECTICUT

PARAMETER	Detects	Count	Average	Average Detected Conc.	Minimum Conc.	Maximum Conc.	MaxLocation	ICVol	> ICVol	RVC	> RVC	SWPC	>SWPC
METALS (UG/L) (cont.)													
Selenium	1	8	1.7	4.1	4.1 J	4.1 J	OU2-MW-313S-07					50	0
Sodium	8	8	67000	67000	13600	234000	OU2-MW-309D-07-AVG						
Vanadium	1	8	0.88	1.7	1.7	1.7	OU2-MW-313S-07						
Zinc	1	8	61.1	463	463	463	OU2-MW-313S-07					123	1
METHANE, ETHANE, ETHENE (UG/L)													
Methane	2	6	20	54.9	9.8	100	OU2-MW-309S-07						
SEMIVOLATILE ORGANIC ANALYSIS (UG/L)													
2,4-Dimethylphenol	1	8	9	36	36	36	OU2-MW-309D-07-AVG						
2-Chlorophenol	1	8	5	4	4	4	OU2-MW-309D-07-AVG						
2-Methylphenol	1	8	6	12	12	12	OU2-MW-309D-07-AVG						
4-Chloro-3-methylphenol	1	8	5	4	4	4	OU2-MW-309D-07-AVG						
4-Methylphenol	1	8	26	170	170	170	OU2-MW-309D-07-AVG						
Benzo(b)fluoranthene	1	8	0.1	0.2	0.2	0.2	OU2-MW-309D-07-AVG					0.3	0
Caprolactam	1	8	5	3	3	3	OU2-MW-309D-07-AVG						
Chrysene	1	8	0.1	0.2	0.16	0.16	OU2-MW-309D-07-AVG						
Naphthalene	1	8	5	4	4	4	OU2-MW-309D-07-AVG						
Phenol	1	8	14	78	78	78	OU2-MW-309D-07-AVG					92000000	0
VOLATILE ORGANIC ANALYSIS													
Chlorobenzene	1	8	6	14	14	14	OU2-MW-309D-07-AVG	6150	0	1800	0	420000	0
Methyl tert-Butyl Ether	1	8	5	4	4	4	OU2-MW-309D-07-AVG						
WET CHEMISTY (MG/L)													
Alkalinity	6	6	59.8	59.8	18	109	OU2-MW-309D-07-AVG						
Carbon Dioxide	6	6	79.9	79.9	44.1 J	114 , J	OU2-MW-309D-07-AVG, OU2-MW-309S-07						
Chloride	4	6	192	278	10	959	OU2-MW-309D-07-AVG						
Ferrous Iron	1	6	0.018	0.03	0.03	0.03	OU2-MW-311D-07						
Nitrate	5	6	2.5	3	0.065	5.08 J	OU2-MW-311S-07						
Sulfate	6	6	44.9	44.9	17.9	82.5	OU2-MW-309D-07-AVG						
Total Organic Carbon	3	6	2.1	3.2	2.8	3.61	OU2-MW-309S-07						

> = number of samples with concentrations greater than the criterion
B = Compound also detected in a quality control blank sample.
EMPC = Estimated Maximum Possible Concentration
J = Quantitation approximate
Note - Table 4-2 includes only detected contaminants.

* = From dilution analysis or Estimated Maximum Possible Concentration
RVC = CT Residential Volatilization Criteria
ICVol = CT Industrial/Commercial Volatilization Criteria
SWPC = CT Surface Water Protection Criteria

TABLE 4-3
DATA SUMMARY AND CRITERIA COMPARISONS - UPGRADIENT BEDROCK
REMEDIAL INVESTIGATION
RAYMARK - OU2 - GROUNDWATER
STRATFORD, CONNECTICUT

PARAMETER	Detects	Count	Average	Average Detected Conc.	Minimum Conc.	Maximum Conc.	MaxLocation	ICVol	> ICVol	RVC	> RVC	SWPC	>SWPC
DIOXINS (PG/L)													
1,2,3,4,6,7,8-HpCDD	1	6	4.2	17.1	17.1 J	17.1 J	OU2-MW-308B-07						
1,2,3,4,6,7,8-HpCDF	2	6	2.5	5.6	2.8 J	8.3 J	OU2-MW-308B-07						
1,2,3,4,7,8-HxCDD	1	6	1.1	1.9	1.9 J	1.9 J	OU2-MW-308B-07						
1,2,3,4,7,8-HxCDF	1	6	1.9	3.9	3.9 JB	3.9 JB	OU2-MW-309B-07						
1,2,3,6,7,8-HxCDD	2	6	1.8	3.2	3 EMPC	3.5 J	OU2-MW-308B-07						
1,2,3,6,7,8-HxCDF	1	6	1.4	3.6	3.6 JB	3.6 JB	OU2-MW-309B-07						
1,2,3,7,8,9-HxCDD	2	6	2	4	4 JB	4.1 J	OU2-MW-308B-07						
1,2,3,7,8,9-HxCDF	1	6	1.5	5	5 JB	5 JB	OU2-MW-309B-07						
1,2,3,7,8-PeCDD	1	6	1.5	4.5	4.5 J	4.5 J	OU2-MW-309B-07						
1,2,3,7,8-PeCDF	1	6	1.7	5.2	5.2 JB	5.2 JB	OU2-MW-309B-07						
2,3,4,6,7,8-HxCDF	1	6	1.2	2.3	2.3 EMPC	2.3 EMPC	OU2-MW-309B-07						
2,3,4,7,8-PeCDF	2	6	1.6	3.5	3.5 EMPC, J	3.5 EMPC, J	OU2-MW-303B-07, OU2-MW-309B-07						
OCDD	4	6	20.1	28.5	6 JB	72 J	OU2-MW-308B-07						
OCDF	2	6	5.2	5.2	4.8 JB	5.6 J	OU2-MW-308B-07						
Total HpCDD	1	6	7	33.8	33.8 J	33.8 J	OU2-MW-308B-07						
Total HpCDF	2	6	3	6.6	2.8	10.5 J	OU2-MW-308B-07						
Total HxCDD	1	6	5.4	22.3	22.3 EMPC	22.3 EMPC	OU2-MW-309B-07						
Total HxCDF	1	6	4.9	14.8	14.8 EMPC	14.8 EMPC	OU2-MW-309B-07						
Total PeCDD	1	6	1.5	4.5	4.5	4.5	OU2-MW-309B-07						
Total PeCDF	1	6	3	8.7	8.7	8.7	OU2-MW-309B-07						
Toxicity Equivalency	3	3	1	1	0.0011 J	1.8 J	OU2-MW-303B-07						
METALS (UG/L)													
Barium	11	11	47.8	47.8	4.5	308	OU2-MW-313B-07						
Calcium	11	11	51200	51200	13600	104000	OU2-MW-313B-07						
Chromium	3	11	1.2	3.1	0.62 J	7.6	OU2-MW-308DB-07					1200	0
Cobalt	2	11	0.61	1.3	1.1 J	1.5 J	OU2-MW-307B-07						
Copper	1	11	0.8	1.7	1.7 J	1.7 J	OU2-MW-309B-07					48	0
Cyanide	1	3	3.7	1.2	1.2 J	1.2 J	OU2-MW-307DB-07					52	0
Iron	4	11	112	274	92.3	394	OU2-MW-311B-07						
Magnesium	11	11	11000	11000	3760	28700	OU2-MW-313B-07						
Manganese	10	11	241	265	23.9	1110	OU2-MW-313B-07						
Nickel	1	11	2.4	3.6	3.6	3.6	OU2-MW-307DB-07					880	0
Potassium	11	11	3970	3970	1460	7860 J	OU2-MW-313B-07						
Selenium	1	11	1.6	3.3	3.3 J	3.3 J	OU2-MW-313B-07					50	0
Sodium	11	11	33800	33800	4780	108000	OU2-MW-313B-07						

TABLE 4-3 (cont.)
DATA SUMMARY AND CRITERIA COMPARISONS - UPGRADIENT BEDROCK
REMEDIAL INVESTIGATION
RAYMARK - OU2 - GROUNDWATER
STRATFORD, CONNECTICUT

PARAMETER	Detects	Count	Average	Average Detected Conc.	Minimum Conc.	Maximum Conc.	MaxLocation	ICVol	> ICVol	RVC	> RVC	SWPC	>SWPC
METALS (UG/L) (cont.)													
Vanadium	1	11	0.78	0.81	0.81 J	0.81 J	OU2-MW-307DB-07						
Zinc	1	11	6.2	9.9	9.9	9.9	OU2-MW-307DB-07					123	0
METHANE, ETHANE, ETHENE (UG/L)													
Ethene	1	10	5.5	10	10	10	OU2-MW-309B-07						
Methane	3	10	9.9	27.3	10	39	OU2-MW-309B-07						
SEMIVOLATILE ORGANIC ANALYSIS (UG/L)													
2-Chlorophenol	1	11	5	2	2 J	2 J	OU2-MW-309B-07						
Caprolactam	1	11	64	650	650 *	650 *	OU2-MW-309B-07						
Di-n-Butylphthalate	1	11	5	2	2 J	2 J	OU2-MW-309B-07					120000	0
VOLATILE ORGANIC ANALYSIS (UG/L)													
1,1,1-Trichloroethane	1	11	15	110	110	110	OU2-MW-307DB-07	50000	0	20400	0	62000	0
1,1-Dichloroethane	1	11	5	2	2 J	2 J	OU2-MW-307DB-07	50000	0	34600	0		
1,1-Dichloroethene	1	11	25	260	260	260	OU2-MW-307DB-07	6	1	1	1	96	1
Acetone	1	11	6	13	13 J	13 J	OU2-MW-501B-07	50000	0	50000	0		
Bromodichloromethane	1	11	5	6	6 J	6 J	OU2-MW-307B-07						
Chlorobenzene	1	11	110	1200	1200 *	1200 *	OU2-MW-309B-07	6150	0	1800	0	420000	0
Chloroform	1	11	7	23	23	23	OU2-MW-307B-07	710	0	287	0	14100	0
cis-1,2-Dichloroethene	2	11	5	5	4 J	6	OU2-MW-216DB-07-AVG						
Toluene	1	11	5	3	3 J	3 J	OU2-MW-309B-07	50000	0	23500	0	4000000	0
Trichloroethene	2	11	8	24	17	31	OU2-MW-216B-07	540	0	219	0	2340	0
WET CHEMISTY (MG/L)													
Alkalinity	10	10	80.2	80.2	18	172	OU2-MW-307DB-07						
Carbon Dioxide	9	10	92.8	100	55.3 J	171	OU2-MW-307DB-07						
Chloride	8	10	43	48	7	96.4	OU2-MW-309B-07						
Ferrous Iron	4	10	0.036	0.068	0.04	0.1	OU2-MW-308DB-07						
Nitrate	9	10	1.8	2	0.19 J	7.56	OU2-MW-303B-07						
Nitrite-N	3	10	0.034	0.055	0.05 J	0.06	OU2-MW-216DB-07-AVG						
Sulfate	10	10	65.2	65.2	22	259	OU2-MW-309B-07						
Sulfide	1	3	0.65	1.4	1.44 J	1.44 J	OU2-MW-309B-07						
Total Organic Carbon	7	10	3.2	4.2	2.51 J	5.17 , J	OU2-MW-307DB-07, OU2-MW-309B-07						

> = number of samples with concentrations greater than the criterion
* = From dilution analysis or Estimated Maximum Possible Concentration
B = Compound also detected in a quality control blank sample.
EMPC = Estimated Maximum Possible Concentration
Note - Table 4-3 includes only detected contaminants.

J = Quantitation approximate
RVC = CT Residential Volatilization Criteria
ICVol = CT Industrial/Commercial Volatilization Criteria
SWPC = CT Surface Water Protection Criteria

TABLE 4-4
CRITERIA FOR COMPARISON - GROUNDWATER
REMEDIAL INVESTIGATION
RAYMARK - OU2 - GROUNDWATER
STRATFORD, CONNECTICUT

Fraction	Parameter	Units	Maximum Upgradient-Bedrock	Maximum Upgradient - Overburden	Maximum Upgradient	RVC	SWPC
ASB	Asbestos	MFL					7
DIOXI	1,2,3,4,6,7,8-HpCDD	PG/L	17.1	5.3	17.1		
DIOXI	1,2,3,4,6,7,8-HpCDF	PG/L	8.3	3.2	8.3		
DIOXI	1,2,3,4,7,8,9-HpCDF	PG/L		5	5		
DIOXI	1,2,3,4,7,8-HxCDD	PG/L	1.9	4.7	4.7		
DIOXI	1,2,3,4,7,8-HxCDF	PG/L	3.9	4.2	4.2		
DIOXI	1,2,3,6,7,8-HxCDD	PG/L	3.5	4.4	4.4		
DIOXI	1,2,3,6,7,8-HxCDF	PG/L	3.6	5.6	5.6		
DIOXI	1,2,3,7,8,9-HxCDD	PG/L	4.1	4.3	4.3		
DIOXI	1,2,3,7,8,9-HxCDF	PG/L	5	7	7		
DIOXI	1,2,3,7,8-PeCDD	PG/L	4.5	6.6	6.6		
DIOXI	1,2,3,7,8-PeCDF	PG/L	5.2	8.6	8.6		
DIOXI	2,3,4,6,7,8-HxCDF	PG/L	2.3	4.7	4.7		
DIOXI	2,3,4,7,8-PeCDF	PG/L	3.5	4.7	4.7		
DIOXI	2,3,7,8-TCDD	PG/L		3.3	3.3		
DIOXI	2,3,7,8-TCDF	PG/L		2.5	2.5		
DIOXI	OCDD	PG/L	72	27.6	72		
DIOXI	OCDF	PG/L	5.6	4.9	5.6		
DIOXI	Total HpCDD	PG/L	33.8	5.3	33.8		
DIOXI	Total HpCDF	PG/L	10.5	6.8	10.5		
DIOXI	Total HxCDD	PG/L	22.3	23.5	23.5		
DIOXI	Total HxCDF	PG/L	14.8	16.3	16.3		
DIOXI	Total PeCDD	PG/L	4.5	5.1	5.1		
DIOXI	Total PeCDF	PG/L	8.7	13.3	13.3		
DIOXI	Total TCDD	PG/L		3.3	3.3		
DIOXI	Total TCDF	PG/L		2.5	2.5		
DIOXI	Toxicity Equivalency	PG/L	1.8	13	13		
M	Antimony	UG/L					86000
M	Arsenic	UG/L					4
M	Barium	UG/L	308	140	308		
M	Beryllium	UG/L					4
M	Cadmium	UG/L					6
M	Calcium	UG/L	104000	222000	222000		
M	Chromium	UG/L	7.6	1.1	7.6		1200
M	Chromium VI	UG/L					110
M	Cobalt	UG/L	1.5	2.2	2.2		
M	Copper	UG/L	1.7	29	29		48
M	Cyanide	UG/L	1.2		1.2		52
M	Iron	UG/L	394	3820	3820		
M	Lead	UG/L					13
M	Magnesium	UG/L	28700	146000	146000		
M	Manganese	UG/L	1110	2030	2030		
M	Mercury	UG/L					0.4
M	Nickel	UG/L	3.6	7	7		880
M	Potassium	UG/L	7860	12000	12000		
M	Selenium	UG/L	3.3	4.1	4.1		50
M	Silver	UG/L					12
M	Sodium	UG/L	108000	234000	234000		
M	Thallium	UG/L					63
M	Vanadium	UG/L	0.81	1.7	1.7		
M	Zinc	UG/L	9.9	463	463		123
MEE	Ethene	UG/L	10		10		
MEE	Methane	UG/L	39	100	100		

TABLE 4-4 (cont.)
CRITERIA FOR COMPARISON - GROUNDWATER
REMEDIAL INVESTIGATION
RAYMARK - OU2 - GROUNDWATER
STRATFORD, CONNECTICUT

Fraction	Parameter	Units	Maximum Upgradient-Bedrock	Maximum Upgradient - Overburden	Maximum Upgradient	RVC	SWPC
O2ETC	Carbon Dioxide	MG/L	171	114	171		
OS	1,2-Dichlorobenzene	UG/L				30500	170000
OS	1,3-Dichlorobenzene	UG/L				24200	26000
OS	1,4-Dichlorobenzene	UG/L				50000	26000
OS	2,4-Dichlorophenol	UG/L					15800
OS	2,4-Dimethylphenol	UG/L		36	36		
OS	2-Chlorophenol	UG/L	2	4	4		
OS	2-Methylphenol	UG/L		12	12		
OS	4-Chloro-3-methylphenol	UG/L		4	4		
OS	4-Methylphenol	UG/L		170	170		
OS	Acenaphthylene	UG/L					0.3
OS	Anthracene	UG/L					1100000
OS	Benzene	UG/L				215	710
OS	Benzo(a)anthracene	UG/L					0.3
OS	Benzo(a)pyrene	UG/L					0.3
OS	Benzo(b)fluoranthene	UG/L		0.2	0.2		0.3
OS	Benzo(k)fluoranthene	UG/L					0.3
OS	Bis(2-Chloroethyl)ether	UG/L					42
OS	bis(2-Ethylhexyl)phthalate	UG/L					59
OS	Caprolactam	UG/L	650	3	650		
OS	Chrysene	UG/L		0.16	0.16		
OS	Di-n-Butylphthalate	UG/L	2		2		120000
OS	Fluoranthene	UG/L					3700
OS	Fluorene	UG/L					140000
OS	Hexachlorobenzene	UG/L					0.077
OS	Hexachloroethane	UG/L					89
OS	Naphthalene	UG/L		4	4		
OS	Phenanthrene	UG/L					0.077
OS	Phenol	UG/L		78	78		92000000
OS	Pyrene	UG/L					110000
OV	1,1,1,2-Tetrachloroethane	UG/L				12	
OV	1,1,1-Trichloroethane	UG/L	110		110	20400	62000
OV	1,1,2,2-Tetrachloroethane	UG/L				23	110
OV	1,1,2-Trichloroethane	UG/L				8000	1260
OV	1,1-Dichloroethane	UG/L	2		2	34600	
OV	1,1-Dichloroethene*	UG/L	260		260	1	96
OV	1,2-Dibromoethane	UG/L				4	
OV	1,2-Dichloroethane	UG/L				21	2970
OV	1,2-Dichloropropane	UG/L				14	
OV	2-Butanone	UG/L				50000	
OV	4-Methyl-2-Pentanone	UG/L				50000	
OV	Acetone	UG/L	13		13	50000	
OV	Acrylonitrile	UG/L					20
OV	bis-(2-chloroisopropyl)ether	UG/L					3400000
OV	Bromodichloromethane	UG/L	6		6		
OV	Bromoform	UG/L				920	10800
OV	Carbon Tetrachloride	UG/L				16	132
OV	Chlorobenzene	UG/L	1200	14	1200	1800	420000
OV	Chloroform	UG/L	23		23	287	14100
OV	cis-1,2-Dichloroethene	UG/L	6		6		
OV	cis-1,3-Dichloropropene	UG/L				6	34000
OV	Dibromochloromethane	UG/L					1020
OV	Ethylbenzene	UG/L				50000	580000

TABLE 4-4 (cont.)
CRITERIA FOR COMPARISON - GROUNDWATER
REMEDIAL INVESTIGATION
RAYMARK - OU2 - GROUNDWATER
STRATFORD, CONNECTICUT

Fraction	Parameter	Units	Maximum Upgradient-Bedrock	Maximum Upgradient - Overburden	Maximum Upgradient	RVC	SWPC
OV	Methyl tert-Butyl Ether	UG/L		4	4		
OV	Methylene Chloride	UG/L				50000	48000
OV	Styrene	UG/L				580	
OV	Tetrachloroethene	UG/L				1500	88
OV	Toluene	UG/L	3		3	23500	4000000
OV	Total Xylenes	UG/L				21300	
OV	Trichloroethene*	UG/L	31		31	219	2340
OV	Vinyl Chloride	UG/L				2	15750
PESTP	Aroclor, Total	UG/L					0.5
PESTP	Aroclor-1016	UG/L					0.5
PESTP	Aroclor-1221	UG/L					0.5
PESTP	Aroclor-1232	UG/L					0.5
PESTP	Aroclor-1242	UG/L					0.5
PESTP	Aroclor-1248	UG/L					0.5
PESTP	Aroclor-1254	UG/L					0.5
PESTP	Aroclor-1260	UG/L					0.5
PESTP	Aroclor-1262	UG/L					0.5
PESTP	Aroclor-1268	UG/L					0.5
PESTP	Dieldrin	UG/L					0.1
PESTP	Heptachlor	UG/L					0.05
PESTP	Heptachlor Epoxide	UG/L					0.05
PESTP	alpha-Chlordane	UG/L					0.3
PESTP	Endrin	UG/L					0.1
PESTP	gamma-Chlordane	UG/L					0.3
WET	Alkalinity	MG/L	172	109	172		
WET	Chloride	MG/L	96.4	959	959		
WET	Ferrous Iron	MG/L	0.1	0.03	0.1		
WET	Nitrate	MG/L	7.56	5.08	7.56		
WET	Nitrite-N	MG/L	0.06		0.06		
WET	Sulfate	MG/L	259	82.5	259		
WET	Sulfide	MG/L	1.44		1.44		
WET	Total Organic Carbon	MG/L	5.17	3.61	5.17		

DIOXI - Dioxins
M-metals
MEE - Methane, Ethane, Ethene
O2ETC - Oxygen, CO_2, Methane
OS-SVOCs
OV-CLP VOCs
PESTP-pesticides/PCBs
WET-other wet chemistry parameters

MLF - Million Fiber per liter
PG/L - Picogram per liter
UG/L - microgram per liter
MG/L - Milligram per liter
RVC - CT Residential Volatilization Criteria
SWPC - CT Surface Water Protection Criteria

* CT RVC listed above are the existing promulgated criteria. CT DEP has proposed new RVC for 1,1-DCE of 190 ug/L and for TCE of 27 ug/L.
Refer to Figure 2-1 for upgradient well locations.
Table 4-4 includes only detected contaminants.

TABLE 4-5
DATA SUMMARY AND CRITERIA COMPARISONS - OVERBURDEN AND RECOVERY
REMEDIAL INVESTIGATION
RAYMARK - OU2 - GROUNDWATER
STRATFORD, CONNECTICUT

PARAMETER	Detects	Count	Average	Average Detected Conc.	Minimum Conc.	Maximum Conc.	MaxLocation	UPGRADIENT OVERBURDEN	>UPGRADIENT OVERBURDEN	ICVol	>ICVol	RVC	> RVC	SWPC	>SWPC
DIOXINS (PG/L)															
1,2,3,4,6,7,8-HpCDD	23	144	2.5	6.5	1.5 , EMPC	44.6	OU1-MW-PC10S-07-AVG	5.3	8						
1,2,3,4,6,7,8-HpCDF	15	144	1.4	3.9	0.83	15.1 J	OU2-MW-530S-07	3.2	6						
1,2,3,4,7,8,9-HpCDF	7	144	1.4	4.1	0.82	7.1 J	OU1-MW-PC05M-07	5	3						
1,2,3,4,7,8-HxCDD	14	144	1.3	3.2	1	6.8 J	OU1-MW-PC05M-07	4.7	4						
1,2,3,4,7,8-HxCDF	10	144	1.2	2.4	0.72	5.9 J	OU2-MW-535S-07	4.2	1						
1,2,3,6,7,8-HxCDD	13	144	1.2	2.8	0.92	7.3 J	OU2-MW-217S-07	4.4	3						
1,2,3,6,7,8-HxCDF	15	144	1	2.7	0.57	7.3	OU1-MW-PC16D-07-AVG	5.6	3						
1,2,3,7,8,9-HxCDD	21	144	1.4	2.6	1.2 , EMPC	5.6 EMPC	OU2-MW-211D-07	4.3	3						
1,2,3,7,8,9-HxCDF	14	144	1.2	2.6	0.95	6.5 J	OU2-MW-504D-07	7	0						
1,2,3,7,8-PeCDD	17	144	1.4	4.3	1.1	8.6 J	OU1-MW-PC05M-07	6.6	6						
1,2,3,7,8-PeCDF	13	143	1.2	3.4	0.71	10.6 J	OU1-MW-PC05M-07	8.6	1						
2,3,4,6,7,8-HxCDF	13	144	1	2	0.82	4.6 J	OU2-MW-504D-07	4.7	0						
2,3,4,7,8-PeCDF	16	144	1.1	3.1	0.82 EMPC	8.9 J	OU2-MW-211D-07	4.7	3						
2,3,7,8-TCDD	5	144	0.97	2.8	1.1	4.8 J	OU1-MW-PC05M-07	3.3	2						
2,3,7,8-TCDF	13	144	0.78	1.9	0.68	3.8 J	OU2-MW-212S-07	2.5	4						
OCDD	41	144	15.6	41.9	2.9	375	OU2-MW-515S-07-AVG	27.6	9						
OCDF	22	144	4.6	10.3	2.1	96.4	OU2-MW-530S-07	4.9	15						
Total HpCDD	24	144	3	9.2	1.5 EMPC	71.1	OU1-MW-PC10S-07-AVG	5.3	8						
Total HpCDF	14	144	1.9	6.4	0.87	37.9 EMPC	OU2-MW-530S-07	6.8	3						
Total HxCDD	9	144	2.6	10.4	1.4	25.1 EMPC	OU1-MW-PC11S-07	23.5	1						
Total HxCDF	11	144	2.8	7.5	0.57	22.4 J	OU2-MW-504D-07	16.3	2						
Total PeCDD	14	144	1.5	3.8	1.1	7.2 J	OU2-MW-211D-07	5.1	4						
Total PeCDF	18	144	1.7	4.4	0.78	16.9 EMPC	OU2-MW-504D-07	13.3	1						
Total TCDD	12	144	1.1	2	1.1	3.6 J	OU2-MW-211D-07	3.3	1						
Total TCDF	17	144	1.5	6.9	0.46 J	35.9 EMPC	OU2-MW-535S-07	2.5	7						
Toxicity Equivalency	63	63	2	2	0.00029 J	18 J	OU2-MW-211D-07	13	3						
METALS (UG/L)															
Aluminum	23	181	7610	59600	411	587000 J	OU2-MW-533D-07								
Antimony	2	181	1.6	1.6	1.2 J	2 J	OU2-MW-505D-07							86000	0
Arsenic	49	178	13.1	42.3	3.2	148	OU2-MW-505M-07-AVG							4	48
Barium	152	179	114	131	2.4	4570	OU4-MW-6-07	140	12						
Beryllium	12	180	1.9	26.1	0.14 J	106 J	OU2-MW-533D-07							4	8
Cadmium	42	180	17.6	74.6	0.33 J	553	OU2-MW-9-07-AVG							6	23
Calcium	179	181	74900	75700	4400	628000	OU1-MW-PC01D-07	222000	10						
Chromium	82	181	10.9	23.5	0.38	603	OU2-MW-4-07	1.1	58					1200	0

TABLE 4-5 (cont.)
DATA SUMMARY AND CRITERIA COMPARISONS - OVERBURDEN AND RECOVERY
REMEDIAL INVESTIGATION
RAYMARK - OU2 - GROUNDWATER
STRATFORD, CONNECTICUT

PARAMETER	Detects	Count	Average	Average Detected Conc.	Minimum Conc.	Maximum Conc.	MaxLocation	UPGRADIENT OVERBURDEN	>UPGRADIENT OVERBURDEN	ICVol	>ICVol	RVC	> RVC	SWPC	>SWPC
METALS (UG/L) (cont.)															
Cobalt	106	180	74.6	126	0.82	2710 J	OU2-MW-211D-07, OU2-MW-533D-07	2.2	85						
Copper	73	180	26.3	63.4	0.48	1300 J	OU1-MW-PC14D-07	29	19					48	14
Cyanide	9	48	4.9	9.6	1 J	48.4 J	OU1-MW-PC14D-07							52	0
Iron	112	181	37900	61200	15.9	2240000	OU2-MW-533D-07	3820	65						
Lead	26	180	3.8	20.1	1.4 J	199 J	OU2-MW-211D-07							13	6
Magnesium	180	181	44300	44500	509	724000	OU2-MW-530S-07	146000	11						
Manganese	171	181	7810	8260	2 J	138000	OU2-MW-111D-07	2030	87						
Mercury	9	181	0.079	0.55	0.08	2.8	OU2-MW-101S-07							0.4	2
Nickel	135	179	294	389	1.2 J	17900	OU1-MW-PC16M-07	7	85					880	11
Potassium	181	181	13300	13300	670	228000 J	OU2-MW-530S-07	12000	46						
Selenium	19	180	3.3	15.6	2.6	51.4 J	OU2-MW-531S-07	4.1	16					50	1
Silver	30	169	0.7	2.1	0.67 J	8.9 J	OU2-MW-530S-07							12	0
Sodium	181	181	195000	195000	3080	5740000	OU2-MW-530S-07	234000	16						
Thallium	26	181	6.2	28.8	2.4	117 J	OU1-MW-PC01D-07							63	3
Vanadium	37	180	1.7	6.3	0.46	52.8 J	OU2-MW-211D-07	1.7	20						
Zinc	90	180	329	650	1.6 J	10800	OU1-MW-PC16M-07	463	21					123	36
METHANE, ETHANE, ETHENE (UG/L)															
Ethane	10	101	7.9	34.1	12	110	OU2-MW-304S-07-AVG								
Ethene	4	101	8.3	88.8	23	230	OU2-MW-304D-07								
Methane	75	101	1590	2140	5.3	20000	OU2-MW-CRA6D-07	100	46						
SEMIVOLATILE ORGANIC ANALYSIS (UG/L)															
2,4-Dimethylphenol	3	182	9	250	1 J	700 *	OU4-MW-6-07	36	2						
2,4-Dinitrophenol	5	183	24	410	2 J	1700 *J	OU1-MW-PC14D-07								
2-Chlorophenol	4	182	6	12	2	20	OU1-MW-PC03D-07-AVG	4	3						
2-Methylnaphthalene	5	183	5	11	1 J	41	OU2-MW-1S-07								
2-Methylphenol	3	183	5	22	7 J	30 J	OU2-MW-110S-07	12	2						
2-Nitrophenol	8	183	6	21	2 J	56 J	OU1-MW-PC14D-07								
4,6-Dinitro-2-methylphenol	5	183	14	28	1 , J	110 J	OU1-MW-PC14D-07								
4-Chloro-3-methylphenol	2	182	5	1	1 J	1 J	OU2-MW-504D-07, OU2-MW-506M-07	4	0						
4-Methylphenol	7	182	9	120	1 J	720	OU2-MW-110S-07	170	1						
4-Nitrophenol	30	183	29	110	1 J	890 *J	OU1-MW-PC14D-07								

TABLE 4-5 (cont.)
DATA SUMMARY AND CRITERIA COMPARISONS - OVERBURDEN AND RECOVERY
REMEDIAL INVESTIGATION
RAYMARK - OU2 - GROUNDWATER
STRATFORD, CONNECTICUT

PARAMETER	Detects	Count	Average	Average Detected Conc.	Minimum Conc.	Maximum Conc.	MaxLocation	UPGRADIENT OVERBURDEN	>UPGRADIENT OVERBURDEN	ICVol	>ICVol	RVC	> RVC	SWPC	>SWPC
SEMIVOLATILE ORGANIC ANALYSIS (UG/L) (cont.)															
Acenaphthene	6	183	6	10	2 , J	27	OU2-MW-503S-07								
Acenaphthylene	3	183	0.1	1	0.45	1.3	OU1-MW-PC02S-07							0.3	3
Benzaldehyde	2	183	6	60	1 J	120 J	OU1-MW-PC14D-07								
Benzo(b)fluoranthene	1	184	0.1	0.3	0.32	0.32	OU2-MW-CRA6D-07	0.2	1					0.3	1
bis(2-Ethylhexyl)phthalate	5	183	5	2	1 J	4 J	OU1-MW-PC11S-07							59	0
Caprolactam	27	183	7	18	2 J	91 J*	OU2-MW-BR-2-07	3	25						
Carbazole	2	183	5	6	3	9 J	OU1-MW-PC02S-07								
Dibenzofuran	3	183	5	4	2	6 J	OU1-MW-PC02S-07								
Diethylphthalate	2	183	6	71	2	140 *	OU1-MW-PC08S-07								
Di-n-Butylphthalate	1	183	5	1	1 J	1 J	OU2-MW-506M-07							120000	0
Fluoranthene	1	183	5	2	2 J	2 J	OU1-MW-PC02S-07							3700	0
Fluorene	4	183	5	6	2	10	OU1-MW-PC02S-07							140000	0
Naphthalene	14	183	7	24	1 , J	260 *	OU2-MW-1S-07	4	6						
Nitrobenzene	1	183	5	1	1 J	1 J	OU1-MW-PC14D-07								
N-Nitroso-diphenylamine	2	183	5	5	5 , J	5 , J	OU2-MW-CRA5S-07, OU2-MW-CRA6S-07-AVG								
Phenanthrene	15	184	0.2	1	0.2	3.1	OU1-MW-PC02S-07							0.077	15
Phenol	9	181	51	930	1 J	8300 *	OU2-MW-110S-07	78	1					92000000	0
Pyrene	2	183	5	2	1 J	4	OU2-MW-207MR-07-AVG							110000	0
VOLATILE ORGANIC ANALYSIS (UG/L)															
1,1,1-Trichloroethane	87	190	1400	2900	1 J	160000 *	OU1-MW-PC02D-07			50000	1	20400	2	62000	1
1,1,2-Trichloroethane	31	184	5	7	1 J	49	OU2-MW-211D-07			19600	0	8000	0	1260	0
1,1-Dichloroethane	107	184	83	140	1 J	1900 *	OU2-MW-CRA2D-07			50000	0	34600	0		
1,1-Dichloroethene	86	190	430	960	1	34000 *	OU1-MW-PC02D-07			6	72	1	85	96	39
1,2-Dichlorobenzene	2	184	5	8	8 J	9 J	OU2-MW-BR-1-07			50000	0	30500	0	170000	0
1,2-Dichloroethane	21	184	5	5	2 J	13	OU2-MW-111D-07			90	0	21	0	2970	0
1,2-Dichloropropane	2	184	5	32	8	56 J	OU2-MW-CRA5S-07			60	0	14	1		
1,4-Dioxane	45	159	140	360	55 J	1800 J	OU2-MW-502D-07								
2-Butanone	1	184	5	37	37 J	37 J	OU2-MW-110S-07			50000	0	50000	0		
2-Hexanone	1	184	5	4	4	4	OU2-MW-515S-07-AVG								
4-Methyl-2-Pentanone	2	184	5	3	2 J	4 J	OU2-MW-506M-07			50000	0	50000	0		
Acetone	11	184	6	6	2 J	9 J	OU2-MW-4M(MW-4D)-07			50000	0	50000	0		

TABLE 4-5 (cont.)
DATA SUMMARY AND CRITERIA COMPARISONS - OVERBURDEN AND RECOVERY
REMEDIAL INVESTIGATION
RAYMARK - OU2 - GROUNDWATER
STRATFORD, CONNECTICUT

PARAMETER	Detects	Count	Average	Average Detected Conc.	Minimum Conc.	Maximum Conc.	MaxLocation	UPGRADIENT OVERBURDEN	>UPGRADIENT OVERBURDEN	ICVol	>ICVol	RVC	> RVC	SWPC	>SWPC
VOLATILE ORGANIC ANALYSIS (UG/L) (cont.)															
Benzene	56	190	47	150	1 J	4600	OU2-MW-CRA6S-07-AVG			530	3	215	3	710	2
Bromodichloromethane	3	184	5	3	1 J	5 J	OU2-MW-203S-07								
Bromoform	1	184	5	1	1 J	1 J	OU2-MW-533S-07			3800	0	920	0	10800	0
Carbon Disulfide	2	184	5	29	4	53	OU2-MW-110S-07								
Chlorobenzene	90	190	110	230	1 , J	6100 *	OU1-MW-PC03S-07	14	47	6150	0	1800	3	420000	0
Chloroethane	24	184	11	54	2 J	380	OU2-MW-522S-07								
Chloroform	51	184	6	7	1 J	61	OU1-MW-PC14S-07			710	0	287	0	14100	0
cis-1,2-Dichloroethene	112	190	160	270	1 , J	2500 *	OU1-MW-PC14S-07								
Cyclohexane	13	184	9	56	2 J	340 *J	OU2-MW-CRA5S-07								
Ethylbenzene	15	184	19	180	2 , J	1600 *	OU1-MW-PC04S-07			50000	0	50000	0	580000	0
Isopropylbenzene	14	184	6	16	1 J	60	OU2-MW-1S-07								
Methyl tert-Butyl Ether	45	184	53	200	1 J	2700 *	OU2-MW-206MR-07	4	20						
Methylcyclohexane	19	184	41	350	1 J	5400 *J	OU2-MW-CRA5S-07								
Methylene Chloride	9	184	8	63	1 J	190	OU1-MW-PC14D-07			50000	0	50000	0	48000	0
Tetrachloroethene	49	190	5	7	1 , J	27	OU2-MW-506D-07			3820	0	1500	0	88	0
Toluene	19	190	200	1900	1 J	35000 *	OU1-MW-PC04S-07			50000	0	23500	1	4000000	0
Total Xylenes	18	184	71	680	1 J	8400 *	OU1-MW-PC04S-07			50000	0	21300	0		
trans-1,2-Dichloroethene	17	184	5	3	1 , J	6 J	OU1-MW-PC11S-07								
Trichloroethene	99	190	490	930	0.7	21000 *	OU2-MW-514D-07			540	30	219	40	2340	9
Vinyl Chloride	80	190	30	70	0.8 J	530 *	OU1-MW-PC14S-07			2	67	2	67	15750	0
PESTICIDE/PCB (UG/L)															
4,4'-DDD	1	183	0.05	0.14	0.14	0.14	OU4-MW-6-07								
4,4'-DDE	1	183	0.051	0.25	0.25	0.25	OU4-MW-6-07								
4,4'-DDT	1	183	0.05	0.063	0.063 J	0.063 J	OU2-MW-4M(MW-4D)-07								
alpha-BHC	4	183	0.028	0.12	0.057 J	0.23	OU1-MW-PC16D-07-AVG								
beta-BHC	2	183	0.025	0.07	0.06	0.08	OU2-MW-210S-07								
delta-BHC	1	183	0.025	0.059	0.059	0.059	OU2-MW-CRA6S-07-AVG								
Dieldrin	2	183	0.026	0.16	0.072	0.25	OU2-MW-302S-07							0.1	1
Endosulfan I	1	183	0.025	0.11	0.11	0.11	OU4-MW-6-07								
gamma-BHC	2	182	0.027	0.2	0.13 J	0.27 J	OU1-MW-PC12D-07								
gamma-Chlordane	1	183	0.025	0.073	0.073	0.073	OU4-MW-6-07							0.3	0
Heptachlor	3	183	0.011	0.06	0.026	0.087 J	OU2-MW-506M-07							0.05	2
Heptachlor Epoxide	6	183	0.016	0.19	0.03 J	0.66 J	OU1-MW-PC12D-07							0.05	3

TABLE 4-5 (cont.)
DATA SUMMARY AND CRITERIA COMPARISONS - OVERBURDEN AND RECOVERY
REMEDIAL INVESTIGATION
RAYMARK - OU2 - GROUNDWATER
STRATFORD, CONNECTICUT

PARAMETER	Detects	Count	Average	Average Detected Conc.	Minimum Conc.	Maximum Conc.	MaxLocation	UPGRADIENT OVERBURDEN	>UPGRADIENT OVERBURDEN	ICVol	>ICVol	RVC	> RVC	SWPC	>SWPC
WET CHEMISTRY (MG/L)															
Alkalinity	96	102	170	181	8 J	732	OU1-MW-PC01S-07	109	58						
Carbon Dioxide	95	102	230	247	11 J	912	OU2-MW-104S-07-AVG	114	72						
Chloride	96	105	152	159	3.49	2550 J	OU2-MW-217D-07	959	2						
Ferrous Iron	65	101	52.1	81	0.03	2030 J	OU2-MW-533D-07	0.03	61						
Nitrate	61	104	6.8	11.6	0.05 J	278 J	OU1-MW-PC14D-07	5.08	23						
Nitrite-N	12	104	0.14	0.89	0.06	3.24	OU1-MW-PC14D-07								
Sulfate	97	105	570	617	5.3	13500	OU2-MW-533D-07	82.5	66						
Sulfide	20	93	0.38	0.83	0.38	2.64	OU1-MW-PC04S-07								
Total Organic Carbon	83	105	13	16.2	2.1	172	OU2-MW-CRA6D-07	3.61	75						
RECOVERY WELL															
METALS (UG/L)															
Aluminum	5	5	103000	103000	3110	408000	OU1-MW-RW1-07								
Arsenic	4	5	18.7	21.9	5.5 J	44 J	OU1-MW-RW1-07							4	4
Barium	4	5	118	142	31.3 J	404	OU1-MW-RW1-07	140	1						
Beryllium	3	5	12.7	20.6	2 J	46.8 J	OU1-MW-RW1-07							4	2
Cadmium	3	5	4.2	6.9	2.4 J	14.4 J	OU1-MW-RW1-07							6	1
Calcium	5	5	87600	87600	32600 J	233000	OU1-MW-RW1-07	222000	1						
Chromium	5	5	125	125	16.9	451	OU1-MW-RW1-07	1.1	5					1200	0
Cobalt	5	5	202	202	32.4 J	704	OU1-MW-RW1-07	2.2	5						
Copper	5	5	2610	2610	85.3	9960	OU1-MW-RW1-07	29	5					48	5
Iron	5	5	45100	45100	12700	162000	OU1-MW-RW2F-07	3820	5						
Lead	3	5	16.5	25.1	14.5	31.7	OU1-MW-RW3-07							13	3
Magnesium	5	5	80300	80300	17600 J	221000	OU1-MW-RW1-07	146000	1						
Manganese	5	5	10400	10400	2390	29000	OU1-MW-RW1-07	2030	5						
Mercury	2	5	0.62	1.5	0.15 J	2.8	OU1-MW-RW1-07							0.4	1
Nickel	5	5	522	522	145	1550	OU1-MW-RW1-07	7	5					880	1
Potassium	5	5	29600	29600	17800 J	45500 J	OU1-MW-RW2F-07	12000	5						
Silver	1	5	1.3	4.8	4.8 J	4.8 J	OU1-MW-RW2F-07							12	0
Sodium	5	5	230000	230000	148000	467000 J	OU1-MW-RW1-07	234000	1						
Thallium	3	5	14.3	22.3	6.8 J	43.2	OU1-MW-RW1-07							63	0
Zinc	5	5	2580	2580	207 J	6260	OU1-MW-RW1-07	463	4					123	5

TABLE 4-5 (cont.)
DATA SUMMARY AND CRITERIA COMPARISONS - OVERBURDEN AND RECOVERY
REMEDIAL INVESTIGATION
RAYMARK - OU2 - GROUNDWATER
STRATFORD, CONNECTICUT

PARAMETER	Detects	Count	Average	Average Detected Conc.	Minimum Conc.	Maximum Conc.	MaxLocation	UPGRADIENT OVERBURDEN	>UPGRADIENT OVERBURDEN	ICVol	>ICVol	RVC	> RVC	SWPC	>SWPC
SEMIVOLATILE ORGANIC ANALYSIS (UG/L)															
2,4-Dinitrophenol	5	5	890	890	88 *	2100 *J	OU1-MW-RW1-07								
2-Methylnaphthalene	1	5	4	2	2 J	2 J	OU1-MW-RW6-07								
2-Methylphenol	3	5	4	3	2 J	4 J	OU1-MW-RW1-07	12	0						
2-Nitrophenol	5	5	120	120	8 J	300 *J	OU1-MW-RW1-07								
4,6-Dinitro-2-methylphenol	4	4	99	99	26	250 *J	OU1-MW-RW6-07								
4-Chloro-3-methylphenol	1	3	5	5	5 J	5 J	OU1-MW-RW2-07	4	1						
4-Methylphenol	3	3	26	26	4 J	45	OU1-MW-RW2-07	170	0						
4-Nitrophenol	5	5	590	590	83 *	1400 *J	OU1-MW-RW1-07								
Acetophenone	1	5	5	3	3 J	3 J	OU1-MW-RW3-07								
Benzaldehyde	5	5	25	25	2 J	64	OU1-MW-RW1-07								
Benzo(k)fluoranthene	1	5	0.2	0.8	0.82 J	0.82 J	OU1-MW-RW1-07							0.3	1
Naphthalene	4	5	4	4	2 J	7 J	OU1-MW-RW6-07	4	2						
Phenanthrene	1	5	0.2	0.4	0.4 J	0.4 J	OU1-MW-RW1-07							0.077	1
Phenol	4	4	18	18	1 J	39 J	OU1-MW-RW5-07	78	0					92000000	0
Pyrene	1	5	4	1	1 J	1 J	OU1-MW-RW1-07							110000	0
VOLATILE ORGANIC ANALYSIS (UG/L)															
1,1,1-Trichloroethane	5	5	17000	17000	260 J	58000 *	OU1-MW-RW6A-07			50000	1	20400	2	62000	0
1,1,2-Trichloroethane	4	5	36	43	8 J	88	OU1-MW-RW1-07			19600	0	8000	0	1260	0
1,1-Dichloroethane	5	5	1700	1700	89	6200 *	OU1-MW-RW6A-07			50000	0	34600	0		
1,1-Dichloroethene	5	5	2900	2900	130	8000 *J	OU1-MW-RW6A-07			6	5	1	5	96	5
1,2-Dichloroethane	3	5	15	21	6 J	50	OU1-MW-RW1-07			90	0	21	1	2970	0
1,4-Dioxane	5	5	3700	3700	120 J	15000 J	OU1-MW-RW1-07								
2-Hexanone	1	5	6	12	12 J	12 J	OU1-MW-RW3-07								
4-Methyl-2-Pentanone	3	5	130	210	89	380 J	OU1-MW-RW6A-07			50000	0	50000	0		
Acetone	1	5	17	67	67	67	OU1-MW-RW2-07			50000	0	50000	0		
Benzene	4	5	21	25	9 J	58 J	OU1-MW-RW6A-07			530	0	215	0	710	0
Carbon Tetrachloride	1	5	6	11	11	11	OU1-MW-RW1-07			40	0	16	0	132	0
Chlorobenzene	5	5	280	280	35	800 *	OU1-MW-RW2-07	14	5	6150	0	1800	0	420000	0
Chloroethane	5	5	240	240	4 J	1000 *	OU1-MW-RW6A-07								
Chloroform	5	5	12	12	4 J	31	OU1-MW-RW1-07			710	0	287	0	14100	0
Chloromethane	2	5	4	2	2 J	3 J	OU1-MW-RW1-07								
cis-1,2-Dichloroethene	5	5	7900	7900	6 J	25000 *	OU1-MW-RW6A-07								
Ethylbenzene	4	5	65	80	13	150 J	OU1-MW-RW6A-07			50000	0	50000	0	580000	0

TABLE 4-5 (cont.)
DATA SUMMARY AND CRITERIA COMPARISONS - OVERBURDEN AND RECOVERY
REMEDIAL INVESTIGATION
RAYMARK - OU2 - GROUNDWATER
STRATFORD, CONNECTICUT

PARAMETER	Detects	Count	Average	Average Detected Conc.	Minimum Conc.	Maximum Conc.	MaxLocation	UPGRADIENT OVERBURDEN	>UPGRADIENT OVERBURDEN	ICVol	>ICVol	RVC	> RVC	SWPC	>SWPC
VOLATILE ORGANIC ANALYSIS (UG/L) (cont.)															
Isopropylbenzene	2	5	4	3	2 J	4 J	OU1-MW-RW1-07								
Methyl Acetate	2	5	9	15	6 J	24	OU1-MW-RW1-07								
Methylcyclohexane	1	5	5	3	3 J	3 J	OU1-MW-RW6A-07								
Methylene Chloride	5	5	190	190	28	440 *J	OU1-MW-RW6A-07			50000	0	50000	0	48000	0
Tetrachloroethene	5	5	500	500	11	2000 *	OU1-MW-RW6A-07			3820	0	1500	1	88	3
Toluene	5	5	1300	1300	31	2700 *	OU1-MW-RW1-07			50000	0	23500	0	4000000	0
Total Xylenes	5	5	260	260	3 J	580 J	OU1-MW-RW6A-07			50000	0	21300	0		
trans-1,2-Dichloroethene	4	5	69	85	6 J	160 , J	OU1-MW-RW3-07, OU1-MW-RW6A-07								
Trichloroethene	5	5	56000	56000	540 *	100000 *	OU1-MW-RW1-07			540	4	219	5	2340	4
Vinyl Chloride	4	5	200	250	29	400 , J	OU1-MW-RW2-07, OU1-MW-RW6A-07			2	4	2	4	15750	0
PESTICIDE/PCB (UG/L)															
4,4'-DDE	2	5	0.13	0.24	0.13 J	0.36 J	OU1-MW-RW1-07								
4,4'-DDT	2	3	0.13	0.16	0.12	0.21 J	OU1-MW-RW3-07								
Aldrin	2	5	0.099	0.21	0.19 J	0.23 J	OU1-MW-RW2-07								
beta-BHC	3	4	1.6	2.1	0.61 J	4.1 *J	OU1-MW-RW1-07								
delta-BHC	1	3	0.041	0.074	0.074 J	0.074 J	OU1-MW-RW6-07								
Dieldrin	2	4	0.052	0.08	0.067 J	0.092 J	OU1-MW-RW3-07							0.1	0
Endosulfan I	2	5	0.13	0.28	0.27 J	0.29 J	OU1-MW-RW2-07								
Endosulfan II	1	3	0.083	0.15	0.15 J	0.15 J	OU1-MW-RW3-07								
Endrin	1	3	0.035	0.056	0.056 J	0.056 J	OU1-MW-RW3-07							0.1	0
Endrin Aldehyde	1	4	0.075	0.15	0.15 J	0.15 J	OU1-MW-RW1-07								
Endrin Ketone	2	5	0.28	0.63	0.16 J	1.1 J	OU1-MW-RW1-07								
gamma-BHC	2	3	0.77	1.1	0.28	2 *J	OU1-MW-RW3-07								
gamma-Chlordane	1	4	0.11	0.35	0.35 J	0.35 J	OU1-MW-RW2-07							0.3	1
Heptachlor	2	5	0.042	0.09	0.039 J	0.14 J	OU1-MW-RW2-07							0.05	1
Heptachlor Epoxide	2	4	0.18	0.35	0.14 J	0.56 J	OU1-MW-RW1-07							0.05	2

* = From dilution analysis or Estimated Maximum Possible Concentration
> = number of samples with concentrations greater than the criterion
EMPC = Estimated Maximum Possible Concentration
J = Quantitation approximate
Note - Table 4-5 includes only detected contaminants.

RVC = CT Residential Volatilization Criteria
ICVol = CT Industrial/Commercial Volatilization Criteria
SWPC = CT Surface Water Protection Criteria

TABLE 4-5a
DATA SUMMARY AND CRITERIA COMPARISONS - MODELED VOCS AND METALS IN ALL AQUIFERS
REMEDIAL INVESTIGATION
RAYMARK - OU2 - GROUNDWATER
STRATFORD, CONNECTICUT

PARAMETER	Detects	Count	Average	Average Detected Conc.	Minimum Conc.	Maximum Conc.	MaxLocation	ICVol	>ICVol	RVC	> RVC	SWPC	>SWPC
METALS (UG/L)													
Arsenic	60	252	11.1	40.4	3.2	148	OU2-MW-505M-07-AVG					4	59
Beryllium	22	254	2.8	30.5	0.14 J	113 J	OU2-MW-533B-07					4	16
Cadmium	59	254	13.2	56	0.3 J	553	OU2-MW-9-07-AVG					6	28
Copper	102	254	75.6	187	0.48	9960	OU1-MW-RW1-07					48	23
Lead	41	256	5.1	26.5	1.4 J	227 J	OU2-MW-211B-07					13	12
Nickel	185	255	269	371	1.2 J	17900	OU1-MW-PC16M-07					880	15
Thallium	40	257	7.8	37.3	2.4	210 J	OU1-MW-PC01B-07					63	7
Zinc	125	254	433	873	1.6 J	10800	OU1-MW-PC16M-07					123	54
VOLATILE ORGANIC ANALYSIS (UG/L)													
1,1,1-Trichloroethane	128	265	1900	4000	1 J	160000 *	OU1-MW-PC02D-07	50000	3	20400	6	62000	2
1,1-Dichloroethene	131	265	680	1400	1	40000 *	OU1-MW-PC02B-07	6	111	1	130	96	65
1,2-Dichloroethane	39	260	5	7	2 J	50	OU1-MW-RW1-07	90	0	21	2	2970	0
Tetrachloroethene	81	265	15	38	1 , J	2000 *	OU1-MW-RW6A-07	3820	0	1500	1	88	3
Toluene	33	265	170	1300	1 J	35000 *	OU1-MW-PC04S-07	50000	0	23500	1	4000000	0
Trichloroethene	147	265	1600	2900	0.7	100000 *	OU1-MW-RW1-07	540	57	219	74	2340	21
Vinyl Chloride	116	265	30	68	0.6 J	530 *	OU1-MW-PC14S-07	2	96	2	96	15750	0

> = number of samples with concentrations greater than the criterion
B = Compound also detected in a quality control blank sample.
EMPC = Estimated Maximum Possible Concentration
J = Quantitation approximate

* = From dilution analysis or Estimated Maximum Possible Concentration
RVC = CT Residential Volatilization Criteria
ICVol = CT Industrial/Commercial Volatilization Criteria
SWPC = CT Surface Water Protection Criteria

TABLE 4-6
DATA SUMMARY AND CRITERIA COMPARISONS - BEDROCK
REMEDIAL INVESTIGATION
RAYMARK - OU2 - GROUNDWATER
STRATFORD, CONNECTICUT

PARAMETER	Detects	Count	Average	Average Detected Conc.	Minimum Conc.		Maximum Conc.		MaxLocation	UPGRADIENT BEDROCK	>UPGRADIENT BEDROCK	ICVol	>ICVol	RVC	> RVC	SWPC	>SWPC
DIOXINS (PG/L)																	
1,2,3,4,6,7,8-HpCDD	10	49	4.5	14.8	1	J	91.2		OU2-MW-215DB-07-AVG	17.1	2						
1,2,3,4,6,7,8-HpCDF	8	49	1.6	3.8	0.53	J	7.4	EMPC, J	OU1-MW-PC02B-07, OU1-MW-PC11B-07	8.3	0						
1,2,3,4,7,8,9-HpCDF	4	49	1.4	3.9	0.36	EMPC	8	J	OU1-MW-PC02B-07								
1,2,3,4,7,8-HxCDD	4	49	1.3	4	0.38	J	9.2	J	OU1-MW-PC02B-07	1.9	3						
1,2,3,4,7,8-HxCDF	5	49	1.2	3.9	0.9		9.1	JB	OU1-MW-PC02B-07	3.9	2						
1,2,3,6,7,8-HxCDD	6	49	1.5	4.7	0.37	J	10.1	J	OU1-MW-PC02B-07	3.5	3						
1,2,3,6,7,8-HxCDF	6	49	1.1	3.8	0.42	J	8.6	JB	OU1-MW-PC02B-07	3.6	3						
1,2,3,7,8,9-HxCDD	5	49	1.4	3.6	0.42	J	7.3	JB	OU1-MW-PC02B-07	4.1	2						
1,2,3,7,8,9-HxCDF	6	49	1.4	4.8	0.45	J	8.6	JB	OU1-MW-PC02B-07	5	3						
1,2,3,7,8-PeCDD	5	49	1.3	4.3	1.6	J	6.9	J	OU1-MW-PC02B-07	4.5	2						
1,2,3,7,8-PeCDF	5	49	1.1	4.9	2.2	EMPC	7.1	JB	OU1-MW-PC02B-07	5.2	2						
2,3,4,6,7,8-HxCDF	6	49	1.2	4.1	0.38	J	11.6	JB	OU1-MW-PC02B-07	2.3	3						
2,3,4,7,8-PeCDF	6	49	1.1	3.6	0.87	J	8.3	J	OU1-MW-PC02B-07	3.5	3						
2,3,7,8-TCDD	2	49	0.97	1.2	0.3	EMPC	2.1	EMPC	OU2-MW-217B-07								
2,3,7,8-TCDF	3	49	0.79	1.5	0.48	J	3	J	OU1-MW-PC10B-07								
OCDD	18	49	32.4	78.5	4.5	J	742		OU2-MW-215DB-07-AVG	72	3						
OCDF	10	49	5.2	11.7	1.2	J	36.3	J	OU1-MW-PC04B-07	5.6	7						
Total HpCDD	11	49	6.8	23.8	2.4	EMPC, J	148		OU2-MW-215DB-07-AVG	33.8	2						
Total HpCDF	8	49	2.5	7.4	0.89	EMPC	20.5		OU2-MW-215DB-07-AVG	10.5	2						
Total HxCDD	6	49	3.7	18.1	1.4	EMPC	40.4	EMPC	OU1-MW-PC02B-07	22.3	2						
Total HxCDF	7	49	3.7	17.8	0.9		37.8		OU1-MW-PC02B-07	14.8	3						
Total PeCDD	7	49	1.5	4.3	1.6	J	6.9		OU1-MW-PC02B-07	4.5	3						
Total PeCDF	7	49	1.7	7.5	0.97	J	15.4		OU1-MW-PC02B-07	8.7	3						
Total TCDD	4	49	1.1	2.5	0.3	EMPC	4.9		OU1-MW-PC01B-07								
Total TCDF	5	49	1.7	10.1	0.7	J	21.1	EMPC	OU2-MW-301B-07								
Toxicity Equivalency	19	19	1.1	1.1	0.00047	J	8	J	OU2-MW-215B-07	1.8	3						
METALS (UG/L)																	
Aluminum	11	52	24700	116000	453		725000	J	OU2-MW-533B-07								
Antimony	1	52	1.6	4.5	4.5	J	4.5	J	OU2-MW-503B-07							86000	0
Arsenic	7	50	6.9	38.1	4.1	J	130		OU2-MW-211B-07							4	7
Barium	41	50	58.2	67	8		645		OU1-MW-PC11BF-07	308	1						
Beryllium	7	50	6.1	42.4	0.58	J	113	J	OU2-MW-533B-07							4	6
Cadmium	14	50	3.2	10.8	0.3	J	64	J	OU2-MW-533B-07							6	4
Calcium	52	52	176000	176000	10600	J	503000		OU2-MW-1B-07	104000	30						
Chromium	24	52	3.8	7.6	0.77	J	56.8	J	OU1-MW-PC01B-07	7.6	9					1200	0
Cobalt	38	52	228	311	0.75	J	3140	J	OU1-MW-PC01B-07	1.5	33						

TABLE 4-6 (cont.)
DATA SUMMARY AND CRITERIA COMPARISONS - BEDROCK
REMEDIAL INVESTIGATION
RAYMARK - OU2 - GROUNDWATER
STRATFORD, CONNECTICUT

PARAMETER	Detects	Count	Average	Average Detected Conc.	Minimum Conc.		Maximum Conc.		MaxLocation	UPGRADIENT BEDROCK	>UPGRADIENT BEDROCK	ICVol	>ICVol	RVC	> RVC	SWPC	>SWPC
METALS (UG/L) (CONT.)																	
Copper	20	50	27	65.8	0.83	J	631	J	OU2-MW-211B-07	1.7	18					48	4
Cyanide	3	22	3.1	1.4	0.53	J	2	J	OU2-MW-502B-07	1.2	2					52	0
Iron	29	52	108000	193000	40.5		2300000		OU2-MW-533B-07	394	21						
Lead	12	52	10.1	40.6	2	J	227	J	OU2-MW-211B-07							13	3
Magnesium	52	52	102000	102000	1160	J	605000		OU2-MW-217B-07	28700	37						
Manganese	51	52	19900	20300	11.4	J	235000		OU1-MW-PC01B-07	1110	27						
Mercury	1	52	0.06	0.26	0.26	J	0.26	J	OU2-MW-211B-07							0.4	0
Nickel	42	52	259	320	1.7	J	3720	J	OU2-MW-533B-07	3.6	33					880	3
Potassium	52	52	16300	16300	1140		113000	J	OU2-MW-217B-07	7860	28						
Selenium	7	50	5.7	30.3	3.9	J	90.3	J	OU1-MW-PC01B-07	3.3	7					50	1
Silver	12	48	1.5	4.9	0.88	J	15	J	OU2-MW-1B-07							12	1
Sodium	52	52	373000	373000	10200		3990000		OU2-MW-217B-07	108000	22						
Thallium	11	52	14.7	61.6	5	J	210	J	OU1-MW-PC01B-07							63	4
Vanadium	15	51	2.9	8.6	0.67		70.7	J	OU2-MW-533B-07	0.81	14						
Zinc	28	50	747	1330	2	J	8790		OU2-MW-4B-07	9.9	21					123	12
METHANE, ETHANE, ETHENE (UG/L)																	
Ethane	3	42	5.6	13.7	11		17		OU2-MW-502B-07								
Ethene	4	42	5.8	13	10		19		OU2-MW-212B-07	10	3						
Methane	28	42	160	238	8		2600		OU2-MW-310B-07	39	18						
SEMIVOLATILE ORGANIC ANALYSIS (UG/L)																	
2,4-Dimethylphenol	2	52	5	1	1	J	1	J	OU2-MW-211B-07, OU2-MW-514B-07								
2,4-Dinitrophenol	1	52	12	11	11	J	11	J	OU1-MW-PC12B-07								
2-Nitrophenol	5	52	6	11	7	J	15		OU2-MW-504B-07, OU2-MW-524B-07								
4-Methylphenol	1	52	5	2	2	J	2	J	OU1-MW-PC11B-07								
4-Nitrophenol	18	52	34	74	1	J	410	*	OU2-MW-504B-07								
Acetophenone	1	52	5	2	2	J	2	J	OU1-MW-PC11B-07								
bis(2-Ethylhexyl)phthalate	5	52	5	2	1	J	4		OU2-MW-305BR-07-AVG							59	0
Caprolactam	11	52	28	110	4		810	*	OU2-MW-4B-07	650	1						
Diethylphthalate	2	52	5	4	1	J	7	J	OU2-MW-213DB-07								
Di-n-octylphthalate	1	52	5	10	10	J	10	J	OU1-MW-PC12B-07								
Phenanthrene	1	52	0.1	0.1	0.12	J	0.12	J	OU1-MW-PC11B-07							0.077	1
Phenol	1	52	5	1	1	J	1	J	OU1-MW-PC06B-07							92000000	0

TABLE 4-6 (cont.)
DATA SUMMARY AND CRITERIA COMPARISONS - BEDROCK
REMEDIAL INVESTIGATION
RAYMARK - OU2 - GROUNDWATER
STRATFORD, CONNECTICUT

PARAMETER	Detects	Count	Average	Average Detected Conc.	Minimum Conc.		Maximum Conc.		MaxLocation	UPGRADIENT BEDROCK	>UPGRADIENT BEDROCK	ICVol	>ICVol	RVC	> RVC	SWPC	>SWPC
VOLATILE ORGANIC ANALYSIS (UG/L)																	
1,1,1-Trichloroethane	36	52	3200	4700	2	J	120000	*	OU1-MW-PC02B-07	110	19	50000	1	20400	2	62000	1
1,1,2-Trichloro-1,2,2-trifluoroethane	1	52	5	7	7	J	7	J	OU1-MW-PC16B-07								
1,1,2-Trichloroethane	18	52	8	13	1	J	96		OU1-MW-PC02B-07			19600	0	8000	0	1260	0
1,1-Dichloroethane	38	52	140	190	1	J	1700	*	OU2-MW-212B-07	2	37	50000	0	34600	0		
1,1-Dichloroethene	40	52	1600	2100	2		40000	*	OU1-MW-PC02B-07	260	15	6	34	1	40	96	21
1,2-Dichlorobenzene	1	52	5	3	3	J	3	J	OU1-MW-PC13B-07			50000	0	30500	0	170000	0
1,2-Dichloroethane	15	52	5	6	2	J	23		OU1-MW-PC16B-07			90	0	21	1	2970	0
1,4-Dioxane	21	49	210	410	67	J	1400	J	OU2-MW-502B-07								
4-Methyl-2-Pentanone	1	52	5	6	6	J	6	J	OU1-MW-PC11B-07			50000	0	50000	0		
Acetone	4	52	52	610	7	J	2400	*J	OU2-MW-4B-07	13	2	50000	0	50000	0		
Benzene	13	52	6	7	2	J	24		OU2-MW-310B-07			530	0	215	0	710	0
Bromodichloromethane	1	52	5	13	13		13		OU2-MW-506B-07	6	1						
Bromoform	1	52	5	4	4	J	4	J	OU2-MW-533B-07			3800	0	920	0	10800	0
Chlorobenzene	36	52	32	44	1	J	210		OU1-MW-PC15B-07	1200	0	6150	0	1800	0	420000	0
Chloroethane	11	52	5	6	1	J	20		OU2-MW-507B-07								
Chloroform	26	52	6	8	1	J	42		OU2-MW-506B-07	23	1	710	0	287	0	14100	0
cis-1,2-Dichloroethene	41	52	220	280	1	J	1600	*	OU2-MW-503B-07	6	37						
Dibromochloromethane	1	52	5	2	2	J	2	J	OU2-MW-506B-07							1020	0
Methyl tert-Butyl Ether	12	52	14	46	1	J	210		OU2-MW-1B-07								
Methylcyclohexane	2	52	5	2	1	J	2	J	OU2-MW-310B-07								
Methylene Chloride	8	52	9	32	6	J	75		OU2-MW-504B-07			50000	0	50000	0	48000	0
Tetrachloroethene	27	52	7	9	1	J	51		OU1-MW-PC16B-07			3820	0	1500	0	88	0
Toluene	8	52	5	3	1	J	7	J	OU1-MW-PC02B-07	3	3	50000	0	23500	0	4000000	0
Total Xylenes	4	52	5	2	1	J	4	J	OU1-MW-PC02B-07			50000	0	21300	0		
trans-1,2-Dichloroethene	12	52	4	2	1	J	4	, J	OU1-MW-PC11B-07, OU2-MW-304B-07, OU2-MW-305BR-07-AVG								
Trichloroethene	42	52	1100	1400	4	J	6900	*	OU2-MW-306B-07	31	37	540	23	219	29	2340	8
Vinyl Chloride	32	52	25	40	0.6	J	230		OU2-MW-504B-07			2	25	2	25	15750	0
PESTICIDE/PCB (UG/L)																	
alpha-BHC	2	52	0.03	0.16	0.095	J	0.23		OU2-MW-304B-07								
alpha-Chlordane	1	52	0.026	0.1	0.1		0.1		OU2-MW-215B-07							0.3	0
beta-BHC	2	52	0.031	0.17	0.081	J	0.26		OU1-MW-PC15B-07								
Dieldrin	1	52	0.033	0.42	0.42		0.42		OU2-MW-215B-07							0.1	1
Endosulfan I	1	52	0.026	0.055	0.055		0.055		OU2-MW-215B-07								

TABLE 4-6 (cont.)
DATA SUMMARY AND CRITERIA COMPARISONS - BEDROCK
REMEDIAL INVESTIGATION
RAYMARK - OU2 - GROUNDWATER
STRATFORD, CONNECTICUT

PARAMETER	Detects	Count	Average	Average Detected Conc.	Minimum Conc.		Maximum Conc.		MaxLocation	UPGRADIENT BEDROCK	>UPGRADIENT BEDROCK	ICVol	>ICVol	RVC	> RVC	SWPC	>SWPC
PESTICIDE/PCB (UG/L) (CONT.)																	
gamma-BHC	2	52	0.033	0.22	0.051		0.39		OU2-MW-304B-07								
Heptachlor	1	52	0.01	0.023	0.023		0.023		OU2-MW-4B-07							0.05	0
Heptachlor Epoxide	5	52	0.015	0.061	0.02		0.18		OU2-MW-215B-07							0.05	2
WET CHEMISTRY (MG/L)																	
Alkalinity	41	43	128	134	10		374	J	OU2-MW-401B-07	172	10						
Carbon Dioxide	40	43	170	181	15.4	J	496	J	OU2-MW-401B-07	171	18						
Chloride	43	45	689	715	1.65		8220	J	OU2-MW-217B-07	96.4	27						
Ferrous Iron	29	43	114	169	0.03		2270	J	OU1-MW-PC02B-07	0.1	21						
Nitrate	29	45	5	7.8	0.065	J	79.9	J	OU1-MW-PC13B-07	7.56	4						
Nitrite-N	7	45	0.12	0.4	0.05		2.07		OU2-MW-211B-07	0.06	5						
Sulfate	43	45	1150	1200	6.41		11800		OU2-MW-533B-07	259	17						
Sulfide	7	39	0.34	0.77	0.5		1.16	J	OU1-MW-PC11B-07	1.44	0						
Total Organic Carbon	32	45	7.9	10.7	2.05		46.3	J	OU2-MW-104B-07	5.17	20						

* = From dilution analysis or Estimated Maximum Possible Concentration
> = number of samples with concentrations greater than the criterion
B = Compound also detected in a quality control blank sample.
EMPC = Estimated Maximum Possible Concentration
J = Quantitation approximate
Note - Table 4-6 includes only detected contaminants.

RVC = CT Residential Volatilization Criteria
ICVol = CT Industrial/Commercial Volatilization Criteria
SWPC = CT Surface Water Protection Criteria

TABLE 4-7
STATISTICAL COMPARISON OF TEMPORAL CHANGES IN GROUNDWATER VOC CONCENTRATIONS THROUGHOUT THE OU2 STUDY AREA
REMEDIAL INVESTIGATION
RAYMARK – OU2 – GROUNDWATER
STRATFORD, CONNECTICUT

	Number of Wells[1]	1997-1999 mean concentration[2] (µg/L)	2002/2003 mean concentration (µg/L)	Wilcoxon Signed Rank (p-value)	Comment[3]
1,1,1-TCA	75	5356	5209	0.000	significant decrease
1,1-DCE	71	1696	2025	0.423	no change
TCE	82	835	1008	0.570	no change
1,2-DCE (Total)	92	302	239	0.000	significant decrease
Toluene	12	2746	2992	0.258	no change
Vinyl Chloride	43	92	76	0.404	no change
Benzene	29	403	192	0.000	significant decrease
Chlorobenzene	42	1196	286	0.005	significant decrease

1 The number of wells in which the VOC was detected in both 1997-1999 and 2002/2003.
2 Several monitoring wells were sampled more than once between 1997 and 1999. The most recent concentration measured at each location was used in this analysis.
3 The difference is not statistically significant if the p-value is greater than 0.05.

TABLE 4-8
STATISTICAL COMPARISON OF TEMPORAL CHANGES IN GROUNDWATER VOC CONCENTRATIONS
IN THE RESIDENTIAL AREA EAST OF FERRY BOULEVARD
REMEDIAL INVESTIGATION
RAYMARK – OU2 – GROUNDWATER
STRATFORD, CONNECTICUT

	Number of Wells[1]	1997-1999 Mean concentration[2] (µg/L)	2002/2003 Mean concentration (µg/L)	Wilcoxon Signed Rank (p-value)	Comment[3]
1,1,1-TCA	17	808	727	0.229	no change
1,1-DCE	19	479	608	0.602	no change
TCE	20	636	667	0.993	no change
1,2-DCE (Total)	21	379	281	0.234	no change
Toluene[4]	3	43	18	------	----------
Vinyl Chloride	7	55	46	1.000	no change
Benzene	4	8	5	0.500	no change
Chlorobenzene	4	33	17	0.625	no change

1 The number of wells in which the VOC was detected in both 1997-1999 and 2002/2003.
2 Several monitoring wells were sampled more than once between 1997 and 1999. The most recent concentration measured at each location was used in this analysis.
3 The difference is not statistically significant if the p-value is greater than 0.05.
4 Sample number is too small for meaningful comparison.

TABLE 4-9
SUMMARY OF INDOOR AIR VOC DATA
REMEDIAL INVESTIGATION
RAYMARK - OU2 - GROUNDWATER
STRATFORD, CONNECTICUT

PARAMETER	Units	Detects	Count	Average	Minimum Of Detects	Minimum Qualifier	Maximum Of Detects	Maximum Qualifier	Raymark Indoor Air Maximum Background
1,1,1-Trichloroethane	UG/M^3	62	69	12	0.291		110		3.77
1,1,2,2-Tetrachloroethane	UG/M^3	1	41	1	38		38		
1,1,2-Trichloro-1,2,2-trifluoroethane	UG/M^3	33	49	0.9	0.541		3.24		0.911
1,1,2-Trichloroethane	UG/M^3	2	44	0.6	0.627	L	9.89		
1,1-Dichloroethane	UG/M^3	20	52	2	0.408		14.7		
1,1-Dichloroethene	UG/M^3	42	69	5	0.272		48		
1,2,4-Trimethylbenzene	UG/M^3	57	66	3	0.0791		33.1		11.1
1,2-Dibromoethane	UG/M^3	1	33	0.5	0.882	L	0.882	L	
1,2-Dichlorobenzene	UG/M^3	2	35	0.4	0.545		0.684	L	
1,2-Dichloroethane	UG/M^3	1	41	0.3	0.326		0.326		
1,3,5-Trimethylbenzene	UG/M^3	38	51	1	0.129		14.8		3.21
1,3-Butadiene	UG/M^3	14	43	0.5	0.12		7.34		0.298
1,4-Dichlorobenzene	UG/M^3	19	50	23	0.0702		811		
2-Butanone	UG/M^3	57	67	5	0.891		98		1.57
2-Hexanone	UG/M^3	1	42	0.3	0.371		0.371		
2-Propanol	UG/M^3	27	56	13	0.42	J	198	JE	151
4-Ethyltoluene	UG/M^3	60	66	4	0.129		69.2	J	9.69
4-Methyl-2-Pentanone	UG/M^3	11	43	0.7	0.12		14		
Acetone	UG/M^3	66	69	11	0.232	BJ	95.3		10.1
Acrylonitrile	UG/M^3	1	12	0.2	0.241		0.241		
Benzene	UG/M^3	69	69	2	0.45		12.5		5.11
Benzyl Chloride	UG/M^3	2	33	0.5	0.532	L	6.26		
Bromodichloromethane	UG/M^3	7	47	1	0.294	L	2.6	L	
Bromoform	UG/M^3	1	5	8	1.49	L,J	1.49	L,J	
Bromomethane	UG/M^3	4	41	0.3	0.0509		0.509	L	
Carbon Disulfide	UG/M^3	9	49	0.3	0.2	L	1.25		
Carbon Tetrachloride	UG/M^3	30	50	0.5	0.375	L	1.07	L	0.741
Chloroethane	UG/M^3	5	46	1	1.33		3.99		1.69
Chloroform	UG/M^3	52	64	1	0.0787	L	5.9		2.08
Chloromethane	UG/M^3	23	44	1	0.645		3.54		
cis-1,2-Dichloroethene	UG/M^3	29	59	2	0.21		16.4		
cis-1,3-Dichloropropene	UG/M^3	1	33	0.3	0.421	L	0.421	L	

TABLE 4-9 (cont.)
SUMMARY OF INDOOR AIR VOC DATA
REMEDIAL INVESTIGATION
RAYMARK - OU2 - GROUNDWATER
STRATFORD, CONNECTICUT

PARAMETER	Units	Detects	Count	Average	Minimum Of Detects	Minimum Qualifier	Maximum Of Detects	Maximum Qualifier	Raymark Indoor Air Maximum Background
Cyclohexane	UG/M^3	24	51	1	0.415		6.92		3.15
Dibromochloromethane	UG/M^3	1	33	1	1.69	L	1.69	L	
Dichlorodifluoromethane	UG/M^3	67	69	8	1.15	L	149		69.7
Dichlorotetrafluoroethane	UG/M^3	4	6	2	1.13	L	4.86		
Ethyl Acetate	UG/M^3	18	33	5	0.41		42.5		15.8
Ethylbenzene	UG/M^3	64	67	4	0.201	L	73.8		6.25
Freon 114	UG/M^3	5	33	0.4	0.0775		0.965	L	
Heptane	UG/M^3	27	48	3	0.412		29.7		
Hexane	UG/M^3	66	69	6	0.815		35.4		22.4
m&p-Xylene	UG/M^3	64	64	11	0.655	L	100		20.6
Methyl tert-Butyl Ether	UG/M^3	46	47	9	0.363		69		25.4
Methylene Chloride	UG/M^3	53	61	24	0.521		910		3.22
Methyl-t-Butyl Ketone	UG/M^3	20	22	20	1.11		173	E	
o-Xylene	UG/M^3	67	69	4	0.24	L	86.9		7.64
Styrene	UG/M^3	19	45	0.7	0.24	L	8.14		
Tetrachloroethene	UG/M^3	34	57	1	0.301	, L	25.3		
Tetrahydrofuran	UG/M^3	24	65	3	1.13		32.7		4.84
Toluene	UG/M^3	69	69	20	0.273	BL	159		30.8
Total Xylenes	UG/M^3	5	5	170	4.8	L	761		
trans-1,2-Dichloroethene	UG/M^3	1	39	0.3	0.447	L	0.447	L	
trans-1,3-Dichloropropene	UG/M^3	1	33	0.3	0.43	L	0.43	L	
Trichloroethene	UG/M^3	39	69	5	0.384	L	52.5		
Trichlorofluoromethane	UG/M^3	61	69	5	1.25		62.3		27
Trichlorotrifluoroethane	UG/M^3	5	5	1	0.718	L	1.68	L	
Vinyl Acetate	UG/M^3	29	34	6	0.674		39		
Vinyl Chloride	UG/M^3	2	69	0.2	0.438		0.618		

B = Analyte is associated with the lab blank or trip blank contamination. Values are qualified when the observed concentration of the contamination in the sample extract is less than 5 times the concentration in the blank.
L = Estimated value is below the calibration range
J = Quantitation approximate
E = Estimated value exceeds the calibration range

TABLE 5-1
FATE AND TRANSPORT MODELING PARAMETERS
REMEDIAL INVESTIGATION
RAYMARK – OU2 – GROUNDWATER
STRATFORD, CONNECTICUT

	Source [a]	Flow line [b]	Y (m)	Z (m)	C_O (µg/L)	x (m)	α_x (m)	α_y (m)	α_z (m)	grad h [c]	Soil Type	K [d] (m/y)	ρ_d (g/cm^3)	n	n_e [e]	Half Life (yr)	K_d (ml/g)
Arsenic	F	1	53	1.4	246	171	5.77	0.289	0.0289	0.0041	silty sand	4280	1.59	0.35	0.25	-	10
	F	2	6.1	2.4	246	213	6.39	0.320	0.0320	0.0040	silty sand	4280	1.59	0.35	0.25	-	10
Cadmium	E	-	65	9	470	800	10.87	1.087	0.0544	0.0014	sand	12130	1.59	0.35	0.25	-	14
Copper	F	1	53	1.4	1360	171	5.77	0.289	0.0289	0.0041	silty sand	4280	1.59	0.35	0.25	-	21
	F	2	6.1	2.4	1360	213	6.39	0.320	0.0320	0.0040	silty sand	4280	1.59	0.35	0.25	-	21
Lead	F	1	53	1.4	101	171	5.77	0.289	0.0289	0.0041	silty sand	4280	1.59	0.35	0.25	-	39
	F	2	6.1	2.4	101	213	6.39	0.320	0.0320	0.0040	silty sand	4280	1.59	0.35	0.25	-	39
Nickel	E	-	65	9	5870	800	10.87	1.087	0.0544	0.0014	sand	12130	1.59	0.35	0.25	-	18
Zinc	E	-	65	9	3010	800	10.87	1.087	0.0544	0.0014	sand	12130	1.59	0.35	0.25	-	3.5
	F	1	53	1.4	970	171	5.77	0.289	0.0289	0.0041	silty sand	4280	1.59	0.35	0.25	-	3.4
	F	2	6.1	2.4	970	213	6.39	0.320	0.0320	0.0040	silty sand	4280	1.59	0.35	0.25	-	3.4
1,1-DCE [f]	D	-	30	10	42000	740	10.57	0.529	0.0529	0.00075	sand	66750	1.59	0.35	0.25	3.96	0.008
TCE	B	-	120	8	88000	768	10.72	0.536	-	0.0015	sand	12130	1.59	0.35	0.25	2.76	0.012

[a] Source as shown on Figure 5-1. Sources on Figure 5-1 are denoted by a capital letters A through F.
[b] Two flow lines were modeled from source F (see Figure 5-1). Flow line 1 trends northeast to Ferry Creek. Flow line 2 trends southeast to the swale between 230 and 250 Ferry Boulevard.
[c] grad h = hydraulic gradient (dimensionless)
[d] K = hydraulic conductivity
[e] n_e = effective porosity (dimensionless)
[f] The 1,1-DCE model is unique, because the centerline of the plume does not intersect the creek bed. The centerline is roughly 18.5 m beneath the creek; consequently, the model forecasts the 1,1-DCE concentration entering Ferry Creek from the upper margin of the plume at x = 740 m, y = 0 m, and z = 18.5 m.

TABLE 5-2
FATE AND TRANSPORT MODELING RESULTS
REMEDIAL INVESTIGATION
RAYMARK – OU2 – GROUNDWATER
STRATFORD, CONNECTICUT

Contaminant	Source	Receptor	Maximum [1,2] Concentration (µg/L)	Approximate Arrival Date [3]
Arsenic	Raymark Waste surrounding MW-102S at 600 East Broadway	Ferry Creek	84	2110
		Swale [4]	24	2130
Cadmium	540 Longbrook Ave. lagoons upgradient from MW-Z	Housatonic River	— [5]	Not predicted to arrive before 2200
Copper	Raymark Waste surrounding MW-102S at 600 East Broadway	Ferry Creek	—	Not predicted to arrive before 2200
		Swale [4]	—	Not predicted to arrive before 2200
Lead	Raymark Waste surrounding MW-102S at 600 East Broadway	Ferry Creek	—	Not predicted to arrive before 2200
		Swale [4]	—	Not predicted to arrive before 2200
Nickel	540 Longbrook Ave. lagoons upgradient from MW-Z	Housatonic River	—	Not predicted to arrive before 2200
Zinc	540 Longbrook Ave. lagoons upgradient from MW-Z	Housatonic River	—	Not predicted to arrive before 2200
	Raymark Waste surrounding MW-102S at 600 East Broadway	Ferry Creek	330	2030
		Swale [4]	97	2040
1,1-DCE	DNAPL beneath Lagoon No. 4 on Raymark Facility Property	Ferry Creek	530	Prior to 1997
TCE	DNAPL beneath acid pits on Raymark Facility Property	Housatonic River	5700	Prior to 1997

1 Maximum sustained groundwater concentration predicted to occur at the point where the centerline of the plume intersects the stream bed. Exception: the centerline of the 1,1-DCE plume passes approximately 18.5 m beneath Ferry Creek; therefore, the forecasted 1,1-DCE concentration applies to the point on Ferry Creek that is 18.5 m above the centerline of the plume.

2 Predicted concentrations are conservative to support human health and ecological risk assessments. (Refer to Section 5.3 for model assumptions and limitations, and the rationale used to select input parameter values.)

3 Calendar year

4 Swale between 230 and 250 Ferry Boulevard that connects to Ferry Creek and contains standing water. The swale is underlain by several feet of organic-rich soil; therefore, the actual concentrations that will eventually discharge to the swale from the 600 East Broadway source will probably be much less than the concentrations predicted by the model (see Section 5.3.4.1).

5 The contaminant plume is moving so slowly and/or the receptor is so far away from the modeled source that the plume may reach its maximum spatial extent before it reaches the receptor.

TABLE 5-3
SIMULATED VERSUS OBSERVED CONCENTRATIONS
REMEDIAL INVESTIGATION
RAYMARK – OU2 – GROUNDWATER
STRATFORD, CONNECTICUT

Observation Point	1,1-DCE (µg/L)		TCE (µg/L)	
	Predicted 1997-2003	Observed 1997-2003	Predicted 1997-2003	Observed 1997-2003
MW-211D	21,300	17,000-21,500	——	——
MW-211M	1700	870-2100	——	——
MW-514D	2400	2400	——	——
MW-212D	4200	1900-9500	——	——
MW-212M	3700	1200-7600	——	——
MW-212S	1000	1000-2100	——	——
MW-111D	3500	3600-6200	——	——
MW-111M	1400	1500-2200	——	——
MW-520S	200	870	——	——
MW-514D	——	——	23,000	21,000
MW-214D	——	——	1300	1300-1600
MW-302D	——	——	5300	5350-6400

TABLE 5-4
SPEARMAN-RHO CORRELATION BETWEEN VOC CONCENTRATIONS IN GROUNDWATER AND VOC CONCENTRATIONS IN SOIL GAS
REMEDIAL INVESTIGATION
RAYMARK – OU2 – GROUNDWATER
STRATFORD, CONNECTICUT

Contaminant	Groundwater sample depth below water table	Soil gas sample depth below ground surface		
		3 ft	**8 ft**	**11-19 ft**
1,1,1-trichloroethane	**1 ft**	0.70 [a]	0.60	0.72
	5 ft	0.75	0.56	0.65
	10-15 ft	0.76	0.53	0.63
	20-25 ft	NR [b]	NR	NR
1,1-dichloroethene	**1 ft**	0.61	NR	0.80
	5 ft	NR	NR	0.64
	10-15 ft	NR	NR	0.57
	20-25 ft	NR	NR	NR
trichloroethene	**1 ft**	0.81	0.67	NR
	5 ft	NR	NR	NR
	10-15 ft	NR	0.53	NR
	20-25 ft	NR	NR	-0.65

[a] Perfect correlation = 1.0.

[b] No relationship ($\alpha = 0.05$).

TABLE 5-5
BIODEGRADATION SCREENING FOR MONITORING WELL MW-303B
REMEDIAL INVESTIGATION
RAYMARK – OU2 – GROUNDWATER
STRATFORD, CONNECTICUT

ANALYSIS	SCREENING		MW-303B	
	CONCENTRATION	SCORE	RESULTS	SCORE
Oxygen*	< 0.5 mg/L	3	2.85	0
Oxygen*	> 5 mg/L	-3	2.85	0
Nitrate*	< 1mg/L	2	7.56	0
Iron II*	> 1 mg/L	3	0.03U	0
Sulfate*	< 20 mg/L	2	56.8	0
Sulfide*	> 1 mg/L	3	0.5U	0
Methane*	> 0.5 mg/L	3	0.005U	0
Oxidation Reduction Potential* against Ag/AgCl electrode	< 50 mV < -100mV	1 2	295	0
pH*	pH < 5 pH > 9	-2	5.97	0
Total Organic Carbon	> 20 mg/L	2	2U	0
Temperature*	> 20° C	1	12.9	0
Carbon Dioxide	> 2 X background	1	138	0
Alkalinity	> 2 X background [2]	1	60	0
Chloride*	> 2 X background [2]	2	57	0
Hydrogen	> 1 nM	3	NA	—
Hydrogen	< 1 nM	0	NA	—
Volatile Fatty Acids	> 0.1 mg/L	2	NA	—
BTEX*	> 0.1 mg/L	2	0.01U	0
Tetrachloroethene [1]		0	10U	0
Trichloroethene* [1]		0 2[3]	10U	0
1,2-Dichloroethene* [1]		0 2[3]	10U	0
Vinyl Chloride* [1]		0 2[3]	1U	0
1,1,1-Trichloroethane*[1]		0	10U	0
Dichloroethane [1]		2	10U	0
Carbon Tetrachloride [1]		0	10U	0
Chloroethane* [1]		2	10U	0
Ethene/Ethane	> 0.01 mg/L > 0.1 mg/L	2 3	0.01U	0
Chloroform [1]		0 2[3]	10U	0
Dichloromethane [1]		0 2[3]	10U	0
TOTAL SCORE				**0**

* Required Analysis.
[1] Concentration is reported in units of µg/L.
[2] Upgradient concentrations: CO_2 = 86.6 mg/L, Alkalinity = 43.3 mg/L, Chloride = 35.0 mg/L.
[3] Points awarded only if it can be shown that the compound is a daughter product.

TABLE 5-6
BIODEGRADATION SCREENING FOR MONITORING WELL MW-307B
REMEDIAL INVESTIGATION
RAYMARK – OU2 – GROUNDWATER
STRATFORD, CONNECTICUT

ANALYSIS	SCREENING		MW-307B	
	CONCENTRATION	SCORE	RESULTS	SCORE
Oxygen*	< 0.5 mg/L	3	3.82	0
Oxygen*	> 5 mg/L	-3	3.82	0
Nitrate*	< 1mg/L	2	3.2	0
Iron II*	> 1 mg/L	3	0.03U	0
Sulfate*	< 20 mg/L	2	31	0
Sulfide*	> 1 mg/L	3	R	—
Methane*	> 0.5 mg/L	3	0.005U	0
Oxidation Reduction Potential* against Ag/AgCl electrode	< 50 mV < -100mV	1 2	203	0
pH*	pH < 5 pH > 9	-2	5.6	0
Total Organic Carbon	> 20 mg/L	2	4.5	0
Temperature*	> 20° C	1	14.8	0
Carbon Dioxide	> 2 X background	1	85.5	0
Alkalinity	> 2 X background [2]	1	20	0
Chloride*	> 2 X background [2]	2	63	0
Hydrogen	> 1 nM	3	NA	—
Hydrogen	< 1 nM	0	NA	—
Volatile Fatty Acids	> 0.1 mg/L	2	NA	—
BTEX*	> 0.1 mg/L	2	0.01U	0
Tetrachloroethene [1]		0	10U	0
Trichloroethene* [1]		0 2[3]	10U	0
1,2-Dichloroethene* [1]		0 2[3]	10U	0
Vinyl Chloride* [1]		0 2[3]	1U	0
1,1,1-Trichloroethane*[1]		0	10U	0
1,1 Dichloroethane [1]		2	10U	0
Carbon Tetrachloride [1]		0	10U	0
Chloroethane* [1]		2	10U	0
Ethene/Ethane	> 0.01 mg/L > 0.1 mg/L	2 3	0.01U	0
Chloroform [1]		0 2[3]	23	0
Dichloromethane [1]		0 2[3]	10U	0
TOTAL SCORE				**0**

* Required Analysis.
[1] Concentration is reported in units of µg/L.
[2] Upgradient concentrations: CO_2 = 86.6 mg/L, Alkalinity = 43.3 mg/L, Chloride = 35.0 mg/L.
[3] Points awarded only if it can be shown that the compound is a daughter product.

TABLE 5-7
BIODEGRADATION SCREENING FOR MONITORING WELL MW-308B
REMEDIAL INVESTIGATION
RAYMARK – OU2 – GROUNDWATER
STRATFORD, CONNECTICUT

ANALYSIS	SCREENING		MW-308B	
	CONCENTRATION	SCORE	RESULTS	SCORE
Oxygen*	< 0.5 mg/L	3	8.00	0
Oxygen*	> 5 mg/L	-3	8.00	-3
Nitrate*	< 1mg/L	2	2.74	0
Iron II*	> 1 mg/L	3	0.03U	0
Sulfate*	< 20 mg/L	2	22	0
Sulfide*	> 1 mg/L	3	R	—
Methane*	> 0.5 mg/L	3	0.005U	0
Oxidation Reduction Potential* against Ag/AgCl electrode	< 50 mV < -100mV	1 2	244	0
pH*	pH < 5 pH > 9	-2	5.8	0
Total Organic Carbon	> 20 mg/L	2	2.5	0
Temperature*	> 20° C	1	14.2	0
Carbon Dioxide	> 2 X background	1	55.3	0
Alkalinity	> 2 X background [2]	1	18	0
Chloride*	> 2 X background [2]	2	18.6	0
Hydrogen	> 1 nM	3	NA	—
Hydrogen	< 1 nM	0	NA	—
Volatile Fatty Acids	> 0.1 mg/L	2	NA	—
BTEX*	> 0.1 mg/L	2	0.01U	0
Tetrachloroethene [1]		0	10U	0
Trichloroethene* [1]		0 2[3]	10U	0
1,2-Dichloroethene* [1]		0 2[3]	10U	0
Vinyl Chloride* [1]		0 2[3]	1U	0
1,1,1-Trichloroethane*[1]		0	10U	0
1,1-Dichloroethane [1]		2	10U	0
Carbon Tetrachloride [1]		0	10U	0
Chloroethane* [1]		2	10U	0
Ethene/Ethane	> 0.01 mg/L > 0.1 mg/L	2 3	0.01U	0
Chloroform [1]		0 2[3]	10U	0
Dichloromethane [1]		0 2[3]	10U	0
TOTAL SCORE				**-3**

* Required Analysis.
[1] Concentration is reported in units of µg/L.
[2] Upgradient concentrations: CO_2 = 86.6 mg/L, Alkalinity = 43.3 mg/L, Chloride = 35.0 mg/L.
[3] Points awarded only if it can be shown that the compound is a daughter product.

TABLE 5-8
BIODEGRADATION SCREENING FOR MONITORING WELL MW-501S
REMEDIAL INVESTIGATION
RAYMARK – OU2 – GROUNDWATER
STRATFORD, CONNECTICUT

ANALYSIS	SCREENING		MW-501S	
	CONCENTRATION	SCORE	RESULTS	SCORE
Oxygen*	< 0.5 mg/L	3	5.2	0
Oxygen*	> 5 mg/L	-3	5.2	-3
Nitrate*	< 1mg/L	2	0.07	2
Iron II*	> 1 mg/L	3	0.03U	0
Sulfate*	< 20 mg/L	2	52.1	0
Sulfide*	> 1 mg/L	3	R	—
Methane*	> 0.5 mg/L	3	0.005U	0
Oxidation Reduction Potential* against Ag/AgCl electrode	< 50 mV < -100mV	1 2	446	0
pH*	pH < 5 pH > 9	-2	5.42	0
Total Organic Carbon	> 20 mg/L	2	3.15	0
Temperature*	> 20° C	1	14.2	0
Carbon Dioxide	> 2 X background [2]	1	71.6	0
Alkalinity	> 2 X background [2]	1	18	0
Chloride*	> 2 X background [2]	2	10	0
Hydrogen	> 1 nM	3	NA	—
Hydrogen	< 1 nM	0	NA	—
Volatile Fatty Acids	> 0.1 mg/L	2	NA	—
BTEX*	> 0.1 mg/L	2	0.01U	0
Tetrachloroethene [1]		0	10U	0
Trichloroethene* [1]		0 2[3]	10U	0
1,2-Dichloroethene* [1]		0 2[3]	10U	0
Vinyl Chloride* [1]		0 2[3]	1U	0
1,1,1-Trichloroethane*[1]		0	10U	0
1,1-Dichloroethane [1]		2	10U	0
Carbon Tetrachloride [1]		0	10U	0
Chloroethane* [1]		2	10U	0
Ethene/Ethane	> 0.01 mg/L > 0.1 mg/L	2 3	0.01U	0
Chloroform [1]		0 2[3]	10U	0
Dichloromethane [1]		0 2[3]	10U	0
TOTAL SCORE				**-1**

* Required Analysis.
[1] Concentration is reported in units of µg/L.
[2] Upgradient concentrations: CO_2 = 86.6 mg/L, Alkalinity = 43.3 mg/L, Chloride = 35.0 mg/L.
[3] Points awarded only if it can be shown that the compound is a daughter product.

TABLE 5-9
BIODEGRADATION SCREENING FOR MONITORING WELLS MW-501D AND MW-501B
REMEDIAL INVESTIGATION
RAYMARK – OU2 – GROUNDWATER
STRATFORD, CONNECTICUT

ANALYSIS	SCREENING		MW-501D		MW-501B	
	CONCENTRATION	SCORE	RESULTS	SCORE	RESULTS	SCORE
Oxygen*	< 0.5 mg/L	3	4.9	0	3.22	0
Oxygen*	> 5 mg/L	-3	4.9	0	3.22	0
Nitrate*	< 1mg/L	2	3.6	0	1.36	0
Iron II*	> 1 mg/L	3	0.03U	0	0.03	0
Sulfate*	< 20 mg/L	2	29.1	0	31.7	0
Sulfide*	> 1 mg/L	3	R	—	R	—
Methane*	> 0.5 mg/L	3	0.005U	0	0.005U	0
Oxidation Reduction Potential*	< 50 mV < -100mV	1 2	371	0	337	0
pH*	pH < 5 pH > 9	-2	5.99	0	7.73	0
Total Organic Carbon	> 20 mg/L	2	2U	0	4.5	0
Temperature*	> 20° C	1	12.7	0	10.3	0
Carbon Dioxide	> 2 X background	1	57.2	0	112	0
Alkalinity	> 2 X background [2]	1	26	0	118	1
Chloride*	> 2 X background [2]	2	54.4	0	35.7	0
Hydrogen	> 1 nM	3	NA	—	NA	—
Hydrogen	< 1 nM	0	NA	—	NA	—
Volatile Fatty Acids	> 0.1 mg/L	2	NA	—	NA	—
BTEX*	> 0.1 mg/L	2	0.01U	0	0.01U	0
Tetrachloroethene [1]		0	10U	0	10U	0
Trichloroethene* [1]		0 2[3]	10U	0	10U	0
1,2-Dichloroethene* [1]		0 2[3]	10U	0	10U	0
Vinyl Chloride* [1]		0 2[3]	1U	0	1U	0
1,1,1-Trichloroethane*[1]		0	10U	0	10U	0
1,1-Dichloroethane [1]		2	10U	0	10U	0
Carbon Tetrachloride [1]		0	10U	0	10U	0
Chloroethane* [1]		2	10U	0	10U	0
Ethene/Ethane	> 0.01 mg/L > 0.1 mg/L	2 3	0.01U	0	0.01U	0
Chloroform [1]		0 2[3]	10U	0	10U	0
Dichloromethane [1]		0 2[3]	10U	0	10U	0
TOTAL SCORE				**0**		**1**

* Required Analysis.
[1] Concentration is reported in units of µg/L.
[2] Upgradient concentrations: CO_2 = 86.6 mg/L, Alkalinity = 43.3 mg/L, Chloride = 35.0 mg/L.
[3] Points awarded only if it can be shown that the compound is a daughter product.

TABLE 5-10
BIODEGRADATION SCREENING FOR MONITORING WELLS GW-J2 AND GW-J5
REMEDIAL INVESTIGATION
RAYMARK – OU2 – GROUNDWATER
STRATFORD, CONNECTICUT

ANALYSIS	SCREENING		GW-J2		GW-J5	
	CONCENTRATION	SCORE	RESULTS	SCORE	RESULTS	SCORE
Oxygen*	< 0.5 mg/L	3	NA	—	NA	—
Oxygen*	> 5 mg/L	-3	NA	—	NA	—
Nitrate*	< 1mg/L	2	168	0	1690	0
Iron II*	> 1 mg/L	3	NA	—	NA	—
Sulfate*	< 20 mg/L	2	688	0	10	2
Sulfide*	> 1 mg/L	3	0.5U	0	0.5U	0
Methane*	> 0.5 mg/L	3	NA	—	NA	—
Oxidation Reduction Potential* against Ag/AgCl electrode	< 50 mV < -100mV	1 2	NA	—	NA	—
pH*	pH < 5 pH > 9	-2	4.22	-2	2.52	-2
Total Organic Carbon	> 20 mg/L	2	89.4	2	131	2
Temperature*	> 20° C	1	NA	—	NA	—
Carbon Dioxide	> 2 X background	1	NA	—	NA	—
Alkalinity	> 2 X background	1	1U	0	1U	0
Chloride*	> 2 X background	2	758	0	5390	2
Hydrogen	> 1 nM	3	NA	—	NA	—
Hydrogen	< 1 nM	0	NA	—	NA	—
Volatile Fatty Acids	> 0.1 mg/L	2	NA	—	NA	—
BTEX*	> 0.1 mg/L	2	25U	0	25U	0
Tetrachloroethene [1]		0	25000U	0	25000U	0
Trichloroethene* [1]		0 2[2]	580000	0	810000	0
1,2-Dichloroethene* [1]		0 2[2]	12000	2	50000U	0
Vinyl Chloride* [1]		0 2[2]	50000U	0	50000U	0
1,1,1-Trichloroethane*[1]		0	74000U	0	77000U	0
1,1-Dichloroethane [1]		2	25000U	0	25000U	0
Carbon Tetrachloride [1]		0	25000U	0	25000U	0
Chloroethane* [1]		2	50000U	0	25000U	0
Ethene/Ethane	> 0.01 mg/L > 0.1 mg/L	2 3	NA	—	NA	—
Chloroform [1]		0 2[2]	45000U	0	45000U	0
Dichloromethane [1]		0 2[2]	50000U	0	50000U	0
TOTAL SCORE				**2**		**4**

* Required Analysis.
[1] Concentration is reported in units of µg/L.
[2] Points awarded only if it can be shown that the compound is a daughter product.

TABLE 5-11
BIODEGRADATION SCREENING FOR MONITORING WELLS RW-3 AND RW-6
REMEDIAL INVESTIGATION
RAYMARK – OU2 – GROUNDWATER
STRATFORD, CONNECTICUT

ANALYSIS	SCREENING		RW-3		RW-6	
	CONCENTRATION	SCORE	RESULTS	SCORE	RESULTS	SCORE
Oxygen*	< 0.5 mg/L	3	NA	—	NA	—
Oxygen*	> 5 mg/L	-3	NA	—	NA	—
Nitrate*	< 1mg/L	2	NA	—	NA	—
Iron II*	> 1 mg/L	3	NA	—	NA	—
Sulfate*	< 20 mg/L	2	NA	—	NA	—
Sulfide*	> 1 mg/L	3	NA	—	NA	—
Methane*	> 0.5 mg/L	3	NA	—	NA	—
Oxidation Reduction Potential* against Ag/AgCl electrode	< 50 mV < -100mV	1 2	NA	—	NA	—
pH*	pH < 5 pH > 9	-2	4	-2	4.5	-2
Total Organic Carbon	> 20 mg/L	2	NA	—	NA	—
Temperature*	> 20° C	1	NA	—	NA	—
Carbon Dioxide	> 2 X background	1	NA	—	NA	—
Alkalinity	> 2 X background	1	NA	—	NA	—
Chloride*	> 2 X background	2	NA	—	NA	—
Hydrogen	> 1 nM	3	NA	—	NA	—
Hydrogen	< 1 nM	0	NA	—	NA	—
Volatile Fatty Acids	> 0.1 mg/L	2	NA	—	NA	—
BTEX*	> 0.1 mg/L	2	1.012	2	3.288	2
Tetrachloroethene [1]		0	200	0	2000	0
Trichloroethene* [1]		0 2[2]	89000	0	76000	0
1,2-Dichloroethene* [1]		0 2[2]	12160	0	25160	0
Vinyl Chloride* [1]		0 2[2]	180	0	400	0
1,1,1-Trichloroethane*[1]		0	24000	0	58000	0
1,1-Dichloroethane [1]		2	1707	2	6200	2
Carbon Tetrachloride [1]		0	10U	0	10U	0
Chloroethane* [1]		2	160	2	1000	2
Ethene/Ethane	> 0.01 mg/L > 0.1 mg/L	2 3	NA	—	NA	—
Chloroform [1]		0 2[2]	4	0	16	0
Dichloromethane [1]		0 2[2]	39	0	440	0
TOTAL SCORE				**4**		**4**

* Required Analysis.
[1] Concentration is reported in units of µg/L.
[2] Points awarded only if it can be shown that the compound is a daughter product.

TABLE 5-12
BIODEGRADATION SCREENING FOR MONITORING WELL RW-1
REMEDIAL INVESTIGATION
RAYMARK – OU2 – GROUNDWATER
STRATFORD, CONNECTICUT

ANALYSIS	SCREENING		RW-1	
	CONCENTRATION	SCORE	RESULTS	SCORE
Oxygen*	< 0.5 mg/L	3	NA	—
Oxygen*	> 5 mg/L	-3	NA	—
Nitrate*	< 1mg/L	2	NA	—
Iron II*	> 1 mg/L	3	NA	—
Sulfate*	< 20 mg/L	2	NA	—
Sulfide*	> 1 mg/L	3	NA	—
Methane*	> 0.5 mg/L	3	NA	—
Oxidation Reduction Potential* against Ag/AgCl electrode	< 50 mV < -100mV	1 2	NA	—
pH*	pH < 5 pH > 9	-2	3	-2
Total Organic Carbon	> 20 mg/L	2	NA	—
Temperature*	> 20° C	1	NA	—
Carbon Dioxide	> 2 X background	1	NA	—
Alkalinity	> 2 X background	1	NA	—
Chloride*	> 2 X background	2	NA	—
Hydrogen	> 1 nM	3	NA	—
Hydrogen	< 1 nM	0	NA	—
Volatile Fatty Acids	> 0.1 mg/L	2	NA	—
BTEX*	> 0.1 mg/L	2	3.419	2
Tetrachloroethene [1]		0	270	0
Trichloroethene* [1]		0 2[2]	100000	0
1,2-Dichloroethene* [1]		0 2[2]	833	0
Vinyl Chloride* [1]		0 2[2]	29	0
1,1,1-Trichloroethane*[1]		0	2400	0
1,1-Dichloroethane [1]		2	370	2
Carbon Tetrachloride [1]		0	11	0
Chloroethane* [1]		2	16	2
Ethene/Ethane	> 0.01 mg/L > 0.1 mg/L	2 3	NA	—
Chloroform [1]		0 2[2]	31	0
Dichloromethane [1]		0 2[2]	360	0
TOTAL SCORE				**4**

* Required Analysis.
[1] Concentration is reported in units of µg/L.
[2] Points awarded only if it can be shown that the compound is a daughter product.

TABLE 5-13
BIODEGRADATION SCREENING FOR MONITORING WELLS PC-14D AND PC-14S
REMEDIAL INVESTIGATION
RAYMARK – OU2 – GROUNDWATER
STRATFORD, CONNECTICUT

ANALYSIS	SCREENING		PC-14D		PC-14S	
	CONCENTRATION	SCORE	RESULTS	SCORE	RESULTS	SCORE
Oxygen*	< 0.5 mg/L	3	2.3	0	0.33	3
Oxygen*	> 5 mg/L	-3	2.3	0	0.33	0
Nitrate*	< 1mg/L	2	278	0	0.05U	2
Iron II*	> 1 mg/L	3	3.2	3	96	3
Sulfate*	< 20 mg/L	2	270	0	718	0
Sulfide*	> 1 mg/L	3	R	—	0.84	0
Methane*	> 0.5 mg/L	3	0.54	3	0.93	3
Oxidation Reduction Potential*	< 50 mV < -100mV	1 2	386	0	-56.1	1
pH*	pH < 5 pH > 9	-2	3.86	-2	6.43	0
Total Organic Carbon	> 20 mg/L	2	97	2	16.9	0
Temperature*	> 20° C	1	14.1	0	16.7	0
Carbon Dioxide	> 2 X background	1	R		275	1
Alkalinity	> 2 X background [2]	1	R		108	1
Chloride*	> 2 X background [2]	2	608	2	91	2
Hydrogen	> 1 nM	3	NA	—	NA	—
Hydrogen	< 1 nM	0	NA	—	NA	—
Volatile Fatty Acids	> 0.1 mg/L	2	NA	—	NA	—
BTEX*	> 0.1 mg/L	2	0.423	2	0.216	2
Tetrachloroethene [1]		0	21	0	10U	0
Trichloroethene* [1]		0 2[3]	11000	0	180	0
1,2-Dichloroethene* [1]		0 2[3]	190	2	2505	2
Vinyl Chloride* [1]		0 2[3]	29	2	530	2
1,1,1-Trichloroethane*[1]		0	130	0	7	0
1,1-Dichloroethane [1]		2	41	2	29	2
Carbon Tetrachloride [1]		0	10U	0	10U	0
Chloroethane* [1]		2	2	2	10U	0
Ethene/Ethane	> 0.01 mg/L > 0.1 mg/L	2 3	0.01U	0	0.108	3
Chloroform [1]		0 2[3]	46	0	61	0
Dichloromethane [1]		0 2[3]	190	0	10U	0
TOTAL SCORE				**18**		**27**

* Required Analysis.
[1] Concentration is reported in units of µg/L.
[2] Upgradient concentrations: CO_2 = 86.6 mg/L, Alkalinity = 43.3 mg/L, Chloride = 35.0 mg/L.
[3] Points awarded only if it can be shown that the compound is a daughter product.

TABLE 5-14
BIODEGRADATION SCREENING FOR MONITORING WELLS MW-302D AND MW-302B
REMEDIAL INVESTIGATION
RAYMARK – OU2 – GROUNDWATER
STRATFORD, CONNECTICUT

ANALYSIS	SCREENING		MW-302D		MW-302B	
	CONCENTRATION	SCORE	RESULTS	SCORE	RESULTS	SCORE
Oxygen*	< 0.5 mg/L	3	2.46	0	0.46	3
Oxygen*	> 5 mg/L	-3	2.46	0	0.46	0
Nitrate*	< 1mg/L	2	4.3	0	1.02	0
Iron II*	> 1 mg/L	3	0.03U	0	0.03	0
Sulfate*	< 20 mg/L	2	156	0	239	0
Sulfide*	> 1 mg/L	3	0.5U	0	0.5U	0
Methane*	> 0.5 mg/L	3	0.005U	0	0.01	0
Oxidation Reduction Potential*	< 50 mV < -100mV	1 2	204	0	110	0
pH*	pH < 5 pH > 9	-2	6.41	0	7.12	0
Total Organic Carbon	> 20 mg/L	2	7.6	0	4.6	0
Temperature*	> 20° C	1	13.2	0	12.7	0
Carbon Dioxide	> 2 X background	1	151	0	108	0
Alkalinity	> 2 X background [2]	1	98	1	110	1
Chloride*	> 2 X background [2]	2	119	2	1210	2
Hydrogen	> 1 nM	3	NA	—	NA	—
Hydrogen	< 1 nM	0	NA	—	NA	—
Volatile Fatty Acids	> 0.1 mg/L	2	NA	—	NA	—
BTEX*	> 0.1 mg/L	2	0.01U	0	0.01U	0
Tetrachloroethene [1]		0	20	0	8	0
Trichloroethene* [1]		0 2[3]	6400	0	1200	0
1,2-Dichloroethene* [1]		0 2[3]	1803	2	160	2
Vinyl Chloride* [1]		0 2[3]	3	2	4	2
1,1,1-Trichloroethane*[1]		0	760	0	220	0
1,1-Dichloroethane [1]		2	980	2	90	2
Carbon Tetrachloride [1]		0	10U	0	10U	0
Chloroethane* [1]		2	8	2	10U	0
Ethene/Ethane	> 0.01 mg/L > 0.1 mg/L	2 3	0.01U	0	0.01U	0
Chloroform [1]		0 2[3]	8	0	10U	0
Dichloromethane [1]		0 2[3]	10U	0	10U	0
TOTAL SCORE				**11**		**12**

* Required Analysis.
[1] Concentration is reported in units of µg/L.
[2] Upgradient concentrations: CO_2 = 86.6 mg/L, Alkalinity = 43.3 mg/L, Chloride = 35.0 mg/L.
[3] Points awarded only if it can be shown that the compound is a daughter product.

TABLE 5-15
BIODEGRADATION SCREENING FOR MONITORING WELL MW-302S
REMEDIAL INVESTIGATION
RAYMARK – OU2 – GROUNDWATER
STRATFORD, CONNECTICUT

ANALYSIS	SCREENING		MW-302S	
	CONCENTRATION	SCORE	RESULTS	SCORE
Oxygen*	< 0.5 mg/L	3	8.4	0
Oxygen*	> 5 mg/L	-3	8.4	-3
Nitrate*	< 1mg/L	2	7.3	0
Iron II*	> 1 mg/L	3	0.03U	0
Sulfate*	< 20 mg/L	2	24.4	0
Sulfide*	> 1 mg/L	3	0.5U	0
Methane*	> 0.5 mg/L	3	0.005U	0
Oxidation Reduction Potential* against Ag/AgCl electrode	< 50 mV < -100mV	1 2	418	0
pH*	pH < 5 pH > 9	-2	6.1	0
Total Organic Carbon	> 20 mg/L	2	2.89	0
Temperature*	> 20° C	1	14.0	0
Carbon Dioxide	> 2 X background [2]	1	36.3	0
Alkalinity	> 2 X background [2]	1	24	0
Chloride*	> 2 X background [2]	2	30.4	0
Hydrogen	> 1 nM	3	NA	—
Hydrogen	< 1 nM	0	NA	—
Volatile Fatty Acids	> 0.1 mg/L	2	NA	—
BTEX*	> 0.1 mg/L	2	0.01U	0
Tetrachloroethene [1]		0	10U	0
Trichloroethene* [1]		0 2[3]	59	0
1,2-Dichloroethene* [1]		0 2[3]	24	2
Vinyl Chloride* [1]		0 2[3]	1U	0
1,1,1-Trichloroethane*[1]		0	74	0
1,1-Dichloroethane [1]		2	16	2
Carbon Tetrachloride [1]		0	10U	0
Chloroethane* [1]		2	10U	0
Ethene/Ethane	> 0.01 mg/L > 0.1 mg/L	2 3	0.01U	0
Chloroform [1]		0 2[3]	2	0
Dichloromethane [1]		0 2[3]	10U	0
TOTAL SCORE				**1**

* Required Analysis.
[1] Concentration is reported in units of µg/L.
[2] Upgradient concentrations: CO_2 = 86.6 mg/L, Alkalinity = 43.3 mg/L, Chloride = 35.0 mg/L.
[3] Points awarded only if it can be shown that the compound is a daughter product.

TABLE 5-16
BIODEGRADATION SCREENING FOR MONITORING WELLS PC-02D AND PC-02B
REMEDIAL INVESTIGATION
RAYMARK – OU2 – GROUNDWATER
STRATFORD, CONNECTICUT

ANALYSIS	SCREENING		PC-02D		PC-02B	
	CONCENTRATION	SCORE	RESULTS	SCORE	RESULTS	SCORE
Oxygen*	< 0.5 mg/L	3	0.57	0	0.27	3
Oxygen*	> 5 mg/L	-3	0.57	0	0.27	0
Nitrate*	< 1mg/L	2	0.05U	2	0.08	2
Iron II*	> 1 mg/L	3	1860	3	2270	3
Sulfate*	< 20 mg/L	2	10400	0	10000	0
Sulfide*	> 1 mg/L	3	0.5U	0	R	—
Methane*	> 0.5 mg/L	3	0.016	0	0.008	0
Oxidation Reduction Potential*	< 50 mV < -100mV	1 2	165	0	205.8	0
pH*	pH < 5 pH > 9	-2	3.82	-2	3.74	-2
Total Organic Carbon	> 20 mg/L	2	36.2	2	21.6	2
Temperature*	> 20° C	1	12.4	0	13.4	0
Carbon Dioxide	> 2 X background	1	2U	0	R	—
Alkalinity	> 2 X background [2]	1	2U	0	R	—
Chloride*	> 2 X background [2]	2	201	2	235	2
Hydrogen	> 1 nM	3	NA	—	NA	—
Hydrogen	< 1 nM	0	NA	—	NA	—
Volatile Fatty Acids	> 0.1 mg/L	2	NA	—	NA	—
BTEX*	> 0.1 mg/L	2	0.027	0	0.011	0
Tetrachloroethene [1]		0	10U	0	10U	0
Trichloroethene* [1]		0 2[3]	130	0	190	0
1,2-Dichloroethene* [1]		0 2[3]	2	2	10U	0
Vinyl Chloride* [1]		0 2[3]	2	2	6	2
1,1,1-Trichloroethane*[1]		0	160000	0	120000	0
1,1-Dichloroethane [1]		2	245	2	338	2
Carbon Tetrachloride [1]		0	10U	0	10U	0
Chloroethane* [1]		2	10U	0	10U	0
Ethene/Ethane	> 0.01 mg/L > 0.1 mg/L	2 3	0.01U	0	0.01U	0
Chloroform [1]		0 2[3]	5	0	11	0
Dichloromethane [1]		0 2[3]	10U	0	10U	0
TOTAL SCORE				**13**		**14**

* Required Analysis.
[1] Concentration is reported in units of µg/L.
[2] Upgradient concentrations: CO_2 = 86.6 mg/L, Alkalinity = 43.3 mg/L, Chloride = 35.0 mg/L.
[3] Points awarded only if it can be shown that the compound is a daughter product.

TABLE 5-17
BIODEGRADATION SCREENING FOR MONITORING WELLS MW-211D AND MW-211B
REMEDIAL INVESTIGATION
RAYMARK – OU2 – GROUNDWATER
STRATFORD, CONNECTICUT

ANALYSIS	SCREENING		MW-211D		MW-211B	
	CONCENTRATION	SCORE	RESULTS	SCORE	RESULTS	SCORE
Oxygen*	< 0.5 mg/L	3	0.77	0	1.1	0
Oxygen*	> 5 mg/L	-3	0.77	0	1.1	0
Nitrate*	< 1mg/L	2	7.2	0	0.05U	2
Iron II*	> 1 mg/L	3	299	3	27	3
Sulfate*	< 20 mg/L	2	286	0	9910	0
Sulfide*	> 1 mg/L	3	0.5U	0	0.5U	0
Methane*	> 0.5 mg/L	3	0.046	0	0.016	0
Oxidation Reduction Potential*	< 50 mV < -100mV	1 2	199	0	219	0
pH*	pH < 5 pH > 9	-2	3.7	-2	3.95	-2
Total Organic Carbon	> 20 mg/L	2	2U	0	2U	0
Temperature*	> 20° C	1	13.5	0	9.3	0
Carbon Dioxide	> 2 X background	1	168	0	2U	0
Alkalinity	> 2 X background [2]	1	10	0	2U	0
Chloride*	> 2 X background [2]	2	233	2	234	2
Hydrogen	> 1 nM	3	NA	—	NA	—
Hydrogen	< 1 nM	0	NA	—	NA	—
Volatile Fatty Acids	> 0.1 mg/L	2	NA	—	NA	—
BTEX*	> 0.1 mg/L	2	0.001	0	0.004	0
Tetrachloroethene [1]		0	1	0	10U	0
Trichloroethene* [1]		0 2[3]	83	0	94	0
1,2-Dichloroethene* [1]		0 2[3]	4	2	4	2
Vinyl Chloride* [1]		0 2[3]	4	2	9	2
1,1,1-Trichloroethane*[1]		0	32000	0	24000	0
1,1-Dichloroethane [1]		2	280	2	630	2
Carbon Tetrachloride [1]		0	10U	0	10U	0
Chloroethane* [1]		2	10U	0	10U	0
Ethene/Ethane	> 0.01 mg/L > 0.1 mg/L	2 3	0.01U	0	10U	0
Chloroform [1]		0 2[3]	12	0	14	0
Dichloromethane [1]		0 2[3]	10U	0	10U	0
TOTAL SCORE				**9**		**11**

* Required Analysis.
[1] Concentration is reported in units of µg/L.
[2] Upgradient concentrations: CO_2 = 86.6 mg/L, Alkalinity = 43.3 mg/L, Chloride = 35.0 mg/L.
[3] Points awarded only if it can be shown that the compound is a daughter product.

TABLE 5-18
BIODEGRADATION SCREENING FOR MONITORING WELL MW-211M
REMEDIAL INVESTIGATION
RAYMARK – OU2 – GROUNDWATER
STRATFORD, CONNECTICUT

ANALYSIS	SCREENING		MW-211M	
	CONCENTRATION	SCORE	RESULTS	SCORE
Oxygen*	< 0.5 mg/L	3	0.49	3
Oxygen*	> 5 mg/L	-3	0.49	0
Nitrate*	< 1mg/L	2	0.05U	2
Iron II*	> 1 mg/L	3	0.03U	0
Sulfate*	< 20 mg/L	2	10300	0
Sulfide*	> 1 mg/L	3	0.5U	0
Methane*	> 0.5 mg/L	3	0.034	0
Oxidation Reduction Potential* against Ag/AgCl electrode	< 50 mV < -100mV	1 2	325	0
pH*	pH < 5 pH > 9	-2	5.38	0
Total Organic Carbon	> 20 mg/L	2	2U	0
Temperature*	> 20° C	1	13.2	0
Carbon Dioxide	> 2 X background	1	2U	0
Alkalinity	> 2 X background [2]	1	2U	0
Chloride*	> 2 X background [2]	2	213	2
Hydrogen	> 1 nM	3	NA	—
Hydrogen	< 1 nM	0	NA	—
Volatile Fatty Acids	> 0.1 mg/L	2	NA	—
BTEX*	> 0.1 mg/L	2	0.01U	0
Tetrachloroethene [1]		0	5	0
Trichloroethene* [1]		0 2[3]	2400	0
1,2-Dichloroethene* [1]		0 2[3]	460	2
Vinyl Chloride* [1]		0 2[3]	32	2
1,1,1-Trichloroethane*[1]		0	3500	0
1,1-Dichloroethane [1]		2	140	2
Carbon Tetrachloride [1]		0	10U	0
Chloroethane* [1]		2	10U	0
Ethene/Ethane	> 0.01 mg/L > 0.1 mg/L	2 3	0.01U	0
Chloroform [1]		0 2[3]	4	0
Dichloromethane [1]		0 2[3]	10U	0
TOTAL SCORE				**13**

* Required Analysis.
[1] Concentration is reported in units of µg/L.
[2] Upgradient concentrations: CO_2 = 86.6 mg/L, Alkalinity = 43.3 mg/L, Chloride = 35.0 mg/L.
[3] Points awarded only if it can be shown that the compound is a daughter product.

TABLE 5-19
BIODEGRADATION SCREENING FOR MONITORING WELLS MW-212M AND MW-212D
REMEDIAL INVESTIGATION
RAYMARK – OU2 – GROUNDWATER
STRATFORD, CONNECTICUT

ANALYSIS	SCREENING		MW-212M		MW-212D	
	CONCENTRATION	SCORE	RESULTS	SCORE	RESULTS	SCORE
Oxygen*	< 0.5 mg/L	3	0.45	3	1.02	0
Oxygen*	> 5 mg/L	-3	0.45	0	1.02	0
Nitrate*	< 1mg/L	2	0.05U	2	0.05U	2
Iron II*	> 1 mg/L	3	7.8	3	133	3
Sulfate*	< 20 mg/L	2	392	0	1110	0
Sulfide*	> 1 mg/L	3	0.5U	0	1.1	3
Methane*	> 0.5 mg/L	3	0.02	0	0.19	0
Oxidation Reduction Potential*	< 50 mV < -100mV	1 2	46	1	65	0
pH*	pH < 5 pH > 9	-2	5.96	0	5.59	0
Total Organic Carbon	> 20 mg/L	2	2U	0	4.2	0
Temperature*	> 20° C	1	11.8	0	13.2	0
Carbon Dioxide	> 2 X background	1	272	1	292	1
Alkalinity	> 2 X background [2]	1	110	1	68.5	0
Chloride*	> 2 X background [2]	2	140	2	211	2
Hydrogen	> 1 nM	3	NA	—	NA	—
Hydrogen	< 1 nM	0	NA	—	NA	—
Volatile Fatty Acids	> 0.1 mg/L	2	NA	—	NA	—
BTEX*	> 0.1 mg/L	2	0.002	0	0.007	0
Tetrachloroethene [1]		0	6	0	2	0
Trichloroethene* [1]		0 2[3]	1700	0	450	0
1,2-Dichloroethene* [1]		0 2[3]	940	2	2000	2
Vinyl Chloride* [1]		0 2[3]	54	2	240	2
1,1,1-Trichloroethane*[1]		0	4900	0	2500	0
1,1-Dichloroethane [1]		2	523	2	525	2
Carbon Tetrachloride [1]		0	10U	0	10U	0
Chloroethane* [1]		2	10U	0	10U	0
Ethene/Ethane	> 0.01 mg/L > 0.1 mg/L	2 3	0.01U	0	0.01U	0
Chloroform [1]		0 2[3]	5	0	7	0
Dichloromethane [1]		0 2[3]	10U	0	10U	0
TOTAL SCORE				**19**		**17**

* Required Analysis.
[1] Concentration is reported in units of µg/L.
[2] Upgradient concentrations: CO_2 = 86.6 mg/L, Alkalinity = 43.3 mg/L, Chloride = 35.0 mg/L.
[3] Points awarded only if it can be shown that the compound is a daughter product.

TABLE 5-20
BIODEGRADATION SCREENING FOR MONITORING WELL MW-212S
REMEDIAL INVESTIGATION
RAYMARK – OU2 – GROUNDWATER
STRATFORD, CONNECTICUT

ANALYSIS	SCREENING		MW-212S	
	CONCENTRATION	SCORE	RESULTS	SCORE
Oxygen*	< 0.5 mg/L	3	0.71	0
Oxygen*	> 5 mg/L	-3	0.71	0
Nitrate*	< 1mg/L	2	1.3	0
Iron II*	> 1 mg/L	3	0.03	0
Sulfate*	< 20 mg/L	2	242	0
Sulfide*	> 1 mg/L	3	0.5U	0
Methane*	> 0.5 mg/L	3	0.005U	0
Oxidation Reduction Potential* against Ag/AgCl electrode	< 50 mV < -100mV	1 2	189	0
pH*	pH < 5 pH > 9	-2	6.26	0
Total Organic Carbon	> 20 mg/L	2	2U	0
Temperature*	> 20° C	1	13.8	0
Carbon Dioxide	> 2 X background	1	278	1
Alkalinity	> 2 X background [2]	1	177	1
Chloride*	> 2 X background [2]	2	144	2
Hydrogen	> 1 nM	3	NA	—
Hydrogen	< 1 nM	0	NA	—
Volatile Fatty Acids	> 0.1 mg/L	2	NA	—
BTEX*	> 0.1 mg/L	2	0.003	0
Tetrachloroethene [1]		0	5	0
Trichloroethene* [1]		0 2[3]	1300	0
1,2-Dichloroethene* [1]		0 2[3]	680	2
Vinyl Chloride* [1]		0 2[3]	62	2
1,1,1-Trichloroethane*[1]		0	3500	0
1,1-Dichloroethane [1]		2	443	2
Carbon Tetrachloride [1]		0	10U	0
Chloroethane* [1]		2	10U	0
Ethene/Ethane	> 0.01 mg/L > 0.1 mg/L	2 3	0.01U	0
Chloroform [1]		0 2[3]	4	0
Dichloromethane [1]		0 2[3]	10U	0
TOTAL SCORE				**10**

* Required Analysis.
[1] Concentration is reported in units of µg/L.
[2] Upgradient concentrations: CO_2 = 86.6 mg/L, Alkalinity = 43.3 mg/L, Chloride = 35.0 mg/L.
[3] Points awarded only if it can be shown that the compound is a daughter product.

TABLE 5-21
BIODEGRADATION SCREENING FOR MONITORING WELL MW-503S
REMEDIAL INVESTIGATION
RAYMARK – OU2 – GROUNDWATER
STRATFORD, CONNECTICUT

ANALYSIS	SCREENING		MW-503S	
	CONCENTRATION	SCORE	RESULTS	SCORE
Oxygen*	< 0.5 mg/L	3	0.49	3
Oxygen*	> 5 mg/L	-3	0.49	0
Nitrate*	< 1mg/L	2	0.05U	2
Iron II*	> 1 mg/L	3	1.74	3
Sulfate*	< 20 mg/L	2	5U	2
Sulfide*	> 1 mg/L	3	0.5U	0
Methane*	> 0.5 mg/L	3	11	3
Oxidation Reduction Potential* against Ag/AgCl electrode	< 50 mV < -100mV	1 2	-117.9	2
pH*	pH < 5 pH > 9	-2	7.14	0
Total Organic Carbon	> 20 mg/L	2	6.6	0
Temperature*	> 20° C	1	9.9	0
Carbon Dioxide	> 2 X background	1	434	1
Alkalinity	> 2 X background [2]	1	433	1
Chloride*	> 2 X background [2]	2	32.4	0
Hydrogen	> 1 nM	3	NA	—
Hydrogen	< 1 nM	0	NA	—
Volatile Fatty Acids	> 0.1 mg/L	2	NA	—
BTEX*	> 0.1 mg/L	2	0.003	0
Tetrachloroethene [1]		0	10U	0
Trichloroethene* [1]		0 2[3]	10U	0
1,2-Dichloroethene* [1]		0 2[3]	10U	0
Vinyl Chloride* [1]		0 2[3]	1U	0
1,1,1-Trichloroethane*[1]		0	10U	0
1,1-Dichloroethane [1]		2	10U	0
Carbon Tetrachloride [1]		0	10U	0
Chloroethane* [1]		2	10U	0
Ethene/Ethane	> 0.01 mg/L > 0.1 mg/L	2 3	10U	0
Chloroform [1]		0 2[3]	10U	0
Dichloromethane [1]		0 2[3]	10U	0
TOTAL SCORE				**17**

* Required Analysis.
[1] Concentration is reported in units of µg/L.
[2] Upgradient concentrations: CO_2 = 86.6 mg/L, Alkalinity = 43.3 mg/L, Chloride = 35.0 mg/L.
[3] Points awarded only if it can be shown that the compound is a daughter product.

TABLE 5-22
BIODEGRADATION SCREENING FOR MONITORING WELLS MW-111M AND MW-111D
REMEDIAL INVESTIGATION
RAYMARK – OU2 – GROUNDWATER
STRATFORD, CONNECTICUT

ANALYSIS	SCREENING		MW-111M		MW-111D	
	CONCENTRATION	SCORE	RESULTS	SCORE	RESULTS	SCORE
Oxygen*	< 0.5 mg/L	3	0.98	0	0.96	0
Oxygen*	> 5 mg/L	-3	0.98	0	0.96	0
Nitrate*	< 1mg/L	2	0.05U	2	0.05U	2
Iron II*	> 1 mg/L	3	4.4	3	235	3
Sulfate*	< 20 mg/L	2	380	0	3280	0
Sulfide*	> 1 mg/L	3	0.5U	0	1.4	3
Methane*	> 0.5 mg/L	3	0.14	0	0.25	0
Oxidation Reduction Potential* against Ag/AgCl electrode	< 50 mV < -100mV	1 2	6	1	258	0
pH*	pH < 5 pH > 9	-2	6.31	0	4.05	-2
Total Organic Carbon	> 20 mg/L	2	8.5	0	21	2
Temperature*	> 20° C	1	12.2	0	12.6	0
Carbon Dioxide	> 2 X background	1	240	1	2U	0
Alkalinity	> 2 X background [2]	1	174	1	2U	0
Chloride*	> 2 X background [2]	2	332	2	210	2
Hydrogen	> 1 nM	3	NA	—	NA	—
Hydrogen	< 1 nM	0	NA	—	NA	—
Volatile Fatty Acids	> 0.1 mg/L	2	NA	—	NA	—
BTEX*	> 0.1 mg/L	2	0.009	0	0.003	0
Tetrachloroethene [1]		0	4	0	8	0
Trichloroethene* [1]		0 2[3]	540	0	1200	0
1,2-Dichloroethene* [1]		0 2[3]	1000	2	1000	2
Vinyl Chloride* [1]		0 2[3]	68	2	68	2
1,1,1-Trichloroethane*[1]		0	1300	0	6500	0
1,1-Dichloroethane [1]		2	375	2	363	2
Carbon Tetrachloride [1]		0	10U	0	10U	0
Chloroethane* [1]		2	10U	0	10U	2
Ethene/Ethane	> 0.01 mg/L > 0.1 mg/L	2 3	0.01U	0	0.01U	0
Chloroform [1]		0 2[3]	3	0	13	0
Dichloromethane [1]		0 2[3]	10U	0	10U	0
TOTAL SCORE				**16**		**18**

* Required Analysis.
[1] Concentration is reported in units of µg/L.
[2] Upgradient concentrations: CO_2 = 86.6 mg/L, Alkalinity = 43.3 mg/L, Chloride = 35.0 mg/L.
[3] Points awarded only if it can be shown that the compound is a daughter product.

TABLE 5-23
BIODEGRADATION SCREENING FOR MONITORING WELL MW-113M
REMEDIAL INVESTIGATION
RAYMARK – OU2 – GROUNDWATER
STRATFORD, CONNECTICUT

ANALYSIS	SCREENING		MW-113M	
	CONCENTRATION	SCORE	RESULTS	SCORE
Oxygen*	< 0.5 mg/L	3	0.41	3
Oxygen*	> 5 mg/L	-3	0.41	0
Nitrate*	< 1mg/L	2	R	—
Iron II*	> 1 mg/L	3	0.8	0
Sulfate*	< 20 mg/L	2	5U	2
Sulfide*	> 1 mg/L	3	0.6	0
Methane*	> 0.5 mg/L	3	3	3
Oxidation Reduction Potential* against Ag/AgCl electrode	< 50 mV < -100mV	1 2	-44.1	1
pH*	pH < 5 pH > 9	-2	6.54	0
Total Organic Carbon	> 20 mg/L	2	2U	0
Temperature*	> 20° C	1	12.3	0
Carbon Dioxide	> 2 X background	1	412	1
Alkalinity	> 2 X background [2]	1	364	1
Chloride*	> 2 X background [2]	2	561	2
Hydrogen	> 1 nM	3	NA	—
Hydrogen	< 1 nM	0	NA	—
Volatile Fatty Acids	> 0.1 mg/L	2	NA	—
BTEX*	> 0.1 mg/L	2	0.01U	0
Tetrachloroethene [1]		0	10U	0
Trichloroethene* [1]		0 2[3]	10U	0
1,2-Dichloroethene* [1]		0 2[3]	1	2
Vinyl Chloride* [1]		0 2[3]	2	2
1,1,1-Trichloroethane*[1]		0	10U	0
1,1-Dichloroethane [1]		2	2	2
Carbon Tetrachloride [1]		0	10U	0
Chloroethane* [1]		2	10U	0
Ethene/Ethane	> 0.01 mg/L > 0.1 mg/L	2 3	0.01U	0
Chloroform [1]		0 2[3]	10U	0
Dichloromethane [1]		0 2[3]	10U	0
TOTAL SCORE				**19**

* Required Analysis.
[1] Concentration is reported in units of µg/L.
[2] Upgradient concentrations: CO_2 = 86.6 mg/L, Alkalinity = 43.3 mg/L, Chloride = 35.0 mg/L.
[3] Points awarded only if it can be shown that the compound is a daughter product.

TABLE 6-1
SUMMARY OF GROUNDWATER HUMAN HEALTH RISK ASSESSMENT
REMEDIAL INVESTIGATION
RAYMARK - OU2 - GROUNDWATER
STRATFORD, CONNECTICUT

Chemicals of Potential Concern	Receptor	Reasonable Maximum Exposure (RME) Risk Estimates						
		ILCR	Risk Drivers				HI	Risk Drivers
			$>10^{-3}$	$>10^{-4}$	$>10^{-5}$	$>10^{-6}$		>1.0
Indoor Air	Resident (adult/child) (current/future)	4.8E-03	Trichloroethene	1,1,2,2-Tetrachloroethane 1,3-Butadiene Benzyl Chloride Methylene Chloride	1,1,2-Trichloroethane 1,2-Dibromoethane Benzene Bromodichloromethane Chloroform Dibromochloromethane Ethylbenzene Tetrachloroethene Tetrahydrofuran	1,2-Dichloroethane Acrylonitrile Carbon Tetrachloride Chloroethane Chloromethane Methyl tert-Butyl Ether Vinyl chloride	37	1,2,4-Trimethylbenzene 1,2-Dibromoethane 1,3,5-Trimethylbenzene 1,4-Dichlorobenzene Benzene Chloroform Total Xylenes Trichloroethene
1,1,2,2-Tetrachloroethane 1,1,2-Trichloroethane 1,1-Dichloroethene 1,2,4-Trimethylbenzene 1,2-Dibromoethane 1,2-Dichloroethane 1,3,5-Trimethylbenzene 1,3-Butadiene 1,4-Dichlorobenzene 4-Methyl-2-Pentanone Acetone Acrylonitrile Benzene Benzyl Chloride Bromodichloromethane Carbon Tetrachloride Chloroethane Chloroform Chloromethane cis-1,2-Dichloroethene Dibromochloromethane Dichlorodifluoromethane Ethylbenzene Hexane Methyl tert-Butyl Ether Methylene Chloride Tetrachloroethene Tetrahydrofuran Toluene Total Xylenes Trichloroethene Vinyl Acetate Vinyl Chloride	Industrial/ Commercial worker (adult) (current/future)	9.70E-04	None	1,1,2,2-Tetrachloroethane 1,3-Butadiene Trichloroethene	Benzyl Chloride Methylene Chloride 1,1,2-Trichloroethane 1,2-Dibromoethane	Benzene Bromodichloromethane Chloroform Dibromochloromethane Ethylbenzene Tetrachloroethene Tetrahydrofuran Carbon Tetrachloride Acrylonitrile	8.9	1,2,4-Trimethylbenzene 1,2-Dibromoethane 1,4-Dichlorobenzene Total Xylenes
Surface Water	Recreational User (adult) (future)	6.8E-08	None	None	None	None	0.0008	None
1,1-Dichloroethene Arsenic Zinc	Recreational User (child) (future)	3.4E-08	None	None	None	None	0.0016	None

TABLE 7-1
FERRY CREEK SURFACE WATER SAMPLES PREVIOUSLY EVALUATED [a]
REMEDIAL INVESTIGATION
RAYMARK – OU2 - GROUNDWATER
STRATFORD, CONNECTICUT

COPC	Range of Detected Concentrations (µg/L)	Frequency of Samples Exceeding ESV	ESV [b] (µg/L)	Maximum Hazard Quotient [c]
Arsenic	3.9 - 75.1	2/14	36	2.1
Chromium	4.4 - 12.3	0/14	50	0.2
Copper	6.3 - 138	7/14	3.1	44.5
Lead	3.7 - 118	2/14	8.1	14.6
Mercury	0.1 - 1.7	2/14	0.94	1.8
Nickel	8.4	1/14	8.2	1.02
Zinc	27.2 - 126	2/14	81	1.6
Aroclor-1262	0.15	1/14	0.03	5.0
4,4'-DDT	0.026	1/14	0.001	26.0

a See *OU3 Ferry Creek – Area I Remedial Investigation Report* (TtNUS, 1999a)

b Ecological screening values (ESVs) in this table are chronic ambient water quality criteria (USEPA, 2002b).

c Maximum hazard quotient = maximum detected concentration ÷ ESV.

COPC = Chemical of potential concern

TABLE 7-2
SURFACE WATER AND SEEP SAMPLES COLLECTED IN 1999
REMEDIAL INVESTIGATION
RAYMARK – OU2 - GROUNDWATER
STRATFORD, CONNECTICUT

Chemical	Location (Seep or Ferry Creek)	Range of Detected Concentrations (µg/L)	Frequency of Samples Exceeding ESV	ESV (µg/L)	Maximum Hazard Quotient [a]
1,1,1-Trichloroethane	Seep samples	3-290	1 of 14	62 [b]	4.7
	Ferry Creek	2-52	0		0.8
1,1-Dichloroethane	Seep samples	5-130	2 of 14	47 [b]	2.8
	Ferry Creek	1-8	0		0.2
1,1-Dichloroethene	Seep samples	2-210	1 of 14	25 [c]	8.4
	Ferry Creek	1-9	0		0.4
1,2-Dichloroethene (total)	Seep samples	2-770	1 of 14	590 [c]	1.3
	Ferry Creek	14-17	0		0.03
2-Butanone	Seep samples	2	0	14,000 [c]	0.0001
	Ferry Creek	ND	0		NA
4-Nitrophenol	Seep samples	4-6	0	300 [c]	0.02
	Ferry Creek	ND	0		NA
Acetone	Seep samples	46	0	1,500 [c]	0.03
	Ferry Creek	ND	0		NA
Benzene	Seep samples	1-11	0	46 [b]	0.2
	Ferry Creek	ND	0		NA
Carbon disulfide	Seep samples	1	1 of 14	0.92 [c]	1.1
	Ferry Creek	ND	0		NA
Chlorobenzene	Seep samples	3-37	0	130 [b]	0.3
	Ferry Creek	ND	0		NA
Chloroethane	Seep samples	1-24	0	230,000 [d]	0.0001
	Ferry Creek	ND	0		NA
Diethylphthalate	Seep samples	2	0	220 [b]	0.01
	Ferry Creek	ND	0		NA
Di-n-butylphthalate	Seep samples	4	0	33 [b]	0.1
	Ferry Creek	ND	0		NA
Phenol	Seep samples	1-3	0	110 [e]	0.03
	Ferry Creek	ND	0		NA
Tetrachloroethene	Seep samples	ND	0	120 [b]	NA
	Ferry Creek	2-3	0		0.03
Toluene	Seep samples	4	0	130 [b]	0.03
	Ferry Creek	ND	0		NA
Trichloroethene	Seep samples	1-380	1 of 14	350 [b]	1.1
	Ferry Creek	3-16	0		0.05
Vinyl chloride	Seep samples	4-55	0	224,000 [f]	0.0002
	Ferry Creek	2	0		0.000009

Notes:
a Maximum hazard quotient = maximum detected concentration ÷ESV.
b USEPA Tier II screening value for fresh water (USEPA, 1996a).
c Tier II secondary chronic value for fresh water (ORNL, 1996).
d Freshwater chronic guideline (USGS, 1997)
e Chronic ambient fresh water quality criteria calculated by the Great Lakes Water Quality Initiative (ORNL, 1996).
f USEPA Region III BTAG ecological screening level for marine water (USEPA, 1995a).
ESV Ecological screening value.
ND Analyte not detected.
NA Hazard quotient not calculated since analyte was not detected.

TABLE 7-3
HAZARD QUOTIENTS FOR FUTURE SEDIMENT PORE WATER AND SURFACE WATER CONCENTRATIONS
REMEDIAL INVESTIGATION
RAYMARK - OU2 - GROUNDWATER
STRATFORD, CONNECTICUT

Chemical	Ecological Screening Value (µg/L)	Sediment Pore Water Concentration Predicted at Seep (µg/L)	Pore Water Hazard Quotient at Seep	Surface Water Concentration Estimated in Ferry Creek or the Housatonic River [a] (µg/L)	Surface Water Hazard Quotient in Ferry Creek or the Housatonic River [a]
Arsenic	36[b]	84	2.3	8.4	0.23
Zinc	81[b]	330	4.1	33	0.41
1,1-DCE	25[c]; 2240[d]	530	21; 0.24	53	2.1; 0.024
TCE	350[e]	5,700	16.3	Unknown; see Section 7.7.2	Unknown; see Section 7.7.2

a Based on groundwater modeling, groundwater plumes of arsenic, zinc, and 1,1-DCE are expected to discharge into Ferry Creek, and TCE is expected to discharge into the Housatonic River.
b Chronic water quality criteria (USEPA, 2002)
c Tier II secondary chronic value for fresh water (ORNL, 1996)
d USEPA Region IV chronic ecological screening value for salt water (USEPA, 2000)
e USEPA Tier II screening value for fresh water (USEPA, 1996)

REFERENCES

REFERENCES

Agency for Toxic Substances and Disease Registry (ATSDR), 1996. Public Health Assessment for Raymark Industries, Stratford, Fairfield County, Connecticut. September 25, 1996.

Aquarian Water Company, 2003. Telephone conversation with customer service representative. Subject: Update on water supply sources, conditions of reservoir use. October 3, 2003.

Bethke, C. and P. V. Brady. 2000. How the Kd approach undermines ground water cleanup. *Ground Water* 38, 435-443.

Bridgeport Hydraulic Company, 1999a. Telephone Conversation with Lab Technician. Subject: Water supply sources, Conditions of reservoir use. Aug. 2,1999.

Bridgeport Hydraulic Company, 1999b. Telephone Conversation with Peter Gallan. Subject: Water supply service for Stratford CT. by BHC. Aug. 24, 1999.

Brown and Root Environmental. 1997, Final Technical Memorandum, Compilation of Existing Data, Raymark – Ferry Creek, Stratford, Connecticut, February, 1997.

Brown and Root Environmental, 1998. Draft Technical Memorandum for OU2. May 1998.

CDM Federal Programs Corporation, 1994. Final RCRA Facility Assessment for Synthetic Products Company Plant #2. Jan. 4, 1994.

Cherkauer, D. A. and J. M. McBride, 1988. A Remotely Operated Seepage Meter for use in Large Lakes and Rivers. Groundwater. Vol. 28 No. 2 Pages 165-171.

Christensen, T.H., N. Lehmann, T. Jackson, and P.E. Holm. 1996. Cadmium and nickel distribution coefficients for sandy aquifer materials. *J. Contam. Hydrol.*, v. 24, p. 75-84.

Conestoga-Rover & Associates (CRA), 1996. Supplemental Hydrogeologic Investigation Report, Synthetic Products Company. May 1996.

Connecticut Department of Environmental Protection (CTDEP), January 1996. Remediation Standard Regulations. Bureau of Water Management, Permitting, Enforcement and Remediation Division, Hartford, Connecticut.

Connecticut Department of Environmental Protection (CTDEP), 1997a. Water Quality Classifications. Adopted April 8, 1997.

Connecticut Department of Environmental Protection (CTDEP), 1997b. Letter to H. Lagautte, Brown and Root Environmental/NUS, from S. Kingsbury, Connecticut Department of Environmental Protection, Natural Resources Center; Subject: Ferry Creek & Selby Pond, Stratford, Connecticut. June 27, 1997.

Connecticut Department of Environmental Protection (CTDEP), 2002. Water Quality Standards. Bureau of Water Management, Hartford, Connecticut. http://www.dep.state.ct.us.

Connecticut Department of Environmental Protection (CTDEP), March 2003. Proposed Revisions Connecticut's Remediation Standard Regulations Volatilization Criteria. Bureau of Water Management, Permitting, Enforcement and Remediation Division, Hartford, Connecticut.

Connecticut Department of Transportation (CT DOT), 1995. Drainage Improvements on U.S. Route 1 to Ferry Creek. Region 1, CONN. Stratford. Project No.: 138-210 U.S. 1. 19 Sheets.

Connecticut Geological and Natural History Survey (CT GNHS), 1985. Bedrock Geologic Map of Connecticut.

Davis, J.A., C.C. Fuller, J.A. Coston, K.M. Hess, and E. Dixon. 1993. *Spatial heterogeneity of geochemical and hydraulic parameters affecting metal transport in ground water.* Environmental Research Brief. USEPA Office of Research and Development, Washington, DC.

Davis, J. A. and D. B. Kent. 1990. Surface complexation modeling in aqueous geochemistry. In *Mineral-Water Interface Geochemistry*, eds. M.F. Hochella and A. F. White, 177-260. Washington, DC: Mineralogical Society of America.

Dominico, P. A. 1987. An analytical model for multidimensional transport of a decaying contaminant species. *J. Hydrol.*, v. 91, p. 49-58.

Environmental Laboratories, Inc. (ELI), 1994. Hydrologic Runoff Analysis, Raymark Industries Inc., Stratford, Connecticut. April 25, 1994.

Environmental Laboratories, Inc. (ELI), 1995. Final RCRA Facility Investigation Report, Raymark, Industries, Inc., Stratford, Connecticut. February 10, 1995.

Federal Emergency Management (FEMA), 1992. "Flood Insurance Rate Map, Town of Stratford, Connecticut, Fairfield, County: Community Panel Nos. 090016-0004", June 16, 1992.

Flint, K. F. 1968. The Surficial Geology of the Ansonia and Milford Quadrangles with map: Geology of the State Geological and Natural History Survey of Connecticut, Scale 1:2400, 2 sheet, 36 p.

Foster Wheeler Environmental Corporation (Foster Wheeler), 1996. Remediation of Raymark Industries, Inc. Part of the Raymark Superfund Site RCRA Cap Design, Stratford, Connecticut, Final Basis of Design/Design Analysis Report. September 1996.

Freeze, R. A. and J. A. Cherry. 1979. Groundwater. Englewood Cliffs, New Jersey: Prentice-Hall, Inc.

Fritts, C. 1965. Milford Quadrangle: Fairfield and New Haven Counties, CT; Bedrock Geological Map, Scale 1:2400.

Gelhar, L. W., C. Welty, and K. R. Rehfeldt. 1992. A critical review of data on field-scale dispersion in aquifers. *Water Resources Res.*, v. 28, no. 7, p. 1955-1974.

Geophysical Applications, Inc., 2002. Geophysical Survey, Raymark Site – OU2 Project, Stratford, Connecticut. November 2002.

Geophysical Applications, Inc., 2003. Borehole Geophysics Logging Report, Wells MW-213DB, -215DB, and -524DB, Raymark Superfund Site, Stratford, Connecticut, July 2003.

Hager-Richter Geoscience, Inc., 1994, Ground Penetrating Radar Survey Raymark Site Stratford, Connecticut. March 1994.

Hager-Richter Geoscience, Inc., 1999a, Seismic Refraction Survey, OU2-Groundwater Study, Raymark Superfund Site, Stratford, Connecticut, September 1999.

Hager-Richter Geoscience, Inc., 1999b, Geophysical Survey, OU4 Ballfield Site, Stratford, Connecticut. June 1999.

Halliburton NUS (HNUS), 1995. Final Remedial Investigation Report, Raymark Industries, Inc. Facility. April 1995.

Howard, P.H. 1989. *Handbook of Environmental Fate and Exposure Data for Organic Chemicals.* Volume 1. Lewis Publishers, Chelsea, MI. 574p.

HRP Associates, Inc. Engineering and Geology, 1997. Closure Plan – Volume I, For Two >90-Day Hazardous Waste Container Storage Areas and two >90-Day Hazardous Waste Storage Tanks. Former Synthetic Products Company, May 1997. (Revised August 1999).

Johnson, P.C., and Ettinger, R.A. 1991. Heuristic model for predicting the intrusion rate of contaminant vapors into buildings. Environ. Sci. Technology, 25:1445-1452.

Johnson, P. C., C. C. Stanley, M. W. Kemblowski, D. L. Byers, and J. D. Colthart. 1990. A practical approach to the design, operation, and monitoring of in situ soil venting systems. *Ground Water Monitoring Review* 10(2): 159-178.

Karikoff, S. W. 1984. Organic pollutant sorption in aquatic systems. *J. Hydraul. Engr.,* v. 110, no. 6, p. 707-735.

Koretsky, C. 2000. The significance of surface complexation reactions in hydrologic systems: a geochemist's perspective. *Journal of Hydrology* 230: 127-171.

Lambe, T. William and Robert V. Whitman pp. 284, 285. 1969. Soil Mechanics, John Wiley and Sons, New York.

Metcalf & Eddy, 2004. Former Raymark Industries Site, Stratford, Connecticut. Post-Remediation Groundwater Monitoring Annual Report, April 2004.

National Oceanic and Atmospheric Administration (NOAA), 1998. Phase II Ecological Risk Assessment, Raymark Industries. May.

National Oceanic and Atmospheric Administration (NOAA), 2002. Climatography of the United States, No. 81: Monthly Station Normals of Temperature, Precipitation, and Heating and Cooling Degree Days 1971-2000; Connecticut.

National Oceanic and Atmospheric Administration (NOAA), 2004. Verified/Historic (Tides) Water Level Data, Station No. 8467150, Bridgeport, CT. http://co-ops.nos.noaa.gov/data_options.shtml?stn=8467150+Bridgeport,+CT

Oak Ridge National Laboratory (ORNL). 1996. Toxicological Benchmarks for Screening Potential Contaminants of Concern for Effects on Aquatic Biota, 1996 Revision. ES/ER/TM-96/R2. Risk Assessment Program, Health Sciences Research Division, Oak Ridge National Laboratory, Oak Ridge, Tennessee.

Pankow, J. F. and J. A. Cherry. 1996. *Dense chlorinated solvents and other DNAPLs in groundwater: history, behavior, and remediation*. Waterloo Press, Portland, OR, 522p.

Pedrosa, M. J. (Chemist), 1983. Letter to J. P. Roper (Town of Stratford, Director of Public Works). Subject: Frog Pond Pump Station. June 13, 1983.

Ranzau, Jr. C.E., Frick, T.W., Norris, J.R., and Martin, J.W. 2000. Water Resources Data, Connecticut. Water Year 1999: U.S. Geological Survey Resources Data Report CT-99-1, 351 p.

Roy F.Weston, Inc., (Weston), 1993. Final Site Inspection Report for Raymark Industries, Stratford, Connecticut. September 30, 1993.

Risk Based Corrective Action (RBCA) 1994. Emergency standard Guide for Risk-based Corrective Action Applied at Petroleum Release Sites. ASTM ES 38-94.

SAIC, 1999. *Draft Raymark Phase III Ecological Risk Assessment Report.* SAIC, Newport, RI. July.

Stone, J.R., Schafer, J.P., London, E.H., and Thompson, W.B., 1992, Surficial Materials Map of Connecticut: U.S. Geological Special Map, Scale I: 125,000, 2 sheets.

Stone, J.R., Schafer, J.P., London, E.H., DiGiacomo-Cohen, M.L., Lewis, R.L., and Thompson, W.B., 1998, Quarternary Geologic Map of Connecticut and Long Island Sound Basin: U.S. Geological Survey Open-File Report 98-371, Scale I: 175,000, 77 pages.

Stratford, Environmental Conservation Office, 1999a Telephone Conversations with Bill McCaan. Subject: Status of well use and well locations in Stratford. CT. Aug – Sept 1999.

Stratford, Health Department, 1999b Telephone Conversations with Ed Knapik. Subject: Status of well use and well location in Stratford. CT. Verifications on well inventories.
Aug – Sept. 1999.

Stratford Health Department, 2003. Telephone conversation with Ed Knapik. Subject – status of well use and well locations in Stratford, CT. October 6, 2003.

Stumm, W. and J. J. Morgan. 1996. *Aquatic Chemistry, 3rd ed*. New York, NY: John Wiley & Sons.

Tetra Tech NUS (TtNUS), 1999a Final Area I Remedial Investigation, Raymark – Ferry Creek OU3. October 1999.

Tetra Tech NUS (TtNUS), 1999b. Final Remedial Investigation, Raymark OU4 – Ballfield. Stratford, Connecticut. April 1999.

Tetra Tech NUS (TtNUS), 1999c. Draft Final Engineering Evaluation/Cost Analysis, Raymark – Shore Road. Stratford, Connecticut. June 1999.

Tetra Tech NUS (TtNUS), 1999d. Draft Addendum Engineering Evaluation/Cost Analysis, Raymark – Shore Road. Stratford, Connecticut. July 1999.

Tetra Tech NUS (TtNUS), 1999e. Technical Addendum to the Engineering Evaluation/Cost Analysis Report, Raymark-Shore Road, Stratford, Connecticut. September 1999.

Tetra Tech NUS (TtNUS), 2000a. Five-Year Review Raymark Industries Inc. Site. Stratford, Connecticut. September 2000.

Tetra Tech NUS (TtNUS), 2000b. Draft Final Area II Remedial Investigation, Raymark – Ferry Creek OU3. November 2000.

Tetra Tech NUS (TtNUS), 2000c. Draft Final Remedial Investigation, Raymark OU2-Groundwater. Stratford, Connecticut. November 2000.

Tetra Tech NUS (TtNUS), 2000d. Draft Final Area III Remedial Investigation, Raymark – Ferry Creek OU3. November 2000.

Tetra Tech NUS (TtNUS), 2004. Final Remedial Investigation, Raymark – OU6 – Additional Properties. April 2004.

United States Army Corps of Engineers (ACOE), 1998. Hydrologic and Hydraulic Analysis, Remedial Investigation and Feasibility Study, Raymark – Ferry Creek – Operable Unit No. 3, Stratford, Connecticut. March 1998.

United States Air Force (USAF). 1989. Benzene. In: The Installation Restoration Program Toxicology Guide. Wright Patterson Air Force Base, OH, p. 18-1 - 18-42.

United States Bureau of Reclamation (USBOR), 1977. Ground Water Manual: A Water Resources Technical Publication.

United States Environmental Protection Agency (EPA), 1986. Guidelines for Carcinogenic Risk Assessment. EPA/630/R-00/004, Risk Assessment Forum, Washington, D.C.

United States Environmental Protection Agency (EPA), 1988. “Interim Final Guidance for Conducting Remedial Investigations and Feasibility Studies Under CERCLA”. EPA/540/G-89/004. October.

United States Environmental Protection Agency (EPA), Region I, 1989a. Draft Final Supplemental Risk Assessment Guidance for the Superfund Program. EPA/901/5-89/001. Boston, Massachusetts. June 1989.

United States Environmental Protection Agency (EPA), 1989b. Risk Assessment Guidance for Superfund - Volume I - Human Health Evaluation Manual (Part A) - Interim Final. EPA/540/1-89/002. Office of Emergency and Remedial Response. December 1989.

United States Environmental Protection Agency (EPA), 1991. Risk Assessment Guidance for Superfund - Volume I: Human Health Evaluation Manual - Supplemental Guidance - "Standard Default Exposure Factors" - Interim Final. OSWER Directive 9285.6-03. Office of Emergency and Remedial Response. March 25, 1991.

United States Environmental Protection Agency (EPA), 1992a. Guidance on Risk Characterization for Risk Managers and Risk Assessors. Office of the Administrator, Washington, DC. February 26, 1992.

United States Environmental Protection Agency (EPA), 1992b. Supplemental Guidance to RAGS: Calculating the Concentration Term. OSWER Publication 9285.7-081. May 1992.

United States Environmental Protection Agency (EPA), Region I, 1994. Risk Updates, Number 2. Waste Management Division, Boston, Massachusetts. August 1994.

United States Environmental Protection Agency (EPA). 1995a. *EPA Region III BTAG Screening Levels*, Philadelphia, Pennsylvania.

United States Environmental Protection Agency (EPA), Region I, 1995b. Risk Updates, Number 3. Waste Management Division, Boston, Massachusetts. August 1995.

United States Environmental Protection Agency (EPA). 1996a. *ECO Update, Ecotox Thresholds*. Volume 3, Number 2, EPA 540/F-95/038, Office of Emergency and Remedial Response, Washington, D.C. January 1996.

United States Environmental Protection Agency (EPA), Region I, 1996b. Risk Updates, Number 4. Waste Management Division, Boston, Massachusetts. November 1996.

United States Environmental Protection Agency (EPA). 1997a. *Ecological Risk Assessment Guidance for Superfund: Process for Designing and Conducting Ecological Risk Assessments*, Office of Solid Waste and Emergency Response, EPA 540-R-97-006. June 1997.

United States Environmental Protection Agency (EPA), 1997b. Health Effects Assessment Summary Tables; FY 1997 Update, EPA 540/R-97-036, National Center for Environmental Assessment, Cincinnati, Ohio. July 1997.

United States Environmental Protection Agency (EPA), 1997c. Exposure Factors Handbook. EPA/600/8-89/043. Office of Health and Environmental Assessment, Exposure Assessment Group. Washington, D.C. August 1997.

United States Environmental Protection Agency (EPA), 1997d. User's Guide for the Johnson and Ettinger (1991) Model for Subsurface Vapor Intrusion into Buildings. Office of Emergency and Remedial Response. Washington, D.C. September 1997.

United States Environmental Protection Agency (EPA), 1997e. Risk Assessment Guidance for Superfund – Volume I, Human Health Evaluation Manual – Part D – Interim Guidance. EPA/540/R-97/0333 OSWER Publication 9285.7-01D. December 1997.

United States Environmental Protection Agency (EPA), 1998. Evaluating Natural Attenuation of Chlorinated Solvents in Groundwater. EPA/600/R-98/128. September 1998.

United States Environmental Protection Agency (EPA), Region I, 1999a. Risk Updates, Number 5. Waste Management Division, Boston, Massachusetts. August 1999.

United States Environmental Protection Agency (EPA), 1999b. Guidelines for Carcinogenic Risk Assessment. NCEA-F-0644, Risk Assessment Forum, Washington, D.C. July 1999.

United States Environmental Protection Agency (EPA), 2001a. Supplemental Guidance to RAGS: Region IV Bulletins, Ecological Risk Assessment. Waste Management Division, Atlanta, Georgia. Originally published November 1995. Website version last updated November 30, 2001: http://www.epa.gov/region4/waste/ots/ecolbul.htm

United States Environmental Protection Agency (EPA), 2001b. Risk Assessment Guidance for Superfund, Volume I – Human Health Evaluation Manual – (Part E, Supplemental Guidance for Dermal Risk Assessment). Working draft of interim guidance. December 2001.

United States Environmental Protection Agency (EPA), 2001c. Trichloroethylene Health Risk Assessment: Synthesis and Characterization (External Review Draft). U.S. Environmental Protection Agency, Office of Research and Development, National Center for Environmental Assessment, Washington Office, Washington, DC, EPA/600/P-01/002A. August 2001.

United States Environmental Protection Agency (EPA), 2002a. Region IX Preliminary Remedial Goals Table. October 2002.

United States Environmental Protection Agency (EPA), 2002b. National Recommended Water Quality Criteria: 2002. EPA-822-R-02-047, Office of Water, Office of Science and Technology, Washington, D.C. November 2002.

United States Environmental Protection Agency (EPA) 2004a. Integrated Risk Information System, On-Line Data Base. February 2004. www.epa.gov/iris.html

United States Environmental Protection Agency (EPA), 2004b. Ecotox On-Line Data Base www.epa.gov/ecotox.

United States Environmental Protection Agency (EPA), Region III, 2004c. Risk-Based Concentration Table. Hazardous Waste Management Division, Philadelphia, Pennsylvania. October 1, 2000.

United States Geological Survey (USGS), 1984a. Milford Quadrangle, Connecticut. 7.5 Minute Series (topographic).

United States Geological Survey (USGS), 1984b. Bridgeport Quadrangle, Connecticut – Fairfield County. 7.5 Minute Series (topographic).

United States Geological Survey (USGS). 1997. Summary of Published Aquatic Toxicity Information and Water Quality Criteria for Selected Volatile Organic Compounds. Open-File Report 97-563, Rapid City, South Dakota.

Van den Berg, M., et. al. 1998. Toxic Equivalency Factor for PCBs, PCDDs, PCDFs for Human and Wildlife. *Environmental Health Perspective*, vol. 106, no. 12. December 1998.

William, G.M. and J.H. Weisburger, 1991. Chemical Carcinogenesis. In Cassarett and Doull's Toxicology: The Basic Science of Poisons, Fourth Edition, M.O. Amdur, J. Doull, and C.D. Klaassen, Eds., Pergamon Press, New York, New York.

Wilson, W.E., Burke, E.L., and Thomas, C.E. Jr., 1974. Water Resources Inventory of Connecticut, Part 5, Lower Housatonic River Basin. Connecticut Water Resources Bulletin No.19 (Prepared by the U.S. Geological Survey), 79 p.

Xu, M. and Y. Eckstein. 1995. Use of weighted least squares method in evaluation of the relationship between dispersivity and field scale. *Groundwater*, v. 33, no. 6, p. 905-908.

Zhang, Y-K and R. C. Heathcote. 2003. An improved method for estimation of biodegradation rate with field data. *Groundwater Monitoring and Remediation* 23: 112-116.