SUPPLEMENTAL PHASE II REPORT

WILMINGTON, MASSACHUSETTS SITE
OLIN CORPORATION

MADEP RTN: 3-0471

JUNE 1997

PREPARED BY

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Massachusetts Department of Environmental Protection **BWSC-108**
Bureau of Waste Site Cleanup

COMPREHENSIVE RESPONSE ACTION TRANSMITTAL FORM & PHASE I COMPLETION STATEMENT

Pursuant to 310 CMR 40.0484 (Subpart D) and 40.0800 (Subpart H)

Release Tracking Number: 3 - 0471

A. SITE LOCATION:

Site Name: (optional) Wilmington - Olin Corporation

Street: 51 Eames Street Location Aid:

City/Town: Wilmington, MA ZIP Code: 01887

Related Release Tracking Numbers that this Form Addresses:

Tier Classification: (check one of the following) ☒ Tier IA ☐ Tier IB ☐ Tier IC ☐ Tier II ☐ Not Tier Classified

If a Tier I Permit has been issued, state the Permit Number: 83004

B. THIS FORM IS BEING USED TO: (check all that apply)

☐ Submit a **Phase I Completion Statement**, pursuant to 310 CMR 40.0484 (complete Sections A, B, C, G, H, I and J).

☐ Submit a **Phase II Scope of Work**, pursuant to 310 CMR 40.0834 (complete Sections A, B, C, G, H, I and J).

☒ Submit a final **Phase II Comprehensive Site Report and Completion Statement**, pursuant to 310 CMR 40.0836 (complete Sections A, B, C, D, G, H, I and J).

☐ Submit a **Phase III Remedial Action Plan and Completion Statement**, pursuant to 310 CMR 40.0862 (complete Sections A, B, C, G, H, I and J).

☐ Submit a **Phase IV Remedy Implementation Plan**, pursuant to 310 CMR 40.0874 (complete Sections A, B, C, G, H, I and J).

☐ Submit an **As-Built Construction Report**, pursuant to 310 CMR 40.0875 (complete Sections A, B, C, G, H, I and J).

☐ Submit a **Phase IV Final Inspection Report and Completion Statement**, pursuant to 310 CMR 40.0878 and 40.0879 (complete Sections A, B, C, E, G, H, I and J).

☐ Submit a periodic **Phase V Inspection & Monitoring Report**, pursuant to 310 CMR 40.0892 (complete Sections A, B, C, G, H, I and J).

☐ Submit a final **Phase V Inspection & Monitoring Report and Completion Statement**, pursuant to 310 CMR 40.0893 (complete Sections A, B, C, F, G, H, I and J).

You must attach all supporting documentation required for each use of form indicated, including copies of any Legal Notices and Notices to Public Officials required by 310 CMR 40.1400.

C. RESPONSE ACTIONS:

☐ Check here if any response action(s) that serves as the basis for the Phase submittal(s) involves the use of Innovative Technologies. (DEP is interested in using this information to create an Innovative Technologies Clearinghouse.)

Describe Technologies:

D. PHASE II COMPLETION STATEMENT:

Specify the outcome of the Phase II Comprehensive Site Assessment:

☒ Additional Comprehensive Response Actions are necessary at this Site, based on the results of the Phase II Comprehensive Site Assessment.

☐ The requirements of a Class A Response Action Outcome have been met and a completed Response Action Outcome Statement (BWSC-104) will be submitted to DEP.

☐ The requirements of a Class B Response Action Outcome have been met and a completed Response Action Outcome Statement (BWSC-104) will be submitted to DEP.

☐ Rescoring of this Site using the Numerical Ranking System is necessary, based on the results of the final Phase II Report.

E. PHASE IV COMPLETION STATEMENT:

Specify the outcome of Phase IV activities:

☐ Phase V operation, maintenance or monitoring of the Comprehensive Response Action is necessary to achieve a Response Action Outcome. (This site will be subject to a Phase V Operation, Maintenance and Monitoring Annual Compliance Fee.)

☐ The requirements of a Class A Response Action Outcome have been met. No additional operation, maintenance or monitoring is necessary to ensure the integrity of the Response Action Outcome. A completed Response Action Outcome Statement (BWSC-104) will be submitted to DEP.

☐ The requirements of a Class C Response Action Outcome have been met. No additional operation, maintenance or monitoring is necessary to ensure the integrity of the Response Action Outcome. A completed Response Action Outcome Statement (BWSC-104) will be submitted to DEP.

SECTION E IS CONTINUED ON THE NEXT PAGE

Massachusetts Department of Environmental Protection **BWSC-108**

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E. PHASE IV COMPLETION STATEMENT: (continued)

☐ The requirements of a Class C Response Action Outcome have been met. Further operation, maintenance or monitoring of the remedial action is necessary to ensure that conditions are maintained and that further progress is made toward a Permanent Solution. A completed Response Action Outcome Statement (BWSC-104) will be submitted to DEP.

Indicate whether the operation and maintenance will be Active or Passive. (Active Operation and Maintenance is defined at 310 CMR 40.0006.):

○ Active Operation and Maintenance ○ Passive Operation and Maintenance

(Active Operation and Maintenance makes the Site subject to a Post-RAO Class C Active Operation and Maintenance Annual Compliance Fee.)

F. PHASE V COMPLETION STATEMENT:

Specify the outcome of Phase V activities:

☐ The requirements of a Class A Response Action Outcome have been met and a completed Response Action Outcome Statement (BWSC-104) will be submitted to DEP.

☐ The requirements of a Class C Response Action Outcome have been met. No additional operation, maintenance or monitoring is necessary to ensure the integrity of the Response Action Outcome. A completed Response Action Outcome Statement (BWSC-104) will be submitted to DEP.

☐ The requirements of a Class C Response Action Outcome have been met. Further operation, maintenance or monitoring of the remedial action is necessary to ensure that conditions are maintained and that further progress is made toward a Permanent Solution. A completed Response Action Outcome Statement (BWSC-104) will be submitted to DEP.

Indicate whether the operation and maintenance will be Active or Passive. (Active Operation and Maintenance is defined at 310 CMR 40.0006.):

○ Active Operation and Maintenance ○ Passive Operation and Maintenance

(Active Operation and Maintenance makes the Site subject to a Post-RAO Class C Active Operation and Maintenance Annual Compliance Fee.)

G. LSP OPINION:

I attest under the pains and penalties of perjury that I have personally examined and am familiar with the information contained in this transmittal form, including any and all documents accompanying this submittal. In my professional opinion and judgment based upon application of (i) the standard of care in 309 CMR 4.02(1), (ii) the applicable provisions of 309 CMR 4.02(2) and (3), and (iii) the provisions of 309 CMR 4.03(5), to the best of my knowledge, information and belief,

> ***if Section B indicates that a Phase I, Phase II, Phase III, Phase IV or Phase V Completion Statement is being submitted***, the response action(s) that is (are) the subject of this submittal (i) has (have) been developed and implemented in accordance with the applicable provisions of M.G.L. c. 21E and 310 CMR 40.0000, (ii) is (are) appropriate and reasonable to accomplish the purposes of such response action(s) as set forth in the applicable provisions of M.G.L. c. 21E and 310 CMR 40.0000, and (iii) complies(y) with the identified provisions of all orders, permits, and approvals identified in this submittal;

> ***if Section B indicates that a Phase II Scope of Work or a Phase IV Remedy Implementation Plan is being submitted***, the response action(s) that is (are) the subject of this submittal (i) has (have) been developed in accordance with the applicable provisions of M.G.L. c. 21E and 310 CMR 40.0000, (ii) is (are) appropriate and reasonable to accomplish the purposes of such response action(s) as set forth in the applicable provisions of M.G.L. c. 21E and 310 CMR 40.0000, and (iii) complies(y) with the identified provisions of all orders, permits, and approvals identified in this submittal;

> ***if Section B indicates that an As-Built Construction Report or a Phase V Inspection and Monitoring Report is being submitted***, the response action(s) that is (are) the subject of this submittal (i) is (are) being implemented in accordance with the applicable provisions of M.G.L. c. 21E and 310 CMR 40.0000, (ii) is (are) appropriate and reasonable to accomplish the purposes of such response action(s) as set forth in the applicable provisions of M.G.L. c. 21E and 310 CMR 40.0000, and (iii) complies(y) with the identified provisions of all orders, permits, and approvals identified in this submittal.

I am aware that significant penalties may result, including, but not limited to, possible fines and imprisonment, if I submit information which I know to be false, inaccurate or materially incomplete.

☒ Check here if the Response Action(s) on which this opinion is based, if any, are (were) subject to any order(s), permit(s) and/or approval(s) issued by DEP or EPA. If the box is checked, you MUST attach a statement identifying the applicable provisions thereof.

LSP Name: Jeffrey D. Bradshaw LSP #: 5086 Stamp:

Telephone: (412) 829-6000 Ext.: 625

FAX: (optional)

Signature:

Date:

COMMONWEALTH OF MASSACHUSETTS
JEFFREY D. BRADSHAW
No. 5086
REGISTERED LICENSED PROFESSIONAL

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Release Tracking Number: 3 - 0471

H. PERSON UNDERTAKING RESPONSE ACTION(S):

Name of Organization: Olin Corporation

Name of Contact: Steve Morrow Title: Principal Environmental Affairs Specialist

Street: 1186 Lower River Road

City/Town: Charleston State: TN ZIP Code: 37310

Telephone: (423) 336-4511 Ext.: FAX: (optional)

☐ Check here if there has been a change in the person undertaking the Response Action.

I. RELATIONSHIP TO SITE OF PERSON UNDERTAKING RESPONSE ACTION(S): (check one)

☒ RP or PRP Specify: ⊗ Owner ○ Operator ○ Generator ○ Transporter Other RP or PRP:

☐ Fiduciary, Secured Lender or Municipality with Exempt Status (as defined by M.G.L. c. 21E, s. 2)

☐ Agency or Public Utility on a Right of Way (as defined by M.G.L. c. 21E, s. 5(j))

☐ Any Other Person Undertaking Response Action Specify Relationship:

J. CERTIFICATION OF PERSON UNDERTAKING RESPONSE ACTION(S):

I, Steve Morrow, attest under the pains and penalties of perjury (i) that I have personally examined and am familiar with the information contained in this submittal, including any and all documents accompanying this transmittal form, (ii) that, based on my inquiry of those individuals immediately responsible for obtaining the information, the material information contained in this submittal is, to the best of my knowledge and belief, true, accurate and complete, and (iii) that I am fully authorized to make this attestation on behalf of the entity legally responsible for this submittal. I/the person or entity on whose behalf this submittal is made am/is aware that there are significant penalties, including, but not limited to, possible fines and imprisonment, for willfully submitting false, inaccurate, or incomplete information.

By: Steve Morrow Title: Principal Environmental Affairs Specialist
(signature)

For: Date:
(print name of person or entity recorded in Section H)

Enter address of the person providing certification, if different from address recorded in Section H:

Street:

City/Town: State: ZIP Code:

Telephone: Ext.: FAX: (optional)

YOU MUST COMPLETE ALL RELEVANT SECTIONS OF THIS FORM OR DEP MAY RETURN THE DOCUMENT AS INCOMPLETE. IF YOU SUBMIT AN INCOMPLETE FORM, YOU MAY BE PENALIZED FOR MISSING A REQUIRED DEADLINE.

EXECUTIVE SUMMARY

INTRODUCTION

Olin Corporation has recently completed an extensive Supplemental Phase II Field Investigation at property located at 51 Eames Street, Wilmington, Massachusetts. This investigation is part of a series of activities that Olin has undertaken since its acquisition of the property in 1980 to identify, understand, and control soil and groundwater contamination that was caused by the property's former owners and operators.

This investigation and the Supplemental Phase II Report and Risk Assessment comply with the Massachusetts Contingency Plan (MCP) (310 CMR 40.000 et. seq.) and the National Contingency Plan, 40 CFR Part 300. Work plans describing the scope of this investigation were approved by the Massachusetts Department of Environmental Protection (MADEP). Under the MCP, Phase II activities are designed to evaluate sufficient information concerning the source, nature, extent and potential impacts of releases of oil and hazardous materials; evaluate the risk of harm to human health, safety, public welfare and the environment; and to determine whether additional comprehensive response actions are needed. This Supplemental Phase II Report summarizes the field activities and analysis that Olin has performed since the completion of its 1993 Phase II Comprehensive Site Assessment (CSA) and presents the results of the supplemental investigation. These activities build upon prior investigations and reports conducted by Olin, including the CSA. Completion of the Supplemental Phase II Investigation involved, among other things, collection of thousands of soil and groundwater samples, bedrock mapping, seismic reflection surveying, fate and transport analysis, installation of 23 new groundwater wells, and geochemical modeling. In addition to the results of the Supplemental Phase II Investigation, this Report presents the results of both human health and ecological risk assessments.

As part of this effort, Olin has communicated with property owners in the affected area about the nature and extent of the contamination, and has implemented a groundwater information program and, where appropriate, a groundwater well closure program to eliminate the risk that could be posed to area residents and workers from inappropriate use of this groundwater resource.

BACKGROUND

The property is located in a heavily industrial section of South Wilmington, Massachusetts. From 1953 until approximately 1980, when Olin purchased the facility, the property was operated by several companies for the manufacture of specialty chemicals for the rubber and plastics industry. The former chemical manufacturing buildings were demolished and removed after Olin ceased operations in 1986. The 53-acre property is largely undeveloped and currently contains an office building, warehouses, a groundwater recovery and treatment, and a closed landfill. The undeveloped portion of the property consists largely of wetlands and a mature forest. Although

the property is generally unused (with the exception of occasional meetings), Olin has a small number of employees on-site for maintenance purposes and to operate the groundwater recovery and treatment system. The property is surrounded by an 8-foot high perimeter fence which is secured when the property is unattended.

The environmental contamination at the facility is the result of several different activities that were conducted at the facility prior to Olin's ownership. A primary source of the contamination was the on-property wastewater disposal activities. Prior to 1970, all liquid wastes resulting from chemical manufacturing operations at the facility were not treated and were discharged into various man-made unlined pits on the property. During the 1970s, although on-property disposal of wastes continued, treatment of the facility's wastewater was introduced and much of it was discharged to a public sewer system after treatment. The remainder was discharged to lined settling ponds after treatment.

The results of investigations show that these liquid wastes traveled down through the soils at the property into the groundwater and have moved through the groundwater beyond the property's borders. Some of these contaminants formed a solution which was heavier than water and which has sunk to the bedrock beneath the aquifer. These contaminants now reside in a dense layer at the base of a portion of the aquifer (the dense layer). Areas of contamination also remain at the property, including contamination in an on- and off-site surface water ditch system, in and around the former chemical production area and tank farm, the location of the former disposal pits, and a calcium-sulfate landfill for treated sludges from wastewater treatment activities. The main contaminants include volatile organic compounds, semivolatile organic compounds, and inorganic compounds, including ammonia, sulfate, chloride, and chromium. The most prevalent contaminants are inorganic compounds.

The sand and gravel aquifer is utilized for some domestic water supplies in this area. The Town of Wilmington operates a municipal water supply wellfield over one-half mile west of the Olin property and on the west side of the Maple Meadow Brook wetland. The wellfield consists of five wells which pump an average of 1.4 million gallons of water per day (gpd). A major focus of the Supplemental Phase II Investigation was to assess the potential for an impact by the dense layer on the municipal water supply. An industrial facility 500 feet west of the Olin property, Altron Corporation, also utilizes groundwater for industrial purposes. In addition, the bedrock is used for water supply by certain individual residences southwest of the property. The bedrock is a low yield source, and the investigation has shown that there is not a hydrogeologic connection between these wells and the groundwater impacted by site contaminants.

Olin closed the facility in 1986. Olin has already acted to protect the public and on-property workers from potential risks in several areas. Olin has installed extraction wells to contain oil and contaminated groundwater in one manufacturing area and has completed an Immediate Response Action with respect to drums buried on the property. These activities are described in reports filed with MADEP.

MAJOR FINDINGS AND CONCLUSIONS OF THE SUPPLEMENTAL PHASE II INVESTIGATION

The Supplemental Phase II Investigation has confirmed most of the findings of the 1993 CSA. The major findings and conclusions concerning the nature and extent of the contamination of the Supplemental Investigation are:

- Inorganic compounds, including ammonia, chloride, and chromium, are the most prevalent class of contaminants associated with the site, and are detected at elevated concentrations (relative to background) in soil, sediment and surface, and groundwater on-property and in sediment and groundwater off-property;

- The primary sources of these inorganic compounds are the former unlined pits. Chromium is also present in a fine-grain precipitate found in the On-Property West and South ditches;

- The dense layer, under the influence of gravity, has come to rest in bedrock valleys beneath the aquifer to the west and, to a lesser extent, to the east of the property. Based on groundwater modeling and direct measurement, the dense layer appears to be moving slowly, if at all, and is restricted to the topographic lows of the bedrock surface, which lie 30 to 120 feet below the ground surface;

- Geochemical modeling and direct measurements of the dense layer and overlying groundwater has shown that contaminants migrate very slowly from the dense layer. Therefore, although the dense layer will continue to be a source of contamination to the surrounding groundwater, it is unlikely that the resultant concentrations will be significantly greater than currently observed; and

- Results of the first 10 months of a monitoring program (through April 1997) indicate that concentrations of certain inorganic compounds detected at the public supply wells that are also contaminants of concern at the site are within the range of concentrations historically detected in samples from these wells.

The Risk Assessment conducted by ABB-Environmental Services has, in accordance with the MCP, evaluated the risk of harm to health, safety, public welfare and the environment posed by the conditions at the site, focusing on the types of contaminants and exposures that are reasonably likely, both under current and future conditions. The Risk Assessment is divided into two documents, addressing human and ecological risks separately. The Risk Assessment process is used to establish whether a level of "No Significant Risk" exists at the disposal site. The Risk Assessment concludes the following:

Human Health

On-Property

- No significant risks are posed to persons currently working at the Olin property.

- No significant risks will be posed to persons working at the Olin property in the future or to the occasional trespasser. A land use restriction will be imposed so that persons conducting any subsurface work at the property, such as utility personnel, do not excavate in impacted areas without taking adequate protections. The land use restriction will also prohibit certain uses of the property, such as residential use, unless further actions are performed to address the contamination.

Off-Property

- No significant risk is posed to an industrial worker at Altron Corporation currently exposed to groundwater through Altron's use of groundwater in its industrial process.

- No significant risk is posed to the neighborhood resident who could currently be exposed to surface water and sediment in the off-property ditches and who currently drinks water from the Maple Meadow Brook municipal wells.

- No significant risk is posed to residents of the Town of Wilmington who obtain their drinking water from the Maple Meadow Brook municipal wells, under current conditions.

- No significant risk will be posed to residents of the Town of Wilmington who obtain their drinking water from the Maple Meadow Brook municipal wells in the future, although modifications to the Town treatment system may be required to adjust for increases in inorganic concentrations. Olin has already implemented an extensive 'Sentinel" groundwater monitoring program which is designed to provide forewarning of potential changes in water quality that would require modifications at the treatment plant or wellfield.

- Because there are portions of the aquifer where contaminant levels exceed Massachusetts drinking water of other standards, under the MCP, it cannot be concluded that no significant risk exists.

Ecological

- No significant risk is posed to terrestrial life both on- and off-property, based upon an evaluation of representative species, the American woodcock, the red fox, and the green heron.

- A condition of no significant risk does not exist for aquatic wildlife both on- and off-property, based upon an evaluation of the impact on green frog reproduction of exposure to sediment and surface water within a portion of the Ditch System.

- Because levels of contaminants in surface water currently exceed Massachusetts ambient water quality standards, it cannot be concluded that no significant risk exists.

CONCLUSION

Under the MCP, Phase II activities, including a Risk Assessment, are intended to enable the party conducting response actions to determine whether additional response actions will be required. Based upon the findings of the Supplemental Investigation and the Risk Assessment that a condition of No Significant Risk has not been achieved for all current and future exposure scenarios, this Report concludes that additional comprehensive response actions will be required at the property. The MCP requires that the identification and evaluation of remedial alternatives be performed in a Phase III evaluation.

ACKNOWLEDGMENTS

We would like to express our sincere thanks to the Wilmington Conservation Commission, Wilmington residents, local industry, and the Town of Wilmington Water Department for their cooperation and understanding in the completion of this large project. Many have given us unrestricted access to their property, including permission to install and sample wells and borings on private land, and have been open with all available sources of information, especially Mike Woods, Rob Antiko, and others that work for the Water Department. The above mentioned groups have openly answered all of our questions and have provided gracious assistance in fulfilling our needs, without which, completion of this investigation would not have been possible.

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ACRONYM LIST

ABB-ES	ABB Environmental Services, Inc.
AQTESOLV	Aquifer Test Solver
ARA	Applied Research Associates, Inc.
ASAS	Applicable or Suitably Analogous Standards
AST	above ground storage tank
ATC	allowable threshold concentrations
ATSDR	Agency for Toxic Substances and Diseases Registry
AUL	Activity and Use Limitations
BEI	backscatter electron image
bgs	below ground surface
C	Celsius
CAS	Columbia Analytical Services
cc/min	cubic centimeters per minute
cfs	cubic feet per second
cm/sec	centimeters per second
cm^2 /sec	square centimeters per second
cm^3 /sec	cubic centimeters per second
CMR	Code of Massachusetts Regulations
CPC	Chemical of Potential Concern
CPDA	Central Pond drainage area
CPS	counts per second
CPT	cone penetrometer testing
CRA	Conestoga-Rover & Associates
CSA	Comprehensive Site Assessment
CSFs	Cancer Slope Factors
CUR	Cancer Unit Risk
DH	Dufresne-Henry, Inc.
DO	dissolved oxygen
DPS	Downgradient Property Status
EDS	energy dispersive detection
EMF	electromotive force
EMPA	electron microprobe analysis
EPC	Exposure Point Concentrations

ACRONYM LIST (continued)

EPH/VPH	Extractable Petroleum Hydrocarbons/Volatile Petroleum Hydrocarbons
ERC	environmental risk characterization
FEMA	Federal Emergency Management Agency
FETAX	Frog Embryo Teratogenesis Assay-Xenopus
FIRM	Flood Insurance Rate Map
FIT	Field Investigation Team
feet/day	feet per day
ft	feet
ft^2 /day	square feet per day
g/cm^3	grams per cubic centimeter
g/ml	grams per milliliter
gpd	gallons per day
gpm	gallons per minute
HEAST	Health Effects Assessment Summary Tables
HI	Hazard Index
HRS	Hazard Ranking System
ID	inside diameter
IRA	Immediate Response Action
IRIS	Integrated Risk Information System
Lake Poly	Lake Poly Liquid Waste Disposal Area
l/kg	liters per kilogram
LSP	Licensed State Professionals
M	molar
m	meter
MA	Massachusetts
MADEP	Massachusetts Department of Environmental Protection formerly known as MADDEN - Massachusetts Department of Environmental Quality Engineering
MCL	maximum contaminant limit
MCP	Massachusetts Contingency Plan
MGD	million gallons per day
mg/kg	milligrams per kilogram

mg/l	milligrams per liter
MMCLs	Massachusetts maximum contaminant levels
mmho/m	millimhos per meter
MRL	minimal risk levels
MSL	mean sea level
mV	millivolts
NAPL	non-aqueous phase liquid
NCEA	National Center for Environmental Assessment
NCP	National Contingency Plan
NFIP	National Flood Insurance Program
NNDPA	n-nitrosodiphenylamine
NPDES	National Pollutant Discharge Elimination System
NPI	National Polychemicals, Inc.
OHM	oil and hazardous materials
OHMPC	oil and hazardous materials of potential concern
ORP	oxidation-reduction potential
PAH	polynuclear aromatic hydrocarbons or polycyclic aromatic hydrocarbons
POTW	publicly owned treatment works
ppb	parts per billion
ppm	parts per million
PVC	polyvinyl chloride
PWS	Public Water Supply
RAP	Remedial Action Plan
Raypath	Raypath, Inc.
RAF	Relative Absorption Factor
RfCs	Reference Concentrations
RfDs	Reference Doses
RI	Remedial Investigation
SC	specific conductance
Smith	Smith Technology Corporation, formerly Smith Environmental Technologies Corporation and BCM Engineers, Inc.

ACRONYM LIST (continued)

SMCL	Secondary Maximum Contaminant Levels
SP	Spontaneous Potential
SWMU	Solid Waste Management Unit
SVOC	semivolatile organic compound
TAL	Target Analyte List
TCE	trichloroethene
TCL	Target Compound List
TDS	total dissolved solids
TELs	threshold effects exposure limits
TIE	Toxic Identification Evaluation
TMP	trimethylpentene
TOC	total organic carbon
TVOCs	total volatile organic compounds
UCL	Upper Concentration Limit
ug/l	micrograms per liter
ug/kg	micrograms per kilogram
um	micrometers
umhos	umhos/cm, umhos, us, us/cm
UR	Unit Risk
USEPA	United States Environmental Protection Agency
USGS	United States Geological Survey
UST	underground storage tank
UTM	Universal Transverse Mercator
VOC	volatile organic compound
WCC	Wilmington Conservation Commission
WDS	wavelength dispersion detection
W&H	Whitman and Howard
WPI	Wetlands Preservation, Inc.
XRD	x-ray diffraction
XRF	x-ray fluorescence

LABORATORY AND DATA VALIDATION QUALIFIERS

Laboratory Qualifiers

General

NR Not required.
U Indicates compound was analyzed for but not detected.
X Other footnotes, see data package.

Organic

A TIC is a suspected aldol-condensation product.
B Analyte is found in the associated blank.
E Concentrations exceed the calibration range of GC/MS.
J Indicates an estimated value below the quantitation limit.

Inorganic

B Value less than contract required detection limit but above instrument detection limit.
E Estimated value due to interference.
N Spike sample recovery not within control limit.
S Value reported was determined by the method of standard additions.
+ Correlation coefficient of MSA < 0.995.
* Duplicate analyses are not within control limits.

Data Validation Qualifiers

b Analyte present at > IDL but < 5 times the amount in associated blank (inorganic analysis only).
j Indicates an estimated value.
n Indicates presumptive evidence of a compound (TICs only).
u Indicates compound was analyzed for but not detected OR for organics the compound was present in an associated blank at a similar concentration and is not considered present in the sample.
uj Indicates an estimated detection limit.
r Value is rejected and is not usable.
x See data package for other footnotes.
[ ] Indicates that the reported result is above instrument detection limit but below the CRDL.

General Qualifiers

nv Data scheduled for validation, but validation not complete

Historic Qualifiers

$ and # Qualifiers listed with historical data (data collected before the CSA), but a description of the was not included with the historic database packages.

Additional Notes

Field data recorded during groundwater sampling were measured from the unfiltered groundwater sample.
Comparison of field data measurements taken from unfiltered groundwater and filtered groundwater show little difference between the numbers.

1.0 INTRODUCTION

MCP 40.0835(4)(a),(b),(c)

This report presents a comprehensive description of environmental conditions at the Olin Corporation (Olin) property in Wilmington, Massachusetts. The results of the Supplemental Phase II Investigations conducted in 1994 through 1997, and preliminary results of ongoing investigations in the Western Bedrock Valley are also presented. The scope of work for most of these activities was described in work plans submitted to the Massachusetts Department of Environmental Protection (MADEP) in August 1993 (BCM, 1993) and June 1996 (Smith, 1996d) and approved by the MADEP in September 1994 and July 1996, respectively. Additional minor work plans have been submitted to and approved by the MADEP throughout 1995 and 1996.

The work on this site is being performed under the Massachusetts Contingency Plan (MCP). In July 1989, the MADEP issued a Waiver of Department Approvals in which the site was considered non-priority. In July 1993, the Waiver was revoked and the site was reclassified as a Tier I site under the current MCP.

1.1 PROPERTY LOCATION AND DESCRIPTION

The Olin property, a 53-acre site associated with a former chemical manufacturing facility, is located at 51 Eames Street in Wilmington, Massachusetts. The property location and surroundings are shown on the United States Geologic Survey (USGS) Topographic Map on Figure 1-1. The property is bounded on the east by the Boston and Maine railroad tracks, on the south by the Woburn/Wilmington town line, on the west by an inactive Boston and Maine railroad spur, and on the north by Eames Street. The approximate Universal Transverse Mercator (UTM) coordinates for the center of the property are 4,710,570 northing and 323,070 easting.

The property is located in an industrial area of the town. The property and most of the surrounding area is zoned as General Industrial. The property is surrounded by extensive industrial land uses to the north and east, and a former municipal landfill for the City of Woburn lies directly south. Heavy and light industrial facilities are also located immediately west of the property. The closest residential land use areas lie further west along Main Street and Cook Avenue. West of the single-family dwellings on Main Street is an extensive wetland system associated with Maple Meadow Brook. The Town of Wilmington has developed a municipal well field consisting of five production wells on the western side of the wetland system over one-half mile west of the Olin property. These supply wells are known as the Town Park, Butters Row No. 1 and No. 2, and Chestnut Street No. 1 and No. 2.

Plate 1-1 is a large scale map detailing the features and topography of the property and surrounding area. The property is accessed from Eames Street and this access is restricted by an 8-foot high perimeter chain link fence, and locked gates when the property is unattended. The property topography is generally flat with surface elevations ranging from approximately 110 feet above mean sea level (MSL) at the southern portion of the property to approximately 75 feet MSL in the central portion of the property.

An interconnected surface water drainage system borders two sides and crosses the center of the property as shown on Plate 1-1. These surface water bodies are named the South Ditch, West Ditch, East Ditch, and Ephemeral Drainage, and are collectively referred to as the Ditch System. The Central Pond lies along the South Ditch in the center of the property and the West Ditch is divided into the Off-Property West Ditch and the On-Property West Ditch.

Existing structures, which are clustered in the northern portion of the property, include a largely unused office building and laboratory, the East and West Warehouses, a guard shack, a small storage building known as the Butler Building, and a building housing the Plant B groundwater treatment system. The Plant B groundwater treatment facility consists of three interceptor wells and water treatment system. The treated water is discharged through the National Pollutant Discharge Elimination System (NPDES)-permitted outfall into the portion of the West Ditch located on the Olin property (Plate 1-1).

The property is not currently used for active manufacturing operations and many of the previous buildings and tanks have been dismantled and removed. The former manufacturing processes, waste disposition, and previous investigations are discussed in detail in the Comprehensive Site Assessment (CRA, 1993) and are summarized below. The existing or former site features which are potential environmental concerns are described in Section 1.3.

1.2 BACKGROUND

1.2.1 Property Ownership and History

The detailed ownership and operational history of the Olin property are discussed in the June 1993 Comprehensive Site Assessment (CSA) Phase II Field Investigation Report by Conestoga-Rover & Associates (CRA). The manufacturing facility, initially consisting of a laboratory and one chemical manufacturing plant, was constructed in 1953. From 1953 until approximately 1971, the facility expanded and was operated under the name of National Polychemicals, Inc. (NPI). Between 1953 and 1968, NPI was owned or operated by three different corporations, American Biltrite Rubber Co., Fisons Limited, and Fisons Corporation. In 1968, Stepan Chemical Corporation purchased NPI, and in 1971, NPI was merged into Stepan Chemical Corporation. Stepan continued to own and operate the facility from 1971 until 1980, when the property was purchased by Olin Corporation. Olin submitted closure plans for plant facilities to the MADEP and United States Environmental Protection Agency (USEPA) in April 1986. Olin closed manufacturing facilities at the site in 1986, but maintains its ownership to the present.

The facility historically manufactured chemical blowing agents, stabilizers, antioxidants and other specialty chemicals for the rubber and plastics industry. The raw materials used at the facility included sodium dichromate from around 1957 until 1967. The manufacturing processes conducted at the facility generated liquid wastes including sulfuric acid, sodium chloride, sodium sulfate, ammonium sulfate, chromium sulfate, and other constituents, including oils and other organic compounds. The environmental contamination at the facility is the result of several different activities that were conducted at the facility prior to Olin's ownership. Some of the major sources of the contamination were the on-property wastewater disposal activities.

Prior to 1970, all liquid wastes were discharged into various unlined pits and ponds in the central portion of the property and into an unlined man-made excavation called Lake Poly Liquid Waste Disposal Area (Lake Poly). Some wastewater sank into the porous soil and some overflowed into the ditch system located on the property. Initial plans from the 1950s labeled Lake Poly as a "filter bed". This name and the design of Lake Poly to include a porous sand bottom reveal that the intent was to discharge wastewater directly into the soil. The results of environmental investigations confirm that wastes were discharged to these unlined pits and that liquid wastes with high concentrations of dissolved inorganics formed a dense layer which sank through the aquifer and pooled on top of the bedrock.

In 1970, Stepan constructed an acid neutralization and treatment system, and installed new Lined Lagoons to replace the unlined pits and ponds. The treated wastewater was discharged to the on-property ditch system until the inter-municipal sewer system was completed in 1972. The remaining waste material, consisting primarily of calcium sulfate, was put into the lined settling lagoons. Periodically, these lagoons were dredged and the calcium sulfate was placed in a landfill on the southwest corner of the property, now known as the Calcium-Sulfate Landfill. Use of the Calcium-Sulfate Landfill was approved by the Commonwealth of Massachusetts in the mid-1970s. After its purchase of the property, Olin re-lined the lagoons 1981 through 1983. The lagoons and the Calcium-Sulfate Landfill were closed after 1986 when operations at the facility ceased.

The discharge into unlined pits, lagoons, and other industrial disposal areas at the property, beginning in the 1950s, caused subsurface contamination both on and off the Olin property. The most aerially extensive subsurface contamination is associated with a dense layer of inorganic contaminants dissolved in the groundwater that is present at the base of the sand and gravel aquifer. This layer is denser-than-water, occupies the bedrock lows, and is chemically and physically distinct from the overlying much less contaminated groundwater. This layer is characterized by high concentrations of sulfate, chloride, ammonia, chromium, and other inorganics. Much of the recent investigation has been focused on obtaining additional information about this dense layer.

Contamination may have also resulted from several other on-site activities. During its efforts to investigate the contamination, Olin has found evidence that drums containing organic chemicals were buried on-property prior to Olin purchasing the property. There are also reports that spills or leaks of chemicals occurred at the property through the years prior to Olin's ownership of the property.

1.2.2 History of Environmental Investigations

The property was originally listed by MADEP in 1987 as a result of the Massachusetts Field Investigation Team (FIT) Phase I Site Inspection Report and subsequent Hazard Ranking System (HRS). The property was then listed as a Non-Priority site by the MADEP under a MCP Waiver of Approvals which was issued in July 1989. CRA was retained by Olin to perform a CSA at the site to assess areas of potential environmental concern. The results of the CSA are presented in a report submitted to the MADEP in June 1993 (CRA, 1993). A risk assessment was prepared for the property by ABB Environmental Services, Inc. (ABB-ES) and submitted to MADEP in 1993 (ABB-ES, 1993a and 1993b). In July 1993, based on the data and results presented in the CSA, the MADEP revoked the Waiver of Approvals and reclassified the property as a Priority site. Under the current MCP, the site was issued a permit as a Tier 1 site on September 20, 1994.

BCM Engineers Inc., which is now a subsidiary of Smith Technology Corporation (Smith), was retained by Olin to prepare the Remedial Action Plan (RAP) for the property based on the results of the CSA and risk assessment. Due to the need for additional data to evaluate remedial alternatives, a work plan for supplemental field investigation tasks was prepared by Smith and submitted to MADEP in August 1993 (BCM, 1993). Conditional approval of the Supplemental Phase II Field Investigation Work Plan was received by Olin in July 1994. A number of additional work plans have been submitted to and approved by the MADEP throughout the Supplemental Phase II process. A detailed chronology of submittals and MADEP approvals or comments since submittal of the CSA in June 1993 is presented in Table 1-1.

1.3 ENVIRONMENTAL CONCERNS

Historical information about facility operations has identified the following areas of potential environmental concern at the site:

- Former Unlined Pits: Lake Poly, East and West Pits, and Acid Pits;
- The dense layer of chromium and other inorganics at the base of the aquifer;
- The Ditch System and associated surface water drainage;

- Plant B Production Area and Tank Farm;
- Drum Disposal Areas; and,
- The Calcium-Sulfate Landfill and Lined Lagoons.

These features (except the dense layer) are shown on the Site Features Map on Plate 1-2. A brief introduction to each of these areas of potential concern is provided below. Other miscellaneous areas, most of which were previously identified as Solid Waste Management Units (SWMUs) in the CSA are not considered to be significant areas of environmental concern. These are generally areas with previous actions or closures including actions by Olin under other regulatory programs, and are described in Section 4.5.

1.3.1 Former Unlined Pits: Lake Poly, East and West Pits, and Acid Pits

As discussed in Section 1.2.1, all liquid wastes at the facility prior to 1970 were discharged into various unlined pits in the central portion of the property: Lake Poly, East and West Pits, and the Acid Pits. These pits were man-made excavations used for liquid waste disposal. The location of the former unlined pits are shown on Plate 1-2. The unlined pits, which were operated prior to Olin purchasing the property, are considered to be the most significant disposal areas. These were the main sources of contaminants to the ditch system and groundwater and are the source of the dense layer.

Lake Poly was an unlined surface impoundment that overflowed into the on-property ditch system and covered about one-fifth of an acre. Lake Poly was located along the western boundary of the facility near the On-Property West Ditch as shown on Plate 1-2. It is believed that Lake Poly was associated with the facility production from the time of plant construction in the early 1950s. A 1969 Pollution Control Study conducted by the Badger Company Inc. for National Polychemicals (Badger, 1969) indicates that the liquid waste stream to Lake Poly included process wastes, cooling tower and boiler blowdown, uncontaminated cooling water, and discharges from yard and floor drains and from the pilot lab. Until 1967, sodium dichromate was used in the Kempore® process and acidic waste containing chromium sulfate was discharged to Lake Poly and the other former acid pits discussed below. Most of these wastes sank through the porous overburden and some overflowed directly into the Ditch System. From about 1969 through 1971, Lake Poly was filled-in and in about 1970 a new wastewater treatment system was constructed. The results of subsurface soil investigations show that Lake Poly soils are impacted with high levels of chromium and other inorganics, and moderate levels of volatile organic compounds (VOCs) and high levels of semivolatile organic compounds (SVOCs).

Prior to 1964, two unlined pits, the East and West Pond, were located in the area now occupied by the two warehouses as shown on Plate 1-2. Interviews with former employees show that these unlined pits received Kempore® process liquid wastes and acidic wastes similar to those discharged to Lake Poly. The location of these unlined pits is based on an October 1960 plan of the facility, and historical aerial photographs from around 1963. These two man-made

excavations covered a combined area of over 15,000 square feet. Building permits and facility plans from 1964 through 1965, suggest that the West Pond was filled in prior to construction of the West Warehouse in 1964 and the East Pond was filled in during construction of the East Warehouse in 1965.

Aerial photographs from 1967 show that a series of three unlined pits, the Acid Pits, existed further south, between the warehouses and the South Ditch. The Acid Pits were three unlined man-made excavations, covering a combined area of over 20,000 square feet. It is believed that these acid pits were constructed in 1964 through 1965 and were replaced by the Lined Lagoons in about 1972. A Wastewater Characterization Study conducted by Marine Research Laboratory (1970) shows that the Acid Pits received liquid acidic wastes including Kempore® process waste which contained chromium sulfate until 1967, when the use of sodium dichromate ceased (Badger, 1969). The Acid Pits were replaced with the Lined Lagoons in approximately 1972. These lagoons received mainly acidic inorganic wastes that did not contain sodium dichromate wastes and were neutralized with lime. These different wastes resulted in a calcium sulfate slurry which was dewatered in the Lined Lagoons and then placed in the Calcium-Sulfate Landfill. The closure of the Lined Lagoons and the Calcium-Sulfate Landfill are discussed in Section 1.3.6.

1.3.2 Dense Layer

The dense layer consists of a denser-than-water solution of mainly sulfate, chloride, ammonia, chromium, sodium, and calcium and occurs in the groundwater at the lowest points within the sand and gravel aquifer, on top of the much less permeable bedrock surface. The dense layer mainly originated from disposal of liquid wastes in the former unlined pits and is generally characterized by specific gravity greater than water, a low pH, a slight to strong green color due to the presence of chromium, at concentrations exceeding 3,000 milligrams per liter (mg/l) at the base of aquifer near the property. The dense layer has moved along the top of the bedrock mainly in the buried bedrock valleys to the west and slightly in bedrock valleys to the east of the Olin property. The historical drilling log in Appendix A from an exploratory well on-property (Test Hole No. 1) in 1964 and a production well at Altron (CRA, 1993) in about 1972 show "green water" was pumped from wells at depths of 32 and 67 feet below ground surface, respectively. The exploratory well on-property was abandoned, and the Altron well was pulled back to a depth of 30 feet bgs and currently serves to provide industrial non-potable water for Altron's use. These observations and other findings presented in this report suggest that this "green water" observed some 25 years ago at the Altron facility was the dense layer and that it had flowed off-property by the early 1970s. The major focus of the Supplemental Phase II Investigation has been to better define the chemical nature and physical constraints of the dense layer.

1.3.3 Ditch System

The Ditch System is an interconnected man-made drainage system for surface water that borders two sides and crosses the center of the property and has a short fenced-in section, off-property, to the west (Plate 1-2). These surface water bodies are named the South Ditch, West Ditch, East Ditch, and Ephemeral Drainage. The West Ditch is subdivided into the On-Property West Ditch,

located within the Olin property, and the Off-Property West Ditch, located off the property to the west. The Central Pond is adjacent to the South Ditch. The South Ditch, West Ditch, and Central Pond are considered to be the major areas of concern due to the presence of elevated inorganics in the Off-Property West Ditch and elevated inorganics and organics in the South Ditch, On-Property West Ditch, and Central Pond.

Overflow from Lake Poly was released into the On-Property West Ditch and downstream to the South Ditch, resulting in the presence of organic and inorganic contamination in these ditches and to a lesser degree in the Ephemeral Drainage. The results of the investigations show that the Lake Poly and the other unlined pits through overflow to ditches and the ground surface also contaminated adjacent low lying areas such as the Central Pond, Central Wetland, and the associated drainage areas. Shallow groundwater in the vicinity of Lake Poly and the other unlined pits would have mainly flowed towards and discharged into the South Ditch and Off-Property West Ditch, but not to the On-Property West Ditch. The chromium-bearing flocculent/sediment present in the South Ditch and On-Property West Ditch is due to both groundwater discharge and historical releases directly into the Ditch System from liquid waste disposal areas.

1.3.4 Plant B Tank Farm and Production Area

The Plant B Area encompasses a former chemical production area and tank farm as shown on Plate 1-2. The former tank farm consisted of six 15,000-gallon storage tanks used for storing raw materials such as bis(2-ethylhexyl)phthalate, n-nitrosodiphenylamine (NNDPA), and "Process Oils". Prior to Olin ownership in 1980, the Plant B Tank Farm sat on grade with no dike or spill containment. As early as 1973, Stepan Chemical Corporation was contacted by the MADEP about a seep of oily material in the East Ditch area adjacent to the Plant B tank farm (CRA, 1993). Analyses performed by Stepan in 1979 confirmed the material contained high concentrations of phthalates. Analysis of the separate-phase oil floating on the water table at interceptor well IW-11 has shown that the oil contains a high percentage of bis(2-ethylhexyl)phthalate and lesser amounts of NNDPA, di-n-octylphthalate, and trimethylpentenes.

Olin installed a pad and curb around the storage tank area in 1981. During construction contaminated soils were encountered and removed from under and around the storage tanks and disposed of off-site in a secure landfill. The results of subsurface soil investigations show that soils are impacted by low levels of VOCs, moderate levels of inorganics, and high levels of SVOCs. In 1982, Olin installed four pumping wells to pump groundwater and stop the oil seep. The groundwater was pumped through carbon and used in the plant as pump seal water. To improve capture of oil, five new wells were drilled closer to the East Ditch fence in 1984 and the old wells were shut down. After the plant was shut down in 1986, the extracted groundwater was treated and trucked to the Greater Lawrence Wastewater Authority Publicly Owned Treatment Works (POTW). Beginning in October 1987, the treated waters have been discharged through a NPDES-permitted outfall. The wells installed in 1984 began to plug by fouling of the screens with iron. In 1988, three large diameter (two 12-inch and one 16-inch diameter) wells were installed to replace the earlier extraction wells. Groundwater pumping and treatment are ongoing.

The treatment system now consists of hypochlorination to remove ammonia and carbon adsorption to remove organics. The water is discharged through the NPDES-permitted outfall into the On-Property West Ditch which eventually flows into the East Ditch.

1.3.5 Buried Drum Areas

Olin submitted an Immediate Response Action (IRA) Completion Statement Report to the MADEP in June 1996 for two buried drum areas on the Olin property. Olin implemented the IRA to reduce risk to on-property works and to evaluate and eliminate any imminent hazardous conditions to the public. One area where drums were disposed of prior to Olin's ownership of the plant was discovered during Olin's repair of Lagoon II in July of 1980. At that time, 11 drums were found and disposed of off Olin's property in a secure facility. The magnetometer surveys in 1990 through 1991 and an extensive test pit program in 1992 through 1993 identified three areas of buried drums and drum debris. These areas are described below and are shown on Plate 1-2.

Drum Area A consists of the area described in the "Test Pit Excavation Summary Report" (CRA, 1992a), where test pits 6, 7, and 8 encountered extensive buried drums and miscellaneous waste. Analysis of waste material samples detected high concentrations of NNDPA and chromium, and moderate concentrations of bis(2-ethylhexl)phthalate and trimethylpentenes.

Drum Area B consists of the area in the vicinity of test pit 21 which contained scattered drums/drum parts, laboratory bottles, and miscellaneous wastes. Low concentrations of VOCs and moderate concentrations of SVOCs (phenols, chlorobenzenes, phthalates, and NNDPA) were detected in the waste material samples from Drum Area B, while much higher concentrations of bis(2-ethylhexl)phthalate were detected in the soil samples.

The Buried Debris Area consists of the area in the vicinity of test pits 18, 19, and 20 which contained mostly soil and rubbish and little drum waste. The analytes in the Buried Debris Area are more typical of Lake Poly and the concentrations are much lower than the other areas. Therefore, the two buried drum areas of concern are Drum Area A and Drum Area B (Plate 1-2).

An Imminent Hazard Evaluation for the drum areas was prepared by ABB-ES in accordance with 310 Code of Massachusetts Regulations (CMR) 40.0950 and submitted to MADEP in November, 1994. The evaluation concluded that the buried drum areas could pose an imminent hazard because the presence of Opex® (dinitrosopenta-methylenetetramine) and Kempore® (azodicarbonamide) may represent a threat of explosion or fire if subjected to heat, flame, or friction. However, it should be noted that such hazardous conditions have not been observed in the greater than twenty years during which these materials have been present. Opex® was used as a nitrogen-releasing expanding blowing agent for the preparation of natural and synthetic rubbers, polyvinyl, chloride plastisols and epoxy, polyester and silicone resins. Kempore® is used as foaming or blowing agent for cellular and thermoplastic resins. In response to the finding that conditions could pose an imminent hazard, Olin implemented a plan to address response actions required by MADEP and subsequently submitted an Immediate Response Action Completion Statement pursuant to 310 CMR 40.0424 of the MCP, which concluded that a condition of

Substantial Release Migration does not exist for Opex® and Kempore®, and that No Imminent Hazard currently exists due to the buried drum areas on the Olin property. Olin currently maintains the fencing surrounding the two buried drum areas and prevents access to these areas, thereby eliminating the need for any further IRAs.

1.3.6 Lined Lagoons and Calcium-Sulfate Landfill

The Lined Lagoons, Lagoon I and Lagoon II, were constructed in 1972 and originally lined with polyvinyl chloride (PVC) liners. Starting at that time, the acid waste streams in the plant were neutralized with lime slurry and the sludge was then sent to the Lined Lagoons for settling. Periodically, this material was dredged and the calcium sulfate sludge was sent to the Calcium-Sulfate Landfill. Groundwater monitoring data and investigations from the late 1970s showed that the Lined Lagoons were leaking and elevated concentrations of chloride and sulfate were detected in wells in the vicinity of the Lined Lagoons. After purchasing the property in 1980, Olin relined Lagoon I in 1981 and Lagoon II in 1983. At this time the lagoons were lined with 36-millimeter Hypalon liners resting on one foot of compacted sand. The results of groundwater monitoring in the early- to late-1980s showed that concentrations of most inorganics and organics in on-property wells decreased steadily during this time period. This data suggests that the original Lined Lagoons were a source of some inorganics such as ammonia, calcium, chloride, sodium, and sulfate and that closure of the Lined Lagoons removed most of this source. Recent shallow groundwater monitoring data from wells downgradient of the Lined Lagoons and Acid Pits continue to show elevated concentrations of inorganics which suggests that aquifer flushing is incomplete and a residual soil source may still be present.

The location of the closed Calcium-Sulfate Landfill is shown on Plate 1-2. The history of the Calcium-Sulfate Landfill began in 1970, after the discontinuance of chromium usage at the property, when NPI implemented the procedure to have a sanitary landfill on-property for disposal of calcium sulfate. This material was a sludge created during the neutralization of sulfuric acid waste by use of a lime slurry, and was discharged to the Lined Lagoons. Over time, this material needed to be periodically dredged, so NPI received approval from the MADEP for plans to construct the Calcium-Sulfate Landfill in January 1974. Stepan received approval for disposal of waste to the constructed landfill in January 1975. After purchasing the property in 1980, Olin continued to use the landfill for disposal of calcium sulfate sludges until 1986. Approximately 75,000 cubic yards of calcium sulfate are contained in the landfill.

When Olin ceased operations in 1986, part of the shutdown included removal of the two Lined Lagoons. All sludge and liners were removed and disposed on the property, at the Calcium-Sulfate Landfill. Olin received approval for the lagoon closure design in January 1986 and implemented a perimeter groundwater monitoring program. Waste placement at the Calcium-Sulfate Landfill ceased in December 1986. Olin applied to the MADEP in 1986 and 1987 to close the Calcium-Sulfate Landfill and received agency approval on both submittals. The Landfill was then capped and closed. After submitting final closure information in February 1988, the MADEP

(formerly known as the MADEQE) informed Olin that although it had approved the landfill cover system twice in the past, it was disapproving the closure due to changes in regulations on capping requirements.

The CSA (CRA, 1993) investigations at the Calcium-Sulfate Landfill have shown that subsurface soils, as would be expected, have high concentrations of sulfate. Groundwater monitoring in 1986 through 1992 around the perimeter of the Calcium-Sulfate Landfill has shown that the groundwater is contaminated with low to moderate concentrations of ammonia, chloride, and sulfate. Recent groundwater monitoring shows that concentrations of indicator parameters are elevated to the east and southwest of the landfill, but not to the west. Groundwater flow in this area is to the northeast and southwest. Recent observations are not significantly different than the earlier data from the late 1980s, and suggest that the Calcium-Sulfate Landfill is contributing inorganics to the groundwater. Other nearby potential sources may include the inactive municipal landfill for the City of Woburn. The results of the January 1996 Initial Site Assessment and recent groundwater monitoring at the Woburn Landfill show that wastes have contaminated groundwater in the area south of the Olin property and therefore the source of some contaminants in this area is unclear.

1.3.7 Shallow Groundwater Concerns

Inorganic compounds, primarily chromium, sulfate, ammonia and chloride, are the main contaminants of concern in the shallow groundwater. These same contaminants are also present in the surface water in the Off-Property West Ditch and South Ditch. VOCs were also detected in the shallow groundwater and the most widespread VOC of concern was trimethylpentene, which was used in the Wytox ADP process beginning in the early 1960s. The sources of contamination in the shallow groundwater are believed to be the liquid waste disposal areas and other disposal areas. As described in Section 1.3.4, Plant B is also an area with shallow groundwater concerns. The shallow groundwater in this area is mainly contaminated with bis(2-ethylhexyl)phthalate and lesser amounts of NNDPA and trimethylpentenes. Olin initiated the ongoing groundwater recovery and treatment program in this area in 1982 to stop the seep of oil into the East Ditch.

1.4 REPORT STRUCTURE

This document provides the results of the Supplemental Phase II Investigations and a comprehensive description of the environmental conditions at the Olin property in Wilmington, Massachusetts. Within the framework of the MCP, these tasks are considered part of Phase II of the CSA (310 CMR 40.0830 through 40.0840). Since a CSA for this site has previously been submitted (CRA, 1993), this document will serve as a supplement and an update to the

conclusions presented in the earlier report. On subjects where no additional information has been obtained, the previously reported conclusions are briefly summarized. Where new data have provided additional insight, the previous conclusions are revised and updated.

The corresponding section of the MCP is cited in each section heading. For example, Section 4 of this report, which addresses the nature and extent of contamination, is addressed under Section 40.0835(4)(f) of the MCP. This citation is therefore included in the heading for Section 4 in this report. This report is also structured to meet the requirements of the National Contingency Plan (NCP) related to preparation of Remedial Investigation (RI) reports. The report is organized as follows:

- Section 2.0 presents the methods and results of each of the Supplemental Phase II tasks, including further characterization of the bedrock surface topography, groundwater levels and aquifer properties, and of the contaminants in surface water, groundwater, surface and subsurface soils, and sediment. The focus of the Supplemental Phase II tasks was further characterization and geochemical evaluation of the dense layer occupying the bedrock valleys and the chromium flocculent present in the Ditch System

- Section 3.0 describes the physical characteristics of the site and includes brief descriptions of previous characterizations of the bedrock geology, overburden geology, hydrogeology, surface water hydrology, and climate.

- Section 4.0 presents the background concentrations for each medium, a summary of the new data for each medium, the nature and extent of contamination at the site, and the major areas to be addressed during development of the Remedial Action Plan (Phase III)

- Section 5.0 presents the contaminant fate and transport characteristics and focuses on the transport processes of contaminants in the Ditch System and the deep groundwater.

- Section 6.0 summarizes the results of the Human Health Risk Assessment and Environmental Risk Characterization.

- Section 7.0 presents the conclusions of investigations.

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2.0 SUPPLEMENTAL PHASE II INVESTIGATION

MCP 40.0835(4)(d)

CHAPTER SUMMARY

This chapter presents the methods and results of the Supplemental Phase II investigations needed to support the human health and ecological risk assessments and to adequately evaluate remedial alternatives (Phase III).

The specific objectives of the Supplemental Phase II Field Investigation (BCM, 1993) were:

- To define the morphology and depth of the bedrock surface in areas further west, east, and southwest of the property where bedrock valleys have been identified;
- To define the limits of the dense, inorganic layer to the west, east, and southwest of the property;
- To evaluate the geochemistry of the dense layer and its effect on migration and potential remedial alternatives;
- To evaluate aquifer characteristics and the potential rate of migration of the dense layer;
- To determine the relationship between the shallow groundwater and surface water in the ditch system, and evaluate the geochemistry of the observed flocculent; and,
- To obtain additional information about the soil quality at the former Lake Poly Liquid Waste Disposal Area (Lake Poly).

Based on the preliminary results of the Supplemental Phase II Field Investigation, another Work Plan (Smith, 1996d) was approved by the Massachusetts Department of Environmental Protection (MADEP) to obtain additional groundwater quality information in the Maple Meadow Brook wetlands area, west of the Olin property. The primary objective of this program was to install wells in the wetlands overlying the Western Bedrock Valley to investigate the nature of the terrain conductivity anomalies detected in the deepest parts of the bedrock valley underlying this area. In addition, an understanding of the relationship between water quality in the supply wells and nearby monitoring wells, in conjunction with other influences such as pumping rates, aquifer water levels, and seasonality was another primary objective.

In addition to these major work plans, a number of other smaller work plans have been submitted to and approved by the MADEP. The objectives of these have generally been to supplement existing information in a specific area, to obtain data in response to MADEP comments on the original risk assessment (ABB-ES, 1993a), and to provide more updated analytical data for use in the current Risk Assessments.

The chronological details of these submittals and approvals are included in Table 1-1.

The key findings of this section are presented below:

- Through various geophysical and field investigation techniques, the Western Bedrock Valley has been determined to extend from the Olin property towards the west, reaching a depth of over 120 feet beneath the wetland surrounding Maple Meadow Brook and terminating slightly north of where the former Middlesex Canal crosses the wetland.

- An Eastern Bedrock Valley, identified in the CSA, does not extend significantly beyond the eastern boundary of the Olin property. A possible Southwestern Bedrock Valley located along the southwestern property line and extending towards Breed Avenue has been determined not to exist.

- Groundwater flow from the Olin property takes place to the east and west. East of a groundwater divide, flow is to the east towards the South Ditch and East Ditch. West of the divide, flow is westward toward Altron, Maple Meadow Brook, and the surrounding wetlands.

- The unconsolidated sand and gravel deposits are heterogeneous with a high average permeability; they form a productive aquifer with the ability to transmit large quantities of water.

- The underlying bedrock has a very low permeability and little storage capacity. It does not form a substantial part of the overall flow system.

- Groundwater velocities measured in the dense layer compared with the overlying groundwater show significant differences, suggesting that the two systems are subject to different flow dynamics. The results for the dense layer suggest very low velocities, or non-uniform flow through the well screens.

- Surface and subsurface soil, surface water, groundwater, and sediment were sampled for numerous programs as defined in this section. The results of these investigations have led to the understanding of the nature and extent of contamination (see Section 4.0).

- The implementation of the Weir Immediate Response Action (IRA) has modified groundwater and surface water flow in the vicinity of the weir. During certain times of the year, a groundwater mound is created in this area, minimizing the discharge of shallow groundwater and reducing creation of the chromium-bearing flocculent. The mound has also locally changed the direction of groundwater flow in the area west of the Off-Property West Ditch to be toward the northwest and Altron, rather than to the southeast toward the ditch.

- The dense layer has migrated under the influence of gravity through the permeable overburden sediments and away from the facility. It is currently present within the Western Bedrock Valley that is located beneath the center of the property, underneath Jewel Drive, the Altron and Koch facilities, and near the GW-59 well cluster. Some of the dense layer has spilled over the bedrock ridge west of Main Street and is present in the bedrock depression at GW-83D. The dense layer is thickest within the bedrock depressions.

- The interface between the dense layer and the overlying shallow groundwater is sharp (within a few feet), with the overlying shallow groundwater containing inorganic concentrations that are generally 100 to 1,000 times less than that found in the dense layer.

- The dense layer itself is stratified, with the highest concentrations of inorganics found at the bottom in the lowest pH regions of the dense layer. The pH of the dense layer is typically less than 5.15. Inorganic concentrations decrease toward the interface and are 10 to 100 times lower in the upper portion of the dense layer.

- Inorganic compounds in the dense layer have separated laterally with ammonia, chloride, and sulfate transported more readily with the dense layer and aluminum, iron, and chromium less mobile due to precipitation and adsorption reactions.

- The chromium detected in samples of groundwater, surface water and the dense layer appears to be entirely in the trivalent form. Previous detections of hexavalent chromium in water samples were in error due to interferences in the analytical methods. This conclusion is consistent with Knowledge of the Thermodynamics and Kinetics of Chromium (IV) and the reduction by iron (II) which is present throughout the groundwater.

- The Ditch System sediments and surface water have been contaminated with inorganics, including chromium.

- Investigation results show that discharge of chromium from the shallow groundwater to the ditches creates an insoluble precipitate or flocculent, with dissolved chromium detected infrequently in groundwater adjacent to the Ditch System. Sources of chromium may include residual soil contamination around the Ditch System and the unlined pits and the dense layer.

- Migration of other analytes of concern to the Ditch System, such as ammonia, chloride, sodium, and sulfate, appears to be limited to portions of the Off-Property West Ditch and areas along the South Ditch near the Central Pond. In general, the concentration of these analytes is substantially lower in surface water locations relative to corresponding groundwater locations, suggesting that natural attenuation and dilution are controlling solute concentrations within the Ditch System.

- The geochemical investigations described in this section characterize the pathways from the dense layer and address MADEP concerns about the mobility of the various constituents, particularly chromium. The understanding of these processes was applied to fate and transport modeling discussed in Section 5.0 and supported the Human Health Risk Assessment evaluation of current and future groundwater exposure pathways (discussed in Section 6.0).

The documentation of the specific data collection activities conducted under all of the work plans and the subsequent findings outlined above are integrated and presented in the sub-sections of this section as outlined below:

- Evaluation of bedrock conditions, including additional seismic reflection surveys and installation of additional borings to improve understanding of bedrock morphology in certain areas, is discussed in Section 2.1;

- Further characterization of the hydrogeology of the overburden aquifer and bedrock, including pumping tests, slug tests, and borehole velocity monitoring, is discussed in Section 2.2;

- Additional surface soil and subsurface soil sampling to aid in the characterization of the contaminants and risk is discussed in Section 2.3;

- A study of the hydrology of the South and West Ditches, including a summary of the Weir IRA, the installation of shallow groundwater wells and piezometers, and evaluation of groundwater and surface water head conditions and surface water flow, is discussed in Section 2.4;

- Groundwater sampling and installation of new wells to provide current information for use in the risk assessment and detailed site characterization in the area of the Maple Meadow Brook wellfield, Calcium-Sulfate Landfill, Plant B, and other areas of the site is discussed in Section 2.5;

- Additional characterization of the dense layer, including specific conductivity profiling, terrain conductivity surveys, cone penetrometer testing, and inductance logging, is discussed in Section 2.6; and

- Evaluation of the geochemistry of groundwater and surface water, including additional groundwater and dense layer sampling, aquifer material geochemistry, installation and sampling of multilevel piezometers, geochemical modeling, and additional sampling of surface water and flocculent in the Off-Property West Ditch and South Ditch, is discussed in Section 2.7.

The following data collection activities were also conducted under Supplemental Phase II Investigation work plans and are discussed in the indicated sections of this report:

- Evaluation and sampling of surface soils, sediment, groundwater, and surface water, to better characterize background concentrations, is discussed in Section 4.1 and in the Phase II Human Health Risk Assessment and the Stage II Environmental Risk Characterization provided in Appendix R and Appendix S, respectively;

- Regional sampling of surface water and sediment for site-specific parameters, inorganics, VOCs, SVOCs, PCBs, and pesticides as part of the Phase II Human Health Risk Assessment discussed in Appendix R and the Stage II Environmental Risk Characterization provided in Appendix S; and

- The Completion Statement for the IRA associated with the two Drum Areas on the Olin property is summarized in Sections 1.3.5 and 4.4.3.

2.1 EVALUATION OF BEDROCK CONDITIONS

In the Comprehensive Site Assessment (CSA), the geometry of the bedrock surface was described in detail in the immediate vicinity of the property and directly to the west. Further west, however, the CSA included very limited information on the bedrock geology and structure, and scattered data points were used to extrapolate information concerning the shape of the bedrock surface. Because of its importance in controlling the migration direction of the dense layer and the presence of inorganic contaminants detected in two of the wells set in bedrock, Supplemental Phase II activities included additional evaluation of the bedrock geology and structure, including the location and shape of the bedrock valleys to the west, southwest, and east of the property.

The depth of the bedrock surface was determined through bedrock mapping, borings, and seismic reflection surveys. The additional borings to confirm bedrock elevations are discussed in Section 2.1.3. The seismic surveys and the resulting map of the bedrock surface topography are discussed in Section 2.1.4.

In addition to defining the bedrock topography, Smith Technology Corporation (Smith) conducted bedrock mapping and borehole geophysics to collect additional information on the rock types and structure of bedrock beneath the property and the surrounding area. The structure of the rock typically controls the frequency and orientation of water bearing zones. Understanding the structure of the rock aids in predicting to what extent the contaminants may have entered the rock and where they might migrate within the bedrock aquifer. Smith identified and mapped geologic units, obtained structural measurements, and evaluated thin sections of rock samples. The procedures and results of bedrock mapping investigations are discussed in Section 2.1.1. The results of borehole geophysical logging are discussed separately in Section 2.1.2 and the bedrock hydrogeology is discussed in the CSA (CRA, 1993) and Sections 2.2 and 3.4. This new information and the previous characterizations of the bedrock provided in the CSA (CRA, 1993) are summarized in Section 3.3.

2.1.1 Bedrock Mapping

2.1.1.1 Methods

To understand migration of the dense layer above the bedrock, it is important to understand the geologic history of the site and the bedrock composition and structure. Since Conestoga-Rover & Associates (CRA) had previously collected information about the orientation of the bedrock fractures and joints at outcrops, Smith investigated the bedrock to obtain specific information on the mineralogy and structure of the bedrock. Smith performed petrographic studies on 50 thin sections of bedrock samples from 12 locations to identify the component minerals and weathering products of the bedrock. Smith also measured the orientations of over 100 fractures at outcrops to determine fracture orientation and frequency. This information was used in conjunction with review of 28 rock cores from the property and surrounding area to establish how the rock texture and fracturing were associated with the various rock types and old fault zones. Smith utilized this information, along with historical regional mapping, magnetic surveys, seismic reflection data, and additional bedrock samples obtained during the Supplemental Phase II Investigations, to provide the results of the bedrock mapping program described below.

2.1.1.2 Major Structural Features and Bedrock Units

The composition and texture of the bedrock reflects the original rock type and the history of tectonic deformation. These in turn control the relative hardness of the rock and the susceptibility to weathering and erosion. The rock hardness affects how the buried bedrock surface was eroded prior to, during, and after the last glacial advance. Prior to glacial advancement, erosion along ancestral tributaries to the Ipswich River had created a bedrock valley in the area around the location of Maple Meadow Brook. During glaciation, the amount of bedrock fracturing also

controlled the amount of erosion, which could have been particularly intense along fracture zones where the glacial ice could have plucked-out large chunks of bedrock. The bedrock structure beneath the site is therefore key to understanding how the topography of the bedrock surface was created and what type of fracturing may be present.

The bedrock beneath the property is the result of a long and active tectonic history. The property lies next to a very old fault, the Bloody Bluff Fault, and the associated area of deformed bedrock called the Burlington Mylonite Zone. The regional structural geology presented on Figure 2-2 shows that the property lies in the Composite Platform of Southeast New England or the Milford-Dedham Zone (Zen, 1983). The Milford-Dedham Zone is used to describe the group of late Proterozoic and early Paleozoic rocks that lie between the Bloody Bluff Fault and the Northern Boundary Fault of the Boston Basin 8 miles south of the property. The area north and west of the Bloody Bluff Fault is called the Nashoba Zone and is predominantly made up of late Proterozoic mafic meta-volcanic and meta-volcaniclastics. In the study area, the Nashoba Zone is represented by the Andover Granite, a younger granitic rock which intruded the meta-sediments. The Milford-Dedham Zone and the Nashoba Zone are separate tectonic plates, that were joined together at the Bloody Bluff Fault millions of years ago, although the actual time that this occurred is still uncertain.

Figure 2-3 presents a local bedrock geologic map that shows the spatial distribution of rock types relative to faulted zones within the study area. The location of faults shown on Figure 2-3 is approximate and is based on the published geologic maps, aeromagnetic surveys, evaluation of rock cores and outcrops, and the bedrock topography determined by borings and seismic surveys (Castle, et. al., 1976; USGS, 1970; Zen, 1983). In this area, the Milford-Dedham zone is represented by three texturally different types of gabbro-diorite which express variable deformation associated with the Burlington Mylonite Zone due to movement along the Bloody Bluff Fault. The movement along the Bloody Bluff Fault over 500 million years ago created mylonites, which are fine-grained rocks formed from coarser-grained rocks being crushed together (granulation). As shown on Figure 2-3, the Bloody Bluff Fault trends northeast-southwest through the Maple Meadow Brook area and a splay of this fault is interpreted to lie between Jewel Drive and the western boundary of the property. The interpreted bedrock geology and relation to these fault zones is discussed below.

Bloody Bluff Fault Zone

The Bloody Bluff Fault Zone is a zone of several closely spaced individual thrust faults. It is believed to have a width of approximately 600 feet in the subject area and may be wider. The trend is northeast-southwest and closely parallels Maple Meadow Brook and passes beneath the Town Park area. In this area, it is reported to be a high angle thrust fault with a northwest dip between 65° and 85°. The major movement along the Bloody Bluff Fault Zone is probably late Proterozoic. Later deformation was brittle and was produced by lateral strike-slip fault movement which is represented by the series of faults at angles to the northeast-southwest

trending series of faults associated with the Bloody Bluff Fault as shown on Figure 2-3. One area of mylonite, mapped between the western side of the property and Jewel Drive, is contained within one of the splays associated with the Bloody Bluff Fault Zone.

The larger area of more deformed, undifferentiated mylonites mapped within the Bloody Bluff Fault Zone beneath the Maple Meadow Brook wetlands includes ultramylonite and siliceous and mafic mylonites. The ultramylonites show severe ductile polydeformation (several events of crushing and movement between mineral grains) which is probably associated with the early movement along the fault. Beneath the Maple Meadow Brook wetlands, the Bloody Bluff Fault Zone encompasses the mylonites formed from deformation of rock composed of volcaniclastic or arkosic sediments (Figure 2-3). These resulting mylonites are generally very fine to fine grained rocks that are very susceptible to weathering and deep erosion. This weakened zone of fractured rock and deeply weathered bedrock was eroded and formed a bedrock low currently occupied by the Maple Meadow Brook wetlands and its associated drainage system.

Burlington Mylonite Zone

The Burlington Mylonite Zone has been characterized as a 3-mile wide zone around the Bloody Bluff Fault Zone within which rocks of variable composition have been severely deformed. Observations in the subject area indicate that some mylonitization (crushing and heating) of rocks away from the Bloody Bluff Fault Zone occurs in specific narrow zones between slightly deformed and/or undeformed rocks. This is typical of areas occupied by competent rock such as the gabbro-diorite beneath the upland areas east of the Maple Meadow Brook as shown on Figure 2-3. The types of gabbro-diorites in the Burlington Mylonite Zone include a normal gabbro-diorite, a gabbro-diorite cataclastic, and a gabbro-diorite with granitic intrusives.

The undeformed or normal gabbro-diorites shown on Figure 2-3 are fine-grained, weakly foliated, and moderately resistant rocks. This type of bedrock forms the low-lying outcrops in the vicinity of the facility including the outcrops north of the South Ditch and those exposed at the eastern boundary of the facility near well cluster GW-51. The cataclastic gabbro-diorite shown on Figure 2-3 form the topographically lower terrain with more gentle slopes. These rock underlie the central portion of the property and the South Ditch in areas where the bedrock surface is more eroded. The areas where the gabbro-diorite is most affected by cataclasis are primarily associated with the fracture zones between Main Street and the east edge of the Maple Meadow Brook wetlands, north and west of Eames Street. The gabbro-diorite with granitic intrusives is the most competent bedrock type and is resistant to weathering. This type of bedrock underlies or is exposed in outcrop at the rounded and knobby hills near the Calcium-Sulfate Landfill and around Cook Avenue, between Jewel Drive and Main Street (Figure 2-3).

2.1.1.3 Description of Bedrock Units

Geologic mapping has identified two classes of rocks which vary both texturally and compositionally: mylonites (and cataclasites) and intrusives. Compositionally, there are two types of mylonites: mafic and siliceous. The mafic mylonites formed from the gabbro-diorite, and the siliceous mylonites potentially formed from a variety of original rock types including granite, quartzite or quartz sandstone, and/or volcaniclastics. Both types of mylonite are physically and chemically altered due to ductile deformation during tectonic stress. Cataclasites are similar to mylonites, but exhibit less developed granulation textures and features. Intrusive rocks beneath the property and the surrounding area include granite and gabbro-diorite rock types which were formed from molten rock which crystallized deep within the earth's crust. These rocks are locally altered due to some differential response to tectonic stresses which has caused textural variations (cataclasis) in the gabbro-diorite and Andover Granite near fault zones. The mineralogy, texture, and weathering of these different rocks are discussed below. A bedrock geologic map is included as Figure 2-3 for reference of spatial distributions of these rock types.

Undifferentiated Mylonites

Mylonites formed from coarser-grained rocks being crushed together (granulation) during movement along thrust faults over 500 million years ago. In the area around Wilmington, the mylonites range from very fine-grained, thinly laminated rocks to very fine-grained rocks with scattered coarser-grained individual crystals (porphyroclastic). Post-mylonite deformation produced brittle fracturing in the mylonites, and later hydrothermal activity filled these fractures with chlorite and calcite.

The mafic mylonites occur within or adjacent to the Bloody Bluff Fault Zone beneath the Maple Meadow Brook wetlands and in the area between Jewel Drive and the western side of the property. These rocks have a groundmass of cryptocrystalline calcite, feldspar, and dark-colored (mafic) minerals such as biotite and hornblende that exhibit mildly porphyroclastic textures, strong fluxion structures (ribbon-like), and mylonite banding. The mafic mylonites were observed in rock cores from wells GW-62BR, GW-71D, and GW-80D which border the Bloody Bluff Fault Zone and the Maple Meadow Brook wetlands and in rock cores from GW-42D, GW-43D, and GW-69D which are located between Jewel Drive and the western boundary of the property.

The siliceous mylonites that originate from quartzite or quartz sandstone are much harder than those that originate from volcaniclastics/arkoses. The mylonitized quartzitic rock makes up the bedrock hills and uplands west of the Maple Meadow Brook wetlands. These rocks are compositionally similar to the granitic mylonite, although they have less mafic minerals and less feldspar. The quartz groundmass of this mylonite is more equigranular and there is more recrystallized quartz present.

The siliceous mylonites that originate from the volcaniclastics/arkosic rock are intensely deformed to a point where they are ultramylonites (minerals crushed together and altered). These mylonites occur within or adjacent to the Bloody Bluff Fault Zone beneath the Maple Meadow Brook wetlands and its associated drainage system. These rocks have a groundmass of cryptocrystalline quartz, calcite, and feldspar which exhibits mildly porphyroclastic textures, strong fluxion structures, and mylonite banding. These rocks are very susceptible to erosion and form the base of valleys filled with glacial and Holocene sediments and are therefore poorly exposed.

This susceptibility to erosion was later enhanced by brittle deformation which resulted in advanced decomposition of the ultramylonite. This later deformation created a thick weathered bedrock or saprolite on top of more competent bedrock with a dislocated fracture system. This fracture system is made up of "dead end" fractures filled with chlorite, sulfide, and calcite minerals and areas of open fractures due to dissolution of calcite. This allowed deep weathering of the bedrock to occur prior to glaciation. Weathered zones of a saprolite composed of chlorite-rich, clayey, silt were observed at all of the wells in the Western Bedrock Valley. At all of the borings, it appears that this saprolite has been partly eroded and incorporated into the matrix of the clayey-gravel till above the weathered bedrock interface.

Normal Gabbro-Diorites

The normal gabbro-diorite has uniform, equigranular texture of mafic minerals and felspar. The major constituents of the gabbro-diorite are horneblende, biotite, and plagioclase feldspar. Where granitic veins or some degree of cataclasis is present, the rock exhibits mild foliation or banding. As mentioned above, the normal gabbro-diorites form the low-lying outcrops in the vicinity of the property including the outcrops north of the South Ditch and those exposed at the eastern boundary of the property near well cluster GW-51.

Cataclastic Gabbro-Diorite

The Cataclastic Gabbro-Diorites are similar to mylonites, but exhibit less developed granulation textures and features. At the property, the cataclastic gabbro-diorite forms the topographically lower terrain beneath the central portion of the property between the South Ditch and the bedrock high at the southern end of the property. Off the property, east and west of Main Street, the gabbro-diorite has been more affected by cataclasis and is altered to a hornblende-biotite schist with fractures parallel and normal to the foliation. At GW-61BR, hydrothermal alteration has filled these fractures with chlorite, sulfide, hematite, and epidote. This variability of deformation is closely related to proximity of fault zones and is believed to be the result of the gabbro-diorite's response to tectonic stresses. This differential response to tectonic stresses is supported by the observed textural variations of the gabbro-diorite near fault zones and by cataclasis within narrow zones of the gabbro-diorite away from the main fault zones. It is possible that the tectonic stresses would have been deflected by the gabbro-diorite and, consequently, were only absorbed along the edges of the intrusive body at the Bloody Bluff Fault Zone and other minor fracture zones.

Gabbro-Diorite With Granitic Intrusives

The gabbro is locally intruded by granite which has reinforced the physical integrity of the gabbro. The gabbro-diorite with granite intrusives has uniform, equigranular texture of mafic minerals and feldpar. The major constituents of the gabbro-diorite are mafic bands of horneblende, biotite, and plagioclase feldspar and light-colored bands of coarser-grained quartz and feldspar. In areas where some degree of cataclasis is present, the rock exhibits mild foliation or banding. The areas underlain by this rock are more resistant to weathering and lie beneath the Calcium-Sulfate Landfill, Cook Avenue, and other uplands south of the property.

Andover Granite/Granitic Mylonites

The composition and texture of the Andover Granite and granitic mylonites mapped in the study are arranged from less deformed granite with medium to coarse-grained quartz and feldpar to granulated and recrystallized quartz and feldspar with a mildly porphyroclastic texture consisting of stressed, fractured, and sericitized feldspar clasts. The groundmass of rock samples near the Bloody Bluff Fault Zone also included irregular bands or laminae of muscovite, chlorite, or partially altered biotite, much of which has been further altered to clays and oxides. Fluxion structures and porphyroclastic textures are characteristic of the more deformed granitic mylonites. The granitic mylonites were observed in the core from wells GW-64D, GW-65D, and CB-3 and in outcrops along Presidential Way, north of the property. At GW-65D, the feldspars and mica making up the granitic mylonite have been partly altered to clays. This intense weathering has allowed the bedrock to be more easily eroded to form a low lying area with a moderate to thick accumulation of overlying glacial sediments.

2.1.1.4 Fracture System Characteristics

The regional fracture systems of the Bloody Bluff Fault Zone and the local subsets associated with the Burlington Mylonite Zone were generated from compressional and extentional states of stress. In this context, the zone of fracturing could penetrate to significant depth. However, low water yields and the observed in-filling of fractures with hydrothermally deposited or weathered minerals shows that the interconnection and water storage of these fractures is very limited. In areas where less mineralogic alteration has occurred or where horizontal exfoliation or stress release fractures are present, the fractures tend to be better connected and more open. Typically, these fractures are more prevalent in the upper portion of the bedrock and diminish in number with depth.

A rose diagram showing the strike orientation of 101 fracture measurements is presented in Figure 2-3. Fifteen different sets of fracture orientations have been measured. These sets are probably representative of five or more independent major stress orientations. Of primary importance is the northeast fracture sets. Regionally, they appear as major topographic features and aeromagnetic anomalies which parallel the shape of the Bloody Bluff Fault Zone. Locally, the northeast fracture set associated with the Bloody Bluff Fault Zone represents the most severely fractured zone. It would not be uncommon to find a 5- to 10-foot zone of intensely sheared rock

following the mylonite foliation (N30-60E). These shear zones typically are made up of individual rectangular fragments of various sizes which create a system of fractures in the bedrock. This system of fractures is accompanied by deep weathering to a clay-rich saprolite which was observed in all the borings in the Western Bedrock Valley near the Bloody Bluff Fault Zone (Figure 2-3). Outside of these sheared zones, the northeast fractures are pronounced and closely spaced.

The other fracture or fault zones are also shown on Figure 2-3. A prominent north trending regional fracture which parallels Jewel Street in the subject area has been interpreted from aeromagnetic data and supported by mapped dextral displacements of lithologic units and other lineaments. It is also along this zone that wells GW-68BR and GW81BR are located. Two other north trending fractures have been interpreted as parallel fault splays from a northeast trending fracture located along the east edge of the Maple Meadow Brook wetlands. The fractures have been interpreted from drill hole data and seismic profiles. There is one set of regional northwest fractures that is also supported by field investigation. This fracture set is manifested in the southeast valley and is supported by weak topographic expressions (lows) and the seismic reflection profiles. Locally, many northwest fractures that have been mapped in outcrop are joints, some of which are very pronounced.

As discussed above, the proximity and relation to the shear zones related to the Bloody Bluff Fault Zone and related splays strongly controls the fracturing and characteristics of the bedrock, and therefore each type of bedrock generally displays a distinct style of fracturing. The normal gabbro-diorite is moderately to strongly fractured and the orientation of fractures is generally sub-horizontal with intersecting near-vertical fractures in areas adjacent to fault zones. The fractures show little to moderate weathering with slight to moderate oxidation of the fracture surface. The cataclastic gabbro-diorite is weakly to moderately fractured and the orientation of fractures is generally sub-vertical with sub-horizontal fractures predominating in areas adjacent to fault zones. The fractures in the cataclastic gabbro-diorite are slightly to highly weathered with chlorite evident on fracture surfaces. The gabbro-diorite with granitic intrusives are weakly to moderately fractured and the orientation of fractures is generally sub-vertical. The fractures in the gabbro-diorite with granitic intrusives are slightly weathered. The undifferentiated mylonites are generally moderately fractured in the area between Jewel Drive and the western side of the property and intensely fractured in the area of Maple Meadow Brook wetlands. Most of the fractures encountered in the undifferentiated mylonites are highly weathered and filled with calcite, chlorite, and clay.

2.1.2 Borehole Geophysics

Downhole geophysical data were collected in the eight open borehole bedrock wells shown on Figure 2-3. These wells include the three bedrock wells, GW-62BRD, GW-65BR, and GW-81BR, installed during the Supplemental Phase II Field Investigations. When used in conjunction with bedrock mapping, seismic reflection, and core logs, the borehole geophysical data helps

provide an understanding of the water-bearing characteristics of the bedrock. Downhole information also supplies a third dimension to the interpretation of the outcrop data and provides indirect information on the presence of inorganic constituents in the flow system.

2.1.2.1 Methods and Overview

Smith contracted Geophysics GPR International of Needham, Massachusetts to collect downhole geophysical data from the previously mentioned bedrock wells. The suite of downhole geophysical tools included logging of caliper, natural gamma, spontaneous potential, single-point resistivity, and temperature. In addition, a downhole camera was used to obtain a video of the appearance of each borehole.

The downhole geophysical logging was conducted on October 24 and 25, 1995. Each well was first logged for temperature due to the sensitivity of this measurement. This was followed by the caliper logging on the same tool. Finally, the electrical and gamma logs were obtained. Real-time results were displayed during logging. Therefore, when any unexpected results were obtained, the logs could be re-run to ensure reproducibility. The video camera logging was conducted on October 31, 1995. Between each well, the geophysical tools and cables were decontaminated to prevent cross-contamination. A summary of the information acquired by each geophysical tool and the general findings are discussed below.

Natural Gamma Logs measure gamma radiation from naturally occurring unstable isotopes mainly of Potassium-40, Uranium-238, and Thorium-232. Of these isotopes, Potassium-40 is the most commonly encountered and is found in feldspars and mica which decompose to clays. Clays also concentrate heavy radioactive elements from processes of ion-exchange and adsorption. According to Keys MacCary (1971), about 90 percent of the gamma-photons detected probably originate within 6 to 12 inches of the borehole wall. The natural gamma logs record an average gamma count which results in a poor thin-bed resolution and which will cause, for example, thin clay-rich beds to look like a silt. Other anomalous shifts may be due to mud cake/scaling on the borehole wall, steel casing, increases in hole diameter, and the gravel pack, bentonite seals, and grout in the annular space.

The natural gamma logs of overburden were in general agreement with drilling logs and were useful in further identifying sand-rich and clay-rich beds which may act as preferential migration pathways or confining units, respectively. In general, discrete, thick beds of a single stratigraphic unit were not observed in the overburden material, emphasizing the well-mixed nature of glacial materials. The natural gamma logs of bedrock intervals also showed lithologic variations that are not identified on the drilling logs. The natural gamma logs of bedrock intervals also showed relatively higher gamma counts per second (CPS) adjacent to intervals which are fractured or more highly altered as identified on the drilling and camera logs and as interpreted from the

caliper and single point resistivity logs. This implies that these intervals contain weathered fractures or more clay alteration products. It is believed that these clay-rich zones are due to more intense weathering along fractured portions of the bedrock.

Caliper Logs measure borehole diameter by mechanical means using three arms. The three arms "feel" the sides of the borehole and measure an average diameter. The average diameter may not include some small or tightly closed fractures and tends to show dipping fractures as a series of inflections.

The caliper logs document the uncased portion of the corehole or borehole in the bedrock. Where a smooth borehole was created by coring, the caliper logs were very useful in identifying fractures or weaker intervals in the bedrock. The caliper logs generally show a decrease in hole diameter immediately below the casing, but in some instances show an increase in hole diameter due to a gap between the bottom of the casing and the top of the rock borehole. The caliper logs were used in conjunction with the single point resistivity logs to identify intervals with low resistivity solely due to increases in borehole diameter.

Single-point resistivity logs measure the total resistance of earth material between the current electrode in the borehole and either the ground electrode at the surface (conventional) or the tool shell/cable ground electrode (differential). Since the radius of measurement is small, 5 to 10 times the electrode diameter, the single-point resistance log mainly measures the resistance of the borehole immediately adjacent to the probe and is therefore strongly affected by the conductivity of the borehole fluid and the borehole diameter. This means that for identification of lithologic changes, the single-point resistance log performs better in fresh water-filled boreholes (where borehole fluid does not short circuit the rock).

The resistance logs were used in conjunction with the natural gamma logs to identify clay-rich intervals which generally had a low resistivity. Resistance logs were also used to identify small fractures by virtue of the fact that a void filled with fresh (or saline) water results in a decrease in resistance on the log; however some of these features were related to wash-outs due to the drilling process. The resistance logs were also used in conjunction with the spontaneous potential logs to tentatively identify intervals with fresh water versus saline water conditions and intervals with low resistivity due to increases in borehole diameter.

Spontaneous potential (SP) logs measure the natural electrical differences in voltage (in units of millivolts [mV]) that develop at the contacts between rock units (shale or clay), formation water, and fluid in the borehole. There are two sources of electrical potentials, the biggest contributor to potential is the electrochemical potential produced at the contact between dissimilar material, although streaming potential caused by flowing water (electrokinetic potential) can be important. The potentials are set up between the formation water and shales. These potentials cause a minute flow of electrical current near the clay-aquifer boundaries which is then measured by the logging probe. The radius of investigation is extremely variable and SP logs are affected by several sources of signal noise.

In very general terms, SP logs in fresh water aquifers are more positive (inflection to right) opposite sands and more negative (inflection to left) opposite clay-rich units. In this case resistivity logs and SP logs generally deflect in the same direction. At the property, intervals with opposite resistivity and SP inflections are interpreted to indicate saline conditions (high dissolved solids) in the groundwater. This interpretation generally agreed with water quality results. However, this interpretation is based on a limited cross section of the bedrock aquifer and the SP responses are less predictable in fresh water-filled boreholes.

Fluid Temperature Logs record borehole fluid temperature and the rate of temperature change with depth and can be extremely useful for evaluating groundwater movement, permeability distribution, and relative hydraulic head. In general, significant temperature changes were only observed in BR-1 and GW-81BR. Minor fluctuations did appear to correspond with fracture zones identified by the caliper and camera logs.

Camera Video Logs produced by a downhole camera are not included in this report but are available for review upon request. The color video logs show the condition of the casing and borehole and displays the depth of the camera in feet below ground surface (bgs). The hydrogeologic and well construction information observed on the camera videos are presented on the camera logs in Appendix B. In several instances, the camera log depths are one to two feet greater than the depth reported on the geophysical logs, since the geophysical logs are based on ground surface and the camera logs are based on the top of casing. The depths to various features discussed below are corrected to ground surface, however for consistency with the video record, the camera logs in Appendix B are not adjusted and the proper adjustment is noted in the header of each log.

The camera logs summarize observations on well construction, fractures, borehole water turbidity, staining, and borehole color. The vast majority of fractures observed on the camera logs were in-filled with a white mineral which was identified as calcite or quartz by petrographic analysis. Due to the presence of in-filled fractures throughout the bedrock, the use of the "cemented" descriptor was limited to fractures which appear partially cemented or in-filled. Fractures are characterized as being cemented, closed, or open. Closed fractures refer to fractures without visible in-filling, but do not appear open. The fractures observed are further categorized on the log as dipping (D), vertical (V), or horizontal (H).

Qualitative descriptors of turbidity included clear, cloudy, and very cloudy. In some cases, the increased turbidity was caused by the movement of the downhole camera. In these cases, turbidity is more of a measure of the video quality. Staining refers to localized discoloration of the casing or the borehole walls due to scaling, precipitation of inorganics, or potentially biological growth. The borehole color is mainly intended to distinguish changes in lithology.

2.1.2.2 Evaluation of Geophysical and Camera Logs

Annotated geophysical logs which include pertinent camera log observations are presented in Appendix B. The report of procedures and results completed by Geophysics GPR International is presented in Appendix C. The detailed summary of the geophysical logs for each well presented below discusses observations on the frequency and nature of fracturing, stratigraphic changes, and observations on the well construction.

GW-61BR. This well is completed to a depth of 92 feet bgs in a cataclastic gabbro-diorite that resembles an amphibole-biotite schist. Structurally, this well is interpreted to lie in close proximity to the fault contact associated with the Bloody Bluff Fault Zone. The majority of fractures observed are in-filled with a white mineral, although a series of open to closed dipping fractures with a 1-foot spacing were noted between 74 and 81 feet bgs. This fracture zone corresponds with inflections in the natural gamma and single-point resistivity logs which show that these fractures lie in a more intensely weathered or altered clay-rich portion of the bedrock. Slight inflections in the differential temperature log observed at 74 feet bgs and 77 feet bgs, and the low resistivity adjacent to fractures at 77 feet bgs and 84 feet bgs, may indicate water bearing fractures. The response of the SP and single-point resistivity logs adjacent to these fracture zones suggest that the borehole water is low in dissolved solids.

The overburden stratigraphy recorded on the drilling log correlates well with the natural gamma response. The natural gamma log identifies sand-rich beds which may be more permeable between 20 and 28 feet bgs and between 36 and 42 feet bgs, and potentially a clayey gravel immediately above the bedrock interface from 46 to 50 feet bgs. The depth to bedrock may be at 50 feet bgs rather than 48 feet bgs reported on the drilling log. The interpreted natural gamma log shows a weathered bedrock interval from 50 feet bgs to 69 feet bgs (see Appendix B for annotated log). The bedrock interval also shows more weathered or altered clay-rich rock between 76 and 84 feet bgs.

The well construction consists of 4-inch polyvinyl chloride (PVC) casing from the surface to a depth of 72 feet bgs. Review of the camera log showed that the increase in hole diameter between 73 and 76 feet bgs is partly due to a ledge where the casing meets the bedrock and the smaller diameter corehole begins and partly due to a series of dipping fractures at 73 to 74 feet bgs which caused a bend in the corehole. The inside of the PVC casing appeared stained on one side between the casing joints at 33 and 43 feet bgs. This staining is probably related to the original condition of the casing, but may indicate a bad casing seal at these locations. The camera video terminates at a piece of discharge tubing at 85 feet bgs, although a sediment pool appears to be visible beyond the tubing at 87 feet bgs.

GW-62BR. This well is completed to a depth of 109 feet bgs in a banded mylonite. Structurally, this well is interpreted to lie very close to a fault contact associated with the Bloody Bluff Fault Zone. The bedrock observed at this location is intensely fractured with intersecting vertical and dipping fractures, although the majority of fractures observed are in-filled with a white mineral. A series of open dipping fractures with a 6-inch spacing were noted between 76 and 84 feet bgs and

open dipping fractures were noted at 96 and 99 feet bgs. These fracture zones correspond with inflections in the natural gamma and single-point resistivity logs which show that these fractures lie in a more intensely weathered or altered clay-rich portion of the bedrock. Slight inflections in the differential temperature log were observed at 77 feet bgs, 80 feet bgs, and 96 feet bgs and may indicate water bearing fractures, although the low yield of this well (about 1 gallon per minute [gpm]) suggests that these fractures are not significant water bearing zones. The response of the SP and single-point resistivity logs adjacent to these fracture zones suggest that the borehole water is high in dissolved solids.

The upper 30 feet of overburden stratigraphy recorded on the drilling log correlates well with the natural gamma response. The natural gamma log identifies sand-rich beds which may be more permeable between 8 and 30 feet bgs and shows an increase in clay content in sands below 30 feet bgs. The lower gamma count between 61 and 66 feet bgs and immediately above the bedrock interface may be a more permeable sand or gravel. The interpreted natural gamma log shows a weathered bedrock interval from 70 feet bgs to 74 feet bgs and a possible change in lithology below 88 feet bgs. The bedrock interval also appears to be more weathered or altered clay-rich rock between 88 and 91 feet bgs and between 96 to 108 feet bgs (see Appendix B for annotated log).

The well construction consists of 4-inch PVC casing from the surface to a depth of 75 feet bgs. Review of the caliper, camera log, and drilling log shows the casing does not extend to the bottom of the 5-inch rock borehole and that this has resulted in an increase in borehole diameter between 75 and 78 feet bgs. Part of the increase in borehole diameter may also be related to wash-outs created during drilling or changes in bit direction caused by the intensely fractured zone between 76 and 84 feet bgs. The low resistivity measured adjacent to this interval may indicate a water-bearing fracture zone immediately below the casing or a poor casing seal. The inside of the PVC casing appeared stained light yellow to orange-brown interpreted as iron precipitate with darker stains at the casing joints at 25 feet bgs and 66 feet bgs. This staining also may indicate a bad casing seal at these locations. The camera video extended from the ground surface to 109 feet bgs, terminating in a shallow pool of sediment.

GW-62BRD. This well is completed to a depth of 145 feet bgs in a banded mylonite. Structurally, this well is also interpreted to lie very close to a fault contact associated with the Bloody Bluff Fault Zone. The bedrock observed at this location is intensely fractured with intersecting vertical and dipping fractures, although the majority of fractures observed are in-filled with a white mineral. A series of open to closed dipping fractures with a 6-inch to 1-foot spacing were noted between 113 and 119 feet bgs, 123 and 127 feet bgs, and 140 and 145 feet bgs. Discrete open dipping fractures were noted at 109 feet bgs and 131 feet bgs. Most of these fracture zones correspond with slight inflections in the natural gamma and single-point resistivity logs which may show that these fractures lie in more intensely weathered or altered clay-rich portion of the bedrock.

Slight inflections in the differential temperature log were observed at 119 feet bgs, 125 feet bgs, 131 feet bgs, and 141 feet bgs. Red-brown streaks 1 to 2 feet long originating or terminating at individual fractures or as stains along the circumference of an individual fracture are interpreted as iron precipitate and were associated with fractures at 109 feet bgs, 123 feet bgs, 126 feet bgs, and 131 feet bgs in the camera logs. These features may indicate water bearing fractures, although the low yield of this well (about 1 gpm) suggests that these fractures are not significant water bearing zones. The dipping fracture at 131 feet bgs appears open and may be the most significant water entry point in the borehole. The response of the SP and single-point resistivity logs adjacent to the open fracture zones suggest the borehole water is high in dissolved solids.

The upper 34 feet of overburden stratigraphy recorded on the drilling log correlates well with the natural gamma log response. The natural gamma log identifies sand-rich beds which may be more permeable between 1 to 34 feet bgs and shows a gradual increase in clay content in sands between 34 and 60 feet bgs. The drilling log for GW-62D indicates that the bedrock starts at 66 feet bgs, while the interpretation of natural gamma log indicates a bedrock depth of 76 feet bgs. The interpreted natural gamma log shows a weathered bedrock interval from 76 feet bgs to 84 feet bgs and a change in lithology below 96 feet bgs. The bedrock interval also appears to be more weathered or altered clay-rich rock between 127 and 132 feet bgs and between 136 and 142 feet bgs (see Appendix B for annotated log).

The well construction consists of 4-inch PVC casing from the surface to a depth of 105 feet bgs. The inside of the PVC casing appeared stained yellow to orange-brown interpreted as iron precipitate with darker stains above and below the casing joints at 65 feet bgs and 76 feet bgs. This may indicate a bad casing seal at these locations. The camera log extended from the ground surface to 145 feet bgs, terminating in a pool of sediment.

GW-65BR. This well is completed to a depth of 145 feet bgs in a medium-grained, weathered granite (Andover Granite) cataclasite. Structurally, this well is also interpreted to lie on the Nashoba Zone which is immediately north of the Bloody Bluff Fault Zone. Due to the turbidity of the water in the borehole at this location, the camera video log was not used to confirm fracture locations. In addition, the air hammer used to advance the borehole produced the rougher borehole walls and made interpretation of fractures by the caliper log difficult. However, the caliper log increases in hole diameter at 113, 117, 120 to 122, 124, 128, 130, 134 and 136 feet bgs generally corresponded with intervals with higher natural gamma counts and are interpreted as fracture zones. These fracture zones correspond with inflections in the natural gamma and single-point resistivity logs which show that these fractures lie in more intensely weathered or altered clay-rich portion of the bedrock. The increase in hole diameter at 134 to 136 feet bgs may be due to open fractures as they correspond with the presence of quartz crystals in cuttings reported on the drilling log. The response of the SP and single-point resistivity logs adjacent to the fractures suggest the borehole water is low in dissolved solids above 120 feet bgs and high in dissolved solids below 120 feet bgs.

The overburden stratigraphy recorded on the drilling log correlates fairly well with the natural gamma response. The natural gamma shows a relatively low gamma count for the peat layer between 8 feet bgs and 11 feet bgs. Increasing clay content is evident in the sands 11 feet above the bedrock interface at 104 feet bgs. The bedrock interval shows more weathered or altered clay-rich rock between 111 and 127 feet bgs and between 129 and 135 feet bgs (see Appendix B for annotated log).

The well construction consists of 4-inch PVC casing from the surface to a depth of 110 feet bgs. The video camera log showed that the borehole and a portion of the casing from 81 feet bgs to 140 feet bgs is filled with a suspension of white to gray, fine-grained material and water, presumably grout and/or cuttings. The top of this material corresponds to a casing joint at 81 feet bgs and may indicate failure of the casing at this depth. The material in the borehole obscured the camera video log and prevented identification of the source of the suspended material. Water quality results for this well show an alkaline pH of 10 to 11 and support the conclusion that the bottom 60 feet of the well is filled with suspended grout. This well was re-developed to remove the suspended material and the re-development is discussed in Section 2.5.5.3.

GW-68BR. This well is completed to a depth of 76 feet bgs in a fine-grained, dark (mafic) banded mylonite that resembles a gneiss with alternating mafic and quartz-rich bands. Structurally, this well is also interpreted to lie in a fault-bounded block that is associated with a splay of the Bloody Bluff Fault Zone that has mylonitized the granite-intruded gabbro-diorite bedrock. No significant fracture zones were noted on the camera or caliper logs, although an open fracture was noted at 74 feet bgs that may be a water entry point. The caliper log shows very minor changes in hole diameter at 66 feet bgs and at 74 feet bgs. Review of the camera log showed no apparent cause of the hole diameter change at 66 feet bgs and a dipping, open fracture at 74 feet bgs. The single point resistivity and SP logs show several inflections potentially related to lithologic variations at 41 feet bgs, 60 feet bgs, and 73 feet bgs. The positive inflection of the spontaneous potential log at 72 to 74 feet bgs and inflections in the temperature log may indicate water inflow or change in water quality at this interval. The water quality of this zone cannot be inferred from the response of the SP and single-point resistivity logs.

The overburden stratigraphy recorded on the drilling log correlates fairly well with the natural gamma response. The natural gamma log shows an increase in clay content in the overburden immediately above the bedrock interface at 11 feet bgs. The interpreted natural gamma log shows weathered or altered clay-rich rock between 31 to 35 feet bgs and 71 to 75 feet bgs and possibly a change in lithology below 35 feet bgs.

The well construction consists of 4-inch PVC casing from the surface to a depth of 31 feet bgs. The video camera log showed that the upper 11 feet of casing is a yellowish gray color and that a slight sheen was present on the surface of the static water column. The relatively lower gamma counts between 31 to 35 feet bgs may in part be related to the grout extending beyond the bottom of the casing (see Appendix B for camera log).

GW-80BR. This well is completed to a depth of 70 feet bgs in a cataclastic to normal gabbro-diorite. Structurally, this well is also interpreted to lie near a fault contact between the normal gabbro-diorite and the cataclastic gabbro-diorite. No significant fracture zones were noted on the camera or caliper logs. The bedrock observed at this location is weakly fractured and only two discrete, dipping fractures were observed. These fractures appeared closed and were noted in the video log at 52 feet bgs and 60 feet bgs. Unlike the other wells logged, these fractures do not correspond with intervals of more intensely weathered or altered clay-rich portion interpreted from the natural gamma and single-point resistivity logs.

Slight inflections in the differential temperature log were noted at 32 feet bgs and 40 feet bgs and changes in the slope of the temperature curve were observed at 46 feet bgs and 54 feet bgs. The inflection in the temperature log at 32 feet bgs corresponds with a gap at the bottom of the casing and very low resistivity and may indicate that water is entering the well at this depth. The inflection in the temperature log at 46 feet bgs corresponds with an interval of softer rock noted on the drilling log and the inflection at 54 feet bgs corresponds with an interval of altered or weathered bedrock. The caliper log shows rough, undulatory borehole walls that reflect the air rotary drilling method used to advance the borehole into the bedrock. The water quality of these zones cannot be inferred from the response of the SP and single-point resistivity logs, however the decreases in resistivity at 32 feet bgs, 38 feet bgs, between 44 and 48 feet bgs, and between 58 and 64 feet bgs may be related to fractures, an increase in clay-content, or change in water quality.

The natural gamma log shows little difference in the response of the overburden and the upper 25 feet of bedrock. The natural gamma log of the bedrock interval shows more weathered or altered clay-rich rock between 15 and 22 feet bgs and between 40 and 48 feet bgs. As noted above, the relatively lower gamma counts between 15 and 22 feet bgs and between 40 and 48 feet bgs correspond to a clay-rich zone identified as weathered bedrock and "soft" on the drilling log.

The well construction consists of 8-inch PVC casing from the surface to a depth of 30 feet bgs. The video camera log showed that the entire length of casing is a yellowish gray color. The video log also shows that a gap is present between the bottom of the casing and the bedrock and that gray to brown scaling was present on the bottom one foot of casing and a thicker gray-brown mass was present on the ledge immediately below the casing where the corehole begins. As indicated above, these observations suggest that overburden groundwater is entering the borehole at the bottom of the casing. The relatively lower gamma counts between 31 to 35 feet bgs may in part be related to the grout that flowed beyond the bottom of the casing during well construction.

GW-81BR. This well is completed to a depth of 50 feet bgs in a fine-grained, dark (mafic), banded mylonite that looks like a gneiss with alternating mafic and quartz-rich bands. Structurally, this well is also interpreted to lie in a fault-bounded block that is associated with a splay of the Bloody Bluff Fault Zone that has mylonized the gabbro-diorite. No significant fracture zones were noted on the camera or caliper logs. Closed to open fractures were observed on the camera log at 28 feet bgs, 35 feet bgs, 41 feet bgs, and 47 feet bgs. The fracture at 41 feet bgs also corresponded with a significant temperature log inflection, although the low yield of this

well (about 1 gpm) suggests that this fracture is a not significant water bearing zone. The single point resistivity and SP logs are limited to the bottom eight feet of the borehole. The low resistivity at 48 feet bgs appears to be due to a minor increase in borehole diameter between 47 and 49 feet bgs. Given the limited interval logged, the water quality cannot be inferred from the response of the SP and single-point resistivity logs.

The overburden stratigraphy recorded on the drilling log correlates fairly well with the natural gamma response. The natural gamma log shows a relatively higher gamma count in the sands above the bedrock interface at 14 feet bgs, which is interpreted as increasing clay content in the sands immediately above the bedrock or as weathered bedrock. The natural gamma log of the bedrock interval does not show any distinctly different lithologic intervals, although relatively higher gamma counts were recorded between 28 and 30 feet bgs.

The well construction consists of 4-inch PVC casing from the surface to a depth of 20.5 feet bgs. The camera log extended from the ground surface to 49 feet bgs, terminating in a pool of sediment. The video camera log showed turbid water at the bottom of the borehole from 41 feet bgs to 49 feet bgs. Water quality results for this well show an alkaline pH of 10 to 11 and supports the conclusion that the bottom 9 feet of the well is filled with suspended grout. This well was re-developed to remove the suspended material and the re-development is discussed in Section 2.5.5.3.

BR-1. This well is completed to a depth of 48 feet bgs in a cataclastic to normal gabbro-diorite. Structurally, this well is also interpreted to lie near a fault contact between the normal gabbro-diorite and the cataclastic gabbro-diorite. No significant fracture zones were noted on the camera or caliper logs. Closed to open fractures were observed on the camera log at 23 feet bgs, 33 feet bgs, and 45 feet bgs. The fracture at 45 feet bgs also corresponded with a slight inflection in the differential temperature log, although the low yield of this well (about 1 gpm) suggests that this fracture is a not significant water bearing zone. The largest inflection in the differential temperature log was also noted at 20 feet bgs, immediately below the casing. This inflection in the temperature log corresponds with a very low resistivity and may indicate that water is entering the well at this depth. Corehole alignment bends at 24 feet bgs, 26 feet bgs, and 28 feet bgs may also indicate the presence of fractures or may be due to lithologic changes. Water quality cannot be inferred from the response of the SP and single-point resistivity logs.

The natural gamma log shows an interval with relatively lower gamma counts between 20 and 34 feet bgs. A drilling log is not available for this well, however the change in gamma counts at 20 feet bgs may represent the bedrock interface.

The well construction consists of 5-inch PVC casing from the surface to a depth of 21 feet bgs. The casing/grout depth on the camera log (21 feet bgs) is about the same as the depth shown on the caliper log. The camera video shows that the corehole extends through grout placed inside the casing to a depth of 21 feet bgs. The grout appears pitted at 6 feet bgs and 11 feet bgs. The coring process eroded the upper portion of the grout creating an oblong corehole at the top which funnels downward until the bedrock. At 19 feet bgs, a gray streak is present on one side of the

grout and thickens toward the bottom of the grout into a gray and red mass. As indicated above, these observations suggest that overburden groundwater is entering the borehole at the bottom of the casing.

2.1.2.3 Summary of Geophysical Results

The geophysical logging of eight open borehole bedrock wells was conducted to provide an understanding of the water-bearing characteristics of the bedrock. The camera and caliper logs show that the bedrock contains many in-filled fractures and relatively few open fractures. Typically two to three open fractures were observed in each well, although the low yield of most of the wells (about 1 gpm) suggests that these fractures are not significant water bearing zones. Most of the fractures occurred in sets or zones which often corresponded with an increase in clay content or the boundary between different lithologic units as interpreted from the natural gamma and resistivity logs.

The natural gamma logs of overburden were in general agreement with drilling logs and were useful in further identifying sand-rich and clay-rich beds which may act as preferential flow pathways or confining units, respectively. Generally, relatively higher gamma counts were recorded adjacent to the overburden and bedrock interface and may indicate the presence of clay-rich glacial deposits, regolith, or a mixture of weathered bedrock fragments and glacial material.

The natural gamma logs of bedrock intervals also showed potential lithologic variations that are not identified on the drilling logs. In general, the granite and mylonite bedrock intervals show relatively higher gamma counts per second (30 to 60 CPS and 10 to 20 CPS, respectively) compared to the quartzite, gneiss, and gabbro intervals (0 to 10 CPS).

The camera and caliper logs were also used to confirm well construction. The caliper logs generally reflect the decrease in hole diameter immediately below the casing, but in some instances show an increase in hole diameter due to an offset between the bottom of the casing and the top of the borehole. A large void observed below the casing in GW-62BR corresponded with a temperature log inflection and low resistivity which may indicate the presence of a water-bearing fracture zone immediately below the casing or a poor casing seal. The stained casing below the water level in GW-80BR and the scaling at the bottom of the casing may also indicate a poor casing seal. The results of geophysical logging at BR-1 may also indicate a poor casing seal. The gray, suspended material observed below 81 feet bgs in the camera log of well GW-65BR and the white, cloudy water below 41 feet bgs at GW-81BR provides an explanation for high pH values in these wells and supports the conclusion that the bottom portion of these wells are affected by grout intrusion. As discussed in Section 2.5.5.3, these wells have been redeveloped.

The resistance logs were used in conjunction with the natural gamma logs to identify clay-rich intervals which generally had a low resistivity. The resistance logs were also useful in identifying small fractures by virtue of the fact that a void filled with fresh (or saline) water results in a decrease in resistance on the log, however some of these features were related to wash-outs due to the drilling process. The resistance log was also used in conjunction with the spontaneous

potential log to tentatively identify intervals with fresh water (low dissolved solids) versus saline water (high dissolved solids) conditions. Intervals with opposite resistivity and SP inflections observed in GW-62BR, GW-62BRD, and the interval of GW-65BR below 120 feet bgs may indicate saline conditions or high dissolved solids in the groundwater. This interpretation generally agreed with water quality results for these wells.

2.1.3 Borings to Define Bedrock Surface

The shape of the bedrock surface beneath the property and the Maple Meadow Brook wetlands is of great importance due to its influence on the migration of the dense layer. Based on a review of the bedrock surface, as interpreted from the seismic reflection data and boreholes for monitoring wells, additional borings were proposed in two areas where interpolated data indicated potentially important bedrock features (Smith, 1996b).

In order to gain more information about the bedrock surface, aquifer materials, and groundwater quality, three borings were drilled on residential properties west of Main Street and groundwater samples were taken at the top of the bedrock surface. In addition, three borings were drilled along the access road to the Chestnut Street pump station.

2.1.3.1 Methods

Prior to commencement of drilling, property access was received from the appropriate property owners along Main Street. The Wilmington Water Department granted access for field activities prior to drilling at the Chestnut Street pump station.

Drilling was performed at all locations using a Rotasonic drill rig manufactured and operated by Boart Longyear of Rochester, Indiana. Rotasonic drilling was selected because it is an efficient method which advances a borehole quickly with minimal generation of drill cuttings and fluids. It allows collection of continuous samples for better understanding of the aquifer materials and can core into rock for confirmation of the bedrock surface. The three boring locations west of Main Street (MB-1, MB-2, and MB-3) were drilled in October through November 1996. The three boring locations along Chestnut Street (CB-1, CB-2, and CB-3) were drilled on November 18 through November 25, 1996. Locations of the borings are shown on Figure 2-1.

During drilling of the boreholes along Main Street, the aquifer materials and conditions were recorded and are presented on logs provided in Appendix A. When rock was encountered, a groundwater sample was collected as described in Section 2.5.3, and drilling continued at least five feet into rock in order confirm the bedrock surface and prevent misidentification of boulders as bedrock. At one of the Main Street locations, MB-2, the Rotasonic drill rig and coring rig were unable to recover bedrock core samples to the desired depth for confirmation of bedrock. Upon completion, the boreholes were tremie-grouted to the surface with a cement/bentonite grout.

The same drilling methods were used for the boreholes along the access road to the Chestnut Street pump station except groundwater samples were not collected, as there is no reason to believe the dense layer is present in this area.

All drill cuttings and water generated during drilling were placed in drums and stored on the Olin property for appropriate classification and disposal. The drilling equipment was steam-cleaned between each of the Main Street locations to prevent cross-contamination. The drilling equipment was steam cleaned before the start of the Chestnut Street borings, but was not steam cleaned between Chestnut Street locations under the assumption that the aquifer around the pump station is relatively clean. After completion of the borehole, a temporary stake was placed in the ground for later surveying.

2.1.3.2 Results

Approximate bedrock depths obtained from the Main Street and Chestnut Street borings are summarized below and were used for the interpretation of the bedrock surface.

Boring Location	Ground levation (ft MSL)	Depth to Bedrock (ft bgs)	Bedrock Elevation (ft MSL)
MB-1	95.3	~ 65	~30.3
MB-2	96.6	~ 55	~41.6
MB-3	94.9	~ 35	~59.9
CB-1	94.4	~ 47	~47.4
CB-2	88.8	~ 44	~44.8
CB-3	82.7	~67	~15.7

Groundwater analytical data obtained from the Main Street borings indicated the dense layer is not present at any of these locations. The analytical data is presented in more detail in Section 4.1.

2.1.4 Seismic Reflection Survey

2.1.4.1 Methods and Overview

A seismic reflection survey uses the velocity of acoustic waves traveling through the subsurface to determine the depth to a reflecting horizon (reflector) and the thickness of the material overlying the reflector. Reflections are caused by abrupt changes in velocity of the acoustic wave at the interface between differing materials. The seismic velocities measured by this technique can be used to calculate the mechanical properties of the subsurface materials, to determine stratigraphic correlation, and to determine the depth of geologic layers such as the bedrock surface.

An acoustic pulse (or "signal") is created in the earth by initiating a seismic source ("shot"). Energy travels from this source into the earth until a reflecting horizon is encountered. Some of the energy from the signal is reflected back to the surface where a series of moving-coil

electromagnetic geophones (geophones) converts the earth motion into electrical signals which are simultaneously recorded by a data logger. The operator can amplify and filter the seismic signals to minimize background interference (extraneous vibrations).

Interpretations are made from travel time curves showing the measurement of the time required for a compressional seismic wave to travel from the shot point to each of a group of geophones. The geophones are located at known intervals (spreads) along the ground surface. The source, geophone type, and spread length used in a reflection survey are determined based on the suspected depth to the reflecting horizon and other field conditions.

2.1.4.2 Study Definition

The seismic reflection survey was performed by Raypath Inc. (Raypath) of Cheltenham, Pennsylvania under subcontract to Smith. Raypath summarized their results of the seismic study in the report included in Appendix D. The locations of the survey lines and bedrock elevations at seismic stations are shown on Plate 2-1. Seismic reflection data were collected during four phases over a period of one and a half years. Down hole velocity surveys were also conducted on 16 wells in order to determine the average velocity gradient to the top of bedrock. The locations and dates of the seismic and velocity survey phases are summarized below.

Area of Interest	Date of Survey Event	No. of lines	Total Length of Lines
Western Bedrock Valley	November 16-December 7,1994	5	4,350'
	July 24-August 6 1995	6	4,450'
	March 10-20, 1996	4	3,150'
Southwest of the property	January 13-24, 1995	2	3,000'
Eastern Bedrock Valley	January 13-24, 1995	4	1,800'
Sitewide - Velocity Surveys	November 23-26, 1996	16 wells	NA

The seismic survey lines are numbered with even numbers for lines running approximately north - south (i.e. BCM-94-6). Lines running approximately east - west are identified with odd numbers (i.e. BCM-94-3). During the course of the seismic study, the "BCM" prefix changed to "SE" for seismic lines performed in the study. Due to swampy conditions, a portion of line BCM-95-10, along the eastern property line of the Olin property, was performed separately between March 20 and 24, 1995.

2.1.4.3 Results

The seismic reflection data are presented as profiles for each of the seismic survey lines. As shown in a sample profile provided on Figure 2-4, the location of the seismic reflector is recorded in units of time. Detailed profiles and discussion about methods and results are included in Raypath's report (Appendix D). The conversion of units of time to actual depth to bedrock is calculated by multiplying the units of time by a velocity value. The velocity value was determined by using data about aquifer materials and depths to bedrock at monitoring wells near the property;

therefore, the interpretation of the seismic profiles is based upon a small sample of aquifer materials. It is assumed that the entire aquifer is similar to the materials logged at the monitoring wells. The seismic reflection profiles provided in Appendix D indicate the presence of intermediate reflectors between the bedrock surface and ground surface. Based on a comparison with drilling logs, the vertical position and character of these intermediate reflectors generally correspond to either gravel layers or till above the bedrock. However, the seismic and velocity surveys were designed to determine the depth of the bedrock surface and are not suitable for determining the depth of these intermediate reflectors.

During the drilling program in the Western Bedrock Valley, bedrock surfaces were encountered at depths which were shallower than determined by the seismic data. This was caused by slower velocities in the thick sequence of fine sediments at the surface. Additional velocity surveys were performed in the wells installed during the Western Bedrock Valley Investigation. The additional velocity data were applied to the existing seismic profile to correct for the change in velocity in the fine sediments.

The data obtained from borings drilled for well installation and soil sampling, the mapping of outcrops, and the seismic reflection surveys conducted in the study area have yielded a detailed picture of the morphology of the bedrock surface. The bedrock surface morphology has been shaped by lithologic differences in the bedrock, structural features such as ancient faults, and millions of years of erosion, and then glaciation. The bedrock surface is highly variable and has a significant amount of relief. A contour map of the bedrock surface which has been prepared using a collation of all data, is presented as Plate 2-1. The cross-sections shown on Figures 2-5 through 2-12 can be used to help interpret the bedrock contour map. These cross-sections are slices through overburden and the bedrock, and provide a vertical picture of the shape of the bedrock surface. The location of each of the cross-sections is shown on Plate 2-1. The major features are summarized below and the detailed description of the major bedrock topographic features is discussed in the following sections.

The primary feature of the bedrock surface is the Western Bedrock Valley that extends from the property towards the west and reaches depths of over 120 feet beneath the wetland surrounding Maple Meadow Brook. Based on the current data, the Western Bedrock Valley terminates slightly north of where the former Middlesex Canal crosses the wetland. An Eastern Bedrock Valley was identified in the CSA, and recent data has shown that this valley does not extend significantly beyond the eastern property boundary. The extent of this valley is depicted on the geologic cross section A-A' on Figure 2-5. The CSA also identified a possible Southwestern Bedrock Valley located along the southwestern property line and extending towards Breed Avenue. A seismic line through this area and the installation of additional wells (the GW-81 pair) has shown that bedrock elevations along the southwest property line are relatively high. Therefore, there is no valley in this area that could connect the Western Bedrock Valley with the bedrock lows on Breed Avenue. The extent of this valley is depicted on the geologic cross section B-B' on Figure 2-6.

Between these valleys are areas where bedrock is at or relatively near the ground surface. These areas are commonly represented by hills. Specific areas underlain by shallow bedrock include the large area of high relief beneath Cook Avenue and the south end of Main Street, the southeastern corner of the Olin property, the hill just south of Town Park, and localized areas of bedrock outcrop along the railroad tracks and Eames Street. For example, Figure 2-7 is a geologic cross section (C-C') which runs across the Western Bedrock Valley and shows that the bedrock rises towards the ground surface and outcrops at the hill crossed by Cook Avenue.

2.1.4.3.1 Western Bedrock Valley

Seismic lines used to delineate the Western Bedrock Valley include the following:

Nov - Dec 1994	July - Aug 1995	March 1996
BCM-94-1	SE-95-3X	SE-96-9
BCM-94-2	SE-95-14	SE-96-22
BCM-94-3	SE-95-16 & SE-95-16X	SE-96-24
BCM-94-4	SE-95-18	SE-96-2X
BCM-94-6	SE-95-20	

Seismic lines SE-96-9, SE-96-22, and SE-96-24 run parallel to Eames Street, Main Street, and Jewel Drive, respectively. Data was not collected on a section of line SE-96-22 extending across the Altron driveway since the driveway could not be blocked. All three lines display an irregular bedrock surface with highly variable bedrock elevations. Line SE-96-9 has an elevation range from approximately 9 to 55 feet above mean sea level (MSL). Bedrock elevation changes along lines SE-96-22 and SE-96-24 are more abrupt with elevation changes of approximately 63 feet and 43 feet, respectively. An important feature to note on line SE-96-22 is a deep hole along the northwestern end of the line with bedrock elevations down to approximately 26 feet MSL.

Seismic lines SE-95-20 and SE-95-18 are located in the wetland west of the intersection of Main Street and Eames Street. Bedrock elevations along these lines range from 54 feet MSL outside the valley to approximately -15 feet MSL in the center of the Western Bedrock Valley.

The profiles for seismic lines SE-95-16, SE-95-16X, BCM-94-3, SE-95-3X, and BCM-94-4 indicate a very irregular bedrock surface with a significant amount of relief. Seismic line SE-95-16X is a continuum of line SE-95-16; therefore for discussion purposes, the two lines will be treated as one. The bedrock surface profile at line SE-95-16 is the most irregular of all the seismic lines. Bedrock elevations range from 53 feet MSL at a bedrock high just southwest of the bedrock valley to -57 feet MSL in the deepest section of the valley; there is approximately 110 feet of relief on this line.

Further northwest in the valley, the bedrock surface remains irregular. The bedrock surface on line BCM-94-3 exhibits 74 feet of relief with two major depressions separated by a bedrock high. Bedrock elevations increase to the west of BCM-94-3 as evidenced by the profile along line SE-95-3X. Elevations rise from approximately 25 feet MSL at the eastern end of line SE-95-3X to

elevations ranging between 40 to 50 feet MSL along the western end of line SE-95-3X. Along line BCM-94-4, the bedrock surface exhibits a shallow depression within a broader low area that has a relatively irregular bottom. The total relief on line BCM-94-4 is approximately 59 feet.

The bedrock surface at lines BCM-94-1 and BCM-94-2 are much less irregular than the previous ten lines. Both lines BCM-94-1 and BCM-94-2 exhibit a relatively smooth bedrock surface dipping gently into the bedrock valley. SE-96-2X is an extension line for BCM-94-2 and shows an important feature: the bedrock elevations rise as one moves west along the line. This provides a closure point for the western edge of the valley.

Lines SE-95-14 and BCM-94-6 are located at the northern end of the valley. Bedrock elevations show an abrupt increase at SE-95-14. A bedrock high exists just northeast of the Butters Row Pumping Station with bedrock elevations greater that 50 feet MSL. The eastern section of line SE-95-14 exhibits a depression with elevations of approximately 17 feet MSL. BCM-94-6 exhibits a relatively smooth bedrock surface with a slight high at the center of the line. The northern end of the bedrock valley occurs on or near seismic line SE-95-14.

Figure 2-7 is a geologic cross section which runs across the Western Bedrock Valley. The location of the profile is shown on Plate 2-1. The bedrock elevation is more than 20 feet lower at C-C' compared to the cross section in the Eastern Bedrock Valley, B-B' (Figure 2-6). The dense layer is present in wells GW-44D and GW-45D. Again to the north and south, the bedrock surface rises towards the ground surface and encloses the dense layer within the Western Bedrock Valley. To the far south, Cook Avenue lies on a hill where the bedrock forms the ground surface.

2.1.4.3.2 Eastern Bedrock Valley

Seismic reflection lines BCM-95-10, BCM-95-10X, BCM-95-12, and BCM-95-7 are located in the area of the Eastern Bedrock Valley. Line BCM-95-10 was divided into two segments due to the presence of a bedrock high and outcrops at the center of the line. The bedrock surface at line BCM-95-10 is relatively smooth with a gentle dip to the south. The total elevation drop along this line, which originates at an outcrop at the northern end, is approximately 10 feet over a distance of 600 feet. Line BCM-95-10X slopes to the north, away from the bedrock high at the southern end. The lowest point occurs at the center of the line approximately coincident with the location of well GW-50D. From that point, the bedrock surface slopes upward to the north. The total elevation difference along this line is 41 feet.

The bedrock surface at line BCM-95-12 is relatively flat with a slight depression at the center of the line. The total relief along this line is approximately 10 feet. The elevation of the bedrock surface in the low is approximately 11 feet higher than the lowest point on line BCM-95-10X. Line BCM-95-7, located on Presidential Way, exhibits a broad gentle low in the bedrock surface which is at a slightly lower elevation than the low point observed on line BCM-95-12. Based on this data, it appears that the Eastern Bedrock Valley ends in the area between lines BCM-95-10X and BCM-95-12.

Figure 2-5 is a geologic cross section which runs down the center of the Eastern Bedrock Valley from the Olin property line to the east. This shows a depression in the bedrock surface beneath GW-50 and the railroad tracks. To the east of this depression, the bedrock surface rises and represents the end of the valley.

2.1.4.3.3 Bedrock Conditions Southwest of Property

Seismic reflection lines BCM-95-5 and BCM-95-8 are located in the area which was tentatively identified as the Southwestern Bedrock Valley in the CSA. The bedrock in this area exhibits a relatively flat, shallow surface with only minor relief. The bedrock surface at line BCM-95-5 slopes slightly to the west with a total elevation difference of 10 feet over a length of 700 feet. The bedrock surface at line BCM-95-8 slopes gently to the south. There is a total elevation difference of 28 feet across the 1,200 foot long line, with most of the elevation drop occurring at the southern end. This indicates the occurrence of a relatively broad bedrock saddle between the Western Bedrock Valley and this area to the southwest. The data also indicate that the bedrock low along Breed Avenue is relatively flat and that well GW-75D is located in the axis of the valley. Based on this data, there is no Southwestern Bedrock Valley, but only a shallow depression southwest of the broad saddle at the southern end of Jewel Drive.

Figure 2-6 is a geologic cross section which runs along Olin's western property line and crosses the Western Bedrock Valley. This valley is shown as a depression beneath wells GW-36, GW-37, and GW-38. To the north and south of these wells, the bedrock rises forming the sides of the Western Bedrock Valley. It is clear from this cross-section that the bedrock depressions to the far south (GW-40) are not connected to the Western Bedrock Valley.

Cross-section D-D' presented on Figure 2-8, crosses the middle portion of the Western Bedrock Valley extending from the GW-61 well cluster to the supply well Chestnut Street 1A/2. To the east and west on this section, the bedrock rises towards the ground surface defining the sides of the Western Bedrock Valley. This section shows two separate lows in the valley. Two additional wells, GW-84 and GW-85, were installed in the bottom of each low to determine if the dense layer was present, as potentially indicated by the terrain conductivity anomalies. These anomalies are discussed in more detail in Section 2.6.2 of this report. Bedrock was encountered at depths of 100 feet below ground surface (bgs) at GW-84 and 85 feet bgs at GW-85. The water supply wells at the Chestnut Street pumping station are clearly distant from the center of the Western Bedrock Valley and screened more than 50 feet above the base of the valley.

Sections E-E' and F-F' presented on Figure 2-9 and Figure 2-10 cross the Western Bedrock Valley further down slope, intersecting Butters Row wells No. 1 and No. 2, respectively. Both of these sections show the shape of the Western Bedrock Valley and its limits to the east and west. Both supply wells are located outside the center of the valley and are screened well above the base of the aquifer. Two new wells, GW-86 and GW-87, were installed in the bottom of each of the two bedrock lows in this area to determine if the dense layer was present. Bedrock was encountered at depths of 85 feet bgs at GW-86 and 104 feet bgs at GW-87.

Section G-G', presented on Figure 2-11, runs down the center of the Western Bedrock Valley past the historical Middlesex Canal, and shows that the Western Bedrock Valley comes to an end as the bedrock rises towards the ground surface north of the canal.

Section H-H', presented on Figure 2-12, runs down the center of the Western Bedrock Valley extending from the property to the eastern edge of Maple Meadow Brook wetland. This section shows that a saddle is present in the Western Bedrock Valley in the area west of Main Street.

2.2 HYDROGEOLOGY

The hydrogeology of the site and surrounding area were discussed in detail in the CSA. However, additional data were needed to provide an understanding of the potential migration of the dense layer. Evaluation of the effect of variable density on the interpretation of hydraulic heads was also necessary. Therefore, additional data was collected during the Supplemental Phase II work that would aid in providing a greater understanding of the hydrogeologic system.

Specific tasks performed in this phase included updating potentiometric data, evaluating the aquifer permeability through re-analyzing old pumping test data and performing slug tests, and measuring groundwater flow directions and velocities directly in wells. The monitoring wells located throughout the site and surrounding area are shown on Plate 2-2. The construction details of these wells are provided in Table 2-2, which also includes the depth to bedrock encountered during drilling.

2.2.1 Potentiometric Data

Monitoring of groundwater and surface water levels in the vicinity of the South Ditch weir has been performed on a regular basis since installation of the weir. This is discussed in detail in Section 2.4. Since the CSA, two rounds of site-wide water levels have been collected, one in October 1995 during seasonal low water conditions, and one in April 1996 during high water conditions. The data, which were used to calculate hydraulic heads, are shown in Tables 2-3 and 2-4. The summer of 1995 was extremely dry in eastern Massachusetts with drought restrictions imposed on water usage. By October 1995, there had been some rainfall to replenish groundwater supplies, but it is likely that water levels were significantly lower than typical for that time of year. In April 1996, groundwater levels were probably at or near the annual high.

The water level data were used to generate groundwater contour maps, one each for the shallow and deep portions of the aquifer for each monitoring date. These maps are shown on Plates 2-3 through 2-6. The division of the sand and gravel aquifer into shallow and deep portions was used in the CSA. In general, shallow wells are screened at or near the water table; deep wells are screened at or near the bedrock. The few intermediate-depth wells are present only where there is a great enough saturated thickness for wells to be located between the shallow and deep portions.

2.2.1.1 Shallow Groundwater

Groundwater contour maps for the water table are shown on Plates 2-3 and 2-5. Shallow groundwater flow patterns are different than shown in the previous CSA due to the presence of the weir in the South Ditch. The weir has created localized mounding in this area with a component of flow to the northwest. East of this area, shallow groundwater flow is generally towards the east in the direction of the ditch systems. In the northern portion of the plant is a groundwater mound that is consistent with previous data. This mound creates a divide where part of the water flows to the east and part to the west. To the west of the Olin property is a large area with a very shallow hydraulic gradient, again consistent with previously collected data. Further west, shallow groundwater flow is towards the Maple Meadow Brook wetlands and the Wilmington water supply wells.

In October 1995, in the vicinity of the municipal supply wells and the Maple Meadow Brook wetlands, groundwater levels were as much as 10 feet lower than previously observed. This was most likely due to the drought conditions in the summer and fall that led to very high use of this wellfield. It appears that groundwater levels had not significantly recovered by the October monitoring date. By April 1996, groundwater levels in this area had recovered to levels similar to those previously observed.

There is no real seasonal difference in shallow groundwater flow directions across the site except in the vicinity of the weir installed across the South Ditch on the Olin property. During dry periods, when groundwater levels are low (October 1995), the surface water held back by the weir has a mounding effect causing groundwater to flow away from the ditches. During wet periods (April 1996) the groundwater levels are higher than surface water behind the weir, and groundwater flow is towards the ditches.

2.2.1.2 Deep Groundwater

Water levels measured from the deep wells are presented in Tables 2-3 and 2-4. However, for some wells, calculation of hydraulic heads is affected by the greater density of the dense layer. Where a well is screened only in the dense layer, the measured groundwater elevation is less than the actual hydraulic head in the aquifer. This effect is due to the fact that it takes a shorter column of water at greater density to equal the head of a column of fresh water. This relationship is shown schematically on Figure 2-13. Therefore, for wells screened in the dense layer, the measured hydraulic heads must be corrected for the effect of density. The specific methodology used for the correction is detailed in Appendix E. These corrections are presented in Table 2-5 which show those wells which intersect the dense layer as well as the approximate corrected hydraulic head value for each well. The correction factor was not needed for wells with the interface near or within the screened interval. Several wells have correction factors in the range of approximately 0.5 feet. The greatest correction factor was 1.6 feet in GW-44D due to the relatively large thickness of the dense layer present in this area. As a test of the sensitivity of this

correction, Table 2-5 also shows the maximum possible correction for each well, assuming the well was filled entirely with the dense layer. The results of this sensitivity analysis indicate that the maximum possible correction ranged from 0.6 to 4.0 feet.

Groundwater contour maps of the hydraulic heads in the deep wells are shown on Plates 2-4 and 2-6. These contours are based on data that include the correction for the effect of density in the deep wells, where applicable. The contours can be used to interpret hydraulic gradients in the absence of the dense layer. General hydraulic gradients for groundwater in the deep portion of the water table aquifer are similar to the shallow portion of the water table aquifer. In the immediate vicinity of the Olin property, the contours show gradients generally eastward and towards the ditch system. To the west of the property, gradients are westward toward the Maple Meadow Brook wetlands and the municipal supply wells. The data from October 1995 show hydraulic heads approximately 10 feet lower than normal in the vicinity of the supply wells. By April 1996, the heads had returned to more typical levels.

Seasonally, the deep groundwater shows the same general groundwater flow patterns as the shallow groundwater. There is no real difference between dry and wet periods except in the vicinity of the weir, and in this area, the hydraulic gradients generally mimic those observed in the shallow groundwater.

2.2.1.3 Vertical Flow Components

The water level data can also be examined to evaluate any vertical components of flow between the shallowest and deepest portions of the aquifer. In general, none of the vertical gradients are particularly strong, with head differences between shallow and deep wells typically on the order of tenths of a foot. For both the October 1995 and April/May 1996 monitoring events, upward vertical gradients generally occur immediately adjacent to surface water bodies (Maple Meadow Brook and the ditches). Downward vertical gradients seem to dominate the remainder of the area although there are a number of wells with no significant head difference between the shallow and deep well. The well pairs immediately adjacent to the water supply wells cannot be evaluated in this way, as the supply wells are pumping from a depth intermediate between the shallow and deep wells. Vertical gradients in the shallow portions of the aquifer would be expected to be downward towards the pumping zone; gradients in the deep portion would be expected to be upward. In addition, vertical components of flow from well pairs that intersect the dense layer cannot be evaluated as the density correction does not allow enough accuracy.

2.2.2 Pumping Test Re-evaluation

Aquifer pumping tests are the best way to quantify large-scale aquifer characteristics such as hydraulic conductivity, transmissivity, and storage. A previous Aquifer Protection Study performed for the Town of Wilmington (IEP, 1990) indicated that several pumping tests had been performed during the development of the wellfield located around Maple Meadow Brook.

Therefore, instead of performing additional pumping tests, it was decided to re-examine the historic data and re-analyze the data using more appropriate methods, for example, those that include the effects of partial penetration.

2.2.2.1 Overview

The wellfield surrounding Maple Meadow Brook currently includes the following five production wells:

Production Well	Date of Installation (IEP, 1990)
Butters Row No. 1	1971
Butters Row No. 2	1979
Town Park	1965
Chestnut Street No. 1	1960
Chestnut Street No. 2 (1A)	1991

The locations of these pumping wells are shown on Plate 1-1. As part of development of this wellfield, aquifer pumping tests were performed at each of these three areas. An additional pumping test was performed at Chestnut Street in 1991 when the No. 2 well was considered. The following table summarizes the pumping tests that were performed and the respective references.

Area	Date of Test	Reference
Butters Row	1964	W&H, 1964
Town Park	1962	W&H, 1962
Chestnut Street	1960	W&H, 1960
Chestnut Street	1961	W&H, 1961
Chestnut Street	1991	Dufresne-Henry, 1992

All information concerning these tests was obtained from the Town of Wilmington Water Department. Because of the age of the studies and the technological limitations of the time, the data sets are far from ideal. Only charts of changes in water level are available, rather than the actual drawdown data. On these charts, water levels at several wells were frequently plotted on the same line, making it unclear at which well the data was actually obtained. In addition, it is often unclear from these reports exactly where surface wells were located. The reports concerning the two Chestnut Street testing programs performed in the early 1960s were not available from the Water Department. Because of the more recent testing in 1991, no further search for the earlier data was conducted.

To help perform the data re-analysis, Smith used the computer program known as the Aquifer Test Solver (AQTESOLV™). AQTESOLV™ (Geraghty & Miller, Inc, 1995) is a sophisticated computer program that generates type-curves for many different types of aquifers and pumping conditions. The type-curves can be visually matched to the data by the user, or the program can be used to determine a least-squares, best-fit match.

The analysis performed by Smith was based on the following assumptions for each test:

- The aquifer was assumed to be unconfined.
- The aquifer thickness was based on the depth to bedrock measurements determined at the monitoring wells installed by Olin in the immediate vicinity of each pumping test.
- Groundwater flow was steady.
- The diameter of the pumping well was very small so that storage in the well could be neglected.
- The aquifer was homogeneous and had a uniform thickness.
- The potentiometric surface was initially horizontal.
- The effects of boundaries were considered, but were shown not to be important; therefore, the aquifer was assumed to be of infinite areal extent.
- The pumping and observation wells partially penetrated the aquifer.

With the above assumptions, Smith utilized the Neuman type-curves (Neuman, 1974), which consider partial penetration in unconfined aquifers and allow for delayed yield.

The conditions of each of the individual pumping tests and the results of the data re-analysis are discussed below.

2.2.2.2 Town Park

Whitman & Howard (W&H) conducted an aquifer pumping test of monitoring well 1 located near the current Town Park well in February 1962. The data collected during the pumping test was presented in the "Report on Pumping Test Town Park Area" (W&H, 1962). Water levels were monitored in the pumping well and several observations wells from February 5 through 19, 1962. Pumping took place at well 1 from approximately 9:00 a.m. on February 5 through 2:30 p.m. February 19, 1962. A nearly constant rate of about 185 gpm was used throughout the pumping test.

Figure 2-14 is a sketch map showing the pumping well and some of the observation wells used during the test. For additional monitoring wells not shown, the water level data were considered unusable due to factors such as minimal drawdown or uncertain location; these wells were omitted from the re-analysis. The remaining observation wells were denoted wells 1 (the pumping well), 1A, 1B, 2, 3, 8, 9, and 13. Information about each well, including the open interval, the distance of each well from the pumping well, and the diameter of the well, is presented in Appendix F. Well logs for the pumping and observation wells can be found in the W&H report (1962). The depth to water was measured in each of these wells during the pumping test. However, water level data for wells 2, 3, and 13 are shown as a single line on a chart. It is virtually impossible to have one set of data for three separate observation wells which are situated various distances from the pumping well. Therefore, the data set was assumed to be for a single well located at a median distance of approximately 200 feet from the pumping well. Data from the remaining wells were more straightforward. The water level data collected throughout the test were obtained from the W&H report (1962) and are summarized in Appendix F.

The results of the re-analysis of this data are shown in Table 2-6, including values for transmissivity, storage, and hydraulic conductivity. The data and match-curves with the AQTESOLV™ output files are included in Appendix F. Calculated transmissivities in the vicinity of Town Park range from 2,400 to 8,600 square feet per day (ft^2/day) with a geometric mean of 4,700 ft^2/day. Using the aquifer thickness of 53 feet gives an average hydraulic conductivity of 90 feet/day. This is lower than the range of values (112 to 271 feet/day) reported by IEP (1990). Values of specific yield ranged from 0.09 to 0.87 with a value of approximately 0.10 being most representative.

2.2.2.3 Butters Row

W&H conducted an aquifer pumping test in the vicinity of the current Butters Row wells in January 1964. The data collected during the pumping test were presented in the "Report on 1963 through 1964 Test Well Investigations" (W&H, 1964). During this test, pumping occurred simultaneously at two wells, denoted wells 2 and 3. Water levels were monitored in the pumping wells and several observations wells from January 8 through 22, 1964. Pumping took place at well 2 from approximately 11:30 a.m. on January 8 through 11:30 a.m. on January 17, 1964. Pumping took place at well 3 from approximately 10:30 a.m. on January 8 through 11:30 a.m. on January 17, 1964. A constant pumping rate of 520 gpm from well 2 was used throughout the pumping test. Well 3 had an initial pumping rate of 510 gpm which was subsequently decreased to a constant rate of 475 gpm.

Figure 2-15 is a sketch map showing the layout of the pumping and observation wells used during the test. Depth to water measurements were collected from five observation wells including well numbers 17, 19, 19A, 19B, and 19C. Locations of observation wells 19B and 19C are uncertain; therefore data from these two wells were not used for the pumping test re-evaluation. The water level data collected throughout the test were obtained from the W&H report (1964) and are

presented in Appendix F. Information about each well including the open interval of the well, the distance of each well from the pumping well, and the diameter of the well is also presented in the Appendix. Well logs for the pumping and observation wells can be found in the W&H report (1964).

During the re-analysis of the Butters Row data, it was not possible to achieve realistic matches with any reasonable type-curves, possibly due to the uncertainty in well locations, the interaction of the two pumping wells, or the presence of the large wetland nearby. Image-well theory was used to try to capture the effect of the wetland, but this was not successful. Data about well locations and screened intervals were not clearly defined in the original report and were contradicted by other references. Since there are so many undefined variables which could not be satisfactorily resolved, Butters Row results were discarded. For reference, IEP (1990) cites a range of hydraulic conductivities of 86 to 371 feet/day.

2.2.2.4 Chestnut Street

In 1991, two aquifer pumping tests, a 24-hour single well pumping test and 48-hour two-well pumping test, were performed at the Chestnut Street location. Pumping tests were performed by Dufresne-Henry, Inc. (DH) and reported in a letter report to the MADEP (1992).

Well identifications and locations at Chestnut Street were conflicting in the 1992 DH report text, figures, and well logs; therefore, several assumptions were made and several of the wells were renamed. In the report text and the well log, the most recently installed pumping well was referred to as well 1A; however, in the site sketch, the well is referred to as New No. 2 well. For purposes of the pumping test re-analysis, the most recently installed well will remain named well 1A. The former pumping well is referred to as Chestnut Street No. 1 throughout the report. In the discussion presented below, the Chestnut Street No. 1 well is referred to as well 1C. In the site sketch map, there are several observation wells shown: wells 1-obs, 1A, 2-obs, 3-obs, B-obs, C-obs, and D-obs. All observation wells will be referred to by number without the "-obs" suffix. Also, observation well 1A is not the same as pumping well 1A; therefore, observation well 1A is referred to as well 1-o, in order to denote the well separately from the pumping well 1A. Finally, since the site sketch map was not drawn to scale, Smith measured the distances in the field between the remaining observation wells and the two pumping wells. As a result of the measurements taken, it has been assumed that observation well 5 and B are actually the same well.

In summary, the two pumping wells will be denoted well 1A and 1C. The seven observation wells are referred to as well 1, 1-o, 2, 3, 4/C, and 5/B. The location of the wells as named for the pumping test re-analysis are shown on Figure 2-16. The water level data collected throughout the two tests were obtained from the DH report (1992) and are summarized in Appendix F. Information about each well, including the open interval of the well, the distance of each well from the pumping well, and the diameter of the well, is presented in Appendix F. Well logs for the pumping and observation wells can be found in the DH report (1992).

24-Hour Constant Rate Pumping Test

DH conducted an aquifer pumping test of pumping well 1A in March 1991. Water levels were monitored in the pumping well and several observations wells from March 13 through 14, 1991. Pumping took place at well 1A from approximately 2:30 p.m. on March 13 through 2:30 p.m. on March 14, 1991. A constant rate of 75 gpm was used throughout the pumping test.

During the 24-hour pumping test, depth to water measurements were taken from the pumping well, 1A, and the observation wells; 2, 3, 4/C, and 5/B. In order to monitor recovery, depth to water data was collected from the wells for an additional two hours after pumping had ceased.

48-Hour Constant Rate Pumping Test

DH conducted a two-well aquifer pumping test in August 1991. Water levels were monitored in the pumping wells and several observations wells from August 28 through 30, 1991. Pumping took place at wells 1A and 1C from approximately 12:00 p.m. on August 28 through 12:00 p.m. on August 30, 1991. A constant pumping rate of 503 and 200 gpm was used throughout the pumping test for wells 1A and 1C, respectively.

During the 48-hour pumping test, depth to water measurements were taken from the pumping wells, 1A and 1C, and the observation wells; 1, 1-o, 2, 3, 4/C, 5/B. In order to monitor recovery rates, depth to water data were collected from the well for an additional 90 minutes after pumping had ceased.

The results of the re-analysis of both data sets are shown in Table 2-6, including values for transmissivity, storage, and hydraulic conductivity. The data and match-curves with the AQTESOLVTM™ output files are included in Appendix F. Calculated transmissivities in the vicinity of Chestnut Street range from 1,300 to 13,000 ft^2/day with a geometric mean of 10,000 ft^2/day. Using a minimum aquifer thickness of 56 feet (which is not well known) gives an maximum average hydraulic conductivity of 174 feet/day. IEP (1990) provides no values for comparison. Values of specific yield ranged from 0.06 to 0.50 with an average of 0.26.

2.2.3 Slug Testing

Aquifer slug tests were performed throughout the area surrounding the Olin property to obtain an understanding of the variability of aquifer hydraulic conductivity. Tests were specifically conducted in wells screened in glacial outwash, ice-contact deposits, and bedrock.

2.2.3.1 Methods

Aquifer slug tests consist of displacing water in a well and monitoring the response as water levels rebound to initial conditions. The rate at which the water levels respond is a function of the hydraulic conductivity of the aquifer. Therefore, slug tests can be used to calculate aquifer hydraulic conductivity.

During the slug test, water can be displaced in the well by adding or withdrawing water, or by emplacing or removing a physical object (called a slug) below the water level. For the slug tests at the site, a cylinder of PVC pipe filled with sand was used to displace the water. Two tests were performed in each well. The first consisted of dropping the slug below the water, causing the water level to rise, and monitoring its fall back to static conditions. The second test was then performed when the slug was removed from the well causing the water level to drop, and monitoring its rise back to static conditions. In some instances, individual tests were repeated. Water level changes were electronically monitoring using pressure transducers and recorded with a data logger. Water levels were recorded at intervals ranging from 0.1 second to 1 minute.

The slug tests were performed between August 22 and 24, 1995 and February 18 and 19, 1997 in the following wells:

Shallow Wells	Deep Wells	Bedrock Wells
	GW-19-D	
GW-48S	GW-48D	
GW-51S	GW-51D	
GW-53S	GW-53D	
GW-61S	GW-61D	GW-61BR
		GW-62BR
GW-63S	GW-63D	
GW-64S	GW-64D	
		GW-68BR
GW-73S	GW-73D	
GW-78-S		
GW-79-S		
	GW-85-D	

Because the water levels responded so slowly in the bedrock, only one test was performed in each bedrock well.

2.2.3.2 Results

The water level data collected during each slug test are presented in Appendix G. These data have been used to generate graphs which show the change in hydraulic head during the tests. These graphs are also included in Appendix G.

The changes in water level over time were used to calculate aquifer hydraulic conductivity. For wells screened in the water table aquifer (shallow and deep wells), the Bouwer & Rice (1976) method was used; for bedrock wells, the method of Cooper, et al. (1967) was used. The calculated hydraulic conductivities are presented in Table 2-7.

The results of the slug tests indicate a wide range of hydraulic conductivities in the unconsolidated aquifer. Calculated values ranged from 2 to 3,000 feet/day. These values are consistent with silty sands, clean sands, and gravelly sands. An average value of hydraulic conductivity for the unconsolidated aquifer of 60 feet/day has been calculated based on a log normal distribution.

The permeability of the bedrock formation is much less than the unconsolidated aquifer. Calculated transmissivities ranged from 0.035 to 2.3 sq. ft./day. The Cooper, et al. (1967) method of slug test analysis also allows the calculation of storage values for the formation. These ranged from 1.3×10^{-4} to 0.01 for the three wells tested.

2.2.4 Borehole Flow Monitoring

As mentioned above, the CSA provided little discussion on the potential for migration of the dense layer. One of the tasks performed as part of the Supplemental Phase II field activities was to obtain groundwater flow velocities measured directly in monitoring wells. The intent of this task was to measure flow velocities in fresh groundwater and in the underlying dense layer and determine to what extent both the velocity and direction of flow were different between the two bodies of water.

2.2.4.1 Methods

KV Associates, Inc. of Falmouth, Massachusetts was subcontracted to perform the flow monitoring in conjunction with Smith. The direction and velocity of groundwater flow were determined using heat-pulse GeoFlowmeter technology developed by KV Associates (e.g., Kerfoot, 1986). Using this technology, groundwater flow is measured by inducing a heat pulse in the center of a monitoring well and monitoring its outward radiation at a set of four paired thermistors. The heat-pulse monitoring system is shown schematically on Figure 2-17. The system works on the theory that as the heat-pulse radiates outward from the source, it will be offset by groundwater flow through the well. The temperature differential measured across the paired thermistors can be used to determine both the direction and rate of groundwater flow. The heat-pulse flowmeter can measure groundwater velocities as low as 0.2 foot/day.

In a similar manner, vertical flow can be measured by placing a packer in the well and forcing all flow through the center of the packer. Thermistors record the movement of the heat pulse through the center of the packer. This information is used to determine whether flow is upward or downward and to calculate a vertical flow velocity. Additional details of this technology are provided in the report prepared by KV Associates in Appendix H.

Groundwater velocities were measured in the field on November 8 through 16, 1995. Readings of horizontal flow were obtained in the following wells:

Shallow Wells	Intermediate Wells	Deep Wells
GT-6S		GT-6D
GW-27S		GW-27D
		GW-30DR
		GW-36
GW-42S		GW-42D
GW-43-S		GW-43D
GW-44S		GW-45D
GW-45S36		GW-45D
GW-50S		GW-50D
GW-58S		GW-58D
GW-62S	GW-62M	GW-62D

Most of these wells have 10-foot well screens. For the shallow wells, each section of screen was divided into two monitoring zones at 3 and 7 feet along the length of the screen. For the deep wells, most of which are screened partially in the overburden aquifer and partially in bedrock, the two monitoring zones were placed halfway along the overburden section of the screen and halfway along the bedrock section of the screen. The monitored intervals are shown schematically on Figure 2-17. Due to the observed variability of readings, flow measurements were taken at each monitoring zone twice (i.e. duplicate measurements).

Information about the make and manufacturer of the well screens was obtained from Soil Exploration, Inc., the drilling company used for the majority of the well installations. According to Soil Exploration, the well screens were 0.010-inch slotted, manufactured by Timco, Inc. The flow monitoring equipment was calibrated for this type of well screen in a sandy formation. Additional details about the calibration process are included in KV Associates report in Appendix H.

Vertical flow was measured in GW-80BR, GW-62BR, GW-62BRD, and GW-68BR in the field on November 8 through 16, 1995. Specific readings were obtained at variable locations throughout the entire open interval of each well. Because of the variability of the readings, multiple readings were obtained at each interval. The instrument used for measuring vertical flow was also appropriately calibrated, as described in the report in Appendix H.

2.2.4.2 Horizontal Flow Results

For each interval in each well monitored for horizontal flow, a direction and a velocity of groundwater flow were obtained. These data are summarized in Table 2-8. As stated above, a duplicate measurement was typically taken. Where the two readings were similar, the results were averaged. Where the two readings were significantly different, the data set that was more

consistent with a uniform flow field was used. Where there was no clearly preferable data set, both results are reported. These data are plotted on Plate 2-7 to provide an areal distribution of groundwater velocities.

In shallow wells, measured velocities were generally between 0.5 and 3.0 feet/day, with an average of 1.7 feet/day. The observed directions of flow were generally as expected: to the west and northwest overall with local flow towards nearby wetlands as shown on Plate 2-7. In general, the quality of the data collected from the shallow wells was reasonably good: most test results were well-reproduced in the duplicates, and separate zones in individual wells had similar results. The few exceptions are most likely associated with local conditions including low flow velocities, the effects of vertical flow components, or passage of trains (especially GW-50S).

Nine of the eleven deep wells were screened within the dense layer. In contrast to the shallow wells, the measured velocities in the deep wells were generally lower. This is shown graphically on the histogram on Figure 2-18. Velocities typically ranged from 0.2 to 2.0 feet/day with a large number of measurements between 0.0 and 1.0 feet/day. The measured directions of flow in the deep wells were typically erratic, and not generally in a direction associated either with the observed hydraulic gradient or with the overall slope of the bedrock surface (see Plate 2-7). The data quality was much poorer than for the shallow wells, with both duplicate tests and separate zones within the same well yielding inconsistent, non-reproducible results. In particular, the tests from the wells located in the center of the dense layer (GW-27D, GW-30DR, GW-36, GW-42D, GW-43D and GW-44D) generated particularly erratic data sets.

From these data, it is clear that the shallow groundwater and the dense layer are subject to very different flow dynamics. The flow in the dense layer is clearly not controlled by the hydraulic gradient as in the shallow groundwater; the observed directions of flow in the deep wells do not coincide with the directions measured in corresponding shallow wells, although hydraulic gradients are similar. The poor data quality associated with wells screened in the dense layer may be associated either with very low flow velocities (below the sensitivity of the instruments) or with non-uniform and/or non-horizontal flow through the well screen for these wells.

2.2.4.3 Vertical Flow Results

Vertical flow was measured at selected intervals in four bedrock monitoring wells (GW-62BR, GW-62BRD, GW-68BR, and GW-80BR). For each interval, each well was monitored for vertical flow, a direction (up or down) and rate of groundwater flow within the borehole. Groundwater flow into a borehole in the bedrock at the point of the entry (fracture) commonly oscillates and therefore two or more measurements were typically taken at each depth interval. This data is summarized in Table 2-9. Where the readings were similar, the results were averaged. Where the readings were significantly different in direction or magnitude, the data set that was more consistent with a uniform flow field was used. Where there was no clearly preferable data set, both results are reported. In general, the quality of the data collected from the bedrock wells was reasonably good in that the magnitude and direction of flow in each tested depth interval were well-reproduced in multiple readings. Exceptions are the slight differences in

magnitude and direction of flows between readings at a depth of 55 feet bgs in GW-80BR and the opposite direction of flows of about equal magnitude observed at a depth of 139 feet in GW-62BRD. As discussed in the report prepared by KV Associates (Appendix H), the differences in flow direction or magnitude observed in GW-62BRD and GW-80BR may be due to the oscillation effects created by "pipe flow" in the fractures.

The average vertical flows measured at the wells and depth intervals were between 0.1 and 0.35 cubic centimeters per minute (cc/min), for an average of 0.2 cc/min or 0.00005 gpm . The highest average vertical flow measured in the bedrock wells was 3.5 cc/min or about 0.0001 gpm downward at a depth of 118.5 feet in GW-62BRD. The very low vertical flow rates observed are consistent with the low transmissivities in the bedrock determined by slug testing (discussed in Section 2.2.3). The observed directions of flow were generally downward and the depth intervals with flow loss or gain usually corresponded to fracture zones identified in the downhole geophysical and camera logging. The magnitude of loss or gain at each fracture zone was too small to identify the most significant fracture zone in each borehole, although in general it appears that only one or two zones contributed to the vertical flow observed in the wells.

2.3 SOIL INVESTIGATIONS

2.3.1 Surface Soil Investigation

Surface soil sampling in the Comprehensive Site Assessment (CSA [CRA, 1993]) included composite samples of ten grid areas across the property and four specific areas (Solid Waste Management Units [SWMUs] 26, 27, 30, and 33) with visibly stained surface soils, as identified on Figure 4-2 from the CSA (CRA, 1993). Further evaluation of these soil results for the Risk Assessment indicated that further characterization of several areas was needed. The locations of these areas are shown on Plate 2-8 and include:

- The two drum areas with impacted subsurface soils;
- CSA grid Areas 1, 8 and 9; and,
- SWMU 26 where surface soils were removed by Olin.

The two areas with contaminated subsurface soil include the Drum Area A and Drum Area B. Samples of drum material and soil were obtained and analyzed during the CSA (CRA, 1993). The purpose of the additional investigation in the Drum Areas was to determine if the surface soils were also contaminated in these areas.

The CSA (CRA, 1993) surface soil results for the grid Area 8 composite sample show elevated concentrations of chromium, sulfate, and bis(2-ethylhexyl) phthalate. One of the discrete sampling locations making up the grid Area 8 composite sample was located at the Central Pond (Plate 2-8). The Central Pond is an area of known contamination and this discrete sampling location may have biased the grid Area 8 composite sample. In addition, grid Area 8 and adjoining grid Area 9 include two distinct areas of surface drainage referred to as the Central

Pond drainage areas and the Central Wetland area as shown on Plate 2-8. Both of these areas potentially received surface runoff from the acid pits and lined lagoons, while the Central Wetland area could have also received runoff from the former manufacturing area of the property. The purpose of the additional surface soil sampling in these areas was to separately characterize these different drainage areas.

The CSA (CRA, 1993) surface soil results for the grid Area 1 composite sample show elevated concentrations of chromium and sulfate. One of the discrete sampling locations making up the grid Area 1 composite sample was located adjacent to the South Ditch which is known to be contaminated with elevated concentrations of chromium. The purpose of additional surface soil sampling in grid Area 1 was to obtain a composite sample that is more representative to the entire area by excluding areas in the immediate vicinity of the South Ditch.

SWMU 26 was described in the CSA (CRA, 1993) as a black area west of Plant D where surface soils were removed. SWMU 26 lies within the area represented by the grid Area 9 composite sample; however no discrete samples were previously collected at this location. Therefore, additional surface soil samples were collected to determine if any contaminated surface soils remain at SWMU 26.

2.3.1.1 Methods

Sampling was performed in accordance with the protocols previously used at the site. Surface soil samples were collected with a dedicated or decontaminated stainless steel hand auger and trowel. All sampling equipment was dedicated or decontaminated in accordance with the procedures established in the CSA Work Plan (Section H.6, CRA, 1991). Representative soil samples were collected from a depth of 0 to 12 inches below ground surface. Discrete samples were transferred to the appropriate sample containers, appropriately labeled, stored at wet-ice temperature, and transported to the laboratory. Compositing of samples, where applicable, was conducted by the laboratory. Samples collected for volatile organic analysis were collected at a depth of 18 inches below ground surface at each location and submitted as discrete samples for analysis. In addition, samples collected for MADEP Extractable Petroleum Hydrocarbons/Volatile Petroleum Hydrocarbons (EPH/VPH) were not composited.

Soil samples were analyzed for Target Compound List (TCL) volatile organic compounds (VOCs) plus 2,4,4-trimethyl-1pentene and 2,4,4-trimethyl-2-pentene (TMPs), semivolatile organic compounds (SVOCs) including n-nitrosodiphenylamine (NNDPA), TCL Pesticides, Target Analyte List (TAL) metals (except from SWMU 26 samples), and site indicator parameters: ammonia, chloride, and sulfate. In addition, four samples from the Central Pond drainage areas and two samples (Areas 8 and 9 for a total of four additional locations) from the Central Wetland area were analyzed for MADEP EPH/VPH. Based on the CSA (CRA, 1993) results, analysis of TAL metals was limited to selected samples: one sample from the Central Wetland area within Area 8, two samples immediately adjacent to the Central Pond, and one

sample from Area 8, outside of the Central Wetland area and the Central Pond drainage areas. The sample location, laboratory sample identification, corresponding CSA (CRA, 1993) grid area location, and analyses performed on each sample are summarized on Table 2-10.

Quality assurance/quality control samples included field rinse blanks, trip blanks, and sample duplicates. Field rinse blanks and sample duplicates were collected on a minimum ratio of one for every 20 soil samples collected. One trip blank was collected for every shipment of VOC samples.

Drum Areas

Two composite surface soil samples were collected from Drum Area A. One composite sample consisted of four discrete sampling locations in the area between test pits 6 and 7 (DRMA-COMPB) and the other consisted of four discrete sampling locations around test pit 8 (DRMA-COMPA)). Samples collected from the Drum Area A were identified as "GA1-DRMA" where "DRMA-" indicates the Drum Area A, G1 through G4 indicate the four discrete locations making up the composite and the prefix "-A" or "-B" indicates which subarea within the Drum Area A.

In the drum areas, four discrete sampling locations were composited from each area that was excavated during the test pit program (see Plate 2-8). In Drum Area B, one composite surface soil sample (DRMB-COMP) was created from four discrete locations in the area around test pit 21. These discrete sampling locations are identified as "G1-DRMB", where "DRMB-" indicates Drum Area B and G1 through G4 indicate the discrete sample locations.

Area 8 and 9

Sampling locations in grid Area 8 and grid Area 9 focused on the Central Pond drainage areas, the Central Wetland area, and adjoining areas as shown on Plate 2-8. In grid Area 9, one composite surface soil sample was collected from four discrete sampling locations in the Central Wetland area. This sample was identified as "A9CW-". The numbered suffix identifies each discrete sample location.

Four grab samples were collected from the Central Wetland area in grid Area 8. These samples were labeled with the prefix "A8CW-" with the numbered suffix indicating the locations of each grab sample.

In the remaining portion of grid Area 8, a series of grab samples was collected to characterize the surface soils associated with different drainage features and the surroundings area. A total of nine surface soil samples were collected from the two separate Central Pond drainage areas. These samples were labeled with the prefix "CPDA-" with the numbered suffix indicating the location of each grab sample.

An additional four grab samples were collected from locations outside of the Central Pond drainage areas. These grab samples were labeled with the prefix "AREA 8-" with the numbered suffix indicating the sample location.

Area 1

In grid Area 1, six discrete sampling locations were composited to represent the area excluding the South Ditch. A greater number of discrete sampling points was collected to ensure that the composite sample was more representative of the entire area for use in exposure evaluations. The sample collected from this area was labeled "AREA 1-" with the suffix identifying each discrete sample location. The composite sample from the six discrete sampling locations was designated as "AREA 1(COMP)".

SWMU 26

One composite sample (SWMU26-COMP) was created from four discrete sampling locations within SWMU 26. Two of the discrete sampling points were located adjacent to boring BH23, where the highest concentrations of contaminants were detected in the subsurface soils. Sample locations were labeled as "SWMU26-" with a numbered suffix identifying each discrete sampling point.

2.3.1.2 Results

The Supplemental Phase II surface soil samples mainly consisted of brown humus and sand, although fill material was observed at Lake Poly and the Drum Areas and waste material was encountered and sampled at several locations in the Central Pond drainage areas, Central Wetland area, and in both Drum Areas. A description of the soil and/or waste material encountered at the surface soil sampling locations is provided in Table 2-11. Five of the sampling locations in the Central Pond drainage area (CPDA-1,-2,-4,-7, and -8) encountered several inches to seventeen inches of white to white-gray sludge. Several of these samples showed relatively higher concentrations of polynuclear aromatic hydrocarbons (PAHs) and sulfate. One sample in the Central Wetland area, A8CW-1, contained a green-gray sludge with elevated concentrations of PAHs. Waste material was observed at two of the grab sample locations for VOCs in Drum Area A, a thin layer of gray material at GA2-DRMA, and a 4-inch layer of white powder at GB4-DRMA. The VOC results for these locations were not elevated relative to the other six sample locations. In Drum Area B, a four inch layer of gray, clay-like material was observed at location G2-DRMB and the analytical results for this sample show slightly higher concentrations of VOCs compared to the other three samples.

The analytical results for Supplemental Phase II surface soil sampling programs are provided in Table 2-12. Table 2-13 provides a summary of maximum concentrations of Supplemental Phase II surface soil sample results for Drum Areas A and B, the Lake Poly Area, and SWMU 26. A comparison of subsurface soil and waste samples collected during the CSA (CRA, 1993) and

Supplemental Phase II surface soil results for Drum Areas A and B are provided in Table 2-14. A summary of Supplemental Phase II surface soil results and comparison with the CSA (CRA, 1993) surface soil sample results for Areas 1, 8, and 9 is provided in Table 2-15.

The surface soil sampling results show that surface soil in the former waste disposal or spill areas; Lake Poiy, SWMU 26, Drum Area A, and Drum Area B, are impacted by the same analytes that were found in subsurface soil or waste material samples, but at generally much lower concentrations (10 to 1,000 times lower). The resampling of CSA (CRA, 1993) composite areas (grid Areas 1, 8, and 9) show generally lower concentrations of most inorganics and organics in grid Area 1, and a mixture of lower and higher concentrations in grid Area 8 and grid Area 9. The results for each area are discussed below.

Data validation of all sample results was performed on selected samples from the Central Pond drainage areas (CPDA-1 through CPDA-9), grid Area 8 (AREA 8-1 through -4), and the Central Wetland (A8CW-1 through -4 and A9CW-1 through -4). Data validation included evaluation of the analytical results for 16 surface soil samples analyzed for TCL Metals, TCL SVOCs, and TCL Pesticides and 19 surface soil samples analyzed for TCL VOCs. In general, data validation has shown that most of the positive results for acetone and methylene chloride are suspect due to method blank contamination, and that most of the positive results are estimated for 4,4'-DDE, 4,4'-DDD, 4,4'-DDT, arsenic, thallium, mercury, and selenium. The Data Validation Reports are provided in Appendix O.

Drum Area A

As shown on Table 2-13, three VOCs, toluene, 1,1,1-trichloroethane, and acetone, were detected at low concentrations in the discrete Supplemental Phase II surface soil samples collected in Drum Area A. The maximum concentrations of these VOCs were all detected in the discrete sample GB4-DRMA at concentrations less than 0.05 milligrams per kilogram (mg/kg). The result for acetone was qualified by the laboratory since it was also detected in the associated blank. The comparison of results on Table 2-14 shows that only one VOC, 1,1,1-trichloroethane, was not previously detected in CSA (CRA, 1993) subsurface soil and waste samples.

Table 2-13 shows that four SVOCs, bis(2-ethylhexyl)phthalate, di-n-butylphthalate, fluoranthene, and NNDPA, were detected in the composite Supplemental Phase II surface soil samples collected in Drum Area A. One SVOC, fluoranthene, was detected at an estimated concentration of 0.45 mg/kg in the Supplemental Phase II surface soil samples, but was not detected in the CSA (CRA, 1993) surface soil samples. As shown on Table 2-14, the maximum concentrations of the CSA (CRA, 1993) subsurface soil and waste samples were 10 to 1,000 times higher than the concentration of the three other SVOCs detected in the Supplemental Phase II surface soil sample: bis(2-ethylhexyl)phthalate at 5.3 mg/kg, di-n-butylphthalate at 0.2 mg/kg, and NNDPA at 2.8 mg/kg (estimated). In addition, the di-n-butylphthalate detection was qualified by the laboratory since this SVOC was also detected in the associated blank.

Inorganics, including aluminum, calcium, chloride, chromium, iron, magnesium, potassium, sodium, ammonia, and sulfate, were detected at elevated concentrations in the composite Supplemental Phase II surface soil samples collected in Drum Area A (Table 2-13). Most of the maximum detected levels of inorganics were found in composite sample DRMA(COMP B). The chromium results were qualified by the laboratory since the spike sample recovery was not within the control limit. As shown in Table 2-14, the maximum concentrations of aluminum (12,900 mg/kg), arsenic (30.9 mg/kg), and magnesium (4,880 mg/kg) were detected at relatively higher concentrations in the Supplemental Phase II surface soil samples in comparison to the CSA (CRA, 1993) subsurface soil and waste samples.

Table 2-13 indicates that pesticides were also detected in Drum Area A at low concentrations (all less than 0.01 mg/kg) in the composite Supplemental Phase II surface soil samples. Seven of the maximum concentrations were detected in composite sample DRMA(COMP B) compared to five in composite sample DRMA (COMP A). There were two pesticides, toxaphene and alpha-chlordane, detected at elevated concentrations in the CSA (CRA, 1993) waste characterization samples (Table 2-14). However, it should be noted that the lowest pesticide detection limit in the CSA (CRA, 1993) subsurface soil and waste sample was higher than the highest detectable level of pesticides in the Supplemental Phase II surface soil samples.

In conclusion, the sample results from Drum Area A show that significantly fewer individual VOCs and SVOCs with generally lower concentrations detected in the Supplemental Phase II surface soil samples in comparison to the CSA (CRA, 1993) subsurface soil and waste samples. The opposite was true for inorganics. Seven inorganics were detected in the Supplemental Phase II surface soil at higher concentrations than were detected in the CSA (CRA, 1993) subsurface soil and waste samples. Two pesticides, toxaphene and alpha-chlordane, were detected at elevated concentrations in the CSA (CRA, 1993) subsurface waste samples whereas, 13 pesticides were detected at lower concentrations in the Supplemental Phase II surface soil samples. The detection of a greater number of pesticides is probably due to the much lower detection limits that were achieved during analysis of the Supplemental Phase II surface soil samples. These results suggest that the subsurface test pit excavations did not adversely impact the surface soils and that the elevated inorganics detected in surface soils are not related to contaminated subsurface soil or waste at Drum Area A.

Drum Area B

As indicated by Table 2-13, six VOCs, including 1,1,1-trichloroethane (0,016 mg/kg) and acetone (0.022 mg/kg), and TMPs were detected in the discrete samples at low concentrations (less than 0.03 mg/kg) in Drum Area B. Acetone was qualified by the laboratory and is considered to be a laboratory artifact since it was also detected in the associated blank sample. These VOCs maximum concentrations were distributed among three of the four discrete samples. As shown in Table 2-14, of the 12 VOCs detected in Drum Area B surface and subsurface samples, 8 maximum concentrations were exhibited by CSA (CRA, 1993) subsurface soil and waste samples

as compared to 4 maximum concentrations detected in Supplemental Phase II surface soil samples (1,1,1-trichloroethane was not previously detected and 2-butanone was only 0.0006 parts per million [ppm] higher).

Table 2-13 indicates that fourteen SVOCs were detected in the composite Supplemental Phase II surface soil sample, collected in Drum Area B, including several phenolic and PAH compounds, phthalates, and NNDPA. Seven of these SVOCs were also detected in the CSA (CRA, 1993) subsurface soil and waste samples at concentrations typically 10 to 1,000 times higher than detected in the Supplemental Phase II surface soil sample. Seven new SVOCs were detected in the composite Supplemental Phase II surface soil sample that were not detected in the CSA (CRA, 1993) subsurface soil and waste samples. These included benzoic acid and six PAHs. All were detected at low, estimated concentrations (less than 1 mg/kg).

As shown on Table 2-13, inorganics including aluminum, calcium, chloride, chromium, iron, ammonia, and sulfate were detected at elevated levels in the composite Supplemental Phase II surface soil sample. As shown by Table 2-14, ammonia, was the only inorganic detected at significantly higher concentrations (two times) than the CSA (CRA, 1993) subsurface soil and waste samples.

Table 2-13 indicates that five pesticides were detected in the Supplemental Phase II surface soil sample at low concentrations below 0.2 mg/kg. Of these five, endosulfan I, was also detected in the CSA (CRA, 1993) subsurface soil and waste samples. Endosulfan I was detected at a slightly higher maximum concentration in the Supplemental Phase II surface soil sample (0.099 mg/kg) than in CSA (CRA, 1993).

In conclusion, the comparison of sample results (Table 2-14) from Drum Area B show that fewer individual VOCs and inorganics with generally lower concentrations were detected in the Supplemental Phase II surface soil samples as compared to the CSA (CRA, 1993) subsurface soil and waste samples. Although seven new SVOCs were detected at low concentrations (less than 1 mg/kg) in the Supplemental Phase II surface soil samples, typically higher concentrations of SVOCs were detected in the CSA (CRA, 1993) subsurface soil and waste samples. A greater number of pesticides were detected at low concentrations in the Supplemental Phase II surface soil samples as compared to the results of the CSA (CRA, 1993) surface soil and waste samples. For pesticides, there were four maximum concentrations. The detection of a greater number of pesticides is probably due to the much lower detection limits that were achieved during analysis of the Supplemental Phase II surface soil samples. This comparison of surface and subsurface results on Table 2-14 suggests that the subsurface test pit excavations did not adversely impact the surface soils.

Area 1

The purpose of the additional surface soil sampling in Area 1 was to obtain a composite sample that was more representative of the entire area by excluding areas in the immediate vicinity of the South Ditch. Therefore, Table 2-15 provides a comparison of the Supplemental Phase II surface

soil samples and the CSA (CRA, 1993) composite sample (AREA 1) analytical results. Generally, the comparison of results shows that the previous sample, AREA 1, exhibited a greater number of SVOCs and inorganics at higher concentrations. The Supplemental Phase II surface soil samples collected in grid Area 1 contained VOCs at low concentrations that were not previously detected in the AREA 1 sample. No pesticides were detected in either the CSA (CRA, 1993) or Supplemental Phase II surface soil samples.

As shown on Table 2-15, six VOCs were detected at low concentrations (less than 1 mg/kg) in the discrete collected in Area 1. No VOCs were detected in the CSA (CRA, 1993) surface soil samples. The VOCs detected in the Supplemental Phase II surface soil samples included 1,1,1-trichloroethane at a maximum of 0.23 mg/kg and acetone at a maximum of 0.036 mg/kg. With one exception, methylene chloride in sample Area 1-1, all maximum detection's of VOCs occurred in sample Area 1-4 which was collected from a peat layer (Table 2-11).

Table 2-15 indicates that only three SVOCs (all phthalates) were detected in the composite Supplemental Phase II surface soil sample collected in grid Area 1. Of these three phthalates, only bis(2-ethylhexyl)phthalate was detected in the CSA (CRA, 1993) composite sample at a concentration 10 times higher than the Supplemental Phase II surface soil sample. The other two SVOCs, diethylphthalate and di-n-butylphthalate, were not detected during the CSA (CRA, 1993) sampling. Both of these compounds were detected at estimated values below 0.05 mg/kg and were also found in the associated blank.

As shown on Table 2-15, 16 inorganics, including chromium, lead, and sulfate, were detected at lower concentrations in the Supplemental Phase II surface soil sample compared to the concentrations detected in the in the CSA (CRA, 1993) sample. Five inorganics were detected in the composite Supplemental Phase II surface soil sample at higher maximum concentrations than detected in the CSA (CRA, 1993). Three of these, mercury (0.11 mg/kg), selenium (0.91 mg/kg), and thallium (1.4 mg/kg) were not detected during the previous CSA (CRA, 1993) sampling. Each of these metals were qualified by the laboratory since the spike sample recovery was not within the control limit. Sodium was only slightly higher in the Supplemental Phase II surface soil sample (50.8 mg/kg) compared to the CSA (CRA, 1993) surface soil sample (44 mg/kg). Ammonia was detected at 143 mg/kg in the Supplemental Phase II surface soil sample which is approximately 3.5 times the concentration detected in the CSA (CRA, 1993) surface soil sample.

Area 8

The purpose of the additional surface soils sampling in Area 8 was to separately characterize the Central Pond drainage areas, the Central Wetland area, and adjoining areas as shown on Plate 2-8. Table 2-15 summarizes the surface soil results for these areas and provides the results of the CSA (CRA, 1993) grid Area 8 sample for comparison. The sample results from Area 8 indicate that several new VOCs, SVOCs, and pesticides were detected at low concentrations in the Supplemental Phase II surface soil samples collected in Area 8 as compared to surface soil samples collected during the CSA (CRA, 1993). Concentrations of those organics detected in the Supplemental Phase II surface soil samples were typically lower than concentrations detected in

the CSA (CRA, 1993) surface soil samples. This was also true for inorganics, which were, with only three exceptions (ammonia, chromium, and nickel), detected generally higher concentrations in the CSA (CRA, 1993) surface soil samples.

Generally, higher concentrations of VOCs, SVOCs, and inorganics were more prevalent in the Central Pond drainage areas than the Central Wetland area and adjoining areas. Pesticides were ubiquitous in grid Area 8 and the maximum concentrations of pesticides were almost evenly distributed among the samples collected from the Central Pond drainage areas, the Central Wetland area, and adjoining areas.

As indicated by Table 2-15, three VOCs (toluene, 1,1,1-trichloroethane, and acetone) were detected at low concentrations in the Supplemental Phase II surface soil samples and were not detected in the CSA (CRA, 1993) surface soil samples. However, as mentioned previously, the detection of acetone in this sample is suspect due to method blank contamination associated with other sample results. The VOCs present at the highest concentrations (less than 0.06 mg/kg) included 1,1,1-trichloroethane and 2,2,4-trimethyl-1-pentene. Most of the maximum concentrations of VOCs were detected in samples from the Central Pond drainage areas. Only one discrete Supplemental Phase II surface soil sample outside of the Central Pond drainage or Central Wetland areas in Area 8, Area 8-2, exhibited a maximum VOC concentration (1,1,1-trichloroethane).

Several new SVOCs, including di-n-butylphthalate, at 1.4 mg/kg, were present in the discrete Supplemental Phase II surface soil samples collected in Area 8 that were not detected in the CSA (CRA, 1993) surface soil sample as shown in Table 2-15. Most of these were PAHs and some were phthalates. Benzoic acid and 1,2,4-trichlorobenzene were also among these new SVOCs. All were detected at low, estimated concentrations below 2 mg/kg. However, one Supplemental Phase II surface soil sample (A8CW-1), collected from the Central Wetland area exhibited significantly higher concentrations of SVOCs (see Table 2-12). This sample was not included in the comparison in Table 2-15 since it was collected from a sludge-like material and is not considered to be representative of surface soil results. Most of the highest SVOC concentrations were detected in the Central Pond drainage areas samples, primarily, CPDA-1. Maximum concentrations of other SVOCs were detected in samples CPDA-4 and AREA 8-4.

The surface soil results in Table 2-15 show that inorganics were generally detected at lower concentrations in the discrete Supplemental Phase II Investigations surface soil samples collected in Area 8 compared to the CSA (CRA, 1993) surface soil sample. Exceptions to this included chromium at 3,010 mg/kg, nickel at 10.2 mg/kg, and ammonia at 363 mg/kg, all detected in Central Pond drainage areas samples. The chromium result was qualified by the laboratory since the duplicate analysis and spike sample recovery were outside of specified control limits.

No pesticides were detected in the CSA (CRA, 1993) surface soil sample, but 14 were detected in the discrete Supplemental Phase II surface soil samples collected in Area 8 as shown on Table 2-15. However, it should be noted that the lowest detection limit in the CSA (CRA, 1993) surface soil sample results was 22 micrograms per kilogram (ug/kg). Most of the pesticides detected in

the Supplemental Phase II surface soil samples were present at concentrations of less than 10 ug/kg, with the exception of gamma-BHC (lindane) at 64 ug/kg, 4,4'-DDT at 40 ug/kg, and dieldrin at 12 ug/kg. The maximum concentrations of pesticides detected appear to be mainly associated with samples from five Central Pond drainage areas locations and two locations outside of Central Pond drainage areas and Central Wetland area (Table 2-12). Only one sample collected in the Central Wetland area exhibited a maximum pesticide concentration.

Area 9

The purpose of the additional surface soil sampling in Area 9 was to further characterize the Central Wetland area. The Supplemental Phase II Central Wetland sample results which are summarized on Table 2-15 show lower concentrations of almost all analytes than were detected in the adjoining grid Area 8 and surrounding portion of grid Area 9. Only two VOCs and one SVOC were detected in the Supplemental Phase II surface soil samples. Four inorganics were detected at higher concentrations than in the CSA (CRA, 1993) surface soil sample. Like Area 8, these included chromium and ammonia, but also included sodium and sulfate. Also similar to Area 8, Area 9 exhibited new detections of pesticides at very low concentrations.

As shown in Table 2-15, the two VOCs detected in the discrete Supplemental Phase II surface soil samples collected in Area 9 included 1,1,1-trichloroethane and trichloroethene. No VOCs were detected in the CSA (CRA, 1993) composite surface soil sample (AREA 9). Both VOCs were detected in sample A9CW-3 at concentrations less than 0.01 mg/kg. A total of 15 SVOCs were detected at low concentrations in the previous CSA (CRA, 1993) sample (AREA 9), while only one SVOC, bis(2-ethylhexyl)phthalate was detected in the grid Area 9 Central Wetland sample at an estimated concentration 0.019 mg/kg. This result was also less than that detected in the CSA (CRA, 1993) surface soil sample.

Table 2-15 shows that four inorganics, chromium, sodium, ammonia and sulfate, were detected in the composite Supplemental Phase II surface soil sample collected in Area 9 at higher concentrations than they were detected in the CSA (CRA, 1993) surface soil sample. Of particular note are ammonia and sulfate which were approximately 8 and 5 times higher, respectively, than the CSA (CRA, 1993) surface soil sample. Seven pesticides were detected at low concentrations (below 0.01 mg/kg) in sample A9CW-(COMP) and all but one pesticide, gamma-BHC (lindane), were detected at estimated concentrations. The greater number of pesticides detected in sample A9CW-(COMP) are attributed to lower laboratory detection limits.

SWMU 26

Supplemental Phase II surface soil results for the SWMU 26 samples show far fewer detected compounds than the results for other surface soils samples collected as part of the Supplemental Phase II Investigations. The results show sulfate and bis(2-ethylhexyl)phthalate are the major chemicals detected in the SWMU 27 surface soil samples and that the other analytes were generally detected at low, estimated concentrations.

As shown in Table 2-13, only two VOCs, 1,1,1-trichloroethane at 0.006 mg/kg and acetone at 0.013 mg/kg, were detected in the discrete Supplemental Phase II surface soil samples collected in SWMU 26. Four SVOCs were detected in the composite Supplemental Phase II surface soil sample collected in SWMU 26. The SVOCs detected at the highest concentrations include benzoic acid (0.79 mg/kg), acenaphthylene (0.045 mg/kg), bis(2-ethylhexyl)phthalate (5.2 mg/kg), and NNDPA (0.3 mg/kg). All but bis(2-ethylhexyl)phthalate were detected at estimated concentrations, but this compound was qualified and considered suspect by the laboratory since it was also detected in the blank.

The inorganic results in Table 2-13 show that chromium was detected at 47.6 mg/kg, but this result was qualified by the laboratory as estimated due to interference. Ammonia was detected at 114 mg/kg and sulfate was detected at 19,400 mg/kg. Four pesticides were detected in the SWMU 26 composite Supplemental Phase II surface soil samples (Table 2-13). Three of these, aldrin, 4,4'-DDT, and endrin, were detected at estimated concentrations below 0.02 mg/kg. Gamma-BHC (lindane) was detected at a maximum of 0.14 mg/kg in the composite Supplemental Phase II surface soil sample.

2.3.2 Soil Investigations at Lake Poly

Prior to 1970, liquid wastes generated at the property were discharged into the unlined Lake Poly located near the western boundary of the property. As discussed in Sections 1.2.1 and 1.3.1, Lake Poly was designed as a "filter bed" for liquid waste streams including process wastes, and discharges from yard and floor drains and from the pilot lab (Badger, 1969; Waghorne-Brown, 1952). Lake Poly was associated with facility production from the time of plant construction in 1953. Until 1967 sodium dichromate was used in the Kempore® process and acidic waste containing chromium sulfate was discharged to Lake Poly and the other pits. In about 1969 through 1971, Lake Poly was filled in and a wastewater treatment system was installed.

The CSA (CRA, 1993) identified subsurface soils contaminated with chromium in the vicinity of the former Lake Poly. However, most subsurface soil samples collected during the CSA (CRA, 1993) were only obtained above the water table and no discrete surface soil samples were collected from within the area of former Lake Poly. In order to further characterize surface and subsurface soils, three surface soil samples were collected and two soil borings were completed to drilling refusal in the area of the former Lake Poly.

2.3.2.1 Surface Soil

Based on the results of the CSA (CRA, 1993), it was determined that additional information was needed for the exposure assessment of surface soils in the area of Lake Poly. In the CSA (CRA, 1993), the discrete surface soil samples for the Area 3 composite sample did not lie within Lake Poly. The additional investigation in the Lake Poly Area was conducted on December 15, 1996. The sample locations shown on Plate 2-8 were designed to characterize the surface soils across Lake Poly.

Methods

Grab samples were collected at three discrete sample locations, LAKE POLY-1, LAKE POLY-2, and LAKE POLY-3, within the perimeter of former Lake Poly. The discrete samples were collected from the ends and midpoint of Lake Poly. Soil sampling was performed following the same procedures described in Section 2.3.1.1.

Results

The sample results for Lake Poly indicate that while VOCs, SVOCs, inorganics, and pesticides were detected in the Supplemental Phase II surface soil samples, concentrations were generally low and, for the organics, typically estimated. One exception was the elevated level of NNDPA in one sample.

The surface soil analytical results are provided in Table 2-12. Table 2-13 shows the maximum concentrations of VOCs detected in the discrete Supplemental Phase II surface soil samples from the Lake Poly Area. Ten VOCs were detected mostly below 0.01 mg/kg; 1,1,1-TCA, detected at 0.011 mg/kg, and acetone, detected at 0.031 mg/kg, were the exceptions. Acetone was qualified by the laboratory since it was also detected in the associated blank sample. The highest number of VOC maximums occurred in sample LAKE POLY-3, but the 1,1,1-TCA and acetone maximums were not exhibited by this sample.

Table 2-13 also indicates that 19 SVOCs were detected in the discrete Supplemental Phase II surface soil samples in the Lake Poly Area. Most of the detected SVOCs were PAHs or phthalates at individual concentrations of 0.3 mg/kg or less. Bis(2-ethylhexyl)phthalate and di-n-butylphthalate were detected in sample LAKE POLY-1 at concentrations of 5.1 mg/kg and 1 mg/kg, respectively. In addition, NNDPA was detected at an elevated concentration of 210 mg/kg in sample LAKE POLY-1. All of the other maximum SVOC concentrations were detected in sample LAKE POLY-2.

Inorganics detected at elevated concentrations include aluminum, chromium, potassium, and sodium (Table 2-13). Of significance was the highest chromium concentration (203 mg/kg) detected in sample LAKE POLY-1, however, this result was qualified by the laboratory since the duplicate analysis and spike recovery were not within control limits. Most of the maximum inorganics were detected in sample LAKE POLY-2.

As indicated on Table 2-13, seven pesticides were detected at low levels in the discrete Supplemental Phase II surface soil samples collected in the Lake Poly Area. Five of the seven were at estimated concentrations. All of these were below 0.01 mg/kg with the exception of 4,4'-DDT which was detected at 0.015 mg/kg in sample LAKE POLY-2. With one exception for gamma-BHC (lindane), all maximum detections were in this sample.

2.3.2.2 Subsurface Soil

Based on the results of the CSA (CRA, 1993), it was determined that additional investigation was necessary to determine if residual sludge is present in former Lake Poly, to obtain additional information about soils present in this area, and to evaluate the vertical extent of contamination. Smith collected stratigraphic information and soil samples from two borings, LPB-1 and LPB-2, in the area of former Lake Poly on January 18, 1995. The location of the soil borings are shown on Figure 2-1.

Methods

During completion of the borings, continuous split-spoon soil samples were collected. Features noted and recorded during drilling of the borings include a stratigraphy, organic/peat layers, visual contamination, groundwater, and the bedrock surface. These data are compiled in soil boring logs provided in Appendix A. The boreholes were grouted with bentonite upon completion. Based on field screening methods and visual observations, selected soils were retained for laboratory analysis. A total of four soil samples were collected: LPB-1(4-6), LPB-1(6-8), LPB-2(7-8), LPB-2(8-9), where the numbers in parentheses represent the sample depth in feet bgs. Soil sampling equipment was washed and rinsed with distilled water between samples in order to prevent cross-contamination. Samples were placed in appropriate containers, labeled, and maintained at wet-ice temperature during transport to the laboratory. Soil samples were analyzed for the following parameters: TCL VOCs, TCL SVOCs, TAL metals, ammonia, chloride, and sulfate.

Results

In general, materials encountered in the two Lake Poly borings consisted of medium-to-dark brown, fine sand with layers containing trace-to-some gravel. Groundwater was encountered at approximately 4 feet bgs in both of the borings. The soil boring logs are provided in Appendix A. In both borings, a layer of crushed stone was encountered, at 7 feet bgs and 6 feet bgs in LPB-1 and LPB-2, respectively. Below the crushed stone layer, relatively darker gray to brown sand and gravel were encountered in both soil borings. In addition, an oil sheen was observed on the sampling spoon and the water in the sampling spoon from 9 to 12 feet bgs in LPB-2. No organic soils were encountered except in small amounts from 8 to 9 feet bgs in LPB-2. Drilling refusal was encountered at 11.5 feet bgs and 10 feet bgs in LPB-1 and LPB-2, respectively. Based on observations during drilling, it appears that there is not a significant amount of residual waste in Lake Poly.

Analytical results for the soil samples collected from the Lake Poly area are summarized in Table 2-16. The analytical results show that the highest concentration and greatest number of organic compounds and the highest concentration of site specific inorganic parameters were found in the deepest soil sample from each soil boring. In general, the highest concentration of VOCs, SVOCs, and inorganics were detected in the 6 to 8 feet bgs sample from LPB-1.

The results of VOC, SVOC, and inorganic analyses indicated the same compounds previously identified by the CSA (CRA, 1993) in the vicinity of Lake Poly. The majority of twelve VOCs were detected in the deepest soil sample from LPB-1 and the most prevalent VOCs included 2,4,4-trimethyl-1-pentene and 2,4,4-trimethyl-2-pentene. The majority of sixteen SVOCs were detected in the deepest soil sample from each boring and the most prevalent SVOCs included phthalates and n-nitrosodiphenylamine. Elevated inorganics, including ammonia, sulfate, vanadium, and chromium, were detected in the deepest soil sample from LPB-1.

The site specific parameters and range of concentrations detected in the Lake Poly soil borings LPB-1 and LPB-2 are summarized below.

Parameter	Concentration Range (mg/kg)	Maximum Concentration Sample Location
2,4,4-Trimethyl-1-pentene	0.093 to 340	LPB-1(6-8)
2,4,4-Trimethyl-2-pentene	0.28 to 210	LPB-1(6-8)
bis(2-Ethylhexyl)phthalate	0.97 to 1,400	LPB-1(6-8)
n-Nitrosodiphenylamine	0.15 to 3,000	LPB-2(8-9)
Chromium	68 to 17,000	LPB-1(6-8)
Ammonia	89 to 10,000	LPB-1(6-8)
Sulfate	< 20 to 260	LPB-1(6-8)
Vanadium	4.8 to 23	LPB-1(6-8)

2.4 CHARACTERIZATION OF SOUTH AND WEST DITCHES

The CSA documented the presence of a white chromium-bearing flocculent in portions of the South and West Ditches on the Olin property. Because of the uncertainties about the conditions leading to flocculent formation and occurrence, the Supplemental Phase II Field Investigations included a study of the ditch system. The study included obtaining additional hydrologic information on the interaction between the groundwater and surface water, additional contamination distribution information, a study of the geochemistry of the system, and information on the seasonal variability in the hydrology and chemistry. The additional chemical characterization is discussed in Section 2.7.6. The section below focuses on the physical system. Plate 2-9 is an enlargement of the area surrounding the South and West Ditches showing additional details at the local scale.

2.4.1 Weir Immediate Response Action

Potential immediate concerns associated with the chromium-bearing flocculent included exposure to the flocculent in the surface water, and possible downstream transport of the flocculent off-property to the East Ditch. To prevent incidental exposure to the flocculent, Olin constructed an 8-foot high chain-link fence around portions of the Off-Property West Ditch. A perimeter fence around Olin's property limits potential exposure to the ditch system within the property.

To prevent downstream migration of the chromium-bearing flocculent, Olin implemented an IRA which was reviewed and approved by the MADEP (described in the "West Ditch Precipitate Action Plan", CRA 1992). A weir was placed across the South Ditch to increase the water level in the West Ditch. In addition to natural flow, the NPDES outfall for the Plant B groundwater recovery and treatment system discharges to the On-Property West Ditch, upstream of the weir at an average flow rate of less than 0.03 cubic feet per second (cfs). The damming and ponding of water behind the weir has allowed the flocculent to settle out of the surface water. In addition, the increased surface water head minimizes the discharge of shallow groundwater to the ditches in this area, and reduces the creation of additional flocculent. Hay bales were placed further downstream across the ditch to prevent downstream transport of flocculent. The flocculent was vacuumed from the ditches upstream of the weir in June 1994 after the dam was built. The weir plate was installed in July 1994. A monitoring program associated with this IRA was begun in April 1994, prior to installation of the weir. The location of the weir is shown on Plate 2-9.

The effectiveness of the Weir IRA is monitored on a regular basis under the following program:

- On a weekly basis, plant personnel inspect the condition of the weir and the water level behind the weir, and note any other observations including flocculent occurrence. Monthly, this inspection is performed in conjunction with a Wetlands Scientist from Wetlands Preservation, Inc. The results of these observations are documented on weekly Weir Inspection Reports which are submitted to the MADEP and Wilmington Conservation Commission (WCC) on a monthly basis.

- Weir sampling began in July 1994. The following wells are included in the weir groundwater sampling: GW-10S, GW-11, GW-19S, GW-24, GW-25, GW-26, GW-35S, GW-39, GW-42S, GW-43S, GW-55S. The locations of these wells can be found on Plate 2-2 . Groundwater samples were collected from select monitoring wells monthly for the first four months after weir installation and quarterly since then. Samples collected from the 11 wells were analyzed for indicator parameters: chloride, ammonia, sulfate, aluminum, chromium, and specific conductance. The results of these analyses are submitted to the MADEP and WCC every six months in the form of a Semi-Annual Status Report.

- Water levels are measured in wells and piezometers surrounding the ditch system at the same time that groundwater samples are collected. These data are used to calculate groundwater and surface water elevations and to construct groundwater contour maps showing flow directions. This information is submitted to the MADEP and WCC in status reports.

Olin has submitted the weir status reports on a semi-annual basis since March 1995 (BCM, 1995; Smith Environmental, 1995, 1996a, 1996b; Smith Technology, 1997). The following points summarize the effect of the weir on the ditch system.

- Installation of the weir has created a localized groundwater mound in the vicinity of the Off-Property West Ditch. During periods of relatively low groundwater levels (summer and fall), shallow groundwater flow is radially away from this mound. As a result, groundwater northwest of the ditches now flows to the northwest where it formerly flowed to the southeast. During periods of relatively high groundwater (winter and spring), groundwater continues to flow towards the ditches as it did prior to installation of the weir.

- Recharge and discharge relationships between the Off-Property West Ditch and the nearby groundwater are variable. However, in general, groundwater discharges to the ditches during periods of high water. During low water periods, the surface water recharges the groundwater where water is present in the ditches.

- Statistically significant trends in shallow groundwater quality have been observed in some wells since the weir was installed. Concentrations of several parameters have decreased during the monitoring program. Seven wells, including all wells adjacent to the Off-Property West Ditch, show decreases in at least one parameter. These changes may have occurred in response to installation of the weir and the consequent recharge of the shallow groundwater system from the ditches.

- Statistically significant increases in parameter concentrations have been observed at GW-43S, located northwest of the West Ditch. Prior to installation of the weir, this area was upgradient of the Ditch; with the mounding caused by the weir, it is seasonally downgradient. It is possible that contaminated groundwater located in the area of the Off-Property West Ditch has been moved to the northwest based on the seasonal changes in groundwater flow direction caused by the weir.

The data collected as part of the IRA is considered in the following discussions of conditions in the South and West Ditches. In addition, it must be noted that all Supplemental Phase II field activities took place after installation of the weir and are therefore representative of post-weir conditions.

2.4.2 Shallow Well and Piezometer Installation

In order to understand shallow groundwater geochemistry and the interaction between the groundwater and surface water, Smith installed additional monitoring locations along the South Ditch. These monitoring locations include four shallow groundwater wells along the ditch, GW-76S, GW-77S, GW-78S, and GW-79S, and seven piezometers, PZ-11 through PZ-17, within the stream bed of the ditch system. The locations of these monitoring points are shown on Plate 2-9 and Plate 2-2. The wells and piezometers were installed between January 16 and 17, 1995.

Smith retained Soil Exploration, Inc. from Leominster, MA to drill and install the four wells along the South Ditch. In accordance with a permit received from the WCC, small trees and brush were removed to create access routes to the proposed well locations in the wetland buffer zone. Prior to drilling, hay bales were staked in place between the drilling location and the wetlands, to control erosion and sedimentation.

Smith recorded stratigraphy, blow counts, and water level during drilling of the boreholes. Each well was constructed of 2-inch inside diameter (ID), schedule 40, PVC well screen and riser. Ten feet of 0.010-inch factory-slotted screen was used in each well. A sand pack was installed to the top of well screen and approximately 6 inches of bentonite seal was installed above the sand pack. A 95 percent neat cement/5 percent bentonite grout seal was emplaced from the top of the bentonite to ground surface. A protective stick-up steel casing with keyed alike lock and concrete pad was installed at each well head. The stratigraphic and construction details are provided on the well logs in Appendix I. Well construction information is summarized in Table 2-2.

After completion, each well was developed by pumping to remove any fine sediment from the well. The development continued until little turbidity was observed or for a maximum of one hour. Water generated during well development and cuttings generated during drilling were contained and disposed at a licensed facility. The drilling rig and the down hole tools were steam cleaned prior to the start of each borehole and prior to leaving the site to prevent cross-contamination.

Piezometers were installed in the stream bed along the West and South Ditches at seven locations, PZ-11 through PZ-17. Each piezometer consisted of steel pipe approximately four feet in length attached to a 2-foot screen. The piezometers were pounded into the stream bed with a sledge hammer until the top of the screen was at least one foot below the stream bed surface. Following installation, the new monitoring wells and piezometers were surveyed for horizontal and vertical control.

In general, materials encountered during the installation of the shallow wells include sand and gravel with little silt and clay. Refusal potentially representing the bedrock interface was encountered at relatively shallow depths in borehole GW-78S.

Water level and groundwater quality data were obtained from these wells to aid in understanding the hydrologic and chemical environment of the South Ditch system. The hydrology is discussed in more detail in the following sections. The chemical conditions are discussed in Section 2.7.6.

2.4.3 Hydrology of the West and South Ditch System

Water level data obtained from wells and piezometers in conjunction with data from the flumes has been used to aid in understanding the hydrology of the ditch system with particular reference to the interaction of surface water and groundwater. Site-wide water levels and groundwater contour maps are provided in Section 2.2 above.

Table 2-17 shows a summary of water level data obtained from the six co-located surface water/groundwater monitoring locations along the length of the Off-Property West Ditch and the South Ditch. These are quarterly data spanning more than two years. Figures 2-19 through 2-28 graphically show the changes in hydraulic heads along the ditch system at these locations. The following general conclusions can be drawn from examination of the hydraulic heads:

- At all monitoring dates, the surface water levels in the Central Pond are higher than groundwater, indicating a potential for groundwater recharge. Water levels in the Pond are higher than nearby water levels in both the groundwater and the South Ditch. Therefore, neither the groundwater nor the ditches are a likely source for the water in the pond; instead, it most likely receives water from rainfall and nearby runoff.

- Within the South Ditch, from the weir to the juncture with the East Ditch, hydraulic head relationships typically indicate the potential for groundwater discharge to the South Ditch in periods of both high and low water level.

- Upstream of the weir, in the Off-Property West Ditch, the interaction between surface water and groundwater is more variable. In general, during periods of low groundwater levels, the surface water, where present, recharges groundwater. During periods of high groundwater levels, groundwater discharges to the Ditch. Despite these generalizations, there is more local and seasonal variability within this section of the ditch.

- The water level data from October 1996 were obtained following a severe storm in which Wilmington, Massachusetts received about 10 inches of rainfall in a three-day period (measured at the Butters Row Treatment Plant). The water levels show groundwater higher than surface water, indicating that the peak storm runoff had dissipated within two days despite the extreme severity of the storm. These data show that the gradient reversal created by excess runoff is dissipated very rapidly, and therefore, the occasions during which the potential for flow from the ditches to the groundwater exists, are infrequent and of short duration.

The hydraulic heads can only be used to evaluate the potential for recharge or discharge between the surface water and groundwater. The extent to which such interaction takes place is dependent on the underlying hydrologic properties such as the aquifer permeability, the hydraulic gradient within the formation, and the permeability of the sediments lining the ditches. The data obtained during the South Ditch flow study described below directly provides a measure of how much water is moving between the surface water and groundwater systems.

2.4.4 South Ditch Flow Study

In order to obtain a better understanding of the flocculent formation, it is necessary to have an understanding of the hydrogeologic conditions and the interaction between the groundwater and the surface water system. Therefore, in December 1995 and March 1996 Smith Environmental submitted 'South Ditch Flume Study" proposals to MADEP in which it was suggested that stream flow measurement stations in the ditch system could be used to identify whether segments of the South Ditch are effluent or influent and to provide an estimation of the rate of gain from groundwater or loss to groundwater.

2.4.4.1 Methods

Flow measurement stations were established at five locations along the South Ditch on May 2 and 3, 1996. The locations of the flow measurement stations are shown on Plate 2-9. At each station, stainless steel flumes were installed in the stream bed to provide a uniform cross-sectional area and thus more accurate flow measurements. The stream flow measurement stations were constructed at points where the channel was narrowest and had the highest banks so that the flumes could be easily installed and the potential for bypass flow was minimized. These locations also have a higher velocity and/or deeper flow than a wider channel location at a given flow rate. This makes it possible to measure lower flow rates, because the velocity meter will not function properly in less than several inches of water and below a minimum velocity.

The flumes were selected to provide the best possible accuracy for the flow measurements without impacting the system that was being studied. Weirs or other similar devices could not be used since they would result in increased head on the upstream side which would alter the interactions between the groundwater and surface water systems. The flumes were installed so as not to change the natural head conditions along the South Ditch.

The flumes were individually sized to fit the width of the ditch at each location. Each flume was constructed of stainless steel plating bent into a "U" shape with squared corners. The length of each flume was sized to fit the width of the South Ditch at each monitoring location. Each flume was 1 foot wide (parallel to the water flow direction) and had a flange on the bottom and both sides that were pushed into the channel bottom and banks, respectively on the upstream side.

The flumes were installed so that the bottoms were placed flush with the bottom of the channel and level. The sides extend vertically upward from the bottoms tight against the stream banks. Sand bags were placed along the upper portion of the sides where the banks sloped away to maintain the stability of the sides and prevent water from flowing around the flume at higher flows.

Flow measurements were taken at each of the flumes on May 2 and 3, 1996, May 21, 1996, June 3, 1996, June 18, 1996, November 12, 1996, December 3, 1996, January 7, 1997, April 16, 1997, and April 30, 1997. The flow measurements on May 2 and 3, 1996 were conducted by Smith personnel and the remaining measurements were conducted by personnel from Wetlands

Preservation, Inc. The cross-section of the stream at each location was divided into several segments. The water velocity was then measured in each of the segments using a Marsh-McBirney direct reading velocity meter. The water depth on the flume and the velocities for each of the segments along the flume were recorded for each flume.

The flow measurements were planned so they did not occur after significant precipitation events. Generally, several days were allowed after precipitation events, before taking flow measurements to assure that baseflow conditions were monitored. In addition, the flow measurements were timed to occur prior to the start of water discharge to the ditch system from Olin's National Pollution Discharge Elimination System (NPDES) permitted outfall on the West Ditch.

The flow volume was calculated at each of the flumes by multiplying the cross-sectional area for each segment by the flow velocity for the respective segment. The flow volume for all of the segments at a flume were then added together to obtain the total flow volume at the flume location. The results of the currently available flow measurements are presented in the following section. Data collection was halted for the summer due to low flow conditions. Date collection resumed in November 1996, when sufficient flow resumed in the ditch system to permit the use of the velocity meter.

2.4.4.2 Results

The results of the flow measurements are presented in Table 2-18 and copies of the flow calculations are included in Appendix J. Flow measurements can only be obtained during certain periods of the year. In the summer, the ditches may be nearly dry, or flows may be so low as to be below the resolution of the instrument. In the winter, the surface of the ditches is frequently iced over. The calculated flow results for several dates are questionable due to extremely low flow conditions. The results for the remaining six dates are shown on the graph presented as Figure 2-29.

The flow measurements from the six monitoring events with data suitable for graphing include two winter events (12/3/96 and 1/7/97) and four spring events (5/3/96, 5/21/96, 4/16/97, and 4/30/97). As described in Section 2.4.2, hydraulic head relationships during these periods suggest the potential for groundwater discharge. The graph on Figure 2-29 shows that the segment of the South Ditch between Flumes 4 and 5 (between the Central Pond and the eastern property line) does gain in flow in five of the six monitoring events. Measured gains range from 27.5 to 60.6 gallons per minute (gpm) with an average of 45.8 gpm. It is also clear from the graph that there is minimal change in flow between Flume 1 and Flume 2 for five of the six events. The change in flow between these two stations ranges from -0.9 to +8.7 gpm with an average of +1.1 gpm. The presence of bedrock outcrops just north of the South Ditch in this area suggest that the extent of the aquifer may be locally limited so that there is little interaction between groundwater and surface water. The flow in the middle two segments of the South Ditch is more variable, being

sometimes gaining, sometimes losing, and sometimes mixed. In addition, there is no clear correlation in these segments between the stage (volume of flow) and whether the stream is gaining or losing. However, the net change between Flume 2 and Flume 4 is small in four of the six monitoring events, ranging from -9.5 to +3.4 gpm with an average of -2.8 gpm.

2.5 GROUNDWATER SAMPLING AND ANALYSIS

In order to supplement groundwater analytical data collected during the CSA, several rounds of groundwater sampling have been performed within the study area. Groundwater samples were collected in order to understand the present groundwater chemistry within the study area. These analytical data will assist in understanding the current groundwater conditions, the mechanisms of transport, and the risk assessment. Groundwater quality data have been obtained from the wells surrounding the Calcium-Sulfate Landfill, the South Ditch (See Section 2.4.3), and from the municipal supply wells. Groundwater has also been sampled from monitoring wells in close proximity to the Central Pond, the weir, and the municipal supply wells. Samples were taken from the wells known to be screened in the dense layer for geochemical evaluation. Groundwater samples were also taken from new wells located in the Maple Meadow Brook wetland in order to determine the extent of the dense layer.

Data collected during the Supplemental Phase II Investigation are summarized in Tables 2-19 through 2-23. Analytical results are discussed in the remainder of this section; however, analytical results associated with the weir monitoring program and all geochemical samples are not discussed below. The data associated with weir monitoring are discussed in weir status reports submitted to the MADEP semi-annually (BCM, 1995; Smith, 1995, Smith, 1996a, Smith 1996b, and Smith Technology, 1997). The methods, analytical results, and geochemical evaluations are discussed in Section 2.7.

2.5.1 Calcium-Sulfate Landfill Wells

Prior to Supplemental Phase II investigations groundwater samples from the wells surrounding the Calcium-Sulfate Landfill were last collected in November 1992 from two of the nine wells. Previous to this, the last complete round of groundwater sampling from this area was performed in May 1989. Since it would be difficult to make a current assessment of groundwater conditions near the Calcium-Sulfate Landfill using data which is over five years old, six of the nine wells surrounding the Calcium-Sulfate Landfill were recently sampled to gather current groundwater analytical data. Three wells (SL-4, SL-7, and SL-8) were not sampled because they are located on bedrock highs and historically do not penetrate a sufficient saturated interval to allow sampling. The remaining wells were sampled on December 17, 1996 and sent for laboratory analysis of calcium and the following site-specific indicator parameters: chloride, chromium, ammonia, and sulfate. Field parameters including temperature, pH, and specific conductance were also recorded. The results of the December 17, 1996 sampling event are presented on Table 2-22 and are summarized below:

	SL-01D	SL-01S	SL-02	SL-03	SL-05	SL-06
Ammonia (mg/l)	nd	nd	nd	0.73	79.3	70.9
Calcium (mg/l)	44.0	nd	81.7	175	343	248l
Chloride (mg/l)	21.6	nd	nd	4.10	28.9	57.6
Chromium (mg/l)	nd	nd	nd	nd	nd	nd
Sulfate (mg/l)	127	17.7l	239	502	1,370	956
pH (Field)	4.78	4.94	5.61	5.26	5.85	5.60

nd = not detected

Concentrations of indicator parameters are relatively low on the west side of the landfill, but elevated concentrations are present in SL-3, SL-5, and SL-6 which are located to the east and south. Concentrations of indicator parameters are also elevated in GW-40D, located at the southwest corner of the landfill. Groundwater flow in this area is to the northeast and southwest. In general, the concentrations observed in 1996 are not significantly different than the earlier data from the late 1980s. This suggests that the Calcium-Sulfate Landfill may be contributing inorganics to the groundwater.

2.5.2 Shallow Groundwater Sampling to Update Risk Assessment

In order to update the evaluation of VOCs in the shallow groundwater in the Human Health Risk Assessment, additional sampling of several shallow groundwater wells was conducted on October 16 and 17, 1996 by ABB-ES. Groundwater samples were taken from the following wells:

Altron B1	GW-42S	GW-69S
Altron B3	GW-43S	GW-76S
GW-10S	GW-44S	GW-77S
GW-25	GW-45S	GW-78S
GW-28S	GW-47	GW-79S

In addition to volatiles, all of the samples were analyzed for TAL metals, chloride, sulfate, ammonia, and bicarbonate. Groundwater analytical results are summarized in Tables 2-19 (volatiles) and Table 2-22 (metals and inorganics). These results were used in conducting the Human Health Risk Assessment which was performed by ABB-ES (see Section 6).

2.5.3 Deep Groundwater Sampling During Drilling

As discussed in Section 2.1.3, six borings were drilled in October and November, 1996 in order to obtain more information about the bedrock surface, aquifer materials, and groundwater quality west of the facility. Three borings were drilled on residential properties west of Main Street (MB-1, MB-2, and MB-3) and three were drilled along the access road to the Chestnut Street pump station (CB-1, CB-2, and CB-3). Groundwater samples were collected from MB-1, MB-2, and MB-3. Groundwater samples were not collected from the Chestnut Street borings as dense layer

is not expected to be present in this area. The analytical results from groundwater samples collected during drilling of these borings are discussed in this section; drilling methods were discussed in Section 2.1.3. Locations of the borings can be found on Figure 2-1.

During drilling, when bedrock was encountered, a tool was sent down the borehole which opened up a three foot screen to allow collection of a groundwater sample at the base of the hole. Prior to the collection of the groundwater sample, the borehole was purged of at least one drilling casing volume to minimize the influence of waters used during borehole advancement. Groundwater samples were measured in the field for pH and specific conductance and the unfiltered samples were sent to the laboratory for analysis of the indicator parameters (ammonia, chloride, and sulfate.

A summary of the analytical results from the borehole groundwater samples is as follows:

	MB-1 10/07/96	MB-2 10/11/96	MB-3 11/12/96
Ammonia (mg/)	3.13	0.301	nd
Chloride (mg/l)	97.6	205	106
Chromium (mg/l)	0.3	nd	1.55
pH (Field)	7.00	6.93	6.70
Sulfate (mg/l)	135	54.7	109

nd = not detected

MB-1 showed elevated concentrations of most of the indicator parameters near or above background levels and therefore may be impacted by site contamination. The chloride results in MB-2 and MB-3 are slightly elevated. However, the concentrations of ammonia in the samples from MB-2 and MB-3 are low, and therefore the groundwater as these locations is unlikely to be impacted by site contamination. Both MB-1 and MB-3 have elevated chromium concentrations, however the pH of these samples is too high to suggest that chromium is actually present in the groundwater at the observed concentrations. The presence of chromium in these samples is likely due to the presence of particulate matter, as chromium occurs naturally in the aquifer material at background concentrations of approximately 15 mg/kg (CRA, 1993), and the samples were unfiltered and taken during drilling.

2.5.4 Miscellaneous Groundwater Sampling

In evaluating the Supplemental Phase II data, several areas of potential environmental concern identified in the CSA (CRA, 1993) were researched. To assess the extent of the contamination, if any, associated with these areas of concern, groundwater analytical data were gathered and compiled. Through this effort of compiling analytical data, it became apparent that several areas lacked current groundwater analytical data. Also, at some locations, groundwater samples which were taken were not analyzed for particular parameters. In order to make accurate assessments of the current status of certain areas and to better understand the mechanisms of transport, additional groundwater analytical data were gathered from selected areas. Additional

groundwater samples were collected from the interceptor wells in the Plant B area, select wells near the ditch system, between Drum Area B and the Central Pond, and at miscellaneous peripheral wells located off the Olin property.

Plant B Area

Groundwater samples were collected from the three interceptor wells near Plant B (IW-11, IW-12 and IW-13) on December 18, 1996 and January 24, 1997. Prior to these samples, the most current analytical data were from a samples of groundwater and floating oil collected in November 1992; therefore recent groundwater samples from IW-11, IW-12, and IW-13 were analyzed for TCL volatiles, TCL semivolatiles, TCL pesticides and the site specific indicator parameters. During groundwater sampling, field parameters were measured and recorded.

The analytical results are shown on Tables 2-19 to 2-22. A summary of the most recent analytical results from the Plant B wells is given below:

	IW-11	IW-12	IW-13
Volatiles (ug/l)			
Ethylbenzene	nd	47 J	nd
Methylene Chloride	6 J	nd	nd
Xylenes, Total	8 J	39 J	nd
Acetone	26 J	nd	nd
2-Hexanone	nd	2,000	nd
2,4,4-Trimethyl-1-pentene	450	560	22
2,4,4-Trimethyl-2-pentene	120	120	5
Total Volatiles	610	2,766	27
Semivolatiles (ug/l)			
Benzoic Acid	nd	3 J	nd
Phenol	nd	nd	5 J
bis(2-ethylhexyl)phthalate	30 JB	2 JB	2 JB
Diethylphthalate	11 J	nd	nd
Di-n-butylphthalate	6 JB	1 J	nd
N-Nitrosodiphenylamine (1)	950	150	94
Total Semivolatiles	997	156	101
Pesticides (ug/l)			
Aldrin	0.77	nd	nd
Endrin	0.0041 J	nd	nd
Heptachlor	0.013 J	nd	nd
Metals-Dissolved (mg/l)			
Chromium	nd	nd	nd
Inorganics (mg/l)			
Ammonia	4.99	3.80	42.3
Chloride	28.9	41.0	25.8
Sulfate	42.4	31.3	159
Physical Properties			
pH (Field)	6.36	6.06	6.30
Sp. Cond. (Field) - umhos	358	348	685
Temperature (Field) - deg. C	12.1	12.4	12.4
TDS calc. from Sp. Cond. (mg/l)	207	202	399

J = Estimated value below quantitation limit
B = Parameter detected in associated method blank
nd = Parameter not detected

The former above ground storage tanks at the Plant B area contained raw materials including process oils, and plasticizers, bis(2-ethylhexyl)phthalate, and n-nitrosodiphenylamine (NNDPA). The recent data summarized above indicates that the Plant B groundwater is contaminated primarily with NNDPA, 2-hexanone, trimethylpentenes, and ammonia. Pesticide contamination is minor and is limited to IW-11. Further discussion of the contamination in the Plant B area, including characterization of the oil and groundwater contamination, can be found in Section 4.2.1.2.

Central Pond

Well GW-7 is located downgradient of the Southeastern Drum Area and former Acid Pits and is proximal to the Central Pond. Historically, the groundwater from this well has had relatively high concentrations of contaminants. During the former groundwater monitoring program in the 1980s, concentrations were decreasing significantly, but as of the last groundwater sampling event in May 1990, concentrations were still relatively high. To evaluate the previously decreasing trend and the current groundwater conditions, a groundwater sample was collected from this well. Groundwater from GW-7 was analyzed for TCL volatiles, TCL semivolatiles, TAL metals, site-specific indicator parameters, and alkalinity. In addition, field parameters were recorded during groundwater sampling. The sample from GW-7 was collected on December 18, 1996. The results are provided on Tables 2-19 to 2-22. The following results are for the indicator parameters: ammonia = 7.99 milligrams per liter (mg/l); chloride = 697 mg/l, chromium = 1.46 mg/l, sulfate = 3590 mg/l, pH (field) = 4.71, specific conductance (field) = 19,200 umhos. The December 1996 sampling shows a decrease in ammonia and an increase in chloride, chromium, and sulfate as compared to the last sampling event in May 1990. However, considering the historical range in concentrations observed in this well, the relative differences between the 1990 and 1996 data are minimal. The high specific conductance, sulfate, and chromium suggest the presence of dense layer in or near this well.

Ditch System

As part of the South Ditch study (see Section 2.4), four shallow wells were installed in January 1995: GW-76S, GW-77S, GW-78S, and GW-79S. In general, all new wells are sampled for TCL volatiles, TCL semivolatiles, TCL pesticides, and site specific indicator parameters. Groundwater sampled from the new wells associated with the South Ditch Study were not analyzed for the above parameters immediately following installation. In order to maintain consistency and a complete data set, groundwater samples from these wells were collected on December 17, 1996, and analyzed for the above parameters. In addition, field parameters were recorded during sampling. The results of these analyses are shown on Tables 2-19 to 2-22. The inorganic results from these wells are discussed in Section 2.7.6.2. The most recent results of major organics from these wells are discussed and summarized below:

12/17/96 (results in ug/l)	GW-76S	GW-77S	GW-78S	GW-79S
Acetone	nd	nd	8 J	13
Total TMPs	206	nd	3.7 J	6 J
Total Phthalates	1.7	0.5 JB	0.5 JB	1.3
NNDPA	12	nd	nd	2 J
Total Pesticides	0.295	nd	0.003 J	nd

J = Estimated value below quantitation limit
B = Parameter detected in associated blank
nd = Parameter not detected

The recent data indicates that GW-77S and GW-78S are contaminated with low concentrations of organics. GW-76S, which is located just off the west side property line near the West Ditch, has elevated concentrations of site-related organics and trace concentrations of pesticides. GW-79S, which is located near the Central Pond, is known to have high concentrations of inorganics (see Section 2.7.6.2), but has relatively low concentrations of organic parameters.

Miscellaneous Wells Off the Olin Property

In May 1997, additional groundwater samples were collected from five wells located at the periphery of the study area: GW-75S and GW-75D to the south, and GW-57D, GW-67S, and GW-67D to the north. These samples were analyzed for the indicator parameters and a select list of metals. The analytical results are included on Table 2-22. The concentrations of parameters in GW-57D, GW-67S, GW-67D, and GW-75S appear unrelated to the Olin property. Ammonia was detected in GW-75D at an elevated concentration of 46.8 mg/l. The high concentration of ammonia in the absence of elevated concentrations of the other indicator parameters associated with the Olin suggests that the ammonia may not be related to the Olin site. Other nearby potential sources may include the inactive municipal landfill for the City of Woburn. Therefore, the source of the ammonia is not clear.

2.5.5 Installation and Sampling of New Wells

Three groups of new monitoring wells were installed during the Supplemental Phase II Investigation. Four shallow wells were installed as part of the South Ditch study discussed in Section 2.4. Six new wells were installed in order to define the current limits and potential migration of the dense layer. Three of these were overburden wells installed at the bedrock saddles leading to the Southwestern and Eastern Bedrock Valleys in order to determine if any dense layer had migrated into these valleys. In addition, three bedrock wells were installed in Southwestern, Eastern, and Western Bedrock Valleys to determine if fractures in the bedrock surface could be a method of transport of the dense layer beyond the limits of the valleys. Plate 2-2 shows the location of these wells labeled as the following:

Aquifer Monitored	Eastern Bedrock Valley	Southeastern Bedrock Valley	Western Bedrock Valley
Shallow Overburden		GW-81 S	
Deep Overburden	GW-80 D	GW-81 D	
Bedrock	GW-80 BR	GW-81 BR	GW-65 BR

In addition, the results of the terrain conductivity survey summarized in the Interim Update Report (Smith, 1996c), indicated several areas where conductivity anomalies were present in the Western Bedrock Valley. The anomalies coincided with the location of the deepest portion of the bedrock valley, as defined by the seismic reflection survey results, and potentially indicated the presence of the dense layer in this valley or the presence of various geologic conditions. In order to provide confirmation of whether the dense layer was present in these areas of the valley, it was proposed that additional monitoring wells be installed and groundwater samples collected (Smith, 1996c).

Five well clusters, GW-82 through GW-86, were installed at these locations to provide data on water quality across the saturated zone. Two to three wells screened at different intervals were installed at the locations as shown on Plate 2-2. The shallow wells were installed to provide water quality and groundwater head data. The intermediate monitoring points are representative of the zone which is pumped by the supply wells. The deep wells were installed to determine the water quality in the deepest portion of the aquifer. Also, the groundwater and potentiometric data obtained from these wells benefited the groundwater model. In addition, a sixth deep well, GW-87D, was installed at a localized bedrock low approximately 250 feet east of GW-86 to assure detection of the dense layer, if present.

2.5.5.1 Property/Wetlands Access

Most of the monitoring wells were located off the Olin property; therefore, Olin obtained access permission from property owners affected by the installation of wells. Due to the placement of some of the wells in the Maple Meadow Brook wetlands area, it was necessary to utilize specialized methodologies to access these well locations which minimized the potential damage and long term impacts to the wetlands. Smith retained K.W. Reese Inc., of Mercersburg, Pennsylvania to build temporary access ways to the well locations by emplacement of large wooden mats. These mats enabled heavy vehicles to drive to the well locations with minimal impact to the wetlands. This method was deemed the most economical and least destructive to the wetlands by the Wilmington Conservation Commission (WCC) and they granted their approval in August 1996.

Prior to well installation, the locations of the access routes were surveyed and adjusted to minimize impact to larger trees in the wetland. A representative of either Wetlands Preservation, Inc. or Smith supervised surveying and installation of the wetland mats to ensure minimal impact to the wetland. At the end of well construction, the mats were removed and the access ways were checked by a representative of Wetlands Preservation, Inc. (WPI) to assure minimal damage to

the wetland, and any problems were recorded. WPI provided the WCC with a letter documenting that all work monitored by WPI was completed in compliance with state regulations and the Order of Conditions. A final report documenting the regeneration of vegetation will be provided to the WCC at the end of the first growing season (Autumn 1997).

In order to ease access to the wells for sampling purposes, a foot path to the well heads will be maintained to ensure accessibility.

2.5.5.2 Monitoring Well Installation

For all monitoring well locations, the aquifer materials encountered during drilling were described and summarized in well logs provided in Appendix I. Characteristics such as color, grain size, sorting, and angularity of the material were recorded. In addition, during soil sampling with split spoons, blow counts were recorded. All soils and water generated during drilling were contained and stored at the Olin property for appropriate characterization and disposal. In addition, the drilling rig and the down hole tools were steam cleaned prior to the start of each borehole and prior to leaving the Olin property.

Wells at Limits of the Bedrock Valleys

Smith contracted Maher Environmental, Inc., located in North Reading, Massachusetts, to install the additional overburden and bedrock monitoring wells at the limits of the bedrock valleys. These wells included: GW-65BR, GW-80D, GW-80BR, GW-81S, GW-81D, and GW-81BR. Both the overburden and bedrock monitoring wells were installed using a Barber Dual Wall Air Rotary drill rig. This method simultaneously advances a temporary steel casing while the borehole is being drilled through the casing with a standard air/water rotary drill. The casing is rotated while being driven into the borehole, making the advancement easier. Soil samples were collected at 5-foot intervals, using a 2-inch split spoon sampler, during the drilling of the deep overburden wells; split spoon samples were not collected at the other monitoring wells in each cluster due to their close proximity to the deep monitoring wells. The temporary steel casing used for the overburden drilling was 8-inch inside diameter (ID) which allowed the construction of 4-inch ID polyvinyl chloride (PVC) wells.

The new monitoring wells were constructed in a manner similar to the pre-existing monitoring wells and construction details are provided on the well logs in Appendix I and summarized in Table 2-2. For overburden wells, once the boreholes were advanced to the desired depth, the well was constructed using 4-inch Schedule 40 PVC with threaded flush joints. Ten feet of factory slotted screen with 0.02-inch slots was installed at the bottom of the borehole. The screen was threaded to Schedule 40 PVC riser (4-inch) which extended to approximately 2 feet above grade. The annular space around the screened interval was backfilled with an appropriately graded silica sand pack as the temporary casing was withdrawn. Care was taken to assure that the sand pack did not bridge in the casing or that the temporary casing was withdrawn too far ahead of the sand pack. The sand pack was extended to approximately 2 feet above the top of the well screen. A 2-foot bentonite pellet seal was then placed above the sand pack. The remaining annular space

was tremie grouted to grade with a cement/bentonite grout mixture. Once the grout was in place, the temporary steel casing was completely removed and additional grout was added as necessary. Due to their location in asphalt pavement, the GW-80 and GW-81 clusters were completed with a flushmount protective locking steel cover and cement pad. The other wells were completed with a locking stick-up protective casing.

Installation of the bedrock wells was accomplished by drilling to bedrock using the above method. Once the temporary steel casing (12-inch) was set to the top of the bedrock, the borehole was advanced approximately 10 feet into the bedrock to ensure a good seal between the overburden and bedrock aquifers. An 8-inch Schedule 80 PVC casing was inserted into the borehole and tremie grouted in place using a cement/bentonite grout. Prior to continued drilling of the open bedrock hole, the grout was allowed to set overnight. A 6-inch open hole was drilled into the bedrock using the air rotary method. The borehole was advanced into the bedrock a sufficient distance to obtain the necessary water yield for groundwater sampling purposes. A steel protective casing with a locking cap was placed over the well head and grouted in place at GW-65BR and flushmount protective covers were installed at the other two bedrock well locations.

Wells in the Western Bedrock Valley

Fourteen monitoring wells were installed at six locations within the Maple Meadow Brook wetlands and include the following wells:

GW-82 S	GW-84 S	GW-86 S
GW-82 D	GW-84 M	GW-86 M
GW-83 S	GW-84 D	GW-86 D
GW-83 M	GW-85 M	GW-87 D
GW-83 D	GW-85 D	

Locations of these wells are shown on Plate 2-2. Boreholes were advanced using the same Rotasonic drilling method employed during installation of the Main Street and Chestnut Street borings (described in Section 2.1.3). Installation of the wells began on September 22, 1996 and was completed on November 24, 1996. Two to three wells were installed at each location in order to monitor different zones of the aquifer: shallow, intermediate, and deep. In general, a deep monitoring well was constructed to the bottom of the unconsolidated zone and extended slightly into the bedrock. Then, an intermediate depth well was installed near the middle of the saturated zone, and finally, a shallow well was installed just below the swamp deposits. Because the depth to bedrock was shallower at some locations and since the thickness of the swamp deposits in the wetlands was much greater than anticipated, either the intermediate or shallow well was not installed at some well clusters. Details about exact well construction are included on the logs in Appendix I and summarized in Table 2-2.

The deep monitoring wells were drilled first and located downgradient of the intermediate and shallow locations. Once each borehole was extended to the top of the bedrock, several soil samples were collected for geochemical evaluation (see Section 2.7). The borehole was then

advanced at least 5 feet into the bedrock to ensure accurate identification of the bedrock surface. Each monitoring well was constructed using 2-inch ID schedule 40 PVC screen and casing with threaded, flush joints and capped with a sanitary seal. The screen was typically 10 feet long and factory slotted with 0.01-inch slot openings. This configuration maintains consistency with the other deep monitoring wells in the unconsolidated sediments throughout the site. The annular space around the well screen was backfilled to approximately 3 feet above the well screen with an appropriately graded clean silica sand. A bentonite pellet seal was emplaced above the sand pack and the remainder of the annular space was tremie grouted to the ground surface with a cement/bentonite grout. The temporary steel casing was withdrawn as the sand pack, bentonite, and grout were emplaced.

The intermediate monitoring well was installed at all of the locations except GW-82. A relatively shallow depth to bedrock at GW-82 allowed a significant portion of the aquifer to be monitored with just two wells. In general, the intermediate well screen was located near the middle of the saturated zone in the unconsolidated sediments. The exact depth and length of the intermediate well screen depended upon depth to bedrock, thickness of swamp deposits, and the depth and thickness of any observed intervals of preferential flow. Once the borehole was advanced to the appropriate depth, several soil samples were collected for geochemical analysis (see Section 2.7). The wells were then constructed with the same materials as described in the deep well, except at some locations a 0.02-inch slot was used instead of the smaller slot because the aquifer contained coarser grained material. An appropriately graded sand pack was used depending on the well slot size.

At most locations a shallow monitoring well was installed a few feet below the swamp deposits. At GW-85, the muck layer was in excess of 30 feet thick; therefore a shallow well was not installed because the screen would be only a few feet from the intermediate well screen. Several soil samples were taken at the base of each borehole for geochemical analysis (see Section 2.7). Shallow wells were installed using the same materials as the deep wells, including a 10-foot, 0.01-inch slot screen.

Once the temporary casing was removed from the ground, a protective steel casing was advanced into the ground around the well. The length of the protective casing advanced into the ground varied from 10 to 30 feet, depending on the thickness of the swamp deposits. After development, additional protective casing and riser were attached to each well to ensure sufficient stick-up above the surface water within the wetland. Enough PVC riser was added to the well to raise the top of the well above the 100-year flood plain, at an elevation of 86 feet MSL. Since the ground surface throughout most of the wetland is at an elevation of approximately 79 to 80 feet MSL, a 7- to 8-foot stick up was required. The annular space between the well and the protective steel casing was filled with the cement/bentonite mixture. Small holes were drilled approximately one foot beneath the top of the steel casing to allow for drainage of any water which may accumulate within the well cap. The wells were fitted with locking caps and secured with a lock.

In order to minimize damage to the protective casing caused by the freeze-thaw effect, a freeze collar was emplaced around each well. The freeze collar consists of a 5-foot piece of 10-inch PVC, which was pounded several feet into the ground. It is designed to isolate the permanent protective casing from ice that forms on the ponded water in the wetland during the winter months.

2.5.5.3 Well Development

After completion, each well was developed by either pumping or the air lift method to remove any fine sediment from the well. The development continued until little turbidity was observed. Specific conductance and pH were monitored during development of the wells in the wetlands to assure the system was at steady-state. Water generated during well development was contained and stored at the Olin property for proper classification and disposal.

At the end of development of the wells installed in the Maple Meadow Brook wetland, an unfiltered groundwater sample was taken for laboratory analysis of sulfate. At GW-83D, this sample was also analyzed for chromium. Analytical results from these samples are included in Table 2-22.

The first groundwater samples taken from GW-81D, GW-81BR and GW-65BR showed pH readings near 11. These pH readings are higher than normal and are potentially the result of inadequate development or residual grout left over in the well from installation. These three wells were re-developed for a minimum of three hours in order to remove the residual fine sediment and grout from the well. Following re-development, the wells were re-sampled.

2.5.5.4 Groundwater Sampling

After the monitoring wells were installed and developed, a preliminary round of groundwater sampling was performed at all of the new wells at least two weeks after development. Groundwater samples were analyzed for pH, specific conductance, and temperature in the field and sent to a laboratory for analysis of TCL volatiles, TCL semi-volatiles, TCL pesticides, TAL metals and indicator parameters. Each of the wells was re-sampled in January 1997; samples were analyzed for the indicator parameters and major ions. Groundwater analytical results are summarized in Tables 2-19 to 2-22.

Volatile organic compounds were detected in the groundwater beneath Maple Meadow Brook. Concentrations are typically highest in the deep wells of the GW-83, GW-84 and GW-85 clusters. Many of the organics in these wells are not associated with operations at the Olin property (such as trichloroethene and tetrachloroethene), or may have originated from a number of sources (acetone, toluene). However, trimethylpentenes were detected at low concentrations (less than 5 ug/l) in GW-84D and GW-85D. In the other wells, fewer organics were detected, and typically at significantly lower concentrations than in GW-83D, GW-84D, and GW-85D.

The results of the analyses for inorganics show that all the deep wells except GW-82D contain elevated concentrations of indicator parameters. The concentrations in GW-83D are consistent with the presence of the dense layer:

Chromium	0.074 and 0.087 mg/l
Chloride	4690 and 4570 mg/l
Ammonia	1650 and 1150 mg/l
Sulfate	approximately 17,000 mg/l
Specific Conductivity	33,900 to 37,000 mg/l

The detected concentrations of sulfate ranged from 3,190 mg/l to 17,600 mg/l. Based on the measured specific conductance and a calculation of the charge balance for the individual samples, it is believed that the higher results (approximately 17,000 mg/l) are more accurate.

Concentrations of inorganics in the other deep wells are not as high as in GW-83D and do not suggest the presence of the dense layer at these wells. Chromium was not detected at any of these wells. The following values of specific conductance were measured:

GW-84D	5950 to 8540 umhos
GW-85D	4440 to 4760 umhos
GW-86D	2890 to 4050 umhos
GW-87D	4220 to 4620 umhos

Concentrations of indicator parameters in GW-82D and all the shallow and intermediate wells were significantly lower than measured in the other deep wells. For example, the following ranges in concentration were observed:

Sulfate	16 mg/l in GW-86M to 265 mg/l in GW-82D
Chloride	52.5 mg/l in GW-82S to 156 mg/l in GW-84M
Ammonia	not detected in GW-84M to 31 mg/l in GW-83M
Specific conductance	269 umhos in GW-86M to 1070 umhos in GW-83M
Chromium	not detected

The data from these wells show that groundwater contamination is present beneath Maple Meadow Brook. However, it is significantly restricted to the deeper portions of the aquifer, and does not seem to be a source of high concentrations of contaminants to the shallower portions of the aquifer.

In April 1997, a regular monitoring of certain of these wells was initiated including GW-84M, GW-84D, GW-85M, GW-85D, GW-86M, and GW-86D. The results from the sampling in April 1997 are included on Table 2-22 and are similar to previous results. These wells are currently scheduled to be sampled monthly through July 1997 and quarterly thereafter.

2.5.6 Municipal Supply Wells

More intensive groundwater monitoring in the area of the supply wells was designed to complement the existing data and provide additional data necessary to evaluate the use of the existing monitoring well network and to forecast the expected concentrations of indicator parameters at the municipal supply wells in the Maple Meadow Brook wellfield.

2.5.6.1 Monthly Sampling and Analysis Program

In order to assess the correlation between analytical results from the municipal supply wells and the nearby monitoring wells, a monthly groundwater sampling program was established. The sampling program was initiated in July 1996 and is scheduled to be completed in July 1997. As part of the study, groundwater samples were collected monthly at the following locations:

Town Park	GW-73S and GW-73D
Butters Row No. 1 and No. 2	GW-64S and GW-64D
Chestnut Street No. 1 and 1A(2)	GW-63S and GW-63D
	GW-62D, GW-62M, GW-62D

Groundwater samples were collected from municipal supply wells when pumping, and were not collected during periods of non-pumping to assess potential variations in groundwater quality during typical operational cycles. All groundwater samples were analyzed for site-specific indicator parameters (chloride, sulfate, chromium, ammonia, specific conductance, and pH) and additional parameters (calcium, sodium, magnesium, and alkalinity) to provide quality assurance on the laboratory analyses. The accuracy of the data was assessed by collecting and submitting blind field duplicate samples for at least two of the supply well/deep monitoring well pairs and two of the upgradient wells during the first two months of the sapling program. The frequency of field duplicates after the first two months was decreased to one blind duplicate.

The monitoring program will be evaluated after one year and if necessary, appropriate changes will be made to the program. Analytical results collected from the program to date (April 1997) are summarized in Table 2-23. Graphs showing concentrations of indicator parameters over time at each of the supply wells are shown in Figures 2-30 through 2-34.

The data collected to date typically show concentrations of indicator parameters within the range observed historically at these wells. For example, prior to July 1996, concentrations of sulfate in Butters Row No. 1 ranged from 43 to 220 mg/l. Between July 1996 and March 1997, concentrations ranged from 91.7 to 168 mg/l. Exceptions to this are as follows:

- At Butters Row No. 1, concentrations of ammonia seem to be somewhat elevated for a few of the sampling events.

- At Chestnut Street No. 1, concentrations of chloride are slightly higher than previously observed for many of the sampling events.

- At Chestnut Street No. 2, concentrations of chloride are also slightly higher than previously observed for a few of the sampling events. In addition, ammonia concentrations are slightly higher for several of the samples.

- At Town Park, concentrations of chloride for two sampling dates are less than one-half of the lowest concentration previously detected.

At this point, no attempt has been made to interpret the significance of any of the results from the supply wells. The data show that concentrations fluctuate, and that there is no obvious, consistent, long-term trend in the group of indicator parameters. However, without the entire year of monitoring, seasonal effects may not be apparent in the limited data set. The data from the supply wells in conjunction with monitoring well data will be analyzed and interpreted after the final scheduled sampling in July 1997. It is anticipated that a report on these results will be submitted to the MADEP in September 1997.

2.5.6.2 Monthly Groundwater and Surface Water Level Monitoring

Methods

The groundwater level and, in the case of the supply wells, the pumping rate at the time of sampling could influence the concentration of indicator parameters. Groundwater levels were recorded prior to sampling each monitoring well, and pumping rates were obtained from the Wilmington Water Department for periods prior to and at the time of sampling. In addition, groundwater levels at GW-64S, GW-65S/D and GW-61S/61D are continuously monitored using pressure transducers attached to a data logger. This data will be used to understand how the groundwater levels fluctuate seasonally and prior to the sampling events. The surface water level in Maple Meadow Brook, adjacent to GW-65D (at the former canal) is also being recorded to assist in the understanding of the interaction between surface water and groundwater levels.

Historic precipitation data was collected to determine how the magnitude of precipitation affects the groundwater and surface water levels. These data will be used to determine exactly which months show the greatest and least amount of precipitation and the magnitude of the fluctuation of precipitation between the seasons. These data will also be useful to determine some of the aquifer hydraulic characteristics when compared to water level data, especially in correlation with storm events.

In order to observe how pumping rates may affect the concentrations observed at the town wells, historic pumping rate data was obtained from the Wilmington Water Department. This information will help determine if the concentration fluctuations are due to seasonal variations, pump rate variation, or some combination of these two variables.

Results

Charts of the groundwater level fluctuations at GW-61S/D, GW-64S, and GW-65S/D are shown on Figures 35 through 37, respectively. The monitoring period presented for GW-64S and GW-65S/D extends from October 10, 1996 through February 20, 1997 and the monitoring period for GW-61S/D extends from September 24, 1996 through November 13, 1996. One of the most striking features on all of the charts is an increase in groundwater elevations of approximately 5 feet on October 21, 1996 during a period of unusually intense rainfall (9.1 inches) which caused flooding of the Maple Meadow Brook wetlands. Field observations of the surface water level in Maple Meadow Brook during this time period indicated a surface water depth of approximately 5 feet at the outflow point at the old canal on October 22, 1996 which dropped by approximately 1 to 2 feet each day over the next several days. These observations demonstrate that the surface water level in the Maple Meadow Brook wetlands has short-term, but significant effects on the groundwater elevations in the monitoring wells surrounding the wetlands.

The groundwater level charts for GW-64S/D (Figure 36) and GW-65S/D (Figure 37) also include the average monthly pumping rate for the Butters Row municipal supply wells. As would be expected, it appears that the groundwater elevations in these wells also fluctuate by several feet in response to changes in municipal supply well pumping rates. At this point to determine the relative importance of these factors, since groundwater levels fluctuate due to both seasonal differences in precipitation/evapotranspiration and variable pumping rates at the municipal supply wells which reflect the seasonal demand for water. Final conclusions will be summarized in a separate report to be submitted to the MADEP at the end of one year of monitoring.

2.5.6.3 Time-Series Sampling

To further evaluate the effects that the pumping rates at the supply wells and well purging prior to monitoring well sampling may have on the variability of indicator parameter results, time-series sampling was conducted.

Method

A time-series sampling test for indicator parameters was conducted at the monitoring well adjacent to the supply wells with greatest variability in concentrations (GW-64D) on July 30, 1996. The well was pumped at a constant rate of approximately 0.5 gpm. This rate was chosen to simulate the purging typically used during past sampling events. Throughout pumping, a series of eight samples was collected (approximately one sample per hour) and submitted for analysis of indicator parameters. Groundwater levels and field parameters were also monitored during the time period. The field data and analytical results are summarized in Table 2-24.

In addition to the monitoring well, time-series sampling for indicator parameters was conducted from October 22 to 26, 1996 at the supply well with the greatest variability in concentrations (Butters Row No. 1. Butters Row No. 1 was pumped at several constant rates (150 to 400 gpm) within the range of historical operating conditions, and a series of five samples was collected (one

sample per different pumping rate) and submitted for analysis of indicator parameters. Field parameters were monitored during the time period. The field and analytical data are summarized in Table 2-25. Groundwater levels in adjacent monitoring wells were also recorded during pumping of Butters Row No. 1.

GW-64D Results

In order to evaluate trends, the analytical data and field data were plotted against volumes of water purged and the results are discussed in terms of the number of well volumes purged from the well. One well volume from GW-64D equals 11 gallons of water. These graphs are shown on Figures 2-38 and 2-39. Several significant features are evident from the time-series sampling at GW-64D.

Field parameters show the highest degree of fluctuation during pumping of GW-64D. Dissolved oxygen content in the groundwater drops significantly from 0.60 mg/l to 0.10 mg/l after purging seven well volumes and remains fairly stable at 0.10 mg/l until approximately 19 well volumes are purged. After 19 well volumes are purged, dissolved oxygen begins to drop again from 0.10 mg/l to 0.05 mg/l for the final reading of the test. Specific conductivity values show a similar decreasing trend, dropping significantly during purging of the first three volumes. Values at the start of pumping were 1,150 umhos compared to 780 umhos after three volumes were purged. Values continued to drop (but not as significantly) after three volumes, but began to stabilize after approximately 6 well volumes were purged. The values remained stable until the end of the test. Values for pH remained steady throughout the test.

In general, laboratory results fluctuated to a minor degree, but did not show significant increasing or decreasing concentration trends which correlate with the volume of water purged from the well. Sulfate concentrations ranged from approximately 195 to 226 parts per million (ppm) throughout the test. Ammonia concentrations varied from slightly more than 7 to 9 ppm throughout pumping and chloride concentrations fluctuated from 121 to 132 ppm. Chromium was not detected throughout the test.

Butters Row No. 1 Results

Prior to the start of the time series test, Butters Row No. 1 was not pumping for a period of approximately 3 weeks. An initial pumping rate of 150 gpm was used at the start of the test on October 22, 1996. It is important to note that the pumping rate was not constant and that, due to operational factors, the pump was turned-off for several hours each day of the test (Table 2-25). In order to evaluate trends, the analytical data, field data, and pumping rate recorded at the time of sampling were plotted against elapsed time of pumping. These graphs are shown on Figures 2-40 and 2-41. Several significant features are evident from the time-series sampling at Butters Row No. 1.

Specific conductivity values (Figure 2-41) show a decreasing trend, dropping significantly from the first sample (1,020 umhos) to the second sample (880 umhos) after 19 hours of pumping at approximately the same pumping rate. The specific conductivity values remained fairly stable during the next two days (at pumping rates of 180 and 300 gpm) then dropped again on the fifth day at the maximum pumping rate of 400 gpm. Dissolved oxygen content in the groundwater dropped significantly from 3.05 mg/l at the start of the test to 0.80 mg/l at the end of the test, although the dissolved oxygen value on the fourth day of the test, at a pumping rate of approximately 300 gpm, was 2.2 mg/l (Table 2-25). A comparison of the dissolved oxygen and temperature graphs on Figure 2-40 shows generally opposite trends, as would be expected. Values for pH remained steady throughout the test and ranged between 5.86 and 6.20 standard units.

In general, graphs of laboratory results for chloride and sulfate show decreasing concentration trends, while ammonia concentrations show no trend over time. Chloride concentrations were highest (130 mg/l) at the start of the test and dropped slightly in concentration (107 mg/l) by the end of the test. Sulfate concentrations were highest (179 mg/l) at the start of the test and dropped slightly in concentration (152 mg/l) by the end of the test, although the lowest sulfate concentration (130 mg/l) was observed on the fourth day of the test at a pumping rate of approximately 300 gpm. Ammonia concentrations varied during the test from slightly more than 12 mg/l to less than 5.5 mg/l. Chromium was not detected throughout the test.

2.5.6.4 Continuous In-Situ Monitoring

Specific conductance measurements have proven to be a useful qualitative indicator of increased concentrations of site-specific indicators. Therefore, an experimental downhole probe was installed in GW-64D to continuously monitor specific conductance. The purpose of this investigation was to observe short-term and long-term fluctuations of specific conductance which may indicate groundwater quality changes in the well. The objective of this monitoring program was to compare this data with the monthly analytical results to provide an indication of variability of these parameters between sampling events.

Methods

On February 19, 1997, a downhole probe manufactured by In-Situ was calibrated according to manufacturer instructions and was then installed in GW-64D adjacent to the well screen. The probe is designed to measure temperature and conductance and convert these readings to a specific conductance value which is then recorded by a data logger programmed to record measurements every six hours. The same data logger used to record groundwater levels from GW-64S was also used to record the conductance measurements from GW-64D; therefore, conductance measurements coincide with groundwater level measurements.

Results

After installation of the conductance probe, the initial reading was 1,500 umhos on February 19, 1997. A comparison of the subsequent readings and field measurements of specific conductivity from groundwater samples collected at GW-64D during the monthly sampling and analysis program (Section 2.5.6.1) shown below, indicate that the probe readings were not accurate and exhibited significant drift over the range of the probe during the period of monitoring. The conductance probe was removed from the well and sent back to the manufacturer for testing and repairs in April 1997. The continuous conductance measurements will be resumed if a more accurate monitoring device is available prior to completion of the one-year monitoring program in June 1997.

Date	Specific Conductivity Probe Reading (umhos)	Specific Conductivity Field Sample Measurement (umhos)
2/25/97	1,500	910
3/25/97	2,300	810
4/23/97	2,300	900

2.5.7 Domestic Well Sampling

Since 1990, Olin has contacted homeowners with domestic water supply wells on an annual basis. Where homeowners have allowed, Olin has obtained samples from these wells and submitted them for laboratory analyses for selected parameters. The data obtained since the CSA was submitted in 1993 are included in Tables 2-19 through 2-22. The samples are denoted on these tables by Wilmington Tax Assessor's Map (M) and Lot (L) number, for example, M-24/L-60. Additional information about these wells is provided in Section 3.4.6 concerning local groundwater usage.

Samples from domestic wells have been obtained on the following dates:

February and May 1994
July 1995
August 1996

The analytical data from the sampled wells has shown no impacts associated with contaminants originating at the Olin property. In 1996, methyl-tertiary-butlyether was tentatively identified in a number of the domestic water samples. This compound is not associated with the Olin property; it is commonly used as an additive to gasoline. These homeowners have been contacted by the MADEP to have their wells re-sampled.

2.6 ADDITIONAL CHARACTERIZATION OF DENSE LAYER

2.6.1 Specific Conductance Profiling

Due to the high dissolved solids content of the dense layer, which results in a high specific conductance, specific conductance measurements in monitoring wells were used, in addition to other groundwater data, for further delineation of the dense layer. Downhole specific conductance profiling was performed in selected monitoring wells between 1993 and 1995 to provide further definition of the limits of the dense layer to the west, east, and southwest of the facility. In addition, the specific conductance profiling was used to identify potential changes in the position of the dense layer by comparing measurements from several dates in selected wells. In addition, specific conductance profiling was used to determine the vertical extent of the dense layer and to correct hydraulic heads for density effects.

Specific conductance is directly related to the concentration of total dissolved solids (TDS) in the groundwater. Increased TDS concentrations result in greater electrical conductance in the groundwater. This relationship is shown in Figure 2-42, which is a graph of specific conductance versus TDS concentrations measured in site groundwater samples. As shown on this graph, there is a very good correlation between specific conductance values and TDS concentrations. The TDS concentrations are in turn directly related to the concentration of the major inorganic ions (ammonia, chloride, sodium, and sulfate) detected in groundwater samples at the site.

2.6.1.1 Methods

Three rounds of specific conductance profiling were performed at wells within and around the extent of the dense layer; in April 1993, January 1995 and October 1995. Collection of the specific conductance measurements occurred several days in advance of any groundwater sampling to ensure that the groundwater column in the well was not disturbed.

At each well, the depth to groundwater was measured and recorded. A downhole specific conductance probe was lowered into the well slowly, taking care to minimize disturbance to the water column. Specific conductance was measured at a maximum of 5-foot intervals from the top of the water column to a few feet above the screen. From a few feet above the screen, specific conductance was measured at increments ranging from a 1-foot maximum to 0.5-inch minimum. Specific conductance was measured with a YSI Model T-L-C (temperature, water level, conductivity) meter with a 150-foot cable. All specific conductance measurements were temperature corrected to 25° Celsius (C). After recording measurements from each well, the probe and cable were washed and rinsed to eliminate cross-contamination.

Specific conductance profiling was performed at the following wells (see Plate 2-2 for well locations) on the dates specified:

	APRIL	JANUARY	OCTOBER		APRIL	JANUARY	OCTOBER
	1993	1995	1995		1993	1995	1995
GW-4-D	X	X	X	GW-61-BR	X	X	X
GW-17-D	X	X	X	GW-62-D	X	X	X
GW-19-D	X	X	X	GW-62-BR	X	X	X
GW-27-D	X	X	X	GW-62-BRD	X	X	X
GW-34-D	X	X	X	GW-64-D	X	X	X
GW-40-D	X	X	X	GW-65-D		X	X
GW-42-S			X	GW-65-BR			X
GW-42-D	X	X	X	GW-67-D		X	X
GW-43-S			X	GW-68-BR	X	X	X
GW-43-D		X	X	GW-69-D	X	X	X
GW-44-S			X	GW-70-D		X	X
GW-44-D		X	X	GW-74-D		X	X
GW-49-D		X	X	GW-80-D			X
GW-50-D	X	X	X	GW-80-BR			X
GW-58-D	X	X	X	GW-81-D			X
GW-61-D	X	X	X	GW-81-BR			X

2.6.1.2 Results

After completion of the field measurements, Smith reviewed the temperature-corrected data and prepared plots of the results for each of the wells. Data from all measurement events were plotted on one figure for each well to allow a temporal data comparison. Specific conductance profiles and the temperature-corrected specific conductivity data for each measurement interval are included as Appendix K.

After generating plots of the profiles, Smith made note of the particular features of each plot. Important features of note included minimum and maximum conductance, general magnitudes of observable conductance spikes, location of the conductance spike in the well profile, and any general trends of the conductance profiles over time. Most of these features are summarized on Table 2-26. The conductance spike referred to on this table and discussed in this section is defined as the maximum increase in conductance between measurement intervals. The elevation of the conductance spike is defined as the elevation of the measurement interval where the maximum increase in conductance occurred on each survey date.

General Observations

In general, the maximum specific conductances measured in the wells ranged from 200 to 70,000 umhos. It should be noted that several of the profiles that show elevated conductance (>25,000 umhos) do not reflect exact readings but represent the maximum value at which the conductance

probe can effectively measure conductance. Based on groundwater analytical data, conductance values taken from wells set in the dense layer range from approximately 15,000 to 100,000 umhos.

As shown in the specific conductance profiles in Appendix K, several of the specific conductance spikes occur at the top of the well screen. These conductance spikes occurred regardless of the presence of dense layer within the aquifer materials and may be due to the interface of stagnant borehole water and aquifer water flowing through the well at the screen. At some well locations outside of the area of contaminant occurrence, the conductance spikes occurred at the bottom of the well screen. These conductance spikes are most likely caused by sediment accumulation in the bottom of the borehole. In general, the interpretation of the specific conductance profiles discussed in this section considered the well construction, magnitude of the conductance spikes, vertical location of the conductance spikes, and the results of groundwater samples, to determine lateral and vertical extent of the dense layer.

Lateral Occurrence of Dense Layer

The maximum specific conductance values from the vertical profiles summarized on Table 2-26 provide a preliminary indication of the occurrence of the dense layer. Based on water quality data, the following wells are known to be completed in the dense layer: GW-27D, GW-42D, GW-43D, GW-44D, GW-58D, GW-69D, and GW-70D. The maximum specific conductance in these wells ranged from 15,000 to 73,000 umhos and the conductance spike in the profiles for these wells was generally over 3,000 umhos (Table 2-26). Based on the range of specific conductance values in known dense layer wells, a specific conductance of above 15,000 umhos was used as a screening tool to identify wells that are potentially screened within the dense layer. The specific conductance values from the profiles of the following wells may indicate the presence of the dense layer or the presence of groundwaters impacted by the dense layer: GW-34D, GW-50D, GW-62BR, and GW-68BR.

The specific conductance profile data were also used to identify well locations where inorganic concentrations may be elevated, but at lower concentrations than what would be associated with the dense layer. Specific conductance profiles with values consistently ranging from approximately 1,500 to 15,000 umhos were used to identify the following wells that have elevated concentrations of inorganics: GW-4D, GW-17D, GW-19D, GW-40D, GW-43S, GW-62BRD, GW-64D, and GW-65BR.

Specific conductance profiling locations where conductance values were less than 1,500 umhos are generally outside of the site or in the cleaner shallow groundwater present above the dense layer. Groundwater analyses from these locations indicate concentrations of inorganic analytes that are consistent with background concentrations or not appreciably different from background concentrations. The locations include the following wells: GW-42S, GW-44S, GW-49D, GW61D, GW-62D, GW-65D, GW-67D, GW-74D, GW-80D, GW-80BR, GW81D, and GW-81BR.

Vertical Occurrence of Dense Layer

The specific conductance profiles provide an indication of the vertical extent of the dense layer when the dense layer interface is within the screened or open-rock portion of the well. The position of the conductance spike (measurement interval with the maximum conductance increase) relative to the well screen and bedrock interface, where applicable, is summarized on Table 2-26. The position of the conductance spike in known dense layer wells was consistently within the screened interval in wells GW-27D and GW-69D. The average elevation from the three surveys of the conductance spike was approximately 61.3 feet above mean sea level MSL at GW-27D and 49.9 feet MSL at GW-69D.

These locations, with the interface between the dense layer and the overlying lower TDS groundwater within the screened interval, also provide an indication of the nature of the interface. At GW-27D, the interface is very sharp with a conductance increase from less than 5,000 umhos to more than 25,000 umhos over a vertical distance of only several feet. At GW-69D the increase in conductance is more gradual, although the conductance increases by 3,000 to 9,000 umhos from 1-foot above to just below the dense layer interface.

Conductance data from within the solid well casing cannot be assumed to be representative of conditions in the aquifer outside of the well casing. Therefore, the portions of the specific conductance profiles in Appendix K, which are based on data within the solid well casing, should not be considered when evaluating the vertical extent and thickness of the dense layer. The position of the conductance spike in known dense layer wells was more frequently above the screened interval in wells GW-42D, GW-43D, GW-44D, GW-58D, and GW-70D. This data was used to correct hydraulic heads for density effects in wells where the dense layer interface was significantly higher than the screened interval of the well (Section 2.2.1).

Temporal Trends

The specific conductance profiles from individual wells were also compared over time to look for observable trends in the data. However, with the limited data set it is difficult to distinguish actual trends from seasonal variability and short-term variations created by purging for prior sampling events. In general, the shapes and magnitudes of the conductance profiles vary relatively little over time. There was a strong consistency between conductance profiles from the three measurement events performed in 1993 and 1995. Four wells, GW-40D, GW-50D, GW-61BR, and GW-62BRD, exhibited consistent decreases in the maximum conductance values, while two wells, GW-49D and GW-62D, showed consistent increases in the maximum conductance values. Temporal groundwater quality data are only available for GW-40D, GW-50D, and GW-62D. These data show no clear trends in inorganic concentrations at GW-50D and GW-62D and decreasing concentrations of chloride and ammonia over time at GW-40D. None of the wells monitoring the dense layer exhibited significant temporal changes. The maximum conductance above and within the screened interval for each measurement event are summarized on Table 2 - 26.

2.6.2 Terrain Conductivity Survey

An electromagnetic terrain conductivity survey was performed to aid in identifying and mapping the presence of inorganic contaminants in the buried bedrock valleys at the site. Terrain conductivity surveying utilizes the application of inductive electromagnetic techniques to directly measure apparent terrain conductivity. The term "apparent conductivity" is used because the measured value is a surface and/or depth-weighted average of conductivity beneath the measurement point. For convenience, apparent terrain conductivity is referred to simply as terrain conductivity throughout this report.

Terrain conductivity depends on the nature of the soil, subsurface porosity, hydraulic permeability, moisture content, depth to and type of bedrock, and the concentration of dissolved electrolytes. The presence of such objects as electric lines, wires, pipes, buildings, tanks, buried metal, fences, and foundations (cultural interferences), may also influence conductivity.

Most physical constituents of a soil are electrical insulators that do not allow appreciable current flow through the soil. The medium that does allow current flow is the relatively conductive interstitial moisture. Water conductivity is determined mainly by ionic content; therefore, variations in soil conductivity can be used to define areas where groundwater may contain a higher concentration of dissolved electrolytes. Due to its high concentration of dissolved solids, the dense layer at the site would have a significantly higher conductivity than the background groundwater.

In general, the conductivity of the earth varies with depth. In a typical vertical profile, the conductivity will initially increase with depth due to increasing soil moisture, becoming constant at the water table due to saturation. If the bedrock underlying soils has a low porosity, the conductivity within the bedrock medium will decrease. In addition, an increased clay content can affect conductivity because of a surface conduction phenomenon that occurs in clay.

2.6.2.1 Survey Methodology

Based on the bedrock morphology, as defined by available data from boreholes and the results of the seismic reflection survey, terrain conductivity survey lines were located to provide spaced cross-sections along the Western Bedrock Valley. The locations of the lines are shown in Figure 2-43. Two survey lines, Lines 10 and 12, were located in the area of the Southwestern Bedrock Valley but the locations of the lines were limited due to cultural interferences. Survey Line 1 was performed on the Olin property across a known area of groundwater contamination to provide a comparison of the conductivity levels with known groundwater concentrations. A terrain conductivity survey was not performed in the Eastern Bedrock Valley, due to cultural interference from the railroad and industries east of the property.

Lines performed in the Western Bedrock Valley required terrain conductivity surveying to be performed in the Maple Meadow Brook wetland. As with all work performed in the wetlands, any clearing of the low brush was performed in accordance with the conditions established by the Wilmington Conservation Commission and/or Woburn Conservation Commission, and was documented by a representative from Wetlands Preservation, Inc.

The terrain conductivity survey was conducted at the site from February 1 through 10, 1995, by Enviroscan Inc. of Lancaster, Pennsylvania, and Smith/BCM. Terrain conductivity surveying at the study area was accomplished using an EM-34-3DL (EM-34) Terrain Conductivity Meter manufactured by Geonics Limited of Mississauga, Ontario, Canada. A detailed discussion on the operation and calibration of the EM-34 is included with the Enviroscan report (Appendix L).

The EM-34 coils may be operated in a vertical configuration or may be laid horizontally on the ground surface during measurements. The orientation of the coils affects exploration depth. When the coils are held vertically, the secondary magnetic field is perpendicular to the plane of the receiver coil. This orientation is known as the horizontal dipole mode. When the coils are held horizontally, the secondary magnetic field is oriented perpendicular to the horizontal. This is known as the vertical dipole mode. Data were collected in both the horizontal and vertical dipole modes at stations spaced approximately 50 feet apart. The following table shows the exploration depths for the coil configurations used.

Intercoil Spacing Meters (m) Feet (Ft)	Exploration Depth Horizontal Dipole	Exploration Depth Vertical Dipole
10 m (33 ft)	0-30 ft	5-50 ft
20 m (66 ft)	0-65 ft	10-95 ft

Four measurements were taken at each station, including horizontal and vertical dipole configurations at 10- and 20-meter intercoil spacing. An Omnidata Series 500 data logger was used with the EM-34 to collect and record the data. The data were downloaded to a portable computer after completion of each line. During the survey, any cultural or environmental features which could result in variation of the data were noted for use in the data interpretation.

2.6.2.2 Terrain Conductivity Results

Terrain conductivity profiles were prepared for each survey line and are included in the report prepared by Enviroscan (Appendix L). In general, typical terrain conductivity values for clean water in saturated sand range from less than 8 to 10 millimhos per meter (mmhos/m) (McNeill, 1980b). The terrain conductivity profiles were examined for the presence of anomalies (areas of terrain conductivity greater than 10 mmhos/m), which could indicate the presence of conductive groundwater. Also, negative terrain conductivity values indicate high conductivity beyond the dynamic range of the instrument; therefore, negative values are also considered anomalous data.

A summary of the data ranges and any observable anomalies are presented in Table 2-27. Conductivity values from Lines 10 and 12, performed in the Southwestern Bedrock Valley, and Line 9, are extremely variable. This is most likely due to cultural interferences caused by overhead wires and buried debris. The conductivity values along the north end of Line 12 may show the presence of a conductivity anomalies dependent on the coil orientation. The negative conductivity values were measured on Line 12 around 850 feet using the 20-meter vertical dipole (20V) configuration and elevated conductivity values were measured around 950 feet using the 10 meter horizontal and vertical (10H, 10V) and 20 meter horizontal (20H) dipole configurations. These anomalies may be associated with either cultural interferences or increased inorganics in the groundwater. As shown on Table 2-27, anomalies that may be attributed to known cultural and surficial features are marked and disregarded for the interpretation. The remaining anomalies are likely caused by subsurface features. These anomalies, with the exception of Line 1, occur within the Western Bedrock Valley.

As expected, the highest observed conductivity (i.e., approximately 20 to 25 mmhos/m) attributed to the dense layer were observed on the Olin property along Line 1 from 0 feet through 200 feet for all coil orientations and from 550 feet through 700 feet for the 10H and 20H coil orientations. The decrease in conductivity between these locations corresponds with a bedrock high where the dense layer is probably absent. Conductivity values from Lines 2 through 8 ranged from a minimum of 2 mmhos/m to a maximum of 19 mmhos/m. In general, most of the lines were within the normal conductivity range for clean saturated sands

Figure 2-43 shows the spatial distribution of terrain conductivity anomalies that exceeded background (or have negative conductivity values) and that were confirmed by groundwater quality data, and were unrelated to cultural or topographic interferences. The terrain conductivity anomalies presented on Figure 2-43 show a broader area of anomalies than previously interpreted in the Interim Update on Supplemental Phase II Field Investigations report (Smith, 1996c). The new interpretation is based on additional groundwater quality data from new wells (GW-83D through GW-87D) and the additional seismic velocity data which further defined the bedrock surface topography (Plate 2-1). The locations with terrain conductivity anomalies correspond to the areas where bedrock is the deepest, as indicated by the seismic reflection survey. The elevated conductivity values appear to be mainly caused by the presence of elevated concentrations of inorganics, although an increase in the thickness of overburden materials due to bedrock lows, an increase in clay content in the overburden materials, and the presence of water bearing fractures within the bedrock surface may also contribute to the elevated or negative conductivity values.

2.6.3 Cone Penetrometer Testing

Electronic cone penetrometer testing (CPT) was performed at six locations within the area of occurrence of the dense layer, to examine the interface between the dense layer and the overlying, relatively unimpacted groundwater. The objective of the CPT was to determine if this interface is relatively sharp (indicating little dispersion), or wide and gradational (indicating greater dispersion of the dense layer). CPT was selected because it permitted vertical delineation of the dense layer with minimal interference to the system being studied.

2.6.3.1 Methods

The CPT survey was initially attempted in June 1995 using a 30-ton, track-driven CPT rig operated by Conetec Inc. of Warren, New Jersey. After attempting to advance the cone at several locations, testing was aborted due to premature refusal caused by large numbers of boulders.

The CPT survey was then performed between September 19 and 22, 1995, by Applied Research Associates, Inc. (ARA) of South Royalton, Vermont. The cone was advanced using a drilling rig contracted to Maher Environmental of North Reading, Massachusetts. The CPT locations are shown on Figure 2-1.

The September 1995 CPT survey was performed using a standard piezocone with added resistivity and pH modules. The resistivity and pH modules were attached in the drive string, immediately above the piezocone. The standard piezocone measures tip resistance, sleeve friction, and pore pressure, which provide an indication of different lithologics encountered. The resistivity module provides a continuous profile of bulk resistivity with depth. This was used to locate the interface of the dense layer, which is highly conductive due to the elevated level of dissolved solids. The pH module provides a continuous record of the pH of the pore fluids. This module was used because the dense layer is also characterized by a low pH.

The piezocone and related modules and drive rods are constructed primarily of stainless steel. The drive rods have an outside diameter of 1.375 inches and the piezocone and attached modules have a slightly larger outside diameter. Rods were added in 5-foot lengths as the CPT was advanced. Electrical wiring to the cone and associated modules extends through the center of the hollow rods to the surface.

A truck-mounted drilling rig was used for advancing the cone penetrometer into the subsurface. The drilling rig was used in place of a standard CPT truck to provide the capability to drill through cobble/boulder zones that would result in cone refusal. After set up of the drilling rig at the desired location, the cone was advanced until refusal was encountered. The cone was then withdrawn from the hole and 3-inch ID temporary steel casing was advanced through the zone of impasse. The cone penetrometer was then reinserted through the casing and advanced until completion of the hole or refusal. The data record over these intervals has gaps where the difficult zone was drilled through prior to the resumption of cone penetration. Due to the difficulty encountered advancing the cone penetrometer, the CPT holes did not all extend to the bedrock surface. The maximum penetration depth after several attempts during each survey are summarized as follows.

MAXIMUM CPT PENETRATION DEPTH
(feet below ground surface)

Location	June 1995 Survey	September 1995 Survey
CPT-1	31	25
CPT-2	25	31
CPT-3	--	32
CPT-4	--	30
CPT-5	--	31
CPT-6	28	26

-- Not surveyed on this date

During the advancement of the cone penetrometer, the tip resistance, sleeve friction, pore pressure, resistivity, and pH were logged continuously and stored in a computerized data logger. Selected information was displayed on a computer monitor to allow real-time data observation.

When the CPT was performed in September 1995, soil samples were collected from selected depths in boreholes drilled adjacent to the CPT holes. These soil samples were submitted for laboratory analyses by Geomega. The specific samples collected and the analyses performed are described in Section 2.7.3.

After the completion of each CPT hole, the cone penetrometer was removed from the hole and a cement/bentonite grout mixture was emplaced in the hole from the bottom upward using a tremie line. The temporary steel casing was removed as the grout was pumped up to the ground surface.

All downhole equipment was decontaminated at the onsite decontamination pad prior to starting the first hole and after completion of each hole. Decontamination water and drilling cuttings were contained in 55-gallon drums and stored at the onsite warehouse until appropriate offsite disposal was arranged.

Graphical logs of each of the parameters recorded during the cone penetrometer testing are provided for each location. Lithologic logs interpreted from the June 1995 CPT data are included with the CPT logs in Appendix M. The lithologic descriptions are interpreted by an algorithm that uses all of the information from the CPT to assess the lithologies encountered. The CPT logs for the September 1995 survey showing tip resistance, sleeve friction, pore pressure, and pH are also included as Appendix M.

2.6.3.2 Results

Due to the limitations of the CPT data caused by the presence of cobbles and boulders in the subsurface, which resulted in the need to predrill significant portions of the vertical section and/or resulted in refusal, three multilevel piezometers were installed (Section 2.7.4). The multilevel

piezometers were installed to provide better definition of the vertical differences in chemical composition above and within the dense layer. A comparison of CPT results with the multilevel piezometer groundwater analytical results are discussed in this section.

The results of the pH measurements performed during the September 1995 CPT program appear to be suspect and do not correlate well with the pH values measured in samples from the multilevel piezometers (MP-1 through MP-3). Therefore, the pH data are not discussed in detail in the remainder of this section.

CPT-1

Between approximately 26 and 30 feet bgs, the resistivity surveyed at CPT-1 in June 1995, decreased from more than 150 ohm-m to approximately 25 ohm-m. This decrease in resistivity correlates well with the conductance increase observed in multilevel piezometer MP-1, which is near the location of CPT-1. This zone reflects the top of the dense layer, which has a significantly higher conductance than the zone with relatively unimpacted groundwater.

The resistivity measured at CPT-1 during the September survey was generally slightly under 10 ohm-m for the interval measured. These values are relatively comparable to those from the June 1995 measurements, but are slightly lower. At a depth of approximately 15 feet bgs, the resistivity was less than 10 ohm-m, which is considerably lower than the same zone from the June 1995 survey. Resistivity then increased below approximately 15 feet bgs which correlates well with the observed decrease in conductance at the same depth in MP-1. This data suggests that the zone above 15 feet bgs represents a slightly more contaminated zone in the shallowest groundwater.

CPT-2

Resistivity within the saturated zone surveyed at CPT-2 in June 1995 ranged from more than 1,000 ohm-m to less than 25 ohm-m. The resistivity was generally between 500 and 600 ohm-m, except for a spike of higher resistivity at approximately 7 feet bgs which is probably due to lithologic differences and/or the degree of saturation. At approximately 22 feet bgs, the resistivity decreased from more than 500 ohm-m to less than 25 ohm-m. This resistivity decrease is offset by several feet as compared to the location of the conductance increase representing the top of the dense layer in MP-2, and is probably associated with the occurrence of cone refusal and the cessation of data collection at this point.

The resistivity measured at CPT-2 during the September survey was generally between 200 and 300 ohm-m below 12 feet bgs. These values are relatively comparable to those from the June 1995 measurements, but are slightly lower. In September, between 11 and 12 feet bgs, the resistivity was less than 50 ohm-m, which is considerably lower than the same zone from the June 1995 survey, although the trend is similar. The observed increase in resistivity at approximately

12 feet bgs correlates well with the observed decrease in conductance at the same depth in MP-2. The zone above 12 feet bgs appears to represent a slightly more contaminated zone in the shallowest groundwater.

CPT-3

CPT-3 was not surveyed during the June 1995 survey program. The resistivity measured at CPT-3 during the September survey was generally under 10 ohm-m. Resistivity data started at a depth of 14 feet bgs due to the need to drill and case off the upper zone. At 15 feet bgs there was a resistivity peak of approximately 100 ohm-m, followed by a rapid decrease to less than 1 ohm-m at 17 feet bgs. This resistivity low, which represents a higher conductance, may indicate a slightly contaminated zone or lithologic changes. The resistivity below this point returns to approximately 10 ohm-m, which is comparable to the resistivity values observed in CPT-1 and CPT-2 where there was a slightly more contaminated zone in the shallow groundwater. The resistivity decrease at 16 feet bgs is too shallow to represent the top of the dense layer. Below this point, the change in resistivity is too minor to be indicative of the dense layer, which would indicate that the top of the dense layer is below the bottom of CPT-3 (31 feet bgs), consistent with the nearest multilevel piezometers.

CPT-4

CPT-4 was not surveyed during the June 1995 survey program. The resistivity data from CPT-4 during the September survey started at a depth of 13 feet bgs due to the need to drill and case off the upper zone. At a depth of 14.5 feet bgs, there was a resistivity low which decreased to approximately 2 ohm-m. This resistivity low, which represents a higher conductance, may indicate a slightly contaminated zone or lithologic changes. From 15 feet bgs to the bottom of CPT-4, the resistivity was generally between 300 and 400 ohm-m. This resistivity data would tend to indicate the presence of a relatively clean groundwater zone, which, if the CPT hole had extended to a greater depth, would be underlain by the dense layer. Data from MP-3, which is located adjacent to CPT-4, indicated that the dense layer occurs approximately 10 feet below the deepest penetration of CPT-4.

CPT-5

CPT-5 was not surveyed during the June 1995 survey program. The resistivity measured at CPT-5 during the September survey was limited to a zone approximately 24 to 28 feet bgs due to the need to drill and case of zones where cobbles and boulders were encountered, resulting in cone refusal. The resistivity in this zone is generally increasing with depth and ranges from 10 ohm-m to slightly more than 100 ohm-m.

CPT-6

The resistivity measured at CPT-6 during the June 1995 survey program was generally between 300 and 500 ohm-m, except for a spike of higher resistivity at approximately 7-feet bgs, which is probably due to lithologic differences and/or the degree of saturation. At approximately 26 feet bgs, the resistivity decreased from 500 ohm-m to less than 100 ohm-m. This resistivity decrease is significantly shallower than would be expected for the top of the dense layer at this location, and is probably associated with the occurrence of cone refusal and the cessation of data collection at this point.

The resistivity measured at CPT-6 during the September survey was limited to a zone approximately 18 to 24 feet bgs due to the need to drill and case of zones where cobbles and boulders were encountered, resulting in cone refusal. The resistivity in this zone is variable, and ranges from less than 10 ohm-m to slightly more than 100 ohm-m.

Summary of CPT Results

The CPT yielded useful resistivity and lithologic data. However, the vertical extent of the data collected was greatly limited by the extremely difficult subsurface conditions, which included widespread occurrence of cobbles and boulders. The resistivity data from CPT-1 and CPT-2 correlate well with the conductance data from MP-1 and MP-2, indicating the occurrence of a zone of slightly higher contamination in the shallow groundwater, which is underlain by cleaner groundwater and the dense layer. The pH data collected during the September 1995 survey have indicated zones of low pH in all of the survey locations, which would be consistent with conditions in the dense layer. However, the vertical position of the low pH zones does not correlate well with the anticipated depth of the dense layer and the field pH data from the multilevel piezometers.

2.6.4 Inductance Logging

Downhole electromagnetic inductance logging (also known as downhole conductance logging) was performed in selected wells to determine the position of the dense layer interface. The inductance logging was performed by lowering a small probe down the monitoring well casing. An electric current in the transmitter coil, which is located near the top of the probe, induces an electric eddy current in the formation adjacent to the well. The induced current produces a secondary electromagnetic field which is measured as a voltage on the receiver coil located near the bottom of the probe. The system is designed so that the voltage in the receiver coil is proportional to the aquifer water conductance. This method "sees" through the PVC well casing, measuring the conductance of the aquifer out to a maximum distance of approximately 2 feet from the center of the well. The focus of the measurement is toward the outer edge of the induced secondary electromagnetic field and is less influenced by the fluid in the well. Pre-purging and post-purging inductance logging conducted during the CSA (CRA, 1993) showed similar results

and demonstrated that the measurements are not appreciably affected by the position of the dense layer in the well. The probe does not have to be immersed in water to obtain accurate data, but would not provide accurate results in the steel-cased upper portions of the wells.

Inductance logging has been effectively used at the site to determine the position of the dense layer interface. The vertical profiles generated by previous inductance logging in the CSA (CRA, 1993) at GW-42D and GW-44D correlated very well with the water quality data generated from the multilevel piezometers. Since the sand and gravel in the aquifer are not very conductive, the measured conductance is mainly determined by ionic content in the pore water. The variations in conductance can be used to vertically define the depth intervals where groundwater may contain a higher concentration of dissolved electrolytes. Due to its high concentration of dissolved solids, the dense layer at the site has a significantly higher conductance than the background groundwater.

The purpose of inductance logging was to delineate the dense layer interface and to determine the thickness and extent of the dense layer. The information collected also provides an indication of water quality variations in the overlying groundwater and provides a baseline data set for the vertical position of the dense layer in the study area.

2.6.4.1 Methods

Inductance logging was performed by Hager-Richter Geoscience Inc. between February 25 and March 12, 1997. Inductance logging was performed on the following 24 monitoring wells:

GW-22D	*GW-36	*GW-44D	*GW-59D	GW-84D
GW-27D	GW-37	*GW-45D	*GW-69D	GW-85D
GW-30DR	GW-38	GW-50D	*GW-70D	GW-86D
GW-34D	*GW-42D	GW-55D	GW-82D	GW-87D
GW-35D	*GW-43D	*GW-58D	GW-83D	

These locations were selected to provide a representation throughout the area of occurrence of the dense layer or immediately adjacent to the dense layer. The locations of these monitoring wells are shown on Plate 2-2. Nine wells (shown with * above) were logged previously by Hager - Richter Geoscience, Inc. during the CSA (CRA, 1993). The wells were logged again to facilitate a comparison and correlation with the earlier data and determine if there has been a change in the position of the dense layer.

The well conductance data were obtained with an MGX II logging system and an electromagnetic induction probe. The probe was calibrated to read formation conductance directly in mmho/m, (numerically equivalent to the SI units, millisiemens per meter, mS/m) with a resolution of 0.02 percent of full scale, a repeatability of +/-2 percent of full scale, and an accuracy of 5 percent of full scale. The data were recorded at intervals of 0.1 feet while the tool was being pulled out of the monitoring well at a rate of about 10 feet per minute. All logs were run with the same instrument settings (scale range 0-1,000 mmho/m) and are presented at two scales: 0 to 50

mmho/m and 0 to 400 mmho/m (see Appendix N). The portions of the inductance logs with values greater than 400 mmhos/m wrap around the value scale so that the scale for these portions of the log is 400 to 800 mmhos/m. Between wells, the downhole equipment was decontaminated by washing the cable and tool with Alconox soap and rinsing with distilled water.

2.6.4.2 Data Evaluation and Results

After completion of field measurements, Smith evaluated the inductance logs to determine depth of the dense layer interface (if present). The presence of the dense layer was determined by comparing the peak conductance of each well logged with the range of dense layer conductance measured in known dense layer wells. The dense layer interface elevations were determined by identifying the segment of the conductance profile with a consistent increase (consistent slope) in conductance with depth, then measuring the depth at the beginning of this slope. Conductance interpreted depth of the dense layer interface, the depth of bedrock (where logging extended to the bedrock surface), and the screened interval of the deep well that was logged and the corresponding shallow well, are provided on the annotated logs in Appendix N. In addition, the annotated inductance logs presented in Appendix N also show the previous conductance profiles of the wells logged during the CSA (CRA, 1993).

Two logs are provided for GW-44D. This well was logged on both the first and last day of the survey as a quality assurance measure. A comparison of these logs provided in Appendix N shows very good agreement between the measured conductance profiles. The interpreted dense layer interface depths for both GW-44D logs are 36.5 feet bgs with an elevation of 47 feet MSL and a dense layer thickness of 24.5 feet.

Occurrence, Elevation, Thickness of the Dense Layer

Interpretation of the inductance logs shows that the dense layer is not present at wells GW-82D, GW-84D, GW-85D, GW-86D, and GW-87D. The dense layer may be present at GW-34D and GW-55D, however the conductance curves from these wells did not show an identifiable interface above the bedrock surface. Sporadic high conductance (> 400 mmho/m) were logged at GW-34D, however the conductance did not consistently increase with depth and a dense layer interface was not identified at this well. The maximum conductance at GW-50D were measured over 10 feet above the bedrock surface and only a slight increase (10 mmho/m) in conductance was observed above the bedrock surface. Relatively low conductances (< 90 mmho/m) were measured at the other wells listed above and the peak conductance frequently occurred well above the bedrock, which is uncharacteristic of the behavior of the dense layer (Appendix N).

The elevation of the dense layer interface for the remaining wells are summarized on Table 2-28. The dense layer interface elevations generally decrease with bedrock elevation and are highest in wells east of Jewel Drive. The dense layer interface elevations suggest that the dense layer occurs in pools with interface elevations varying from 58.1 to 62.9 feet MSL in wells on-property, from

59.1 to 68.6 feet MSL in wells between the property and Jewel Drive, from 28.1 to 47 feet MSL in wells near Main Street and Eames Street, and at -22.1 feet MSL at GW-83D, which is below a bedrock saddle west of Main Street.

Since the vertical extent of the dense layer is controlled by the bedrock surface, the thickness of the dense layer was calculated by subtracting the bedrock elevation from the interface elevation. The thickness of the dense layer is summarized in Table 2-28 and contoured on Plate 2-10. A comparison with the bedrock surface contours (Plate 2-1) and the dense layer thickness contours (Plate 2-10) shows that most of the dense layer lies in the upper Western Bedrock Valley and is confined to the east and west by bedrock saddles. A relatively small amount of the dense layer is confined to the bedrock low at the GW-83 well cluster and does not extend to the north or west. Since the dense layer is controlled by the bedrock surface, the dense layer is thickest at bedrock lows, thins as the bedrock elevation increases, and does not extend out of the Western Bedrock Valley (Plate 2-1 and Plate 2-10). An anomalous small lobe of dense layer less than 5 feet thick is present at the GW-55 well cluster. The higher interface elevation at GW-55D (70.2 feet MSL) and lower conductance (50 mmho/m) compared to adjacent dense layer wells suggests that the dense layer at GW-55D may be a remnant of past conditions. The presence of the dense layer at GW-55D may suggest that (1) the dense layer interface historically reached elevations high enough to flow between bedrock highs east and west of GW-55 (Plate 2-1) and was left in place as the dense layer receded back into the Western Bedrock Valley, or (2) overflows from liquid waste disposal areas such as Lake Poly were transported down the Ditch System, ponded in the low lying area around GW-55, and then infiltrated through the aquifer to form an isolated pool of dense layer on top of the bedrock surface.

Summary of Conductance Measurements

The range of conductance measured within the dense layer and overlying groundwater are presented on Table 2-28. In general, the highest dense layer conductance (410 to 800 mmho/m) were measured in wells in the bedrock lows with the thickest dense layer (GW-42D, GW-44D, GW-45D, and GW-59D). On-property, the dense layer conductance ranged from a minimum of 70 mmho/m at GW-55D to a maximum of 490 mmho/m at GW-35D. In general, the maximum conductance does not correlate with water quality data and may instead reflect the thickness of the dense layer at each well. The maximum conductance of the dense layer at GW-83D (210 mmho/m) is generally lower than that measured in wells with similar dense layer thicknesses; GW-30DR (10.2 feet thick, 320 mmho/m), GW-36 (12.5 feet thick, 370 mmho/m), and GW-43D (12 feet thick, 380 mmho/m). However, the maximum conductance of the dense layer at GW-83D is similar to the maximum conductance of wells at the periphery of the dense layer in the upper Western Bedrock Valley: GW-22D (190 mmho/m), GW-27D (190 mmho/m), GW-58D (120 mmho/m), GW-69D (210 mmho/m), and GW-70D (240 mmho/m). This supports a conceptual model where GW-83D represents the outer edge of the dense layer that had spilled over the bedrock saddle west of Main Street.

The background conductance of the groundwater overlying the dense layer presented in Table 2-28 range from 10 to 65 mmhos/m. The background conductance for groundwater presented in this table was determined by approximating a best fit line through the consistently lowest conductance measured in the groundwater above the dense layer. On-property, the wells with the greatest range of groundwater conductance and highest background conductance (GW-27D, GW-30DR, GW-35D, GW-37, GW-50D, and GW-55D) generally do not correspond to the wells with the highest dense layer conductance (with the exception of GW-30DR and GW-35D). The inductance logs for these wells also show generally elevated conductance at shallow depths, often with conductance spikes at discrete depth intervals (Appendix N). The higher background conductance in these wells and presence of conductance spikes well above the dense layer interface may reflect the presence of preferential flow zones or the presence of near-surface sources (other than the dense layer), which contribute to the background conductance in the shallow groundwater.

Comparison of 1997 and 1992 CSA Inductance Logs

The annotated inductance logs presented in Appendix N also show the 1992 CSA conductance profiles as a dashed line on the following nine well logs: GW-36, GW-42D, GW-43D, GW-44D, GW-45D, GW-58D, GW-59D, GW-69D, and GW-70D. The 1992 well conductance data were obtained with a Geonics EM39 logging system. The 1997 inductance logs were conducted with a more sophisticated logging system and generally show improved vertical resolution. In general, all of the 1992 inductance logs are very similar to the 1996 inductance logs, with the exception of the logs for GW-69D. At GW-69D, the inductance log from 1992 shows generally lower conductance values in the dense layer than the inductance log from 1997, although the depth of both the dense layer interfaces appears to be similar. None of the adjacent wells show an increase or decrease in dense layer interface elevations and slight differences in the magnitude and depth of the peak dense layer conductance are attributed to the improved resolution of inductance logging instrumentation. The 1992 results for GW-69D are believed to be spurious. It should be noted that GW-69D was logged in December 1992, while the other eight wells were logged in August 1992. The similarity between dense layer interface elevations measured in the 1992 and 1997 surveys in the other eight wells suggests that the dense layer has not moved significantly during this time period.

2.7 SITE GEOCHEMICAL CHARACTERIZATION

2.7.1 Presentation of Geochemical Investigations

The following sections present the methods and results of investigations implemented in accordance with the following work plans which were developed to address MADEP concerns about deep groundwater transport of contaminants, particularly chromium:

- "A Geochemical Approach to Mitigating Chromium" (PTI, 1994) included further assessment of the existing geochemical database and evaluation of chromium transport hypotheses, application of site-specific analytical methods, and two groundwater sampling events that were focused on the geochemistry of the dense layer. This work plan was submitted to the MADEP in November 1994 and was approved by the MADEP in December 1994.

- 'Multilevel Piezometer Work Plan" included geochemical characterization of the dense layer and profiling the vertical chemistry of both the dense layer and overlying groundwater. This work plan was submitted to MADEP in November 1995 (Geomega, 1995), and approved by the MADEP in December 1995.

- 'Supplemental Phase II Investigations Work Plan" included geochemical investigations of the surface water and flocculent. This work plan was submitted to the MADEP in August 1993 (BCM, 1993), and approved by the MADEP in September 1994.

The methods and results of the tasks implemented under these work plans are discussed in Section 2.7.2 through 2.7.6. The groundwater sampling events and site-wide characterization of geochemical conditions are discussed in Section 2.7.2. This section identifies the major ions in the groundwater and quantifies the geochemical factors controlling chromium mobility at the site, such as the pH, oxidation-reduction potential (ORP), and the buffering capacity (acidity or alkalinity) of the groundwater. This section also reviews the dense layer and groundwater results to show how the overall distribution of inorganics varies in terms of total dissolved solids and examines the mobility of the main inorganic consistuents in the dense layer and overlying groundwater.

The sampling and analysis of aquifer materials are discussed in Section 2.7.3. In this section the geochemistry of chromium precipitates found in the aquifer matrix are evaluated to reveal the composition of solids which control the solubility of inorganics in the aquifer (Section 2.7.3.1). This section also examines the solubility of a laboratory-created precipitate from a sample of the dense layer and looks at dense layer reactions with the aquifer matrix through adsorption/partitioning studies. One of the partitioning studies performed the focused on the adsorption of chromium and used samples from the most concentrated part of the dense layer (Section 2.7.3.2) Another partitioning study examined the fate of the major inorganic consistuents in the dense layer and used samples from the periphery of the dense layer and from the deep groundwater upgradient of the public supply wells (Section 2.7.3.3).

Further characterization of the dense layer chemistry and the interface with overlying groundwater using multilevel piezometers is discussed in Section 2.7.4. This section describes the installation and sampling of the multilevel piezometers which were designed to determine the vertical distribution of inorganics in the dense layer and overlying groundwater. The groundwater analytical results discussed in Section 2.7.4.2 document the sharp contact and large concentration gradient between the dense layer and the overlying groundwater. The vertical and lateral

distribution of major ions in multilevel piezometer samples is examined in Section 2.7.4.3. This section further documents the chemical distinction between the dense layer and overlying groundwaters and examines the observed diffusive pathways/mechanisms from the dense layer.

The further evaluation of the dense layer analytical results through geochemical modeling is discussed in Section 2.7.5. The modeling utilized site data to describe geochemical conditions and processes in the aquifer to determine if chemical conditions favor the formation of certain precipitates and how these processes affect solute transport. The results of geochemical modeling (Section 2.7.5.3) predicted the formation of the same amorphous hydroxides solids observed in samples of aquifer material and determined that chromium complexes were not present at concentrations that would affect chromium solubility. This modeling and the results of the other geochemical investigations formed the basis for fate and transport modeling discussed in Section 5.0.

Additional sampling and characterization of the surface water, shallow groundwater, and flocculent in the Off-Property West Ditch and South Ditch are discussed in Section 2.7.6. The geochemical evaluations in this section were built on the results of groundwater head and surface water flow measurements in the Off-Property West Ditch and South Ditch (Section 2.4.4) and were used to determine how the shallow groundwater interacts with the surface water system to create a chromium-bearing flocculent. The results of groundwater and surface water sampling, presented in Section 2.7.6.2, documents the geochemical conditions which lead to flocculent formation and the changes in surface water and shallow groundwater chemistry due to the influence of the weir. Testing of flocculent samples (Section 2.7.6.2.4) and geochemical modeling (Section 2.7.6.2.5) revealed the composition of the flocculent and successfully simulated the mechanism of flocculent formation. These studies and the other investigations have resulted in a conceptual model of flocculent formation discussed in Section 2.7.6.3.

2.7.2 Site Groundwater Geochemical Characterization

In the CSA report (CRA, 1993), a strong correlation was noted between the spatial distribution of inorganics and the pH in the site groundwater. That report concluded that this correlation was due to the limited solubility of chromium at near-neutral pH levels, which is consistent with the known geochemical behavior of chromium (Griffin et al., 1977) and other metals including and aluminum. Furthermore, that report concluded that the conditions of the site groundwaters may affect the fate and transport of inorganics through the aquifer. This in turn affects the characterization of pathways and development of risk scenarios.

The geochemical characterization presented below specifically addresses MADEP concerns for chromium in the dense layer and describes the geochemical conditions which control chromium mobility in the aquifer. For example, at high oxidation-reduction potential (referred to interchangeably as Eh or ORP) conditions , chromium may occur as the hexavalent chromium [(chromium(VI))] species. However, as discussed below, chromium(VI) is not stable at the Eh and pH conditions observed in the site groundwater and surface water, and so the overwhelmingly predominant species is chromium(III). Because the chromium(III) ions are more

strongly adsorbed to aquifer materials than chromium(IV) ions, chromium(III) species will migrate downgradient more slowly than chromium(VI) species when the aquifer is at neutral pH and in most cases chromium(III) is attenuated by precipitation and adsorption processes. To understand the fate and transport of dense layer constituents through the groundwater system, the Supplemental Phase II Investigations obtained samples to evaluate the site geochemical conditions of the groundwater system.

2.7.2.1 Methods

2.7.2.1.1 Collection of Groundwater and Surface Water Samples

Prior to collecting groundwater samples, three well volumes of groundwater (minimum) were purged from the wells using a peristaltic pump and dedicated Tygon tubing. Deeper wells, such as GW-62BR, were sampled using a decontaminated, submerged turbine pump. The last 10 percent of the purge water was diverted through a flow-through cell to measure the dissolved oxygen (DO), pH, specific conductance (SC), temperature, and ORP measured as Eh. The flow-through cell was used to exclude oxygen from the sample during measurement of the field parameters, and to minimize volatilization of dissolved gasses, which may affect the geochemistry of the sample, particularly the pH and DO. Samples were collected after it had been determined that the field parameters measured in the flow-through cell had stabilized.

Surface water samples were collected through a short length of new Tygon tubing, using a peristaltic pump and methods analogous to the sampling of groundwater. Field parameters were recorded immediately following stabilization of measurements for dissolved oxygen.

Both filtered and unfiltered samples of the surface and groundwaters were collected in appropriately preserved sample bottles, as described in the work plan (BCM, 1993). Splits of the filtered samples were collected at each well and analyzed colorimetrically (Hach, 1989) at the well head for ferrous iron and total iron with ferric iron determined by difference. Alkalinity was analyzed using a photospectrometer and digital titrator, respectively. Samples were then forwarded to Pace Laboratories, Westbrook, Maine, where they were analyzed following procedures consistent with the Massachusetts Contingency Plan, as described in the work plan (BCM, 1993).

2.7.2.1.2 Chromium Analytical Methods and Evaluation of Species Present

Other than the analysis for chromium(VI), the laboratory analytical methods described in the work plan (BCM, 1993) were used. While the majority of the chromium(VI) analysis performed for this study used the analytical method prescribed in the work plan (U.S. EPA Method 6010), groundwaters collected during the multilevel piezometer study were analyzed by an alternative method (U.S. EPA Method 7196), which was less prone to matrix interferences.

During the Phase II field activities conducted in 1992, deep groundwater was sampled and analyzed for both chromium(III) and chromium(VI) concentrations (CRA, 1993). These analyses indicated that chromium(VI) consistently comprised approximately 10 percent of the total dissolved chromium concentration, particularly in the more acidic regions of the dense layer (CRA, 1993). Samples collected in groundwater sampling rounds in 1995 also showed chromium(VI) concentrations ranging from approximately 1 to 10 percent of the total dissolved chromium (Section 2.7.2.3), while samples collected in 1996 showed no appreciable chromium(VI).

However, in addition to containing chromium, the site groundwater also contains appreciable concentrations of iron(II) with concentrations generally increasing with lower pH (Section 2.7.2.3). Numerous studies have shown that the occurrence of chromium(VI) is chemically incompatible with the presence of iron(II) because it is well established that these two species undergo an oxidation-reduction reaction that converts chromium(IV) to chromium(III) according to the following reaction:

$$Cr(VI) + 3Fe(II) \rightarrow Cr(III) + 3\,Fe(III)$$

This reaction is stoichiometric and reaches completion in a matter of seconds (Eary and Rai, 1988; Fendorf and Li, 1996; Saleh et al. 1989).

The incompatibility of chromium(VI) and iron(II) in the same solution made the analytical results for chromium(VI) in the 1992 and 1995 sampling rounds suspect (CRA, 1993), hence, their methodologies were assessed in more detail. In the CSA samples, chromium(VI) was determined by a procedure that involved the addition of lead, as $PbSO_4$ dissolved in acetic acid, to a sample aliquot (EPA Method 7195). The addition of lead theoretically should result in the removal of chromium(VI) from solution as a lead chromate ($PbCrO_4$) precipitate, which is then separated and analyzed for chromium by ICP (EPA Method 6010). However, high concentrations of sulfate and cationic metals are known to interfere with this method (EPA Method 7195) by forming precipitates that coprecipitate and/or adsorb chromium(III) as well as chromium(VI), resulting in false positive determinations for chromium(VI). The dense layer contains up to 91,000 mg/l sulfate and up to thousands of mg/l of cations, such as aluminum, chromium(III), iron(II), manganese(II), sodium, and ammonium, making it probable that the chromium(VI) concentrations reported in groundwater for the 1992 and 1995 sampling were in error because of matrix interferences (i.e., coprecepitation of chromium(III) in $PbS0_4$).

More recently, during the 1996 sampling of the multilevel piezometers (see Section 2.7.4), groundwaters were analyzed for chromium(III) using diphenylcarbazide, a chromagenic agent for chromium(VI) (EPA Method 7196). Selected splits of dense layer samples also were analyzed using the aforementioned lead sulfate precipitation method for comparison to samples analyzed by Method 7196. Whereas the lead sulfate precipitation method indicated chromium(VI) concentrations up to 185 mg/l, no detectable chromium(VI) was found by the colorimetric method in any sample (detection limits ranging from 0.01 to 0.05 mg/l).

These results indicate that the majority of previous chromium(VI) detections in groundwater samples were erroneous and probably caused by co-precipitation of chromium(III) with precipitates of lead sulfate, metal hydroxides, and possibly other metal sulfates (Katahdin Laboratories, 1996). The redox and pH conditions observed in the vast majority of samples support the existence of chromium(III). Additionally, the presence of dissolved chromium as entirely chromium(III), rather than partially as chromium(VI) in the groundwaters, is consistent with current knowledge of the kinetics of redox reactions between dissolved chromium(VI), iron(II), and manganese(II).

2.7.2.1.3 Data Evaluation Tools

All analytical data, including field- and laboratory-collected data, were input into an electronic database where they could be accessed easily for further evaluation. The goals of these evaluations were to 1) identify the geochemical conditions that might affect the fate and transport of dense layer constituents, 2) determine how the major ion chemistry of groundwater changes across the site, and 3) determine whether the bulk chemistry of groundwater is changing with time, and if so, whether these changes indicate that dense layer constituents are migrating downgradient. The tools used to achieve these goals are listed as follows.

Eh/pH

The Eh/pH diagram is used to assess the geochemical stability of dissolved and solid phases in the groundwater system. The diagram demonstrates graphically the Eh-pH conditions at which a particular mineral or dissolved species will be thermodynamically stable. It can be used to predict the likely effect of changing the system pH or Eh on the solubility of a particular contaminant. A demonstration of how these diagrams are used to evaluate the geochemical conditions of an aquifer is discussed in Section 2.7.2.2.

Major Ions

The predominant ions found in most natural waters typically include the major cations (calcium, potassium, magnesium, and sodium) and the major anions (chloride, sulfate, carbonate, and bicarbonate). Piper diagrams are used to project the concentrations of these major constituents onto a two-dimensional plot, based on the ratios of the cations and anions in individual samples. The locations of individual samples on a Piper diagram, together with the total dissolved solid concentration of that sample, provide a graphical representation of the major-ion "fingerprint" of that sample. The results of the Piper diagram analysis are provided in Section 2.7.2.3.

Temporal Plots

Temporal plots (which are *x-y* scatter graphs on which are plotted the concentrations of individual constituents as function of time) are used to evaluate changes in the chemical composition of the groundwater over time If, for example, the concentrations of sulfate, sodium, and ammonia in the groundwater from a well located at the toe of the dense layer increases steadily with time, it may

indicate that the sulfate, sodium, and ammonia of the dense layer are migrating downgradient. Conversely, decreasing concentrations may indicate that attenuation of the dense layer is occurring.

2.7.2.2 Indicators of General Geochemical Conditions

The major factors controlling the fate and transport of inorganics through the aquifer can be categorized as hydrologic factors or geochemical factors. The limiting hydrologic factors include the high viscosity and density stratification of the dense layer in bedrock lows which minimizes interaction with and advective transport through more permeable portions of the aquifer (Section 2.2).

The primary geochemical factors controlling the rate of inorganics migration through the aquifer system include the propensity for inorganics to precipitate as mineral phases or sorb to aquifer materials. The main parameters that affect the mobility of chromium in these two processes are pH, alkalinity, Eh, and the dissolved oxygen of the system. Other geochemical factors relate to the concentration of reactive ions in the dense layer. The main parameters were measured at the well head during sampling, as was specific conductivity (SC), which provides a gross indicator of the total dissolved solids (TDS) content of the groundwater (Table 2-29). This section describes the methods used to quantify these parameters in the site groundwater and surface water systems.

2.7.2.2.1 Specific Conductivity

Specific conductivity (SC) is a measure of the electrical conductivity of water at 25°C, and is useful as a gross indicator of TDS content. Most dissolved constituents have an electrical charge, and their presence facilitates the conductance of electricity through the water. Therefore, the higher the concentration of dissolved constituents, the easier it is to conduct electricity through the water. In most waters, the TDS values in mg/l range between 0.55 and 0.75 times the SC values in umhos (Driscoll, 1989), depending on which ions make up the bulk of the TDS load.

The relation between specific conductance and TDS is useful at the site, because the TDS of groundwater has been measured only 142 times since the inception of site groundwater investigations, compared to 866 SC measurements. Because of the strong statistical correlation (almost 1:1) between the TDS and SC in site groundwater (R^2=0.92, Figure 2-42), the TDS can be estimated for those samples where only the SC was measured.

Histograms showing the distribution of all available groundwater SC measurements made at the site indicate that the average SC in each well ranges between 8 and 33,000 umohs in the shallow groundwater, and 107 and 71,000 umhos in the deep groundwater (Figure 2-44). Groundwater collected from the shallow portion of the aquifer were normally distributed about a median value of 490 umhos (Figure 2-44). However, within the deep groundwater system, a

bimodal distribution of SC values was evident: the upper mode corresponds to the wells screened in the dense layer, with SC values ranging from 10,000 to 71,000 umhos, and the lower SC modal value ranges from approximately 110 to 10,000 umhos.

2.7.2.2.2 pH

The pH is a measure of the hydrogen ion concentration in a water (H^+), and is proportional to the hydroxide (OH^-) concentration (i.e., the higher the pH, the more OH^- in the solution and the less acidic the water). Because increasing hydroxide concentration (increasing pH) may result in precipitation of metal hydroxides [e.g., $Fe(OH)_3$ and $Cr(OH)_3$], an important first step in understanding the solubility of many dense layer constituents is to describe the pH conditions of the site groundwater system.

Histograms showing the distribution of all available groundwater pH measurements except the wells with known grout contamination (GW-65B, GW-81BR, and GW-81D) are shown on Figure 2-45. These histograms show that the groundwater pH ranges from 3.5 to 9.5. Groundwater collected from the shallow portion of the unconfined aquifer are normally distributed about a mean pH value of 6.2 (Figure 2-45). However, within the deep groundwater system, a bimodal distribution of pH values was evident. The lower mode corresponds to the wells screened in the dense layer, with pH ranging from 3.5 to 4.5. The upper mode has an approximate pH of 6.75, and ranges from approximately 5 to 9.5 and represents deep wells not screened in the dense layer. Eight percent of the wells screened in the deeper portions of the overburden aquifer had average pH values higher than 8, including GW-18D, GW-52D, GW-20, GW-81D. These wells are located in different regions of the site, and are separated by numerous wells with groundwater pH values that are much lower. Therefore, it is likely that these high-pH groundwaters are highly localized, and do not represent the deep groundwater system as a whole.

2.7.2.2.3 Alkalinity/Acidity

Alkalinity and acidity are measures of the propensity of a system to remain at a particular pH (i.e., the buffering capacity of the system). Highly alkaline systems require a large quantity of acid to lower the system pH. Conversely, systems with high acidity require a large quantity of base to raise the system pH. By measuring the alkalinity/acidity of groundwater, an assessment may be made as to whether its pH would be likely to change as it migrates through an aquifer and is exposed to sources of base or acid such as unaffected groundwater and aquifer matrix minerals.

Alkalinity

The alkalinity of a sample is determined in the field by assuming that all of the buffering capacity present is derived from dissolved carbonate (CO_3^{2-}) or bicarbonate (HCO_3^-) species depending on the pH. Given the common occurrence of carbonates in aquifer materials, and the prevalence of carbon dioxide in the atmosphere (which readily dissolves into water), this assumption is valid in

most natural waters (Stumm and Morgan, 1981). The alkalinity of site groundwater was determined by titrating the samples with acid to a pH of 4.5. Samples whose initial pH was 4.5 or less were not analyzed, as their alkalinity by definition, is zero (Table 2-29).

Groundwater alkalinity ranged from 0 to 1,900 mg/l as equivalent calcium carbonate ($CaCO_3$). Based on the assumption that modal pH is an indicator of background pH at the site (Figure 2-46), the alkalinity of groundwaters with pH similar to the average shallow groundwater (i.e., pH 6.0–6.4) ranged between approximately 24 and 800 mg/l $CaCO_3$, with an average alkalinity of 150 mg/l $CaCO_3$ (Figure 2-46). This alkalinity range is typical for groundwater in aquifers of this type, and is similar to values observed in background groundwaters in the area of the site (Davis et al., 1994). As would be expected, the alkalinity of the groundwater generally depended on the pH, with higher-pH groundwater having higher alkalinities. There were notable exceptions to this observation. Specifically, five wells (GW-18D, GW-20, GW-65BR, GW-81D, and GW-81BR) had pH values of 9 or greater, but had alkalinities of <300 mg/l as $CaCO_3$. As discussed above, the higher alkalinity at GW-65BR GW-81D and GW-81BR is due to residual cement grout, while the anomolously elevated alkalinity of the other wells may be attributed to the adjacent Calcium-Sulfate Landfill.

Acidity

As mentioned previously, the dense layer has an acidic pH. Determination of the total acidity of the dense layer is important for making an assessment of the extent to which natural neutralization reactions in groundwater might attenuate its migration in the aquifer by raising its pH. As the dense layer mixes with groundwater having higher pH values, and as it comes into contact with aquifer materials that have the capacity to buffer the system pH via silicate hydrolysis reactions, the pH of the dense layer will increase. The amount of increase will depend on the alkalinity of the groundwater with which it mixes (as discussed previously), the alkalinity of the aquifer matrix, and the acidity of the dense layer groundwater. If the dense layer is highly acidic, a great deal of alkalinity will be required to raise the system pH. The acidity of the dense layer was found to be 0.43 M, or 24,500 mg/l as $CaCO_3$, which is unusually high in comparison to natural groundwaters, and is more than 100 times higher than the alkalinity of the shallow groundwaters. Thus, an equal volume of the dense layer would have to mix with approximately 100 times more shallow groundwater to achieve a neutral pH.

The total acidity of the dense layer solution was determined by base titration of a sample obtained from monitoring well GW-42D. This well is completed in the dense layer and samples obtained from it have high acidity and high dissolved metal concentrations (Table 2-30). For determination of total acidity, 1.5 liters of a dense layer sample was titrated by the stepwise addition of 50 percent NaOH, while continuously sparging with $N_2(g)$ to exclude air. The titration was conducted slowly over the course of eight hours to allow time for equilibration with any precipitates that may have formed as the solution was neutralized.

The dense layer sample from GW-42D used in the titration experiment was initially dark green. The addition of NaOH during titration immediately resulted in the formation of a light green precipitant, characteristic of chromium hydroxide [$Cr(OH)_3$(am)] (Figure 2-47). As more NaOH was added, a brown precipitate also formed, probably iron(III) hydroxide [$Fe(OH)_3$(am)]. The mixture of the two solids produced an olive-green slurry by the end of the titration.

Based on the titration curve, the total acidity of the dense layer to pH 7.0 was determined to be 0.43 M. In addition to containing strong acid anions like chloride and sulfate, the dense layer contains a high concentration of mineral acidity in the form of dissolved cationic metals [i.e., iron(II), chromium(III), and aluminum, (Table 2-30), whose precipitation as hydroxides releases H^+ ions and strongly buffers the solution pH as shown by the formation of iron hydroxide in the following reaction:

$$Fe^{+3} + 3H_2O = Fe(OH)_3(am) + 3H^+$$

The high mineral acidity of the dense layer indicates that neutralization of the dense layer solution would require reaction with strong bases or reaction with large amounts of weak bases.

The aquifer system at the site does contain some weak bases in the form of silicate minerals that may be consuming a fraction of the dense layer acidity through dissolution reactions. The glacial till and gneissic bedrock contain layer silicates, including biotite, chlorite (the mineral), and illite. Biotite, in particular, dissolves more rapidly than other silicates (e.g., feldspars, kaolinite, quartz) under acidic conditions (Stumm and Morgan, 1996), and may consume acid by reactions under moderately oxidizing conditions such as:

$$KMgFe_2AlSi_3O_{10}(OH)_2 + 3H_2O + 7H^+ = K^+ + Mg^{+2} + 2Fe^{+2} + Al(OH)_3(am) + H_4SiO_4^0$$

Evidence that such reactions are occurring in the aquifer is provided by the occurrence of iron(II) in addition to low concentrations of iron(III) in the dense layer and equilibria with aluminum hydroxide [Al(OH)3(am)] solubility (Section 2.7.5). Additionally, concentrations of silicic acid show a substantial increase at low pH and exceed the solubility limit for amorphous silica [SiO_2(am)] at pH < 4, whereas in the more neutral pH portions of the aquifer, conditions of near equilibria with quartz or chalcedony are existent (Figure 2-48). These data indicate that increased silicate mineral dissolution is occurring in the acidified parts of the aquifer and has partially neutralized the acidity of the dense layer solution. However, silicate hydrolysis reactions, such as that shown for biotite above, are slow, hence the dense layer can be expected to remain acidic for an extended period of time, consistent with its persistence in the aquifer to the current time.

2.7.2.2.4 Dissolved Oxygen and Eh

The dissolved oxygen (DO) content of a groundwater is important, as it may affect the solubility of some of the dense layer constituents. For example, ferrous iron (Fe^{2+}) may react with oxygen to form ferric iron (Fe^{3+}), which in turn precipitates as $Fe(OH)_3$ thereby lowering the

concentrations of iron in solution. Because iron is a major component of the dense layer, it is necessary to understand the oxygen content of the aquifer to understand the geochemical fate of iron (and other dense layer constituents).

Dissolved Oxygen

The DO contents of both the shallow and deep groundwaters, and in the dense layer, are low, ranging from 0.0 to 7.5 mg/l, with the vast majority of groundwater samples containing little or no detectable oxygen (Table 2-29). In a few dense layer samples, the field-measured DO of the dense layer was greater than 45 mg/l, higher than would be expected in any natural system. In fact, the solubility of oxygen in water at the temperature of the site groundwaters is four times lower at 11.2 mg/l @ 11.6°C). These readings are anomalous and are probably due to a reaction between the DO electrode and the dense layer constituents. Given that the groundwater upgradient (GW-50), downgradient (GW-64D), and directly above the dense layer (GW-42D) contained less than 1 mg/l DO (Table 2-29), the true DO content of the dense layer is probably also less than 1 mg/l.

Eh

Eh can be conceptualized as a measure of the availability of reactive electrons in an aqueous system, with electrons being far less available at high Eh than at low Eh. Because the exchange of electrons changes the oxidation state of an element (e.g., the reduction of Fe^{3+} to Fe^{2+}), the Eh (together with the pH) can be used to assess the likely oxidation state of an element. Oxygen is the most common terminal electron acceptor, so oxygen content has a large effect on the Eh. When oxygen is present, the Eh is typically high. However, at the site the DO concentrations are generally low, and the dissolved ions (mainly the $FE^{+2}/FE(OH)_3$ couple) and pH play a greater role in determining the availability of reactive electrons or Eh of the system.

The Eh of site groundwater ranged between 2 and 490 mV, indicating that site groundwaters are slightly to moderately oxidizing. A probability plot of each Eh value recorded in the deep and shallow groundwater systems (Figure 2-49) indicates no appreciable difference between the Eh of the deep and shallow groundwater systems.

2.7.2.2.5 Eh and pH as General Indicators of Chromium Geochemistry

As stated previously, the primary purpose of evaluating the pH and Eh of the groundwaters at the site is to obtain information regarding the potential oxidation/reduction reactions in the dense layer. Ideally, the Eh measured in the field would correspond to a "system Eh," which could be used to determine the theoretical oxidation state of the redox-sensitive components in the system, such as Fe(II)/Fe(III) and chromium(III)/chromium(VI) (Langmuir, 1971). Unfortunately, the measured concentrations of such components do not always correspond to the theoretical concentrations based on this system Eh, particularly when multiple redox couples are present in the system (Lindberg and Runnells, 1984).

To determine whether field-measured Eh values correspond to system Eh in site groundwater, theoretical Eh values were calculated from the measured well-head concentrations of Fe^{2+} and Fe^{3+}, and these data were plotted on an Eh-pH diagram with the measured Eh values (Figure 2-50). This diagram can be used to determine the most stable mineral phase under various Eh-pH conditions. The results indicate that while the Eh values calculated from the Fe(II)/Fe(III) couple were generally lower than the field-measured Eh, both the measured and calculated Eh values fall well below the stability field for chromium(VI) (with the exception of grout-impacted samples collected from GW-81D and GW-65-BR), indicating that chromium(III) is the most likely stable chromium phase in the site groundwater. These data confirm that the 1995 laboratory analyses which show no detectable chromium(VI). While groundwater from wells GW-81D and GW-65BR did fall within the stability field of the chromium(VI) species, CrO_4^{2-}, chromium was not detected in these wells (detection limit 0.015 mg/l). In addition, the presence of the reduced species such as ammonia, Fe^{2+}, and Mn^{2+} is further evidence that the groundwater system is not sufficiently oxidized to support the existence of chromium(VI) (Brookins, 1988).

In addition to the Eh-pH diagram, a second method was used to assess the stability of the two chromium species. The thermodynamic equilibrium model EQ3NR (Wolery, 1993) was used to determine whether an increase in the Eh of groundwater from GW-45D to a level above that measured in October 1995 (431 mV) would result in formation of chromium(VI). To answer this question, the chemical data collected from this well during October 1995 were input into the model, and the model stepped through a range of possible Eh values (–200 mV to 850 mV). These OPR values bracketed the range of Eh values observed in site groundwater (2 to 490 mV). The results, shown on Figure 2-51, confirm the conclusions stated in Section 2.7.2.1.2, that chromium(VI) is unlikely to occur in site groundwater. At a pH of 4 and a total chromium concentration of 680 mg/l (the conditions of GW-45D during the October 1995 sampling event), the Eh of groundwater would have to be elevated to over 850 millivolts (mV) before the chromium(VI) concentration would exceed the detection limit of 0.01 mg/l (U.S. EPA Method 7196). Given the lack of oxygen in the groundwater, and the presence of other reduced species (e.g., NH_4^+, Fe^2, and Mn^{+2+}), site groundwaters are unlikely to ever achieve Eh values high enough for chromium(VI) to form.

2.7.2.3 Major Ions as Total Dissolved Solids

Major ions are defined as groundwater constituents that contribute to the majority of a water's charge balance and TDS load in water. For example, if a spoonful of table salt (NaCl) is added to a glass of deionized water, the major ions in that glass of water would be sodium (Na^+) and chloride (Cl^-). The positively charged ions (cations) in the glass would consist predominantly of the total Na^+ concentration, and the amount of negatively charged ions (anions) would consist predominantly of the total Cl^- concentration. The TDS in the glass would essentially be the mass of salt added to it. In most natural waters, the majority of the charge balance and the TDS can be accounted for by measuring the concentration of eight major constituents: sodium, calcium, magnesium, potassium, sulfate, chloride, carbonate, and bicarbonate. At the site, ammonia is also a major constituent since it is so prevalent in the groundwater and dense layer.

The concentrations of these major ions span 3 to 6 orders of magnitude between the dense layer and overlying groundwater. In general, the concentration of each of the major ions is strongly correlated to sulfate concentration in the dense layer and groundwater. For example, Figure 2-52 shows that ammonia and sodium are proportional to the sulfate concentrations. Furthermore, the concentration of ammonia and sodium in the site groundwater and the dense layer are distributed across the range of observed concentrations and do not cluster in separate groups of low concentration or high concentration samples. These data suggest that the intermediate-TDS groundwaters (in the middle of the concentration range on Figure 2-52) are actually mixtures of two types of groundwater, the dense layer and low TDS groundwater.

These two end-member types of groundwater may mix by diffusion or advection processes. Advective mixing would result in a physical mixing of groundwater and the dense layer, in the same way that two glasses of water would mix when poured together in a pitcher. If one of these two glasses contained salt water, the ions dissolved in the salt water would become evenly distributed in the resulting mixture. Alternatively, diffusive mixing of the dense layer with the background groundwater would be analogous to what occurs in a glass that is half full with salt water with a layer of fresh water floating on top of it. The ions that are in the salt water will diffuse, or migrate, into the fresh water without actually mixing the two waters together. Because different ions diffuse across a boundary between fresh and salt water at different rates, the relative proportions of ions in the salt water would be different than the fresh water. One tool used to determine if a disproportionate mixing of two waters has occurred is the Piper diagram.

2.7.2.3.1 Piper Diagrams of Major Ions and Total Dissolved Solids

Piper diagrams are a systematic method of projecting the concentrations of these nine variables (including ammonia) onto a two-dimensional graph, facilitating the evaluation of complex data sets. Piper diagrams also provide a graphical means of determining whether a particular water is a mixture of two other waters. For example, if the composition of a mixture of two distinctly different waters is plotted on a Piper diagram, together with the compositions of the unmixed waters (i.e., end members), the mixture would plot along a line (the tie line), connecting the two end members. The location of a mixture along the tie line indicates how much of each end member is contained within the mixture. If a mixture falls exactly halfway between the two end members, it is composed of 50 percent of each. To extend the example to the site groundwater, if the compositions of the dense layer and background groundwaters are well known, then a mixture of these two end members would plot along a line that connects the two end members. It is important to note that the Piper diagrams reflect the relative proportions of major ions to determine mixtures and end members and do not reflect the magnitude of their concentration.

As mentioned above, for Piper diagrams to be useful, the compositions of the end members (e.g., dense layer, shallow groundwater, and surface water) must be consistent (i.e., predictable), and unique (i.e., different from one another). Because traditional Piper diagrams do not include ammonia, the Piper diagrams used in this investigation were modified to include this parameter. Furthermore, when using traditional Piper diagrams, it is often difficult to resolve the difference between groundwaters that are dominated by sulfate and chloride (i.e., when bicarbonate is

present at low concentrations relative to sulfate and chloride). Therefore, a variant of the Piper diagram method was used to evaluate the differences among the water chemistries at the site. For this investigation, Piper diagrams were constructed with the sulfate- and carbonate-system axes reversed, and the ammonia concentrations were integrated into the magnesium-system axis (Figures 2-53 and 2-54).

The Piper diagrams presented in Figures 2-53 and 2-54 used the average concentration of each constituent, at each location, over time. The resulting diagrams indicate that there are three distinct types of groundwater subjacent to and downgradient from the property: consisting of the dense layer, intermediate-TDS groundwaters, and low-TDS groundwaters. The classification of which type of groundwater a well sample represents is dependent on both the relative proportions of ions shown by the position of well on the Piper diagrams and the magnitude of inorganic concentrations as depicted by circles representing the TDS concentration. The circle surrounding the wells on Figure 2-53 indicates the average TDS concentration groundwater samples, which was calculated from the total concentration of constituents included in the Piper diagram. The diameters of each circle are proportional to the TDS concentration of each sample.

The Piper diagrams show that the dense layer well samples all show similar major ion chemistry, have low pH, and contain little or no carbonate or calcium relative to the other constituents. The domain of the dense layer samples or wells impacted by the dense layer is indicated by the green shading on Figures 2-53 and 2-54. For example, GW-6D and GW-42D have ion chemistry in the domain of the dense layer. However, GW-6D has relatively low TDS concentrations and the deep groundwater at this location reflect residual chemistry imparted by the dense layer. To differentiate between the dense layer and dense layer 'impacted" wells with low to intermediate-TDS concentrations, the samples from wells screened in the dense layer are identified on Figure 2-53 as those with visible TDS circles. TDS values >10,000 mg/l were found only in deep groundwater, and all these fall within the area shaded green on the Piper diagrams. These wells are GW-22D, GW-27D, GW-30DR, GW-35D, GW-36, GW-37, GW-38, GW-42D, GW-43D, GW-44D, GW-45D, GW-50D, GW-59D, GW-62BR, and GW-69D. The deep groundwater geochemistry at these locations is either dominated by the dense layer or is strongly influenced by past or current contributions from the dense layer.

When the TDS scale is large enough to encompass these wells, it is not possible to easily discriminate between the TDS circles for wells containing waters with TDS <10,000 mg/l. Therefore, Figure 2-54 was constructed by plotting only the TDS values <10,000 mg/l. These non-dense layer waters plot in two general areas of Figure 2-54, dependent on TDS content. The groundwaters plotting in the areas shaded with yellow generally had low TDS values. The average of samples from the town water-supply well (indicated by 'town well" on Figure 2-53) plot within the domain of the low-TDS groundwater indicating that water pumped from the town wells is different from the dense layer and intermediate TDS groundwater. Other wells within the low TDS domain generally include wells located outside or at the periphery of the Disposal Site or wells completed in shallow groundwater above the dense layer.

The intermediate-TDS shallow groundwater (approximately 1,000 to 9,000 mg/l) that fall within the blue-shaded area of Figure 2-54 include those from wells GW-3, GW-4, GW-8, GW-10D, GW-17S, GW-22S, GW-26, GW-43S, GW-50S, GW-56S, GW-76S, GW-78S, and GW-79S. Groundwater from these wells generally had TDS values between the dense layer wells and the wells in the yellow-shaded region of Figure 2-54. These wells are located in a contiguous area east of Jewel Drive, north of the Milcom/Preferred Equipment building, south of Janis Research, and extending eastward beneath the central portion of the property as far as the east ditch. Given their location and composition, these groundwaters are likely to have been affected by the dense layer. In general, these wells are located at or downgradient of the former liquid waste disposal areas and at the shallow groundwater discharge points along the Ditch System.

The wells screened in deep groundwater that have intermediate-TDS values include GW-17D, GW-19D, GW-34D, GW-40D, GW-58D, and GW-64D. With the exception of GW-40D, all of these wells are located adjacent to the dense layer. GW-40D is located in the southwestern bedrock valley, and although it is not adjacent to the dense layer, it has a major ion composition similar to dense layer which reflects contributions of similar constituents from the Calcium Sulfate Landfill. The average TDS concentration presented for GW-40D (approximately 1280 mg/l) is close to the range of TDS observed in low TDS groundwaters.

2.7.2.3.2 Segregation of Major Ions from the Dense Layer

If the intermediate-TDS groundwater (shaded blue on Piper diagrams) are derived from the mixing of dense layer and low-TDS groundwaters, they should plot between these two end members on the Piper diagrams (i.e., between the green and yellow-shaded areas on Figure 2-54). The fact that they do not suggests that the individual major dense layer components may be transported away from the dense layer at different rates. As the goals of this investigation were to fully characterize the dense layer pathways and to determine how the dense layer constituents will be transported through the aquifer, it is important to determine whether certain dense layer constituents are migrating at disproportionate rates.

An additional utility of Piper diagrams is that the anion and cation chemistry of water can be evaluated separately. The anion chemistry of the intermediate-TDS groundwater is similar to that of the dense layer (Figure 2-54). Furthermore, the anion composition of most of the intermediate-TDS waters lies between the dense layer and the low-TDS groundwater. This suggests that anions (sulfate, bicarbonate/carbonate, and chloride) have migrated away from the dense layer at equivalent rates.

An evaluation of the cation chemistry of the groundwater indicates that the intermediate-TDS groundwater does not fall along the mixing line between the dense layer and low-TDS groundwater (Figure 2-54). Instead, the intermediate-TDS groundwater is disproportionately enriched in ammonia, magnesium, or both. To determine which of these two constituents is being enriched, the average ammonia and magnesium concentration of each of the wells were plotted on Figure 2-55. If the intermediate-TDS groundwaters were a result of simple mixing of dense layer with low-TDS groundwaters, the composition of the intermediate-TDS groundwaters would plot

closely to the ideal mixing line shown on Figure 2-55. Instead, the ammonia concentrations increase at a greater rate than do the magnesium concentration (i.e., the points fall well above the ideal mixing line), indicating that ammonia is disproportionately enriched in the intermediate-TDS groundwaters.

It is important to note that when ammonia (NH_3) dissolves in water, it may form either ammonia (NH_3^0) species or one of many ammonium species (e.g., NH_4SO_4 or NH_4). The factor that most influences the form of ammonia in groundwater is the groundwater pH as demonstrated by the following equation:

$$NH_4^+ = NH_3^o + H^+ ; pka = 9.2$$

This equation indicates that at a pH of 9.2 there will be equal concentration of both NH_3^0 and NH_4^+, but below a pH of 9.2 that NH_4^+ is the dominant species. At higher pH levels, ammonia will exist as ammonia species, which may volatilize from groundwater as ammonia gas, providing the primary exposure pathway for humans. At low pH and new-neural pH, ammonia primarily exists in ionic form and the ammonium species in dissolved form in the groundwater. Thus, it is important to determine which species is present is site groundwaters.

To accomplish this, the geochemical model MINTEQA2 was used to calculate the speciation of ammonia in dense-layer groundwater (GW-45D over a pH range of 4 to 11. The results indicated that the range of pH values observed in the vast majority of site groundwater (i.e., less than pH 8) support the formation of ammonium (e.g., NH_4^+) as the predominant species. Therefore, volatilization of ammonia gas [$NH_3^o(g)$] from groundwater pumped from this aquifer would be minimal.

2.7.2.4 Characterization of Overall Site Groundwater Quality

The above geochemical results demonstrate that there are three distinct types of groundwater subjacent to and downgradient from the property: the dense layer, intermediate-TDS groundwaters, and low-TDS groundwaters. The results also show that TDS concentrations correspond directly to the concentration of the major inorganics and is a good means of evaluating the overall site groundwater quality. The above geochemical discussions described three methods of estimating the TDS of the groundwater systems: measuring it directly in the laboratory, summing the concentrations of the major constituents, or estimating if using the specific conductance of the groundwater. Although the first method (direct measurement) is the most accurate, TDS was not measured during the early investigation of the site. The second method is accurate only when a complete analysis of the groundwater was performed. Earlier investigations used an abbreviated inorganic analyte list, and it is therefore not possible to estimate the TDS values for these samples. The strong correlation between TDS and specific conductance in the site groundwater supports using the third method to provide a good estimate of the TDS distribution in the groundwater, both through time and spatially.

Plate 2-11 shows the TDS concentrations in the shallow and deep (including the dense layer) and provides an overall picture of groundwater quality at the site. This figure shows that the shallow groundwater with TDS values >1,000 mg/l is restricted to an area beneath the central portion of the property, extending west of the property to Jewel Drive (Plate 2-11). The vast majority of the shallow groundwater west of the property contain <1,000 mg/l TDS. The highest-TDS deep groundwater (>20,000 mg/l) shown on Plate 2-11 is contained within the bedrock valley (Plate 2-1) and extends as far as GW-83D. East of Main Street, the dense layer is approximately 800 feet wide, but it narrows as it enters the steeper, narrow portion of the bedrock valley west of Main Street (Plate 2-11).

A cross-sectional TDS isocontour map (Figure 2-57) drawn along cross section line I-I' (Plate 2-11) shows that TDS values >1,000 mg/l occur in both shallow and deep groundwater beneath the Olin property, but are restricted to the base of the aquifer west of the property (note that the cross-section is generalized and well data were projected onto line of section I-I'). Furthermore, the dense layer with the highest TDS values (>20,000 mg/l) occurs in two pools at the base of the aquifer. The first begins beneath the property, and extends westward to a point between GW-45D and GW-58S. A much smaller volume of dense layer is restricted to a deep (over 120 feet bgs) depression in the bedrock surface in the vicinity of GW-83D. Excluding the dense layer samples, the majority of groundwater west of the property contain <500 mg/l TDS.

Also shown in Figure 2-57 are groundwater pH isocontours along cross section H-H'. Comparison of the TDS isocontour cross section with the pH isocontour cross section suggests that the dense layer at GW-83D has a higher pH than does dense layer with a similar TDS concentration closer to the site. To determine if this is true, the average TDS of groundwater from each well was plotted against pH in Figure 2-58. Also included on this figure are the TDS and pH values of four wells (MP-1 [port 4], GW-43D, GW-70D, and GW-83D). These groundwaters have TDS values of 30,000–37,000 mg/l, but they are located at different distances downgradient from the site (Plate 2-2), with MP-1 onsite, and GW-83D located the farthest from the site. Figure 2-58 indicates that as the dense layer migrates downgradient the pH increases as the dense layer mixes with groundwater and is neutralized by the aquifer material. Because some dense layer constituents, such as chromium and aluminum, have limited solubility at near-neutral pH values, this neutralization process is likely to limit their dissolved concentration mobility.

The effect of dense layer neutralization on chromium mobility is evident in the third cross section presented on Figure 2-57. Although dense layer is present at GW-83D, there is relatively little chromium detected in this well (0.074 mg/l on 10/26/96). This relation is also demonstrated by Figure 2-59, which shows the average TDS and chromium concentrations for each well, including the same four wells for which the pH and TDS were compared in Figure 2-58. Although the TDS values in these four wells were similar (30,000 to 37,000 mg/l), the wells close to the site (MP-1[port 4] and GW-43D) had chromium concentrations 3 to 4 orders of magnitude higher than in wells farther away from the site (GW-70D and GW-83D), confirming the general immobility of chromium in near-neutral pH groundwater.

2.7.2.5 Summary General Geochemical Conditions

- Close to the Olin property, the dense layer is typified by a dense, low-pH groundwater (as low as 3), with a high TDS concentrations (maximum measured concentration was 120,000 mg/l at GW-421) and high chromium concentrations (maximum measured concentration was 4,700 mg/l at MP-2, port 1).

- Over the range of Eh and pH values observed in the site groundwater, chromium(VI) is thermodynamically unstable, and is not be expected to occur. Matrix interferences during the analysis of chromium(VI) using EPA Method 6010 has likely resulted in false detections.

- As the dense layer migrated along the base of the bedrock valley, it reacted with the aquifer materials and is partially neutralized. The resulting increase in pH, (increasing from 3.5 to 5), effectively reduces the solubility and mobility of chromium. Consequently, the western-most portion of the dense layer contains high TDS concentrations (e.g., 35,000 mg/l) but very little chromium.

- Although measurement of DO is prone to interferences in the dense layer matrix, the dense layer is surrounded by groundwater containing <1 mg/l, suggesting that the dense layer is relatively anoxic.

- Measurement of Eh in both the dense layer and the surrounding groundwater indicates that both are slightly to moderately oxidizing and these Eh conditions are mainly a result of dissolved ions (e.g., Fe^{+2}) and pH which play a greater role in determining the availability of reactive electrons than the DO content of the system.

- Evaluation of the major ion chemistry using modified Piper diagrams indicates that the intermediate-TDS groundwaters may be enriched in ammonia (present as mostly dissolved ammonium) with respect to the other dense layer constituents, indicating that diffusion is an important mechanism by which dense layer constituents are transported into the intermediate-TDS groundwaters. Although diffusion may be important as a mass transfer mechanism, it is not effective at transferring high concentrations into the overlying groundwater, as evidenced by the sharp interface at the top of the dense layer, and significantly lower concentrations of these solutes in overlying groundwater.

- Geochemical modeling of groundwater at GW-83D (refer to section 5.2.2 and Appendix Q) indicates that chromium precipitation with ferric hydroxide is favorable and that the concentrations of chromium at GW-83D should decrease over time as chromium-bearing solids precipitate. The existence of these precipitates has been confirmed by electron microprobe analysis of the aquifer matrix at GW-83D.

2.7.3 Geochemical Characterization of Aquifer Material

The Supplemental Phase II Investigations included sampling and analysis of aquifer material to provide information of geochemical processes affecting dense layer chemistry and the fate and transport of chromium in the aquifer. First, in Section 2.7.3.1, the results of aquifer material testing provides a basis for understanding 1) what precipitates or adsorbed inorganics are present in the aquifer matrix, and 2) how the composition of these reaction products varies with depth. Then in Section 2.7.3.2, the results of experiments on dense layer and aquifer material samples from near the property are used to 1) quantify the solubility of chromium-bearing precipitates and 2) examine the role of partitioning in the fate of chromium in the dense layer. The results of similar testing of samples from both the periphery of the dense layer and intermediate TDS groundwater to quantity the adsorption of chromium and other inorganics is discussed in Section 2.7.3.3.

2.7.3.1 X-Ray Fluorescence and Electron Microprobe Analyses

To determine the effects of the dense layer on the chemical composition of aquifer materials and to provide information to aid in the interpretation of the geochemical processes affecting dense layer chemistry, samples of subsurface aquifer materials were collected and characterized by x-ray fluorescence (XRF) and electron microprobe analysis (EMPA). X-ray fluorescence analyses provide total elemental concentrations in solid samples. The electron microprobe is used to analyze the elemental composition of discrete solid particles or points within a particle; information that can be indicative of mineralogy and associations of specific elements with different solid phases such as authigenic precipitates.

2.7.3.1.1 XRF and EMPA Sample Collection and Methodology

For these analyses, samples of aquifer material were obtained at depth using a split-spoon sampler during drilling from a borehole located near multilevel piezometer MP-2. A total of 10 samples were collected over depths of 8.5 to 33 feet; a range that extends from "clean" aquifer materials in shallow groundwater (i.e., unexposed to the dense layer) into intermediate groundwater, and the dense layer zone at this location.

Splits of the aquifer samples were rinsed repeatedly with deionized water to remove dissolved constituents that might be present in the pore spaces and which might affect the chemical analyses of the aquifer solids. The XRF samples were then dried, sieved to less than 2 millimeters ground

to approximately 10 micron in a tungsten carbide grinder, and placed into a sample holder for insertion into the x-ray generating machine. The samples were then exposed to an x-ray beam and analyzed for total chromium and total iron. The concentration of an element in a sample is proportional to the intensity of the secondary x-rays with characteristic wavelengths generated by irradiation of the sample by the primary beam.

Samples for the EMPA were also dried, sieved to less than 2 millimeters in diameter, and then embedded in epoxy. The epoxy mounts were polished under kerosene to obtain a smooth cross-sectional surface through the grains and to prevent dissolution of water soluble minerals, and carbon-coated (Davis et al., 1993). The carbon coat provides a conductive path for the electron beam to escape the impinged sample. In EMPA, the electron beam (about 1 micrometer width) can be positioned on a particle to produce secondary x-ray emissions whose intensity and wavelength are indicative of the elemental composition of the solid. The proportion of the elements can be used to qualitatively determine which types of minerals or amorphous solids are present.

2.7.3.1.2 XRF and EMPA Results

Results from the XRF analyses show that total chromium concentrations increased from 10 to 20 mg/kg at 0 to 15 feet bgs to more than 1,000 mg/kg at 33 feet bgs (Figure 2-60). Total iron concentrations increase from approximately 0.5 percent by weight for the sample from the upper interval (0 to 18 feet bgs) to approximately 3.6 percent for the sample from 33 feet bgs. The samples from 15 to 18 feet bgs, where total chromium concentrations started to increase, is above the elevation where the dense layer currently resides in this area but the groundwater does contain approximately 0.2 mg/l chromium, as indicated by the sampling results from the nearby multilevel piezometer MP-2. The samples from 33 feet depth are within the upper part of the dense layer.

The increases in the chromium and iron concentration in the aquifer solids are consistent with the stratification of the dense layer (see Section 2.7.4) and indicate that geochemical precipitation reactions have removed chromium and iron from solution and incorporated it into solid phases.

To determine the identity of the chromium-bearing solid phases, the three deepest samples from 16 to 33 feet were analyzed by EMPA. These analyses showed the presence of chromium(III) hydroxide [$Cr(OH)_3$] generally in association with iron(III) and aluminum hydroxides [$Fe(OH)_3$ and $Al(OH)_3$] (Figures 2-61 and 2-62).

The metal hydroxides are most commonly present as coatings on layer silicates and less commonly as discrete particles. Figure 2-61a shows a layer silicate, resembling chlorite, from the 16 feet depth with a coating of iron-aluminum-hydroxide that contains 0.3 to 3.9. percent by weight chromium. Iron hydroxides containing a few tenths of a percent by weight chromium were also found as surficial coatings on clay particles (Figure 2-61b). Figure 2-62a shows a discrete particle comprised of an iron-aluminum hydroxide that contains 1.1 percent by weight chromium in the sample collected from 33 feet, which is at the top of the dense layer.

Other types of precipitates found in the aquifer samples include barite [$BaSO_4$], gypsum [$CaSO_4$ $2H_2O$] (not shown) and complex Al-sulfates of indeterminate type, possibly jurbanite [$AlSO_4OH{:}5H_2O$], basaluminite [$Al_4SO_4(OH)_{10}{:}5H_2O$], $AlOHSO_4$, or alunite [$KAl_3(SO_4)_2(OH)_6$] (Figure 2-62b).

The typical occurrence of the hydroxides as coatings on layer silicates indicates that hydrolysis reactions have caused local neutralization of the dense layer to the extent that the precipitation of secondary solids, including metal hydroxides and sulfates, has occurred. The hydroxides serve as solubility controls for several cationic metals [i.e., chromium(III), iron(II), and aluminum] present as solutes in high concentrations in the dense layer. Additionally, the sulfate-containing solids may be a result of precipitation reactions that occurred as the dense layer solutes reacted with the aquifer materials. The aluminum sulfates are normally found only under acid conditions and would not be expected to occur naturally in the glacial sediments comprising the aquifer.

The physical identification of the types of solids incorporating metals, especially chromium, is important because their solubilities (the amount of a solute that can exist in solution) will limit the metal concentrations in solution, depending on the chemical conditions. The chemical conditions within the dense layer are conducive to the formation of metal hydroxide and sulfate solids. Geochemical reactions causing neutralization of the dense layer are discussed in more detail in the geochemical modeling study presented in Section 2.7.5.

2.7.3.2 Chromium Solubility and Partioning Studies

To address MADEP concerns for the mobility of chromium from the dense layer, the Supplemental Phase II Investigations also included experimental studies to provide information on geochemical processes affecting the fate and transport of chromium in the aquifer. Experimental studies to determine the total acidity of the dense layer discussed in Section 2.7.2.2 above, were used to assess the extent to which natural neutralization reactions might attenuate its migration through the aquifer. Other experimental studies discussed in this section were conducted to determine the solubility of the chromium(III) solids formed by neutralization of dense layer solutions and the partitioning of chromium(III) between solid and solution phases.

2.7.3.2.1 Solubility of Chromium-Containing Dense Layer Precipitate

Partial neutralization of the dense layer by hydrolysis reactions with aquifer minerals should result in the precipitation of chromium hydroxide solids similar to those observed in the EMPA study (Section 2.7.3.1). Because the solubility of the chromium(III) hydroxide formed by dense layer neutralization will control the concentration of chromium(III) in solution, solubility experiments were conducted with samples of the precipitate slurry created during the total acidity determination (Section 2.7.3.2.1).

The procedure for the solubility experiments was as follows. Aliquots of the precipitate slurry were placed into 50-millimeter polycarbonate centrifuge tubes filled with 0.1 molar (M) NaCl previously purged of air with $N_2(g)$. Aliquots of either 1 M HCl or 1 M NaOH were added to

adjust the pH of the solutions to provide a range of pH for solubility measurements. After the pH was adjusted, the tubes were filled completely to prevent contact with air, capped, and placed into a constant temperature bath set at 25(±2)°C. The tubes were agitated daily to keep the solutions mixed and to resuspend the solids to speed the approach to steady-state conditions. A total of 11 tubes over the pH range of 3.7 to 8.2 were prepared.

The tubes were each sampled after 16 days of equilibration by opening them and inserting a $N_2(g)$ bubbler to preclude the incursion of air. Approximately 10 milliliters of sample was removed, filtered at 0.45 micrometers and acidified to pH <2 with 0.1 M HCl. The pH of the solution was measured at the time of sampling. The concentrations of total dissolved chromium in the solution samples were determined by ICP.

Results from the solubility experiments show a trend with pH that is generally consistent with the theoretical solubility of chromium hydroxide [$Cr(OH)_3(am)$] calculated using thermodynamic data from Rai et al. (1987) (Figure 2-63). This result indicates that chromium(III) concentrations in dense layer solution undergoing neutralization reactions with aquifer solids can be predicted by the solubility of chromium(III) hydroxide; an important consideration for modeling the fate and transport of chromium(III) in the groundwater system contacting the dense layer.

2.7.3.2.2 Chromium(III) Partitioning between Solution and Solid Phases

Chromium(III) solutes entering the aquifer by diffusion away from the dense layer or mixing with overlying groundwater may be partitioned to the solid phase through adsorption reactions, depending on the solution chemical conditions. Because partitioning has an important effect on solute fate and transport for conditions of undersaturation with chromium(III) hydroxide solubility, a series of experiments were conducted to obtain partition coefficients for the chromium(III) solutes. This study used samples from the most highly concentrated portion of the dense layer and aquifer matrix samples collected near the Olin property. Partioning studies discussed in Section 2.7.3.2.3 were performed on samples from both the periphery of the dense layer and intermediate TDS groundwater collected from beneath Maple Meadow Brook wetlands.

In the partitioning experiments, a sample of the dense layer from monitoring well GW-42D was used as the source of soluble chromium(III) (see Table 2-30). The aquifer solids were obtained from the borehole (CPT-2) located near multilevel piezometer MP-2, also used for the XRF and EMPA studies. Solids from a depth of 8.5 to 9.5 feet, which is well above the dense layer at this location, were used in all experiments. The aquifer solids are typical of the area, consisting primarily of sand-sized quartz and silicate rock fragments. Before use in the experiments, fragments >2-millimeter diameter were excluded by sieving because this size fraction contributes only minimally to the adsorptive capacity.

The experiments were conducted at dilutions of the dense layer solution of 50, 100, 200, 500, and 1,000 times, using deionized water as the diluant. These dilutions simulate concentrations in the groundwater resulting from dispersion and dilution distal to the dense layer. Prior to use in experiments, the diluted dense layer solutions were capped to exclude air and allowed to sit for

approximately 30 to 60 minutes to check for the formation of precipitates, and, thus, ensure that the solutions were not oversaturated with the solubility of any metal hydroxides whose precipitation during the experiments would confound interpretation of the results. For each dilution, 7 to 8 individual tubes with solid:solution ratios ranging from 1:3.5 to 333 were prepared.

The experiments were conducted by placing a weighed amount of aquifer material into a 50-milliliter plastic centrifuge tube. An aliquot of diluted dense layer solution sufficient to completely fill the tube to eliminate any head-space was then added and the tube capped. The tubes were placed on a rotary mixer that turned the tubes end-over-end at a rate of 3 to 4 revolutions per minute. After 24 hours of mixing at room temperature (19 to 22°C), the tubes were opened and a 25-milliliter sample was extracted from each, and the solution pH was measured. The samples were filtered at 0.45 micrometer and acidified to pH <2 with 1 N HCl to preserve them for subsequent analysis of total dissolved chromium concentrations by ICP. For purposes of interpretation, all dissolved chromium in the experimental solutions was assumed to be chromium(III) as was found for the dense layer in general. The rate of chromium(III) oxidation by oxygen is so slow (estimated half-life for chromium(III) of 1.5 to 9 years, Saleh et al. 1989; Schroeder and Lee, 1975) that it is unlikely to have produced measurable chromium(VI) over the 24-hour time frame of the experiments.

Results from the partitioning experiments demonstrate a sharp increase in the removal of chromium(III) from solution with increasing solid to solution ratio (Figure 2-64; Table 2-31). This result implies that the chromium was removed from either adsorption and/or by precipitation reactions.

A convenient parameter for describing the amount of solute removal by adsorption is the partitioning coefficient, K_d, defined in general terms by:

$$K_d = \frac{x/m}{Ceq}$$

where x is the mass of constituent adsorbed, M is the mass of the aquifer matrix to which it is adsorbed, and Ceq is the concentration of the consituent in the groundwater. Partition coefficients for chromium(III) were derived from the difference in chromium(III) in solution resulting from adsorption for the range in pH obtained from diluting the dense layer solution by 50 to 1,000 times (Table 2-31).

The derived K_{ds} show a strong dependence on pH, increasing sharply from about 10 at pH <4.3 to 200 to 300 at higher pH (Figure 2-65). This result is similar to that obtained by Griffin et al. (1977) who observed a sharp increase in chromium(III) adsorption to clay minerals at pH >4. The increase in K_d with pH is probably caused by deprotonation of solid surfaces with increasing pH, resulting in greater affinity of hydrolyzed chromium(III) species [i.e., $CrOH^{+2}$, $Cr(OH)_2^{+}$] for the solid surfaces.

Adsorption of the positively charged chromium(III) species against positive surface charges existent in the acidic experimental suspensions implies strong inner-sphere bonding between active sites and the chromium(III) ions. At pH >4.7, K_d determination is difficult because of the increasing probability of chromium hydroxide precipitation which limits soluble chromium(III) to near detection-level concentrations. Chromium(III) hydroxide precipitation is probably initiated by the formation of hydroxide polymers on oxide mineral surfaces that coalesce into surface precipitates and have a solubility similar to bulk chromium hydroxide (Charlet and Manceau, 1992). This process may be effective in removing chromium(III) from solution under conditions of slight undersaturation with the solubility of chromium hydroxide thereby causing the sharp increase in K_d with pH.

The experimental results indicate that dissolved chromium(III) is partitioned strongly onto the aquifer solids at pH >4.3 either through adsorption and/or precipitation reactions. These reactions remove chromium(III) from solution under condition of undersaturation with the solubility of chromium hydroxide partially attenuating its mobility in the groundwater system. This result is important for understanding the processes likely to affect the fate and transport of chromium(III) derived from the dense layer in the groundwater.

2.7.3.3 Additional Aquifer Material Partitioning/Adsorption Studies

As discussed above in Section 2.7.3.2.2, as groundwater migrates through the aquifer, dissolved constituents may be adsorbed by the aquifer materials, removing them from solution. Section 2.7.3.2.2 focused on the fate of chromium in the most concentrated portion of the dense later, this section examines fate and transport of the dense layer constituents which reside immediately upgradient from the public supply wells. Since partitioning behavior is highly site specific, the adsorption tests were designed to test aquifer matrix in potential preferential flow zones in the aquifer beneath Maple Meadow Brook wetlands and to closely simulate site conditions (i.e., pH and redox conditions).

As discussed in the previous section, several contributing laboratory studies were used to determine the site geochemical conditions including the dense layer acidity (Section 2.7.2.2), the solubility of precipitates (Section 2.7.3.2.1), and the composition of these precipitates (Section 2.7.3.1.2). The resulting data, coupled with our understanding of the geochemical conditions in the groundwater system (Sections 2.7.2 through and 2.7.6) were used to develop the conceptual model of groundwater migration and geochemical model of the fate and transport of dense layer constituents (Section 5).

2.7.3.3.1 Sample Collection, Preparation, and Analysis

Two groundwater samples (GW-83D and GW-87D) were collected by Smith personnel and shipped on ice to PTI's Boulder laboratory. The groundwater samples were selected for testing based on their ammonia, sulfate, and chromium concentrations and their proximity to the municipal supply wells. Smith personnel also shipped two samples of aquifer material (GW-86M

and GW-84S), which were frozen prior to shipment. This aquifer material was collected from sampling intervals that were representative of the likely flow paths in the aquifer to the municipal supply wells.

Dense layer constituents occur primarily at the base of the aquifer. If these constituents were to be captured by the municipal wells, which are screened in the shallow portions of the aquifer, they would be transported through shallow and intermediate-depth aquifer materials. Therefore, aquifer samples from each of these two depth intervals (GW-84M and GW-84S, respectively) were included in the adsorption tests. GW-86M aquifer material, which was collected from 64 to 66 feet bgs, consisted of medium to coarse sands. The GW-84S aquifer material was collected from 44 to 45 feet bgs and consisted of a fine sand.

Aquifer materials were placed in a nitrogen-filled glove box (<1.0 percent oxygen) and allowed to thaw overnight. Excess water was then poured off the samples. Aquifer materials were placed in sealed 4-liter bottles and placed in a tumbler for one hour to homogenize the sample. A 500-gram sample of each aquifer material was collected for determination of Cr, NH_4^+, SO_4^{2-}, Fe, Al, Mn, and total organic carbon (TOC). In addition, triplicate samples of each were taken for determination of moisture content.

The groundwater was filtered in the lab immediately prior to use in these tests under a nitrogen atmosphere. Approximately 8 liters of each groundwater were pumped through a 0.45-micrometer filter, maintaining a nitrogen atmosphere above the pre- and post-filter samples. Splits of each water were used to measure pH, dissolved oxygen, specific conductance, temperature, and concentrations of sulfate, ferrous iron, and total iron. Samples of each groundwater were placed into pre-preserved bottles and sent to Columbia Analytical Services (CAS), in Kelso, Washington, for determination of initial water quality (Table 2-32).

K_d Determination

Subsamples of each aquifer material were mixed with each groundwater in 1-liter HDPE bottles at four different water:solid ratios by mass (1:1, 3:1, 9:1, and 19:1), for a total of 16 tests (Table 2-33). Each bottle was sealed in the glove box and equilibrated in a rotating tumbler for 12 hours. Following settling, each bottle was filtered to <0.45 micrometer in the glove box. The pH, dissolved oxygen, specific conductance, temperature, and sulfate, ferrous iron, and total iron concentrations of each filtrate were then measured. Each filtrate was placed in the appropriate pre-preserved bottles, removed from the glove box, and sent to CAS for determination of the analytes in Table 2-32.

In addition, the following quality control tests were performed:

- Two tests were conducted using only the groundwater without soil: these data were used to determine the amount of sorption or precipitation of constituents of concern onto the test apparatus.

- Two tests were conducted using deionized water and aquifer material, to determine the amount of potential desorption from the solid phase (soil blank).

- One test was performed using deionized water only, to determine if constituents were being released from the testing apparatus (equipment blank).

The K_d was calculated from the ratio of the adsorbed concentration to the aqueous concentration. The formula used to calculate K_d is:

$$K_d = \frac{x/m}{C_{eq}} = \frac{(C_o - C_{eq})\,V/m}{C_{eq}}$$

where C_o is the concentration of the analyte in the groundwater blank, C_{eq} is the concentration in the final solution, V is the volume of groundwater used in the test, and m is the mass of aquifer material used in the test.

2.7.3.3.2 Adsorption Study Results

The pH of sample GW-83D, which was initially 4.87, increased to 5.26 in the groundwater blank, and varied between 5.84 and 5.08 following equilibration with the aquifer materials (Table 2-34). The pH of sample GW-87D dropped from 7.47 to 7.16 when exposed to the test apparatus, and varied between 7.52 and 6.99 when equilibrated with the aquifer materials. For both groundwater samples, the variability in pH was independent of the soil:water ratio, suggesting that the variability was not due to interactions between the aquifer materials and the groundwater. The SC decreased slightly as the soil:water ratio increased (Table 2-35). This may have been a result of adsorption of dense layer constituents which were not analyzed in this study. Dissolved oxygen in the slurries was consistently below detection limits, indicating that oxygen was effectively excluded from the apparatus.

The measured compositions of aquifer materials used in these experiments are summarized in Table 2-35. Neither sample contained any detectable organic carbon, and only GW-86M had detectable ammonia (1.05 mg/kg). Both aquifer materials contain low concentrations of iron and aluminum, suggesting that these samples contain little amorphous iron hydroxide or gibbsite [$Al(OH)_3$], both strong sorbents of chromium (EPRI 1984). Chromium concentrations in these two aquifer samples (6.6 and 5.4 mg/kg in GW-86M and GW-84S, respectively) are below concentrations typically found in sedimentary rocks; 11 to 90 mg/kg (Mason and Moore, 1982).

Each groundwater sample was subjected to the anoxic adsorption procedure in the absence of any aquifer material (groundwater blank). Comparison of these samples with the initial groundwater chemistry indicates that there were appreciable changes in the groundwater chemistry as a result of sorption to the apparatus (HDPE bottles). Therefore, the groundwater blank chemistry was assumed to be C_o in equation 2.

Results of the anoxic K_d tests are presented in Table 2-36. The data in this table were used to calculate the K_d values presented in Table 2-37. These data indicate that, of the three primary constituents being evaluated (ammonia, sulfate, and chromium), ammonia was consistently adsorbed with K_d values ranging up to 2.1 liters per kilogram (l/kg) and averaging 0.4 l/kg, with ammonia concentrations increasing in only 2 of the 16 tests performed. The $K_{d\text{-}NH4+}$ was independent of the system pH.

The sulfate concentration increased or remained unchanged in 11 of 16 tests performed, indicating that sulfate adsorbed to aquifer materials only at the highest solid:solution ratios tested.

Of the eight tests performed using groundwater containing detectable chromium concentrations (i.e., GW-83D, which contained 0.058 mg/l chromium), five showed no change or an increase in chromium concentration. Literature values for $K_{d\text{-}Cr(III)}$ in sand aquifers range between 1.7 and 1,700 l/kg, with an average value of 70 l/kg. The high concentration of iron in GW-83D (2,030 mg/l) may partly explain adsorption study results. If a small amount of the total iron load adsorbed onto the aquifer material it would compete with chromium for adsorption sites. If iron and chromium were present at similar concentrations like those observed in dense layer samples from GW-42D, the chromium would compete with the iron for the adsorption sites. But because iron is present at GW-83D at concentrations 35,000 times higher than the chromium, it is able to effectively compete for the adsorption sites. Nevertheless, chromium migration will be retarded due to precipitation of amorphous hydroxides (e.g., iron hydroxide) and adsorption of chromium on to these hydroxides.

In addition to chromium, sulfate, and ammonia, the concentrations of other dense layer constituents also were measured. The concentrations of aluminum, iron, and manganese were measured because, if they were to precipitate from solution as hydroxides during the adsorption procedure, the effectiveness of the aquifer material as an adsorbent could be enhanced. Aluminum concentrations in the adsorption test leachates decreased when groundwater sample GW-83D was used (Table 2-36). Furthermore, this decrease was proportional to the amount of soil used during the adsorption test (Table 2-36), and was independent of the system pH (Figure 2-67), suggesting that this decrease was due to the adsorption of aluminum rather than precipitation of gibbsite [$Al(OH)_3$] following changes in the system pH. The average $K_{d\text{-}Al}$ was 4.7 l/kg. With the exception of the 1:1 soil-to-water test, aluminum concentrations in GW-87D groundwaters decreased from 0.07 mg/l to <0.05 mg/l for all other tests performed with this groundwater, indicating minimal adsorption (Table 2-37). The post-adsorption aluminum concentration in the 1:1 soil-to-water ratio using GW-87D groundwater was 6.27 mg/l.

With the exception of two of the samples tested, the iron concentrations decreased slightly during the adsorption test (Table 2-36). Like aluminum, this decrease was independent of the system pH (Figure 2-68), but unlike aluminum, it was also independent of the soil-to-water ratio used during the test. These data illustrate why it is important to generate site-specific measurements of adsorption behavior, as the adsorption capacity of the both of the aquifer materials depended on the groundwater being tested. Both aquifer materials tested had a much stronger affinity for iron dissolved in groundwater from GW-87D (average $K_{d\text{-}Fe}$= 120 l/kg) than for GW-83D

groundwaters (average $K_{d\text{-}Fe}$= 0.3 l/kg). This difference may be due to the large disparity in the iron concentration of these two groundwaters (16.1 and 2,030 mg/l, respectively), or may be due to the complexation of iron in GW-87D with sulfate, thus reducing the efficiency of adsorption.

For all but two of the tests, manganese concentrations decreased in amounts proportional to the amount of aquifer material used. The average K_{dMg} was higher for tests using GW-83D groundwater (44 l/kg) than for those using GW-87D groundwater (1.4 l/kg).

2.7.3.3.3 Summary of Adsorption Study Conclusions

The adsorption tests using the aquifer matrix and dense layer samples from Maple Meadow Brook wetlands indicate that low levels of chromium occurring along the leading edge of the dense layer (e.g., 0.06 mg/l at GW-83D) are sparingly adsorbed by the aquifer matrix. As would be expected, ammonia, chloride, and sulfate, show similar results that indicate that adsorption is not a dominant mechanism controlling their migration rates away from the less concentrated dense layer. One possible reason for this lack of adsorption is the high concentrations of other cations that occur along periphery of the dense layer (e.g., iron concentration at GW-83D is 2,030 mg/l). These other cations may initially out-compete the chromium for adsorption sites. However, continued transport and precipitation of amorphous hydroxides would tend to create more adsorption sites and continue to scavange chromium from the groundwater through precipitation and/or adsorption reactions.

The data from this study also indicate that sulfate is not adsorbed onto the site aquifer materials. In addition, while the tests indicate that the site aquifer materials do have the capacity to adsorb ammonia, it is only weakly adsorbed ($K_{d\text{-}NH4+}$ = 0.4 l/kg). Therefore, adsorption is not likely to affect the rate of ammonia migration through the aquifer.

The adsorption tests also indicate that aluminum was adsorbed by aquifer materials $K_{d\text{-}Al}$ = 4.7 l/kg). Iron was strongly adsorbed by aquifer materials from GW-87D groundwaters ($K_{d\text{-}Fe}$ = 120 l/kg), but weakly adsorbed from GW-83D groundwaters ($K_{d\text{-}Fe}$ = 0.3 l/kg). Manganese demonstrated the opposite behavior, with aquifer materials capable of strongly adsorbing manganese from GW-83D groundwaters (K_{dMn} = 44 l/kg) and only weakly adsorbing manganese from GW-87D groundwaters (K_{dMn} = 1.4 l/kg).

2.7.4 Multilevel Piezometers

Three multilevel piezometers were installed at the study area to obtain an accurate vertical characterization of groundwater chemistry above and within the dense layer to understand potential future transport of analytes of concern. Prior data from standard monitoring well clusters indicated that there were two layers in the groundwater comprising a less dense surficial groundwater overlying a green-colored dense layer. The dense layer was acidic (pH from 3 to 4), with a specific conductivity as high as 100,000 umhos , specific gravity up to 1.12 grams per milliliter (g/ml), and elevated chromium, chloride sodium, ammonia, iron, manganese and sulfate

concentrations. Conversely, the shallow groundwater had neutral pH, specific gravity near background, and specific conductivity and constituent concentration values as much as five orders of magnitude lower than the dense layer.

Initially, six CPT were performed during September 1995 to improve understanding of the dense layer interface with overlying groundwater. The presence of cobbles and boulders in the alluvial aquifer prevented the cone penetrometer pushes from penetrating the dense layer. However, the results suggested the possible existence of a transition layer with low pH (2 to 4), and near background constituent concentrations based on specific conductivity and resistivity measurements. Multilevel piezometers were selected to provide better vertical resolution than the existing monitoring well clusters and the cone penetrometer tests.

2.7.4.1 Methods

2.7.4.1.1 Multilevel Piezometer Installation

Three Waterloo System multilevel piezometers from Solinst Canada were installed at the study area adjacent to or near existing monitoring wells and prior CPT holes at locations shown on Plate 2-2. Multilevel piezometer MP-1 was located in the vicinity of the former Lake Poly liquid waste disposal area, former acid pits, former lagoons, and other pits, adjacent to CPT-1 and approximately 240 feet northeast of monitoring well GW-36. Multilevel piezometer MP-2 was located within the center of the upper Western Bedrock Valley, adjacent to CPT-2 and monitoring well GW-42D, in which the highest concentrations of dissolved solutes have been detected at the study area. Multilevel piezometer MP-3 was located within a deep closed bedrock depression the Western Bedrock Valley, downgradient from MP-2, adjacent to CPT-4, and approximately 140 feet northeast of monitoring well cluster GW-44 (Plate 2-2).

The spacing of the sampling ports in the multilevel piezometers was originally designed based on inductance logs of nearby monitoring wells and CPT data. Where inductance log and CPT data indicated significant changes in groundwater parameters, the sampling ports were spaced 1.5 feet apart for increased resolution, increasing to 5.5 feet where groundwater parameters were expected to be constant. Piezometer design was modified in the field to accommodate the encountered depth to water and bedrock (Figure 2-69).

The boreholes were drilled in March 1996 with a Barber Dual Wall Air Rotary drill rig using a standard air rotary hammer, while simultaneously advancing an 8-inch ID, rotating, temporary steel casing. The boreholes for MP-1 and MP-2 were drilled until bedrock was encountered. The borehole for MP-3 was terminated 67 feet bgs prior to reaching bedrock, but penetrated at least 10 feet into the dense layer based on observation of the cuttings. This depth was approximately 14 feet deeper than the prior estimated depth to bedrock. All soils and water generated during drilling were contained and stored at the Olin property for characterization and appropriate disposal. In addition, the drill rig and the down hole tools were steam cleaned prior to the start of each borehole and prior to leaving the study area.

The multilevel piezometers were constructed of prefabricated 2-inch ID Schedule 80 PVC sampling ports and blank casing sections which were assembled then lowered inside the temporary steel. Figure 2-70 shows a photograph of a typical length of one of the piezometers. Either 1/4-inch or 3/8 x 1/2-inch tubing of sufficient length to reach approximately 3 feet above ground surface was attached to each sampling port. The 3/8 x 1/2-inch tubing was of sufficient diameter to allow collection of water level data using a Solinst water level probe (Model No.M-102/P3). The annular space around the piezometers was backfilled with clean No. 3 silica sand adjacent to sampling ports and separated by either granular bentonite or bentonite chips to preclude vertical flow between sampling ports. The temporary steel casing was withdrawn as the backfill materials were emplaced to prevent collapse of the natural formation on the borehole. For MP-1 and MP-2, sand and granular bentonite were emplaced with a Solinst Sand-Bentonite Injector using dry compressed air to drive the materials down an HDPE tube to the elevation at which they are to be emplaced, thus preventing bridging. For MP-3, sand and bentonite chips were emplaced manually from ground surface because the height of the water column in the borehole hampered use of the Injector. The sampling tubes in each multilevel piezometer were threaded through manifolds, with each port uniquely numbered for easy identification during sampling. Ports were numbered in sequential order from Port 1 as the deepest port to the highest numbered port as the shallowest port. Multilevel piezometer MP-1 was completed with a stick-up, while MP-2 and MP-3 were completed with flush mounts.

Each sampling port was developed by connecting silicon tubing to the sampling tubes and pumping with a peristaltic pump until little or no turbidity was observed in the water. Several ports were clogged with sand grains or bentonite that were cleared by reversing the pump flow direction and injecting clean, potable water for up to 20 seconds at a time. Following clearing of a clog, the port was pumped for up to 10 minutes to ensure that all injected water was removed. Despite multiple efforts to develop recalcitrant ports, two ports (9 and 15) in MP-1 and one port (5) in MP-2 could not be developed. Water generated during well development was contained and stored at the Olin property for characterization and appropriate disposal. Following development, the sampling ports were allowed to equilibrate with ambient aquifer conditions for at least 3 weeks before sampling.

2.7.4.1.2 Sampling and Field Analysis

Selected ports of MP-1 were sampled on April 18, 1996, with all developed ports on all three multilevel piezometers sampled during May 20-23, 1996. Multilevel piezometers were purged prior to sampling by pumping groundwater from each port through oxygen impermeable silicon tubing connected to a peristaltic pump, directly into a polycarbonate flow-through cell in which pH, oxidation-reduction potential (Eh), dissolved oxygen (DO), and specific conductivity probes were mounted using air-tight fittings. Groundwater was allowed to fill the cell, and purging continued until the electrode readings stabilized prior to sampling. Groundwater readings typically stabilized after two to five gallons of water had been flushed through the system. Dissolved oxygen was measured using an Orion Model R-810 DO meter with barometric pressure and temperature compensation; specific conductivity was measured using a Fisher Accumet specific conductivity meter; temperature, pH, and Eh were measured using an Orion Model 290A

pH-concentration meter equipped with an Orion pH triode probe model 9107 for pH and temperature readings, and a Fisher Accumet platinum/silver-silver chloride electrode model 13-620-82 for relative Eh readings, to which the reference exposure was calibrated against ZoBell's solution (Nordstrom, 1977).

The redox state of groundwater is typically measured using a platinum electrode. However, Lindberg and Runnells (1984) demonstrated that the redox state of groundwater measured using a platinum electrode cannot be used to predict the valence state of a specific species. Therefore, the concentrations of the ferrous-ferric couple, total nitrate+nitrite-ammonia couple, chromium(III)-chromium(VI) couple, and dissolved oxygen were used to characterize groundwater redox conditions. Samples for ammonia analysis were filtered and acidified to pH<2 with sulfuric acid and stored at 4±2°C, while those for total nitrate+nitrite analysis were stored unacidified. Samples of filtered water for laboratory chromium(VI) analysis were collected in clean polyethylene containers, stored at 4±2°C. In addition to laboratory analysis of redox analytes, filtered groundwater samples were analyzed on-site for chromium(VI), total iron, and ferrous iron using Hach AccuVac vials and a Hach DR/2000 spectrophotometer (Hach Company, Loveland, Colorado). Samples were analyzed colorimetrically for ferrous iron following complexation with 1,10-phenanthroline while a second aliquot was analyzed for total iron following reduction of ferric iron to ferrous iron, with ferric iron determined by difference. Method detection limits for ferric iron and total iron were 0.01 mg/l. chromium(VI) was measured spectrophotometrically following complexation with 1,5-diphenylcarbohydrazide. The method detection limit for chromium(VI) was 0.01 mg/l.

2.7.4.1.3 Laboratory Analyses

General groundwater chemistry, described by major cation (aluminum, calcium, iron, magnesium, potassium, sodium, manganese, [all analyzed by EPA Method 200.7/6010] and ammonia [EPA Method 350.1]), anion (chloride [EPA Method 325.2], total nitrate+nitrite [EPA Method 353.2] and sulfate [EPA Method 375.4]), and trace metal (antimony, arsenic, barium, beryllium, cadmium, calcium, chromium, cobalt, copper, lead, nickel, selenium, silver, thallium, vanadium, zinc [all by EPA Method 200.7/6010], and mercury [EPA Method 245.1]) concentrations, was determined for use in the geochemical model MINTEQ4 to evaluate metal solubility and dissolved metal speciation (Section 2.7.5). Alkalinity was determined by titration to the bicarbonate endpoint (APHA method 406C).

Groundwater samples for chromium(VI) were analyzed within 24 hours colorimetrically (EPA Method 7196) and/or by coprecipitation followed by inductively coupled plasma spectroscopy (ICP) (EPA Method 6010). chromium(III) was determined by difference between total chromium and chromium(VI). Groundwater samples from selected ports were analyzed for specific gravity according to ASTM methodology.

2.7.4.2 Results and Discussion

2.7.4.2.1 Summary of Field and Laboratory Analyses

The analytical data collected during the April and May sampling events are in general agreement (Tables 2-38 and 2-39). However, chloride concentrations in ports 1 and 2 of MP-1 during May were reported as 91,000 and 69,000 mg/l, respectively, compared to 17,000 and 13,000 mg/l in April. In addition, chloride in ports 1 and 2 of MP-1 for May are inconsistent with specific conductivity data that did not increase in accord with the reported increase in chloride concentration. For example, in port 1 the specific conductivity increased from 81,500 umhos to 90,200 umhos, while port 2 decreased from 74,200 umhos to 70,900 umhos. Thus, chloride data from the May sampling event are unrepresentative of the actual dense layer chemistry at MP-1, and were therefore not used in the evaluation presented below. Reported ammonia concentrations between the April and May sampling events differed for the lower six ports in MP-1 (Tables 2-38 and 2-39). However, reported ammonia concentrations have been variable at many monitoring wells at the study area.

Specific gravity typically increased with depth at all locations (Figure 2-71), reaching a maximum in MP-2, with the lowest peak specific gravities in MP-3. These results were consistent with the observed constituent concentrations (i.e., in general, the highest peak concentrations were observed in MP-2, and the lowest peak concentrations were observed in MP-3). Similarly, measured specific conductivity values were consistent with observed constituent concentrations. The values were low in the shallow groundwater, ranging from 282 umhos to 2,690 umhos , but increasing to as high as 102,600 umhos in the dense layer (Figure 2-72; Table 2-40).

Dissolved oxygen concentrations measured in MP-1 and MP-2 were <1.0 mg/l, exhibiting little or no trend with depth (Figure 2-73; Table 2-40) and were consistent with earlier results (Section 2.7.2.2.4). Any oxygen dissolved in infiltrating water near these locations is rapidly consumed by biological or geochemical reactions at shallow depths. However, in MP-3, there appears to be a decreasing trend of oxygen concentration from the water table to approximately 12 feet below the water table. Dissolved oxygen may penetrate further into the aquifer at this location due to the relatively clean shallow groundwater at MP-3.

pH was slightly acidic to slightly alkaline (5 to 8) in shallow groundwater, generally decreasing with depth, and becoming acidic (3 to 4) in the dense layer (Figure 2-74; Table 2-40). At all three multilevel piezometers, redox conditions ranged from very slightly reducing to slightly oxidizing (Figure 2-75; Table 2-40). There were no apparent trends with depth, although at the pH of these groundwaters, the conditions were mildly oxidizing in the shallow portion of the aquifer and mildly reducing in the dense layer.

For the other redox-sensitive couples, i.e., ferrous/ferric iron (Figure 2-76; Table 2-41), chromium(III)/chromium(VI) (Figure 2-77; Table 2-39), and total nitrate+nitrite-ammonia (Table 2-39), the reduced form generally dominated over the oxidized form. In a large majority of the sampling ports, ferrous iron exceeded 80 percent of the total iron detected in the groundwater

samples (Table 2-41), while contrary to previous site data, no chromium(VI) was detected in any groundwater sample according to results of the colorimetric (EPA Method 7196) analysis. Detections of chromium(VI) with the coprecipitation (EPA Method 6010) analysis, both in the current and previous sampling events, were due to analytical interferences in the coprecipitation method as discussed in Section 2.7.2.1.2.

Reported concentrations for laboratory-measured and field-measured total dissolved iron were in good agreement (Figure 2-76). In shallow groundwater at MP-1 and MP-3, total dissolved iron was either not detected or detected at concentrations only slightly above the detection limit, while at MP-2 iron was detected at concentrations ranging from 2.1 to 38 mg/l in shallow groundwater. At each location, total dissolved iron concentrations in the dense layer exceeded 10 mg/l, reaching a maximum of 3400 mg/l in MP-1 (port 1).

At each multilevel piezometer location, total dissolved chromium concentrations in the dense layer exceeded 4 mg/l (Figure 2-77) with a maximum of 4,700 mg/l in MP-2 (port 1). In MP-1 and MP-3, chromium was not detected in shallow groundwater, while in MP-2 total dissolved chromium was detected in shallow groundwater at concentrations decreasing with elevation from 0.028 mg/l at 78.6 feet MSL to 1.5 mg/l at 60.1 feet MSL.

Similar to iron and chromium, concentrations of other metals, including magnesium, sodium, and aluminum (Figure 2-78; Table 2-39), and anions, including sulfate and chloride (Figure 2-79; Table 2-39), were at a maximum in the dense layer, decreasing rapidly by approximately two to four orders of magnitude in the transition to shallow groundwater.

Sulfate and calcium concentrations were elevated in the uppermost ports (ports 16 through 18) of MP-1, decreasing with depth before increasing into the dense layer (Figure 2-79; Table 2-39). The elevated concentrations in the upper 10 feet at MP-1 are probably attributable to remnants of calcium sulfate from the former lined lagoons. Gypsum sludge, which consists primarily of calcium sulfate, is noted in the boring log for monitoring well GW-19D (CRA, 1993) and has a solubility of about 2,000 mg/l at a neutral pH. The slightly elevated concentrations of magnesium, sodium, and chloride may also be associated with the gypsum sludge. The elevated constituent concentrations were responsible for the slightly elevated specific conductivity values measured in the shallow ports of MP-1 (Figure 2-72; Table 2-40).

Concentrations of several constituents in the ports above the dense layer at MP-2 were elevated in comparison to MP-1 and MP-3. Chromium was detected in all ports of MP-2, with concentrations generally increasing with depth (Figure 2-77; Table 2-39). Although not detected in the two shallowest ports of MP-2, aluminum concentrations also increased with depth. Conversely, concentrations of constituents including iron, magnesium, sodium, chloride, sulfate, and ammonia decreased with depth to the dense layer (Figures 2-76, 2-78, and 2-79; Table 2-38). In general, these data suggest that aluminum and chromium in the shallow groundwater may be contributed from the dense layer or from residual reaction products of the dense layer, while the

other consistuents are also contributed from source which interacts with the shallow groundwater. The different observed trends in vertical concentrations are also likely to be the result of a combination of several possible hydrogeologic, geochemical, and temporal (e.g., seasonal) factors.

The hydrogeologic properties of the aquifer at MP-2 may also impact the constituent distribution by encouraging preferential contaminant flushing. The geologic boring log for adjacent monitoring well GW-42D indicates a general coarsening of the sediments with depth below 80.4 feet MSL (CRA, 1993), which implies an increasing hydraulic conductivity with depth. In this case, the intermediate portion of the aquifer, between approximately 60 and 70 feet MSL at MP-2, may have a higher groundwater flow velocity than the shallower portion of the aquifer, and thus, be flushed of constituents faster. This would result in the lower concentrations of sodium, chloride, sulfate, ammonia, magnesium, and iron observed in the intermediate ports of MP-2 in comparison with those in the shallower ports of MP-2 (Table 2-39). The increasing trends of chromium and aluminum concentrations with depth at MP-2 (Figures 2-77 and 2-78; Table 2-38), would appear to contradict this hypothesis, however, the concentration of these constituents are controlled by pH which decreases with depth (Figure 2-74; Table 2-40).

The constituents detected above the dense layer at MP-2, particularly the aluminum and chromium were probably the result of remnant mixing into shallow groundwater flow from the former unlined pits that were eliminated when the lined lagoons and treatment system were installed in the early 1970s, relined in the early 1980s, and later removed in 1986. The previous geochemical evaluations show that the dense layer, which was probably at a higher elevation when the unlined pits active, reacts and precipitates chromium-bearing aluminum hydroxides. It is believed that a remnant reaction zone may be contributing aluminum and chromium to the shallower ports at MP-2. Subsequently, clean infiltrating groundwater has flushed significant constituent mass from the shallow groundwater in the vicinity of MP-1, but insufficient time has passed to flush it from the vicinity of MP-2 and into the Ditch System.

Other factors which may have impacted the constituent distribution observed at MP-2 include temporal variations in the waste stream sent to the unlined pits (for example, cessation of chromate use in 1967), dilution of intermediate zone groundwater by waters from the adjacent ditch due to purging of GW-42S, contributions from residual calcium sulfate observed at MP-1, and localized changes in groundwater flow directions due to installation of the South Ditch weir in July 1994.

Groundwater elevations at the multilevel piezometer locations did not have a clear trend (Table 2-42). At MP-1, after correction for the density of the groundwater, there was a slight downward gradient through the water column. At MP-2, there is no vertical gradient between the two monitored ports open to the shallow groundwater. At MP-3, there is a slight downward gradient between the monitored ports open to the shallow groundwater. At MP-2 and MP-3, there is an upward gradient from the dense layer to the shallow groundwater.

2.7.4.2.2 Groundwater Profile Definition

Figures 2-71 through 2-79 show vertical profiles of various parameters in each of the multilevel piezometers. These profiles clearly show the sharp contrast between the dense layer located at the base of the aquifer and the overlying cleaner groundwater.

Water quality within the dense layer is typically acidic (pH <5.15), with high concentrations of sulfate, sodium, chloride, iron, magnesium, aluminum, and specific conductance. Ammonia is also elevated, but not as consistently as the other parameters. Within the dense layer, specific gravity is significantly higher than background groundwater (up to 1.10 g/cm^3). Flow within the dense layer is highly density-dependent with movement taking place down bedrock slopes under the influence of gravity.

Above the dense layer the groundwater quality is significantly different with higher pHs, and concentrations of contaminants are several orders of magnitude less than in the dense layer. The specific gravity is approximately 1.00 g/cm^3. Flow within this zone is advective and is typical of groundwater flow under the influence of a simple hydraulic gradient.

Between the dense layer and the overlying cleaner groundwater is a zone of transition between these two very different waters. Moving upward through the transition layer, concentrations drop significantly, pH increases, and specific gravity decreases. The transition layer also marks a transition between the underlying dense layer where flow is dominated by density effects and the overlying cleaner water where flow is dominated by advection. It is likely that within the transition layer, both flow mechanisms are in operation to variable degrees. The approximate elevations of these different layers based upon multilevel piezometer chemical results is presented on the table below.

Top of Layer Elevations Based on Multilevel Piezometers (ft MSL)

	Elevation Range for Top of Layer (ft MSL)		
Layer	**MP-1**	**MP-2**	**MP-3**
Dense Layer	54.9 to 57.4 b/w/ ports 4&5	54.14 to 57.14 b/w ports 4 & 6	45.5 to 47 b/w ports 6 & 7
Transitional Layer	57.4 to 58.9 b/w ports 5 & 6	57.14 to 58.64 b/w ports 6 & 7	47 to 48.5 b/w ports 7 & 8

The vertical profiles in Figures 2-71 through 2-79 show a sharp interface between the dense layer and the overlying groundwater (i.e., a thin transition layer). This relatively sharp interface in the presence of high concentration gradients is consistent with the results of some recent theoretical and experimental work.

The narrow thickness of the transition layer is consistent with recent studies which suggest that diffusion and dispersion across high concentration gradients is retarded due to non-Fickian behavior. For example, laboratory experiments with $CuSO_4$ performed by Carey et al. (1995) required effective Fickian diffusion coefficients up to 70 percent less than the infinite dilution diffusion coefficient to match observed profiles. Similarly, Hassanizadeh and Leijnse (1995) found it necessary to reduce the longitudinal dispersivity by a factor of three so that a Fickian model could match breakthrough curves observed in a one-dimensional laboratory experiment with high NaCl concentration gradients. Thus, mass loss from a stationary dense layer to shallow groundwater may be sufficiently slow such that constituent concentrations in the overlying groundwater (e.g., chromium) are diluted below levels of concern. This hypothesis is supported by non-detections of chromium, iron, and aluminum in MP-1 port 7 (Table 2-39), which is between 1.5 and 3.0 feet above the top of the transition layer. In addition, chromium, iron, and aluminum are not detected in MP-3 port 11, which is between 4.5 and 6.0 feet above the top of the transition layer. These observations further support the hypothesis that the slightly elevated constituent concentrations observed in shallow groundwater at MP-2 are not totally due to diffusion or dispersion from the dense layer but are probably due to contributions from remnant reaction zones, residual sources on-property, or from incompletely flushed mixtures of liquid wastes and shallow groundwater from the former unlined pits.

2.7.4.3 Major Ion Chemistry In Multilevel Piezometer Samples

The major ion chemistry of groundwater samples collected from the multilevel piezometers was evaluated using the modified Piper diagrams discussed in Section 2.7.2.3 (Figures 2-80 through 2-85). These diagrams demonstrate that the chemistry of dense-layer groundwaters is distinct from that of the overlying groundwaters, and that these chemical differences indicate diffusive constituent transport mechanisms from the dense layer into a transitional layer.

Evaluation of Major Ions at MP-1

Figures 2-80 and 2-81 show the major ion chemistry of groundwaters collected from MP-1. The data points represent the average constituent concentrations from the two sampling events. The anomalously high chloride concentrations detected in ports 1 and 2 during May of 1996 (Section 2.7.4.2) were excluded. The chemistry of the groundwaters collected from MP-1 was compared with that of the groundwaters collected from the wells (shown on Figures 2-53 and 2-54, and indicated by the shaded areas on Figures 2-80 through 2-85). This comparison indicated that the major ion chemistry of the dense layer groundwater collected from the lowest four ports of MP-1 was different from that of dense layer groundwater samples from wells such as GW-42D (Figure 2-80). The groundwaters from the uppermost ports of MP-1 had chemistries that were similar to the intermediate-TDS groundwaters collected from the wells (Section 2.7.2.3). Similar major ion chemistries were observed in the groundwaters collected from MP-2 and MP-3 (2.7-38 through 2.7-41). Because the groundwaters from GW-42D plot between the lower ports of the piezometers and the intermediate-TDS groundwaters on the modified Piper diagrams, this suggests that the groundwaters collected from GW-42D may be mixture of these two water types

(i.e., the groundwaters from GW-42D do not provide pure samples of the dense layer). Therefore, the samples collected from the lower ports of the multilevel piezometers may provide a more accurate characterization of the dense layer than is provided by the wells alone.

The TDS circles on Figure 2-44 indicate that groundwater collected from the lower four ports of MP-1 have substantially higher TDS values than groundwater from other ports. These data indicate a sharp boundary between the groundwater of port 4 and port 5. In addition to the changes in pH and specific conductance observed between ports 4 and 5 (Table 2-40), Figure 2-80 indicates that the major ions are transported across this boundary at differing rates, with ammonia and chloride being disproportionately enriched in the groundwater of port 5. (If all these constituents were being transported across this boundary at the same rates, then the groundwater would plot at the same location on the Piper diagrams, but would have smaller TDS circles as a result of dilution). The mechanism for this enrichment of ammonia is likely to be differing diffusion rates (discussed in Section 2.7.2.3). Port 5 groundwater also had much higher TDS values than those from ports 7–13, immediately above it (Figure 2-81). This trend is also evident in the measured specific conductivity values shown in Figure 2-72.

The increase in TDS and specific conductivity observed in the uppermost ports of MP-1 (Figures 2-72 and 2-80) corresponded to a disproportionate enrichment of calcium and sulfate in these waters (Figure 2-80). This increase may indicate that as rainwater infiltrates site soils, it dissolves calcium and sulfate from residual lagoon material (principally $CaSO_4$), which has been found in soils of some on-property bore holes.

Evaluation of Major Ions at MP-2

The samples from MP-2 fall into three general groups (Figures 2-80 and 2-81): the dense-layer found in the lower four ports (those with visible TDS circles on Figure 2-81), groundwater enriched in chloride (ports 10–12), and all other samples. The chloride-enriched samples have the lowest observed specific conductance values (Table 2-40) and occur in the middle portion of the groundwater column (Figure 2-72). The occurrence of this lower-TDS groundwater at intermediate depths in the vicinity of MP-2 may be due to the infiltration of low-TDS surface waters from the Offsite West ditch into the groundwater system (Figure 2-57) or may be due to difference in flushing created by hydrogeologic conditions. The higher concentrations of constituents found in the shallow groundwaters were either contributed from remnant reaction zones, residual sources on-property, or from incompletely flushed mixtures of liquid wastes and shallow groundwater from the former unlined pits.

The remaining groundwater from this piezometer (ports 6–9 and 13–17) had compositions that were similar to the intermediate-TDS groundwater described in Section 2.7.2 (Figure 2-82), suggesting that these dissolved constituents were either transported away from the dense layer, or derived from the initial release event (i.e., as the dense liquid wastes infiltrated the aquifer at the unlined pits).

Evaluation of Major Ions at MP-3

As with piezometers MP-1 and MP-2, the dense-layer in MP-3 was restricted to the lower four ports of the piezometer, with a sharp transition in major ion chemistry between the port 6 groundwater and the groundwater that overlie it (Figures 2-84 and 2-85). The systematic compositional changes that occur in the groundwater in the vertical profile sampled by piezometers MP-2 and MP-3 were very similar between these two locations, suggesting that similar mechanisms dispersed the dense-layer constituents throughout the two water columns.

2.7.4.4 Summary of Multilevel Piezometers Conclusions

The multilevel piezometer investigation contributed considerably to the understanding of pathways from the dense layer and the future fate and transport of the major inorganic consistuents detected at the site. The multilevel piezometers provide direct evidence of a sharp concentration gradient between the dense layer and overlying groundwater as predicted by geochemical modeling of site conditions. The chemical results also document that the density differences have resulted in minimal interaction of the dense layer and overlying groundwater. The results demonstrate that slow diffusion dominates transport pathways and that the mobility of the cationic metals such as chromium are further limited by precipitation and adsorption reactions, while differences in the reactivity has resulted in effectively greater diffusion of more mobile ions (ammonia, chloride, and sulfate). Further testing of the entire vertical profile of the groundwater and the dense layer at the three multilevel piezometers has shown that chromium(III) is predominant species and not chromium(VI). Other conclusions which contribute to the overall understanding and assessment of transport pathways are summarized below.

- Analytical results show that while the greatest mass of inorganic contamination is within the dense layer, there is a sharp drop in concentrations within a few feet above the dense layer and the concentrations in the overlying groundwater are generally 100 to 1,000 times less. These dramatic compositional changes show that mixing may be limited by non-Fickian diffusion processes.

- The compositional changes that occur across this boundary also indicate that the more mobile dense-layer constituents are transported disproportionately across it. The disproportionate dense-layer constituent concentrations also suggest that diffusion is an important, although long and slow, transport mechanism at the interface between the dense layer and intermediate groundwater.

- Calcium sulfate sludge found in some onsite soils may continue to affect the quality of the shallowest groundwaters in the vicinity of MP-1. The increase in TDS and specific conductivity observed in the uppermost ports of MP-1 also corresponded to a disproportionate enrichment of calcium and sulfate in these waters which may indicate that as rainwater infiltrates site soils, it dissolves calcium and sulfate from residual lagoon material.

- The presence of chromium throughout the groundwater column in the vicinity of MP-2 indicates that all groundwater at this location has been affected by the wastes released at the property, either during the initial infiltration event, or as a result of constituent transport away from the dense layer at the base of the aquifer.

- Comparison of the multilevel piezometer data with the data from the deep wells that are screened across the dense layer (e.g., GW-42D) indicates that the wells are sampling a mixture of dense-layer groundwater and the overlying intermediate-TDS groundwaters.

2.7.5 Geochemical Processes Affecting Solute Fate and Transport

The XRF studies described above (Section 2.7.3) showed an increase in chromium concentration in aquifer solids with depth into the dense layer. The EMPA showed that the increase in chromium was the result of the precipitation of chromium hydroxide as grain coatings and discrete particles. Other precipitates of metal hydroxides and metal sulfates were also found that are indicative of reactions between the acidic dense layer solution and aquifer solids. These observations are very important because they provide evidence that geochemical processes of neutralization and solubility have significantly attenuated the transport of metal ions, in the groundwater contacting the dense layer.

To further investigate the specific processes affecting solute transport, a geochemical modeling study was conducted. Geochemical models are used to determine whether chemical conditions favorable for the formation of specific precipitates, such as chromium hydroxide, are existent in the groundwater based on comparisons of solute concentrations with theoretical solubility relationships derived from thermodynamic data. The models can be used to substantiate whether the formation of the specific solids is thermodynamically possible for the groundwater conditions and to predict the maximum concentrations likely to occur under equilibrium conditions. In combination with direct chemical analyses of solids by EMPA, the geochemical models can be used to describe the processes affecting the fate and transport of solutes in an aqueous system.

2.7.5.1 Geochemical Modeling Methods

The MINTEQ4 geochemical model was used in this study. This version was developed at Battelle Pacific Northwest Laboratory for the U.S. Environmental Protection Agency (USEPA) (Eary and Jenne, 1992; Peterson et al. 1987). MINTEQ4 was chosen over the MINTEQA2 model maintained by the USEPA (Allison et al., 1990) because it contains updated equilibrium constants for chromium(III) and chromium(III)-iron(III) hydroxides from the work of Rai et al. (1987) and Sass and Rai (1987) which are relevant to the mobility and solubility of chromium in the dense layer.

To depict the chemistry of the groundwaters relative to equilibrium conditions, chemical compositions from samples collected from monitoring wells in 1992 and 1995, and multilevel piezometers in 1996 were entered into the MINTEQ4 geochemical model. The model was then used to calculate the activities of individual aqueous species [e.g., Cr^{+3}, $CrOH^{+2}$, $Cr(OH)_2^+$, $CrSO_4^+$, etc.]. These activities were then summed to give a total species activity, such as total chromium(III) activity; a quantity that could be compared to various solubility relationships. Individual ion activities were calculated within MINTEQ4 using the extended Debye-Hückel B-dot model (Helgeson, 1969; Smith, 1988).

Equilibrium constants were corrected for the effect of temperature by either Maier-Kelley power functions for equilibrium constants or the van't Hoff equation subject to the availability of empirical coefficients or reaction enthalpies, respectively, within the MINTEQ4 database (Eary and Jenne, 1992; Smith, 1988). Water temperatures in the monitoring wells range from 9 to 18°C, hence the effect of temperature on most of the pertinent solubility equilibria is negligible.

2.7.5.2 Modeling Assumptions

The redox environment of the aquifer is only slightly oxidizing in that most of the dissolved iron is present as iron(II), however, sulfur is present as sulfate and no sulfide has been found (Section 2.7.2). Dissolved oxygen concentrations determined with the multilevel piezometers were generally less than 0.5 mg/l, although near surface samples from MP-3 ranged as high as 2.5 mg/l (Section 2.7.4). Additionally, detailed analyses of the dense layer discussed in Section 2.7.2.1, indicated that the chromium is present only in the trivalent oxidation state (chromium(III)).

Based on this information, all dissolved chromium was specified as chromium(III) in the MINTEQ4 runs. The concentrations of dissolved iron(II) and iron(III) were taken directly from the measured values where available. Where unavailable, dissolved iron was specified to be iron(II) because this oxidation state is predominant in the dense layer (Section 2.7.2). Dissolved manganese was specified to be manganese(II) because manganese(III) is not stable in aqueous solution and manganese(IV) is only present under very oxidizing conditions. Dissolved sulfur was specified to be present only as sulfate because sulfide or other forms of sulfur have not been reported for the groundwaters.

2.7.5.3 Geochemical Modeling Results and Discussion

The MINTEQ4 geochemical model was used to calculate the activities of the aqueous metal species for comparisons to probable metal hydroxide solubility controls. Results for chromium(III) show that the upper limit for chromium(III) activity in the dense layer is consistent with the solubility of chromium hydroxide [$Cr(OH)_3(am)$] (Figure Geo2-87). In fact, a linear regression of log[total chromium(III) activity] versus pH for pH < 6 and log[total chromium(III) activity] > $10^{-6.5}$ M, which eliminates nondetect samples, results in a slope of 2.15 (r^2=0.87); a value that is close to the theoretical value of 2.0 expected for the reaction

$$Cr(OH)_3(am) + 2H^+ = CrOH^{+2} + 2H_2O$$

which dominates aqueous chromium(III) speciation under acidic conditions (Rai et al. 1987). The lower bound for chromium(III) activity is less well defined but is generally represented by the solubility of a chromium(III)-iron(III) hydroxide solid solution with a formula of $(Cr_{0.69}Fe_{0.31})(OH)_3(am)$ (Figure 2-86). This composition is the highest chromium-containing solid solution for which solubility data are available for comparison (Sass and Rai, 1987).

The dense layer solutions generally contain higher concentrations of chromium(III) than iron(III) (Section 2.7.2), hence can be expected to form hydroxides with high chromium(III) to iron(III) ratios upon neutralization. Sass and Rai (1987) were unable to synthesize hydroxides with chromium(III) to iron(III) ratios greater than 0.69 to 0.31 by precipitation from solution, indicating that solutions with high proportions of chromium(III) to iron(III) may form end-member solids rather than a complete solid solution. Additionally, iron(III) hydroxides were probably present in the aquifer naturally and may have served as nucleation sites for chromium(III) hydroxide formation as suggested by the low chromium(III) contents in iron hydroxide phases observed in the EMPA study.

Solubility control by chromium hydroxide was also indicated by the results of the experimental solubility study conducted with neutralized dense layer solution. Additionally, chromium-enriched surface coatings and particles of iron and aluminum hydroxides were found to be present in the aquifer materials by EMPA. These studies in combination with the geochemical modeling results indicate that chromium(III) concentrations in the groundwater are limited by chromium hydroxide solubility.

The occurrence of other metal hydroxides in the aquifer materials implies that chemical conditions under which iron, aluminum, and manganese concentrations are solubility limited also exist in the groundwater system. This hypothesis was tested by comparisons of metal activities calculated by MINTEQ4 for the groundwaters against theoretical solubility curves.

The comparison for aluminum shows that its activities in the groundwater are bounded by the solubilities of amorphous aluminum hydroxide [$Al(OH)_3(am)$] and microcrystalline gibbsite [$Al(OH)_3(uc)$] with a few exceptions for pH > 5.5 (Figure 2-87). These solids are typically found to control dissolved aluminum concentrations in many aqueous systems. For example, crystalline and microcrystalline gibbsite have been postulated as solubility controls for aluminum in surface waters (Driscoll and Schecher, 1988; Driscoll *et al.*, 1984). In acid mine drainage areas, aluminum concentrations have been found to be bounded by the solubilities of amorphous aluminum hydroxide and microcrystalline gibbsite (Nordstrom, 1982; Nordstrom and Ball, 1986; Sullivan *et al.*, 1988).

Iron(III) concentrations in acidic systems are generally expected to be controlled by amorphous iron(III) hydroxide [$Fe(OH)_3(am)$] solubility (Langmuir and Whitmore, 1971). A plot of the total iron(III) activities in the site groundwaters indicates that solubility control by amorphous iron(III) hydroxide may be valid for pH < 4.5, but there is substantial deviation from the theoretical solubility curve at higher pH (Figure 2-88). The low pH solutions also show the increase in

amorphous iron(III) hydroxide solubility that would be predicted from the formation of iron complexes (e.g., [$FeSO_4^+$ and $Fe(SO_4)^-_2$])under the acidic conditions of the dense layer, which contains up to 90,000 mg/l sulfate.

The deviances from the solubility curve at pH > 4.5 may reflect disequilibrium with amorphous iron(III) hydroxide solubility or just the variability of its solubility. Langmuir and Whitmore (1971) report a range in the solubility product for amorphous iron(III) hydroxide solubility from -37.3 to -43.3 ($K = \log[Fe^{+3}][OH^-]^3$), which they attribute to aging differences in crystallinity of iron(III) hydroxide in different environments. Also, much of the dissolved iron in the groundwater is present as iron(II), which is unlikely to be converted to iron(III) because of the absence of oxygen in the groundwater. Iron(II) concentrations are generally limited by the solubilities of iron(II) carbonates under near neutral to alkaline pH conditions. However, there is insufficient carbonate in the groundwaters to precipitate iron(II) carbonate solids, hence they cannot be expected to act as solubility controls for dissolved iron.

Manganese(II) activities in groundwaters show similarities to iron(III) in that they indicate a linear decrease up to about pH 5.5, but are scattered at higher pH (Figure 2-89). Under acidic conditions, the manganese(II) activities show an upper limit consistent with the solubility of manganite ($\gamma MnOOH$) and a lower limit approximated by birnessite (δMnO_2) solubility.

Manganite is known to form from the oxidation of Mn(II) and is favored over feitknechtite ($\beta MnOOH$) in sulfate-containing solutions (Hem and Lind, 1983; Kessick and Morgan, 1975). Monitoring data indicated that the dense layer is only slightly oxic and contains high sulfate concentrations, hence only limited oxidation of Mn(II) to form $\gamma MnOOH$ may have occurred.

The major anion in the dense layer is sulfate, hence it may be expected that conditions favorable for the precipitation of metal sulfates may exist in portions of the aquifer contacted by the dense layer. To investigate this possibility, the MINTEQ4 was used to calculate saturation indices (SIs) for the potential metal sulfate solids to provide information on the potential for these solids to form. The SI for a solid is defined by

$$SI = \left(\frac{Ion\ Activity\ Product}{K_{sp}} \right)$$

where K_{sp} is the solubility product of the solid. A solution that is oversaturated with respect to the solubility of a solid will have an SI >0, indicating that the solid can precipitate but cannot dissolve. An SI <0 indicates that the solid can dissolve but cannot precipitate, and an SI = 0 means the solution is exactly at equilibrium with the solubility of the solid for the given set of conditions. The SI is helpful for identifying potential solubility controls for multicomponent systems that cannot easily be portrayed in solubility diagrams such as those used for the metal hydroxides.

The presence of the Calcium-Sulfate Landfill near the site (CRA, 1993) and high sulfate concentrations in the dense layer make it likely that conditions of gypsum solubility have been reached in the more acidic portions of the dense layer. This conjecture is confirmed by a plot of the SI for gypsum [$CaSO_4{:}2H_2O$] against pH. The gypsum SIs indicate a trend from undersaturation at pH > 4.5 to slight oversaturation at pH < 4.5 (Figure 2-90a). A calcium sulfate-type phase, presumably gypsum, was found by EMPA of a sample of aquifer material from the dense layer (Section 2.7.3.1), consistent with the solubility relationship shown in Figure 2-90a. The trend in gypsum SIs indicates that sulfate concentrations may be limited by gypsum solubility only in the most acidic portions of the dense layer.

A hydrated aluminum sulfate phase was also observed in the EMPA of the aquifer material from the dense layer zone. Results from the MINTEQ4 modeling indicate that solutions with pH <5.5 were all oversaturated with jurbanite [$AlSO_4OH{:}5H_2O$] solubility (Figure 2-90b). Solutions with pH >4 were also greatly oversaturated with basaluminite [$Al_4SO_4(OH)_{10}{:}5H_2O$] solubility (Figure 2-90c). Thus, a region of overlap exists for the formation of these two aluminum sulfate solids. Basaluminite is metastable compared to another aluminum sulfate solid, alunite [$KAl_3(SO_4)_2(OH)_6$], over its probable stability range, but alunite generally requires temperatures greater 25°C for formation whereas basaluminite precipitates directly from low temperature solutions (Nordstrom, 1982). Consequently, for the acidic dense layer solutions with temperature less than 19°C, basaluminite may be a more likely precipitate than alunite.

Jurbanite, basaluminite, and an unnamed hydroxy aluminum sulfate phase have been postulated to be solubility controls for dissolved aluminum in acidic systems comparable to those found in the dense layer-affected aquifer (Karathanasis et al. 1988; Nordstrom, 1982; van Breemen, 1973). However, for the dense layer and related groundwaters analyzed here, the persistence of strong oversaturation with the solubilities of these aluminum sulfate solids indicates that although conditions are favorable for their formation, they do not serve as effective solubility controls for either aluminum or sulfate.

2.7.5.4 Summary of Geochemical Processes Affecting Solute Fate and Transport

The results of geochemical modeling show that in the dense layer, the concentrations of aluminum, chromium(III), iron(III), manganese(II), calcium, and sulfate are controlled by the solubilities of their respective oxyhydroxide and sulfate solids. The significance of the equilibria modeling is that solubilities can be used to describe source terms at different locations in the aquifer and to determine the maximum concentrations of solutes possible. A portion of the solutes diffuse upwards towards the shallow aquifer where they mix and react with aquifer materials, resulting in a loss of metals from solution because of hydroxide precipitation. Consequently, the transport rate in the vertical direction is diffusion limited, resulting in only a narrow layer of elevated solute concentrations residing just above the dense layer. In the horizontal direction, the mobile anions are transported downgradient with groundwater movement, whereas the reactive solutes, such as aluminum, chromium(III), iron(III), and

manganese(II), are partitioned to the solid phase, and hence are removed from solution. Other modeling conclusions which contribute to the overall understanding and assessment of transport pathways are summarized below.

- Both geochemical modeling of the groundwaters and experimental results indicate that chromium(III) concentrations will be limited by the solubility of chromium hydroxide in the aquifer system. Above a pH of about 5.8, the solubility of chromium hydroxide is below the MCL for dissolved chromium (0.1 mg/l).

- Analyses of aquifer solids by EMPA showed the presence of the metal hydroxide solids indicated to be solubility controls by the geochemical modeling results. This finding provides strong evidence that the formation of metal hydroxides such as $Cr(OH)_3$ has partially attenuated the transport of metal solutes in the groundwater.

- The results of partitioning experiments indicate that chromium(III) will be strongly attenuated by adsorption and/or precipitation to the aquifer material under conditions of undersaturation with chromium hydroxide solubility, particularly for pH > 4.4.

- Manganese, iron, and silica are present in high concentrations in the dense layer. Given that these solutes were not known to have been released in the original waste stream, they were probably derived from the dissolution of silicate and oxyhydroxide minerals present in the aquifer. Dissolution reactions generally consume acid, and hence have probably moderated the pH of the dense layer solutions to some extent, although they have also contributed mineral acidity to the dense layer.

- The slow rates at which silicate mineral dissolution reactions consume acidity and the persistence of the acidic dense layer solutions in the aquifer indicate that natural attenuation reactions are unlikely to significantly reduce the dense layer acidity.

- Dissolved chromium in the groundwater is present only as chromium(III). Previous analytical results indicating the presence of chromium(VI) were in error because of matrix interferences. This result is consistent with knowledge of the thermodynamics and kinetics of chromium(VI) and reduction by iron(II), which is present throughout the groundwater.

- The geochemical modeling indicates that there are no chromium complexes present in the groundwater that are likely to increase the solubility of chromium hydroxide.

- As with chromium(III), the geochemical modeling results indicate that aluminum concentrations are limited by the solubility of aluminum hydroxide.

- At pH <4.5, the concentrations of iron(II) and manganese(II) are also limited by the solubilities of their respective oxyhydroxide solids. At higher pH, iron(II) and manganese(II) do not appear to be solubility limited, possibly because of variabilities in thermodynamic constants or incomplete oxidation under the near anoxic conditions of the groundwater.

- At pH <4, sulfate concentrations are limited by the solubility of gypsum. However, at higher pH, the groundwaters are gypsum undersaturated, probably because of low calcium concentrations in the groundwaters mixing with the dense layer solutions. The groundwaters are oversaturated with various aluminum sulfate solids but these appear to be ineffective solubility controls for both sulfate and aluminum.

- Differences in the reactivity of different solutes with the aquifer material has resulted in a separation of the mobile anions from the less mobile cationic metals in areas downgradient from the dense layer. The concentrations of metal solutes [i.e., chromium(III), aluminum, iron(II), and manganese(II)] are limited by the pH-dependent solubilities of metal hydroxide solids, hence are largely confined to the low pH regions of the dense layer. The concentrations of the anions (i.e., chloride and sulfate) are not appreciably limited by solubility controls, hence their concentrations are affected primarily by dilution.

2.7.6 Geochemical Investigation of Surface Water and Flocculent

The geochemistry of the Off-Property West Ditch and South Ditch surface water and groundwater was investigated to determine the origin and composition of flocculent observed in portions of the Ditch System streambed. Process wastes released from the former liquid waste disposal areas are considered to be the main source of inorganic contaminants to Ditch System surface water (CRA, 1993). Additional sources of inorganic contaminants to the Ditch System include historical overflows of Lake Poly to the On-Property West Ditch; leaching of contaminants from surface and subsurface soils in the vicinity of the former liquid waste disposal areas to shallow groundwater; and discharge of contaminated shallow groundwater to Ditch System surface water during periods of high groundwater (or low surface water) levels.

Inorganic analytes of concern in the dense layer and site groundwater include ammonia, chloride, sulfate, chromium, and acidic pHs. Many of these analytes also occur at elevated levels of concern in ditch shallow groundwater and the surface water associated with the Ditch System (CRA, 1993), indicating that historical discharge of contaminated site groundwater to the Ditch System may be impacting the surface water environment. Of particular concern, is the presence of chromium-bearing flocculent in the Off-Property West Ditch and South Ditch streambeds (Smith, 1995). Laboratory analysis of the flocculent shows that it is composed of many of the same analytes found in groundwater and the dense layer, suggesting that the origin of the flocculent may be the result of groundwater discharge to the ditches (BCM ,1993).

Field observations describe the flocculent as a white to red, fluffy, low-density solid that resides on ditch streambed sediments. When the flocculent is disturbed, it becomes easily suspended in the water column and transported downstream by the current. In chemical terms, a flocculent forms through the process of flocculation, i.e., the agglomeration or aggregation of suspended colloidal (<10 um diameter) particles resulting in the formation of a solid-phase material or "flocculent" (Stumm and Morgan, 1981; Sax and Lewis, 1987). In the Ditch System, the flocculent is hypothesized to originate as a precipitate of metal oxyhydroxides from solution as a result of the discharge and mixing of slightly acidic, metal-bearing groundwater and near-neutral surface water (BCM, 1993). While the geochemical precipitation of metal hydroxides in response to increases in pH is a well established phenomena in laboratory experiments, the exact mechanism(s) of flocculent formation in the natural environment may be the result of several geochemical processes, such as precipitation, co-precipitation and/or adsorption that act in concert to produce the flocculent material observed at the site.

As described previously in Section 2.4, a weir was installed at the west end of the South Ditch in 1994 for the purpose of trapping flocculent to prevent downstream transport. In June 1994, flocculent was vacuumed from the ditches upstream of the weir prior to the installation of the weir plate in July 1994. The apparent reappearance of flocculent in May 1995 (Smith, 1995) suggests that the mechanism(s) causing flocculent formation may still be active, although sporadic flushing of residual colloidal material near the streambed, scouring and redeposition of "old" flocculent, and the occurrence of non-chromium-bearing precipitates could also partly explain the reappearance of flocculent material.

Shallow groundwater hydraulic head data for 1995 from wells along the Ditch System show that the influence of the weir is seasonal and highly localized. During periods of low water levels (summer and fall) ponded surface water extends a limited distance to the northwest of the weir and the northern portion of the Off-Property West Ditch is seasonally dry. During periods of high groundwater levels (winter and spring) the shallow groundwater discharges to the Ditch System. Groundwater and surface water levels collected before the weir was installed show that these high groundwater and discharge conditions represent a return to the Ditch System's original flow pattern (before the weir was installed).

Hydraulic heads measured in shallow groundwater wells, piezometers, and surface water along the Off-Property West Ditch and South Ditch during 1995 and 1996 show that these portions of the ditch are primarily groundwater discharge areas (see Table 2-17 "Groundwater and Surface Water Levels Along West and South Ditches"). In contrast, recharge to groundwater was shown only to occur near sample location GW-76S/SW-11 (located in the north branch of the Off-Property West Ditch) during three months in 1995 (which then became a groundwater discharge area during all of 1996), and at the Central Pond location GW-79S/SW-16, where heads are higher than shallow groundwater wells for most of 1995 and 1996. Hydraulic heads at all other sample locations clearly indicated that conditions of groundwater discharge to the ditches occurred for most of 1995 and 1996; with only short-lived seasonal deviations (e.g., GW-50S/SW-15 during May 1995; GW-77S/SW14 during July 1995; and GW-42S/SW-12 during October 1995).

Based on these data, it is hypothesized that flocculent formation is the result of the continued discharge of groundwater to surface water, and that the concentrations of metals and indicator analytes may be sufficient to produce the flocculent material observed in the ditches.

To test this hypothesis, additional data collection was undertaken during the Phase II Supplemental Investigation to characterize the chemistry of the flocculent material, establish its mode of formation, and to determine its geochemical fate. The focused data collection also provided a basis for proposed Phase III feasibility evaluations by determining: 1) the source of the flocculent, 2) the chemical reaction(s) that is causing flocculent formation, 3) the chemical composition of the flocculent, and 4) the hydraulic and chemical relationship between shallow groundwater and surface water in the Ditch System that contribute to flocculent formation (BCM, 1993).

2.7.6.1 Methods

Chemical composition data for the Ditch System surface water, shallow groundwater and flocculent obtained during the 1995 South Ditch Supplemental Phase II Field Investigation (Smith, 1995; BCM, 1995) in conjunction with surface water and flocculent data collected by Geomega in 1996 were used in this investigation. The South Ditch Supplemental Phase II Investigation provided inorganic chemical data from six co-located surface water (SW-11, SW-12, SW-14, SW-15, SW-16, SW-17) and shallow groundwater (GW-42S, GW-50S, GW-76S, GW-77S, GW-78S, GW-79S) sampling locations along the Off-Property West Ditch and South Ditch (Figure 2-91).Figure 2-91 These locations were sampled quarterly (February, May, July, and October) during 1995 and submitted for field and laboratory chemical analysis as described in Section 2.7.2.1.1 above. The suffix "-95" was added to the 1995 surface water sampling locations on figures showing all surface water locations to distinguish these sampling points from previous CSA locations (CRA, 1993). The sampling locations listed above are shown on Figure 2-91. These data are summarized in Tables 2-43 through 2-46 in this section.

Four additional surface water samples (SD-1, SD-2, SD-3, and SD-4) and a sample of the Central Pond were collected by Geomega in April 1996 to augment the Supplemental Phase II Investigation (Figure 2-91). These samples were submitted for the standard suite of metals and anions. All samples were field filtered using a 0.45 um filter prior to submittal to the laboratory for analysis. In addition, an unfiltered sample split from the Central Pond was submitted for analysis of total metal concentrations in addition to dissolved concentrations to assess the presence of colloids/particulates in the Pond. All additional surface water data is presented in Table 2-47; a comparison of unfiltered and filtered surface water chromium results are present in Table 2-48, Central Pond data is presented in Table 2-49.

A total of nine flocculent samples were collected from the Ditch System (Figure 2-91). One sample of whitish-colored flocculent (WF-1) was collected in May 1995 from the South Ditch immediately downstream from the Central Pond. A red-colored flocculent (RP-1) was also collected in May 1995 from the central branch of the Off-Property West Ditch. Both flocculent

samples were collected in a 5 gallon carboy using a peristaltic pump, vacuum filtered onto pre-weighed paper filters, and allowed to dry at room temperature. These flocculent samples were submitted for laboratory analysis of total metals and selected anions, and also submitted to the University of Colorado for electron microprobe analysis (EMPA). Chemical analyses of these samples are presented in Tables 2-50 and 2-51.

Six flocculent samples (Floc-1, Floc-2, Floc-3, Floc-4, Floc-5 and Floc-6) were collected in April 1996 from successive locations along South Ditch (Figure 2-91). These samples were also collected by vacuuming the flocculent from the streambed using a peristaltic pump onto pre-weighed paper filters. Five of the flocculent samples (Floc-1 through Floc-5) were submitted for total chromium analysis by EPA SW-846 method 6010 (ICP). The sixth flocculent sample (Floc-6) was submitted for electron microprobe analysis (EMPA) at the University of Colorado.

Data obtained from the laboratory chemical analysis and EMPA of flocculent samples were used to characterize the chemical composition and morphology of the flocculent material. Initial attempts to characterize the flocculent using x-ray diffraction (XRD) methods established the flocculent as a complex amorphous solid, but because of its amorphous nature was unable to identify the structure or chemical stoichiometry of the flocculent. Electron microprobe analysis was used instead to identify the stoichiometry and morphology of the flocculent because of its ability to analyze amorphous solids as well as provide photomicrographs of the flocculent.

All flocculent samples submitted for EMPA were analyzed at the Laboratory for Geological Studies at the University of Colorado, Boulder, following the procedure described by Davis et al. (1993). Briefly, the flocculent samples were mounted in an epoxy mold, allowed to cure at room temperature, and polished at low speed using kerosene to prevent dissolution of water-soluble phases. The surface of the mold was then cleaned with isopropyl alcohol and coated with a thin layer of carbon. Metal-bearing particles were identified using a combination of energy dispersive detection (EDS), wavelength dispersive detection (WDS), and backscatter electron image detection (BEI). Images of the flocculent particles were obtained using BEI and recorded as photomicrographs. The chemical composition of the flocculent was established relative to EMPA standards and provide a relative estimation of the stoichiometry of solid-phases comprising the flocculent.

The results of field, laboratory and EMPA analyses of surface water, groundwater, and flocculent were selectively used as input to a geochemical model to simulate the interaction between ditch shallow groundwater and surface water and determine if a flocculent-like material could be formed from the mixing of these waters. Modeling was performed using the geochemical model PHREEQC (Parkhurst, 1995). Version 1.4 of PHREEQC was chosen to model flocculent formation because of its capability to simulate a range of geochemical reactions relevant to the Ditch System including aqueous speciation and saturation index calculations, mixing of solutions, mineral equilibria, and surface complexation reactions. The results of the PHREEQC modeling are presented in Table 2-52.

2.7.6.2 Results and Discussion

2.7.6.2.1 Indicator Analytes

Ditch surface water and shallow groundwater indicator analytes (specific conductance, sulfate, chloride, sodium, ammonia, nitrate, pH, and Eh,) and major metals (aluminum, chromium, and iron) were examined using histogram plots to evaluate the concentrations of these analytes relative to the location of the weir. The histogram plots presented on Figures 2-91 through 2-100 and Figure 2-102 show the concentration of analytes on the y-axis and the sampling locations from upstream to downstream on the x-axis. Surface water sampling location SW-16 is in the Central Pond.

Specific conductance plots show increases in conductivity in surface water and groundwater in the South Ditch locations relative to the Off-Property West Ditch locations (Figure 2-91). Specific conductance is a measure of the electrical conductivity of a solution due to the combined effects of all ionic species in solution and therefore is an excellent indicator of the total concentration of dissolved analytes in solution. The specific conductance in Off-Property West Ditch surface water and groundwater locations upgradient of the weir are generally similar, with values ranging between 200 and 1,000 umhos. Conductivity increases by approximately 1000 umhos in the South Ditch surface water locations, reaching a maximum of 2,440 umhos (July 1995) at SW-17 (immediately downgradient of the weir). Conductivity values remain relatively stable at about 2,000 umhos in the Central Pond (SW-16), and then decrease to approximately 1,000 umhos at the downstream location SW-15. The observed decrease in conductivity at SW-15 indicates that significant attenuation of solute concentrations has occurred downstream of the Central Pond. SW-15-95 is located below a portion of the ditch with a lower gradient and slower surface water velocity. Conductivity in shallow groundwater also increases in the South Ditch locations relative to the Off-Property West Ditch locations, with moderate increases of approximately 500 to 600 umhos observed at GW-78S, reaching a maximum of 5,320 umhos at GW-79S (adjacent to the Central Pond). The elevated conductivity values in SW-16 and GW-79S probably reflect the hydraulic connection hypothesized to exist between groundwater and surface water at this location. Conductivity then decreases by as much as 3,800 umhos between GW-79S and well GW-50S (October 1995), indicating that attenuation and/or dilution has occurred in the downgradient portions of the South Ditch shallow aquifer. In general, the highest conductance typically occurs in July, and the lowest occurs in February.

Sulfate concentrations in ditch surface water and groundwater exhibit a very similar concentration profile to specific conductance (Figure 2-92). Correlation plots of sulfate versus specific conductivity give a least squares regression (r^2) value of 0.85 for surface water and 0.99 for groundwater, indicating a strong correlation exists between the two parameters. As with specific conductance, sulfate concentrations increase significantly in the South Ditch locations relative to the Off-Property West Ditch locations, reaching maximum concentrations at SW-16 (1,000 mg/l) and GW-79S (2,000 mg/l), and then decreasing significantly at downgradient locations SW-15 and GW-50S (Tables 2-43 and 2-45). Further decreases in sulfate levels in South Ditch surface water during 1996 (see Table 2-47) relative to 1995 may be in part due to source reduction,

dilution, or the attenuating effects of precipitation of iron-, aluminum- and chromium-bearing sulfate flocculent. It is important to note the elevated sulfate in both GW-79S and SW-16 (Central Pond). While 1995 and 1996 hydraulic head data implies that surface water recharges groundwater at this location, the sulfate data suggests the opposite, i.e., that groundwater may actually be discharging to the Central Pond. Concentrations of sulfate probably remain high in the Pond because of the restricted exchange of water between the Pond and the South Ditch, which prevents flushing of dissolved constituents to the ditch. April 1996 sampling of the Central Pond (Table 2-49) indicated that sulfate concentrations have decreased (630 mg/l) relative to 1995 levels (870 to 1,000 mg/l).

Sodium concentrations in surface water exhibit a gradual increase from the Off-Property West ditch locations to the South Ditch locations, but overall remain at relatively constant levels (Figure 2-93). In contrast, sodium concentrations in shallow groundwater remain fairly constant at approximately 25 to 50 mg/l in upgradient wells GW-76S through GW-78S but then exhibit substantial increases in downgradient South Ditch wells GW-79S (up to 350 mg/l) and GW-50S (up to 310 mg/l), suggesting that a source of sodium in groundwater (probably the southern-most edge of the dense layer) may exist along this reach of South Ditch. The gradual increases observed in surface water may be due to groundwater discharge of sodium at these locations. However, sodium that is discharged to the South Ditch may be attenuated by incorporation into flocculent; sodium was detected at a concentration of 8,700 mg/kg in a flocculent sample (WF-1) collected from the South Ditch (Table 2-50).

Chloride concentrations in surface water samples are generally higher than chloride concentrations in groundwater samples (Figure 2-51). A least squares regression shows that the occurrence of chloride is not correlated to sulfate in surface water (r^2= 0.004). However chloride concentrations in groundwater exhibit a trend similar to sulfate (r^2= 0.79) and specific conductances suggesting the occurrence of these analytes may be correlated. These data suggest that chloride and sulfate in groundwater are contributed from a similar source, while chloride concentrations in surface water are controlled by other sources such as non-point source contributions and treated groundwater discharges containing chloride from the NPDES outfall on the On-Property West Ditch (Figure 2-94).

All Off-Property West Ditch wells (GW-76S, GW-42S, GW-77S) and one South Ditch well (GW-78S) show chloride concentrations that average approximately 50 mg/l; whereas downgradient South Ditch wells GW-79S and GW-50S average 270 mg/l and 185 mg/l, respectively, indicating that a source of chloride may be discharging upgradient or at these locations. Elevated chloride in groundwater is probably residual chloride contamination from the liquid wastes or disposal of other wastes, which appears to be located upgradient of both GW-79S and GW-50S, and that are slowly being flushed from the system as contaminants migrate from the area. As discussed above, the chloride in surface water may be the result of groundwater discharge as well as the application of road salt in nearby streets and parking lots and NPDES discharges of treated groundwater (chloride approximately 200 mg/l).

Ammonia (as ammonium ion, NH_4+) occurs at relatively low to moderate levels (range: <0.05 to 35 mg/l) in Off-Property West Ditch surface water, but then increases substantially in South Ditch locations SW-17 (up to 170 mg/l) and SW-16 (up to 97 mg/l). Concentrations then decrease significantly at downgradient location SW-15 (15 to 28 mg/l), indicating that attenuation, oxidation, or dilution of ammonia is occurring between SW-16 and SW-15. Concentrations of ammonia are relatively low in Off-Property West Ditch shallow groundwater (ranging from <1 mg/l in GW-77S to 92 mg/l in GW-76S), but then increase in the South Ditch wells, reaching a maximum of 380 mg/l in GW-79S (Figure 2-95). Ammonia decreases substantially in downgradient well GW-50S (68 to 140 mg/l). Ammonia in the Central Pond (SW-16) is much lower (2.3 mg/l to 11 mg/l) than the levels in adjacent groundwater (GW-79S), suggesting that little ammonia is discharged to the Central Pond by groundwater, it is being diluted significantly or taken up by vegetation. The absence of dissolved iron at SW-16-95 suggests the primary process is oxidation, although head relationships show that groundwater is not a significant source of recharge to the Central Pond. Nitrate concentrations in surface water exhibit a gradual increase from Off-Property West Ditch location SW-11 (0.33 mg/l) to the Central Pond (up to 11 mg/l in SW-16), then decrease to 2.8 mg/l in SW-15 (Figure 2-96). Nitrate in groundwater is highest in South Ditch wells GW-79S (up to 8.5 mg/l) and GW-50S (up to 5.7 mg/l). The nitrogen cycle and transformation of ammonia can occur by several mechanisms: by oxidation of ammonia (NH_4^+) to nitrate (NO_3^-), which occurs in the presence of O_2, or by nitrifying bacteria in a distinct two step process. In the bacterial process, NH_4^+ is first oxidized to nitrite (NO_2^-), then NO_2^- is oxidized to NO_3^- (Chapelle 1993) with subsequent bacterial mediated denitrification oxidizing organic carbon (Drever 1988):

$$2NO_3^- + 2.5C_{org} + 2\,H^+ = N_2 + 2.5CO_2 + H_2O.$$

This process consumes H^+ ions and buffers groundwater acidity. This hypothesis is supported by the presence of total organic carbon at concentrations up to 22 mg/l in well GW-79S and up to 12 mg/l at SW-16. Microbial generation of nitrogen gas (N_2) and carbon dioxide (CO_2) as the result of this denitrification reaction may explain the release of small gas bubbles that were observed emanating from the South Ditch streambed when sediment was gently disturbed. Ammonia in groundwater is strongly correlated to sulfate ($r^2 = 0.92$). The high concentrations of ammonia (NH_4^+) and correlation to sulfate are probably due to the historical disposal of ammonium sulfate ($[NH_4]_2SO_4$) in the liquid waste disposal areas (CRA 1993).

Field measurements of pH in surface water (range 5.05 to 7.35) are typically one-half to one standard unit higher than pH in groundwater (Figure 2-97). Surface water pH exhibited seasonal variations; where the lowest values were measured in February and the highest values were measured in July. Low pH in surface water is roughly correlated to periods of groundwater discharge. The overall higher pH values of surface water is probably due to the lower partial pressure of $CO_{2(g)}$ (and corresponding lower concentration of H_2CO_3), a condition that is typical of surface waters relative to groundwaters (Sposito, 1989). In general, Off-Property West Ditch location SW-12 had the overall lowest pHs, which may be due to the influence of discharging

acidic, shallow groundwater at this location, or due to the presence of organic acids from decomposing organic matter. Near neutral pHs at SW-17 may be due to the influence of near-neutral pH water discharged to the On-Property West Ditch as part of Olin's NPDES permit.

Field measurements of pH in shallow groundwater indicate slightly acidic to near neutral conditions; ranging from 4.66 (GW-42S) to 6.52 (GW-79S). A seasonal trend in pH was observed in shallow groundwater that is similar to surface water, with pH lowest in February and May, becoming more alkaline in July when surface water temporarily recharges groundwater at several locations, and decreasing again in October. The lowest pH (4.66) in shallow groundwater was measured in well GW-42S (Table 2-45). This well is paired with deep well GW-42D, and is adjacent to multilevel piezometer MP-2, both of which penetrate the dense layer. The low pH at this location may be a function of the proximity of the well to the transition layer, or to naturally-occurring organic acids derived from the decay of vegetation matter that is present in this part of the Off-Property West Ditch. The highest pH (6.52) in groundwater was measured in well GW-79S, located adjacent to the South Ditch Central Pond. The high pH at this location may be due to seasonal recharge to groundwater from the Central Pond.

Seasonal trends in field measured alkalinity in groundwater and surface water are shown on Figure 2-98. Alkalinity in shallow groundwater ranged from 0.76 mg/l (GW-77S) to 235 mg/l (GW-79S), but typically remained between 25 mg/l to 150 mg/l, with the higher measured values corresponding to more alkaline pH conditions (Table 2-46). Surface water alkalinity showed a similar range of values (Table 2-44), with the exception of one outlier at SW-16 during July 1995 (1,900 mg/l; not shown on Figure 2-98). In general, alkalinity is highest in downstream South Ditch wells GW-79S and GW-50S, which also shows elevated levels of other indicator analytes. Alkalinity in surface water is also highest in South Ditch locations. Values of field measured alkalinity are slightly higher than laboratory measured bicarbonate alkalinity, suggesting that additional acid-consuming species, such as organic anions, may be contributing to the overall alkalinity.

Eh measurements of surface water (Table 2-44) were generally higher than shallow groundwater, consistent with an environment in contact with atmospheric $O_{2(g)}$ (Baas-Becking et al. 1960). While negative Eh values were measured in four shallow wells in February, a negative Eh was recorded in only two surface water locations (SW-11 [-5 mV] and SW-14 [-110 mV]) during this period. Positive Eh values in surface water generally provide a stable environment for oxidized redox species such as ferric iron, sulfate, and nitrate. Oxidizing conditions in ditch surface water may encourage the formation of flocculent through the precipitation of amorphous ferric hydroxide. Field measurements of Eh in shallow groundwater range from a low of -150 mV in well GW-42S to a high of +440 mV in well GW-76S (Table 2-46). Eh measurements in all shallow wells indicate slightly to moderately oxidizing conditions (Figure 2-99). Eh varied widely in some wells (e.g., GW-42S and GW-77S) while other wells (e.g., GW-50S and GW-76S) exhibited consistent Eh values between sampling periods. The lowest groundwater Ehs, and the only negative Eh values recorded during the year occurred in February, indicating that mildly reducing conditions may be seasonal.

In summary, these plots show that concentrations of indicator analytes in surface water and groundwater can generally be divided into two populations; one population comprises surface water and groundwater locations in the Off-Property West Ditch which are located hydraulically upgradient of the weir; and the second population comprises those locations in the South Ditch and Central Pond, which are hydraulically downgradient of the weir. The Off-Property West Ditch locations are characterized by lower levels of specific conductivity, sodium, sulfate, chloride, ammonia, and nitrate relative to the South Ditch locations. Whether these trends are a consequence of the weir, or because the South Ditch locations are closer to the former acid pits and therefore may be receiving discharging dissolved contaminants is inconclusive, because of a lack of historic time-series data at these locations (groundwater data only extends back through 1995).

2.7.6.2.2 Chromium and Other Metals

The relationship between aluminum, chromium, and iron in ditch surface water is shown on Figure 2-100. Dissolved chromium was detected infrequently in ditch surface water during 1995 (Table 2-43) and below detection during 1996 (Table 2-47). chromium was detected once in the Central Pond (SW-16) at a concentration of 0.035 mg/l (February 1995), and once in SW-17 at 0.017 mg/l (October 1995). Total (unfiltered) chromium in SW-16 (0.042 mg/l) and SW-17 (0.016 mg/l) indicate that particulate/colloidal chromium may be present at SW-16 but not at SW-17. However, a sample from the Central Pond in 1996 contained total chromium at 0.02 mg/l and dissolved chromium at <0.015 mg/l, suggesting that particulate/colloidal-bound chromium may occasionally occur at low concentrations in the Pond. Total chromium was detected in SW-15 (May 1995) at 0.023 mg/l, however the filtered sample was non-detect (<0.015 mg/l) also indicating the presence of particulate chromium at low concentrations.

Chromium(VI) was not detected in surface water samples collected during 1995 or 1996. The absence of chromium(VI), and sporadic low-level detections of chromium(III) in surface water is consistent with Eh-pH conditions across the site. Field measurements of Eh and pH in surface water plotted on a chromium Eh-pH diagram (Davis and Olsen 1995) show that all Eh measurements plot below the chromium(VI) stability boundary (Figure 2-101), indicating that chromium(VI) is thermodynamically unstable in surface water conditions, thus explaining its absence. Most surface water samples either plot within the stability fields of the chromium solid phases $Cr_{0.25}Fe_{0.75}(OH)_{3(am)}$ and/or $Cr(OH)_{3(am)}$. These solid phases have been observed in flocculent samples examined by EMPA, establishing that these solids are probably controlling chromium solubility in the Ditch System environment.

Chromium was detected in only three of six shallow wells adjacent to the Ditch System (GW-77S, GW-78S, and GW-79S), at levels generally less than 0.050 mg/l (Table 2-44; Figure 2-101). The highest level of chromium was detected in South Ditch well GW-78S (0.22 mg/l) during October 1995. This well also had 0.25 mg/l of total (unfiltered) chromium indicating that trace levels of particulate/colloidal chromium may also be present (Table 2-48). All unfiltered samples at GW-78S had detections of total chromium (range of 0.25 mg/l to 1.9 mg/l), raising the possibility that discharge of particulate/colloidal chromium from groundwater to surface water could contribute

to flocculent formation in this part of the South Ditch. Field detections of ferric(III) iron in this well (range 1.04 to 3.1 mg/l) also increases the potential for precipitation of insoluble chromium(III) ferric hydroxide having the general formula ($Cr_xFe_{1-x}(OH)_3$). The near-neutral pHs at adjacent surface water sampling location SW-17 (range of 6.41 to 7.35), in conjunction with oxidizing Ehs in surface water could facilitate the formation of a flocculent comprised of chromium(III), iron(III), and aluminum oxyhydroxides. This hypothesis is supported by a low detection (0.017 mg/l) of chromium at SW-17, confirming that chromium flocculents are essentially insoluble in ditch surface water, even when present in adjacent groundwater.

Off-Property West Ditch well GW-77S also had detections of total chromium (0.033 to 0.054 mg/l) and dissolved chromium (0.088 to 0.22 mg/l) (Table 2-48), indicating that groundwater in this area may be an additional source of chromium to flocculent formation in the Off-Property West Ditch. Chromium was not detected in the corresponding surface water location (SW-14), confirming the insolubility of flocculents of chromium hydroxides in the surface water environment. South Ditch well GW-79S had three detections of total chromium (0.019 to 0.45 mg/l), but only two trace level detections of dissolved chromium (0.017 and 0.023 mg/l), indicating that this well is probably a minor contributor of chromium to flocculent in the South Ditch. In wells that had detections of chromium, there is a positive correlation between chromium concentration and sulfate (r2 = 0.92) for those wells with sulfate concentrations <800 mg/l. This correlation drops significantly ($r^2 = 0.007$) when all sulfate and chromium data are considered due to the high percentage of non-detections of chromium in groundwater.

While chromium was not detected in Off-Property West Ditch well GW-42S, it was detected in the nearby multilevel piezometer MP-2. Chromium was detected in ports 11-17, at a range of 0.028 to 0.24 mg/l (Section 2.7.4). These ports sampled groundwater at the same elevation in the shallow aquifer as does the screened interval in GW-42S. The presence of chromium in MP-2 could be due to the improved vertical resolution afforded by the multilevel piezometers. Disturbance of the aquifer during sampling of the multiports is minimized due to the minimal purging that is required; or it could be due to aquifer trauma, in which previously immobile chromium was mobilized during the drilling and installation of the well, in which case subsequent sampling of MP-2 should show decreases in chromium as the aquifer re-equilibrates. In either case, the absence of chromium in adjacent surface water (SW-12) indicates that any chromium that may be present in shallow groundwater does not appear to be impacting surface water quality at this location.

The absence of chromium in shallow groundwater in samples from most weir monitoring wells (Section 2.4) suggests that the chromium detected in some wells (GW-77S, GW-78S, and GW-79S) may be derived from a localized source of "old" chromium. In this context, "old" chromium refers to chromium derived from the former liquid waste disposal areas or from chromium-contaminated soil, but not derived from the present-day dense layer or intermediate TDS groundwaters. This chromium has been retained in soil or aquifer matrix and is slowly mobilized by leaching, desorption, dissolution, or ion-exchange reactions that release low

concentrations of chromium to solution after contact with infiltrating rain/snow or cleaner groundwater. This chromium reaches the ditch wells at low (ug/l) concentrations, and a few sampling events show that it may also exist in particulate form (Table 2-48).

Chromium(VI) was detected only once in ditch shallow groundwater; South Ditch well GW-78S had a detection of chromium(VI) of 0.35 mg/l in July 1995 (Table 2-45). However, the validity of his detection is doubtful given that chromium(VI) was below the detection limit (<0.015 mg/l) during the other three sampling periods, and there was no chromium(VI) detected in corresponding surface water location SW-17. Field measurements of shallow groundwater Eh and pH plotted on an Eh-pH diagram demonstrate that all groundwater points lie below the chromium(VI) stability boundary (Figure 2-101), similar to surface water. Approximately one half of the points lie within the stability field of $Cr_{0.25}Fe_{0.75}(OH)_{3(am)}$ or $Cr(OH)_{3(am)}$ solid phases. These phases were observed by EMPA in flocculent samples from the Off-Property West Ditch and South Ditch. These observations suggest that as groundwater discharges to the surface water system, the Eh-pH condition shifts in response to higher surface water pH and Eh, resulting in decreased chromium solubility and thereby precipitating insoluble hydroxides.

Iron concentrations in ditch surface water are generally below 1 mg/l for both ferrous (Fe^{+2}) and ferric (Fe^{+3}) iron (Tables 2-43 and 2-44), consistent with a surficial environment in contact with atmospheric O_2, in which Fe^{+2} is rapidly oxidized to Fe^{+3} and subsequently precipitated as amorphous ferric hydroxide [$Fe(OH)_{3(am)}$]. However, occasional detections of iron at concentrations greater than 1 mg/l have been measured in surface water (e.g., SW-11 at 4.3 mg/l, SW-12 at 4.8 mg/l, and SW-17 at 4.1 mg/l) when surface water levels are low (Figure 2-100). Iron was not detected in filtered samples from the South Ditch Central Pond (detection limit = 0.025 mg/l), but was detected at a concentration of 0.082 mg/l in an unfiltered sample (Table 2-49), indicating that iron occurs in particulate/colloidal form in the pond.

Iron concentrations in shallow groundwater spanned a wide range of concentrations during 1995 (Tables 2-45 and 2-46). One Off-Property West Ditch well (GW-42S) and two South Ditch wells (GW-78S and GW-79S) all had iron concentrations much greater than 1 mg/l, whereas wells GW-76S, GW-77S, and GW-50S had iron concentrations substantially less than 1 mg/l. Elevated iron in GW-78S (up to 9.2 mg/l) and in GW-79S (up to 39 mg/l) (Figure 2-102) is consistent with the presence of elevated levels of indicator analytes at these locations (see previous section). The absence of iron in the Central Pond (SW-16) is probably due to the ambient oxidizing conditions (i.e., high Eh and dissolved oxygen) measured there, which facilitates the precipitation of iron that may be derived from groundwater near GW-79S (Table 2-44). Colloidal iron concentrations in the Central Pond support this scenario. Iron in Off-Property West Ditch well GW-42S is elevated relative to adjacent wells GW-76S and GW-77S, suggesting that increased iron levels (primarily ferrous iron) may be due to seasonally low pHs and the acid solubilization of iron-rich sediments that characterize this section of the Off-Property West Ditch.

Aluminum was rarely detected in ditch surface water (Table 2-43), due to the near-neutral pH conditions under which aluminum hydroxides are insoluble. The highest aluminum detected (2.4 mg/l) occurred at SW-16 (Central Pond). An unfiltered sample at this location contained

aluminum at 3.1 mg/l, indicating aluminum is also present in particulate/colloidal form. Aluminum was also detected in SW-11 (0.23 mg/l); SW-12 (0.22 and 0.31 mg/l); SW-14 (0.11 mg/l); and SW-17 (0.14 mg/l) (Figure 2-102). Aluminum concentrations in 1996 South Ditch surface water samples (Table 2-47) remained relatively constant (range: 0.24 to 0.26 mg/l) compared to the 1995 data, and may represent ambient background levels of aluminum in surface water.

Aluminum was detected at significantly higher levels in shallow groundwater samples than surface water samples at concentrations ranging from 0.044 mg/l (GW-50S) to 1.4 mg/l (GW-77S). Groundwater pH is more acidic than surface water at these locations, possibly solubilizing aluminum hydroxide. Aluminum was also detected in Off-Property West Ditch wells GW-42S and GW-77S, and in South Ditch well GW-78S (Table 2-47).

Low (ug/l) concentrations of dissolved aluminum, chromium, and iron in ditch surface water samples indicate that the majority of these metals may be sequestered in oxyhydroxide solids (i.e., flocculent formation) that scavenge chromium and other trace metals discharging from shallow groundwater to the Ditch System. Other metals such as antimony, arsenic, barium, beryllium, cadmium, copper, lead, manganese, and zinc in ditch surface water and Central Pond, were below detection or present at very low concentrations, indicating that natural attenuation mechanisms of precipitation and sorption effectively remove metals from solution, thereby decreasing their mobilities.

2.7.6.2.3 Influence of Weir on Ditch Analyte Concentrations

The weir was installed in July 1994 to address surface water transport of chromium-bearing sediments and also to suppress shallow groundwater discharge leading to the formation of chromium-bearing flocculent. The effectiveness of the weir on ditch water quality is difficult to assess because little surface water/groundwater data is available prior to mid-1994 when the weir was installed. However, one well (Off-Property West Ditch well GW-42S) has data extending back to August 1991. Plots of pH, SO_4, Cr, Al, and Fe concentrations from this well were examined to assess the influence of the weir on flocculent-related analyte levels at this location.

Time series plots of chromium and sulfate show large decreases in concentrations prior to the installation of the weir, indicating that these parameters may have been decreasing before the weir was installed (Figure 2-103). A similar plot of aluminum, iron, and pH show little change in concentration in aluminum over time, while pH appears to gradually decrease over time (Figure 2-104). In contrast, iron concentrations increase after installation of the weir. These data suggest that the influence of the weir on flocculent-related analyte concentrations at GW-42S are minimal or non-existent, and that the weir probably has had little effect on flocculent production in Off-Property West Ditch wells and even less influence on wells that are located in the South Ditch area.

2.7.6.2.4 Flocculent Chemistry and Morphology

Flocculent has been observed in both the Off-Property West Ditch and South Ditch streambeds. The nature and chemistry of the flocculent were investigated in May 1995 and April 1996 when samples were collected and submitted for both laboratory and EMPA. Laboratory chemical analyses of samples collected in 1995 (termed "White Floc", "Red Precipitate", and "SED-17,11") are presented in Table 2-50. Laboratory analysis of total chromium in floc samples collected in April 1996 (Floc 1 to Floc 5) are presented in Table 2-51. It should be noted that the concentration of aluminum, chromium, and iron detected in flocculent samples are generally higher than detected in the CSA (CRA, 1993) sediment samples. These higher concentrations reflect the biased nature of sampling procedures which collected only material from a thin layer of flocculent resting on the bed of the ditches.

EMPA results show that several chromium-bearing hydroxide and sulfate solid phases were detected in the flocculent. These phases consist of several complex amorphous hydroxide mixtures containing aluminum, chromium, and iron in sample WF-1 ("white floc"), and a complex Fe-Cr-Al-sulfate phase with a composition similar to the soil mineral redingtonite in sample RP-1 ("red precipitate") (Figure 2-105). Solid phase mixtures of these metals typically occur because chromium(III) readily substitutes for aluminum and iron(III) due to the similar atomic radii of the three elements (Al = 1.43 D, Cr = 1.25 D, and Fe = 1.26 D) and identical charge (Bartlett and James, 1988). In addition, chromium(III) can form solid solutions with amorphous ferric hydroxide (Sass and Rai, 1987), exhibiting a typical stoichiometry of $Cr_{0.25}Fe_{0.75}(OH)_3$ (Eary and Rai, 1988).

Sample WF-1 exhibits many needle-like intergrowths of aluminum-, chromium-, and iron-hydroxides within a silicate matrix. This sample contains abundant aluminum hydroxide [$Al(OH)_3$], in which chromium(III) was present between 0.8 percent to 2 percent by weight. The morphology of RP-1 shows an amorphous mixture of aluminum and iron hydroxides and sulfate phases rimmed by, and intergrown with, the redingtonite. Chromium concentrations increase toward the edges of the flocculent, where it appears as amorphous coatings on the exposed surfaces, indicative of secondary precipitation or adsorption reactions that scavenge chromium from solution. Sample Floc-6 from the South Ditch consisted of several mixed hydroxides of aluminum, chromium(III), and iron (Figure 2-106). There is also an Fe-Cr sulfate phase in Floc-6 that contains small particles of a Pb-Cr oxide. The lead in the Pb-Cr oxide phase is probably scavenged from naturally-occurring lead minerals or impurities in the sediments, because lead was not a waste product at the site, nor have there been any detections of lead in surface water or groundwater at the site (detection limit = 0.005 mg/l).

The presence of these hydroxide and sulfate phases is consistent with earlier hypothesized surface water/groundwater geochemical interactions (BCM 1993), specifically the reaction of dissolved aluminum, iron(III) and chromium(III) to form a series of amorphous solid precipitates that co-precipitate or sorb other metals. The existence of these precipitates in the flocculent samples indicates that these phases are thermodynamically stable under the ambient neutral, oxidizing conditions of the ditch environment.

Laboratory analysis of the White Floc (WF-1), Red Precipitate (RP-1), and SED-17,11 samples clearly establish the wide range of metals that may be adsorbed or coprecipitated from the water column during flocculent formation (Table 2-51). Chromium concentrations up to 10 percent by weight in particles were detected by EMPA in the Red Precipitate sample. This sample also had the highest total chromium content (35,000 mg/l) and highest iron content (180,000 mg/l). Microprobe analysis also detected other metals, i.e., calcium, potassium, and silica at lower concentrations. It is likely that several of these elements (such as silicon and potassium) are probably artifacts of the sampling process in which some streambed sediments (clays) were attached to the flocculent. Additional metals detected in WF-1 include many metal species known to occur in the dense layer, including Ba (4600 mg/kg), Ca (2700 mg/kg), Cu (25 mg/kg), Mg (380 mg/kg), Mn (46 mg/kg), Na (8700 mg/kg), V (11 mg/kg), and Zn (3400 mg/kg). Most of these metals were also detected in the Red Precipitate, and to a lesser extent in SED-17,11. The high concentration of chromium (35,000 mg/kg) in the Red Precipitate is probably due to the very high level of iron (180,000 mg/kg) which promotes the formation of a $Cr_xFe_{1-x}(OH)_3$ solid solution, which is less soluble than the pure chromium(III) hydroxide (Sass and Rai, 1987). Chloride (210 mg/kg) and sulfate (740 mg/kg) were also detected in the Red Precipitate.

In general, chromium concentrations in flocculent decrease with distance in the downstream direction along South Ditch, indicating that the source of chromium to the ditch is upstream of the Pond (Figure 2-107). This interpretation is consistent with the regular detections of chromium in South Ditch well GW-78S, indicating that groundwater in the vicinity of this well is a source of chromium to the flocculent in the South Ditch. Departures from this trend are seen in floc sample 'Floc-3" (18,000 mg/kg), which was collected immediately downstream of the Central Pond, and to a lesser extent Floc-5 (5,200 mg/kg) which was collected at the east end of South Ditch (Table 2-51).

2.7.6.2.5 Geochemical Modeling of Flocculent Formation

Geochemical modeling using the code PHREEQC (Parkhurst, 1995) was performed to determine if mixing of ditch surface water and groundwater would result in the precipitation of solids with a composition similar to that of the flocculent. To simulate this phenomena, the model was set-up to allow all oversaturated solids to precipitate from solution following mixing of groundwater and surface water. Adsorption of barium, calcium, manganese, and sulfate onto ferrihydrite [$Fe(OH)_3$] was then modeled to simulate the geochemical scavenging effects of ferrihydrite (Dzombak and Morel 1990). Adsorption was limited to these species because of the availability of established adsorption parameters in the PHREEQC data base, and the regularity of detections of these species during 1995. The mass of each precipitated phase, as well as the mass of adsorbed metals, were summed to determine the mass of flocculent that could theoretically be produced.

Surface area, adsorption density, and other input values for ferrihydrite were taken from Dzombak and Morel (1990) or from the default values in PHREEQC. The mass of ferrihydrite in the flocculent was calculated by PHREEQC after allowing ferrihydrite to precipitate from solution

following mixing with surface water. This mass (in grams) was then used as input to the adsorption routine to calculate surface complexation. System Eh and redox equilibria were determined using field measured ratios of ferrous to ferric iron and laboratory values of total iron; if field iron speciation data were absent, then the field measured Eh was used. Field measurements of temperature, pH, and alkalinity and laboratory results for metals and anions were also used as input. Input data used the field and laboratory chemical data are presented in Tables 2-43 through 2-46.

Because the flocculent contains high concentrations of chromium and iron, only those wells that had consistent detections of chromium (GW-77S and GW-78S) and/or iron (GW-42S) were modeled. Modeling was also confined to those periods during which groundwater was known to discharge to surface water. For the GW-77S/SW-14 and GW-78S/SW-17 locations, data from October 1995 were used; for the GW-42S/SW-12 location, May 1995 data were used because hydraulic heads at most SW/GW locations in the Ditch System clearly indicate conditions of groundwater discharge during most of 1995 and 1996, with only short-lived seasonal deviations. When hydraulic head data for 1996 are compared to the flow rate data obtained during the flume study of South Ditch conducted in 1996, it is apparent that groundwater discharge (baseflow) probably accounts for a majority of flow in the ditch except during periods of significant surficial runoff (e.g., snowmelt or rain). Based on this relationship, a conservative ratio of 1:1 surface water to groundwater was chosen for the mixing routine.

Results of PHREEQC mixing simulations are shown on Table 2-52. Mixing of groundwater from well GW-42S and surface water SW-12 (Off-Property West Ditch) resulted in the precipitation of approximately 7 milligrams of flocculent per liter of mixed fluid, composed of ferrihydrite [$Fe(OH)_3$] and basaluminite [$Al_4(OH)_{10}SO_4$]. GW-42S was supersaturated with respect to ferrihydrite (SI = 1.07) and basaluminite (SI = 0.14); surface water was oversaturated with respect to ferrihydrite (SI = 1.69) only. Sulfate was the only phase adsorbed in any quantity (1.9 mg), although small amounts of barium (3×10^{-5} mg), calcium (7×10^{-3} mg), and manganese (1×10^{-3} mg) were also adsorbed to the ferrihydrite. The total mass of flocculent (7 mg) produced was calculated by summing all precipitated and adsorbed masses. Chromium was absent in both groundwater and surface water at these locations, therefore the chromium that was detected in flocculent sample RP-1 in this part of the ditch may have been scavenged from old chromium present in the sediments.

Mixing of groundwater from GW-77S and SW-14 (Off-Property West Ditch) resulted in the precipitation of approximately 1.4 mg/l of flocculent, composed of a mixture of ferrihydrite and gibbsite [$Al(OH)_3$]. GW-78S was oversaturated with respect to ferrihydrite (SI = 0.53), gibbsite (SI = 1.2), and basaluminite (SI = 3.9); SW-14 was oversaturated with respect to ferrihydrite (SI = 1.5) only. Sulfate was the dominant adsorbed phase (0.4 mg); followed by calcium (0.13 mg) and trace amounts of adsorbed barium (8×10^{-5} mg) and manganese (2×10^{-2} mg). Low concentrations of chromium are present in groundwater at GW-77S, however, groundwater and the resulting mixture of groundwater and surface water are undersaturated with respect to chromium hydroxide, suggesting that chromium in Off-Property West Ditch flocculent is incorporated into the flocculent by coprecipitation or adsorption processes.

Mixing of groundwater from GW-78S and SW-17 resulted in the greatest amount of flocculent produced (11.8 mg/l), consisting of 0.2 milligram of amorphous chromium hydroxide [$Cr(OH)_3$], 9.5 milligrams of ferrihydrite, 2 milligrams of basaluminite, and 0.03 milligram of barite ($BaSO_4$). GW-78S was oversaturated with respect to ferrihydrite (SI = 1.3), gibbsite (SI = 1.6), and basaluminite (SI = 6.9). SW-17 was oversaturated with respect to $Cr(OH)_3$ (SI = 0.7), ferrihydrite (SI = 2.4), gibbsite (SI = 0.9), basaluminite (SI = 3.8), and barite (SI = 0.86). Sulfate was the major adsorbed species (6.3 mg), with trace amounts of adsorbed calcium (8×10^{-2} mg), barium (1×10^{-6} mg), and manganese (7×10^{-4} mg) also present. The oversaturation and precipitation of amorphous chromium hydroxide is consistent with the presence of $Cr(OH)_3$ mixtures observed in South Ditch flocculent samples WF-1 and Floc-6.

By summing all of the flocculent produced at these three locations (approximately 20 mg/l of mixed fluid) and normalizing it to the total flow measured in the South Ditch, a hypothetical annual mass of flocculent can be estimated. Flow data obtained during 1996 from the South Ditch shows wide variations in total flow, ranging from flow too low to measure, to a maximum flow of 1,030 gpm (Smith 1996). Assuming an average yearly flow rate of 10 gpm (38 liters per minute) or 2×10^7 liters per year, results in approximately 4×10^8 milligrams or 880 pounds of flocculent per year. For a total stream Ditch System reach of 2000 feet and an average ditch streambed of 5 feet wide results in a total ditch streambed area of approximately 10,000 feet^2. This would result in a flocculent density of 0.09 pounds (41 grams) of flocculent per feet^2 of streambed.

These calculations demonstrate that the current levels of metals in groundwater and surface water, when mixed in the right proportions, can produce a substantial quantity of flocculent over the course of a year. There may be periods when no flocculent is produced (during periods of surface water recharge to groundwater, or during periods of high surface water flow when dilution is highest) and periods when a lot of flocculent may be produced (dry months when groundwater discharge contributes the most to streamflow). However, the process of flocculation in the Ditch System acts to keep dissolved metal concentrations in surface water (especially aluminum, chromium, iron, and sulfate) to a minimum by entraining them in the flocculent matrix.

2.7.6.3 Conceptual Model of Flocculent Formation

In this investigation, a conceptual model (Figure 2-108) of the relationship between former source areas (i.e., unlined pits and lagoons) at the Olin site and the Ditch System (represented schematically by the South Ditch stream) was constructed to help portray the transport of chromium in groundwater, and the potential sources of metals that result in the formation of flocculent in this portion of the Ditch System and similar areas.

As shown on the conceptual model, (1) dense, acidic, high ionic-strength liquid waste containing chromium(III) sulfate ($Cr_2[SO_4]_3$) and other analytes (Ca^{+2}, Na^+, Cl^-, NH_4^+, SO_4^{-2}) at high concentrations (CRA 1993), migrated from the liquid waste disposal area to the base of the aquifer where it accumulated as the dense layer (2). Groundwater flowing through the area of

vertical waste migration subsequently transported dissolved constituents by advection toward the South Ditch forming a dissolved plume (3). The low pH of the liquid waste facilitated reaction with magnesium- and iron-bearing oxide and silicate minerals (e.g., hematite, biotite, and illite) in the aquifer matrix, released these metals to solution. Eventually this dissolved plume with high TDS and TSS concentrations began to discharge to the South Ditch in response to vertical hydraulic gradients (4), forming a geochemical reaction zone at the interface between the acidic groundwater and near-neutral surface water (5a), and leaving residual soil contamination in the vicinity of the unlined pits and in places in the aquifer that accumulated inorganic consistuents and their reaction products. In these zones (5a and 5b), it is hypothesized that some of the aluminum, chromium, and iron precipitated out and accumulated in response to the higher pH and Eh and hydrogeologic conditions. Discharge and subsequent mixing of the metal-bearing groundwater with surface water, followed by the oxidation of ferrous iron to ferric iron and complexation with chromium(III) and aluminum resulted in continuous flocculent formation (6).

After the liquid waste disposal ceased, continued groundwater flow flushed the residual dissolved plume and possibly colloidal solids from the upper aquifer, leaving behind the dense layer (7). A portion of the dissolved metals were retained in the aquifer matrix as a result of sorption, precipitation, and ion exchange reactions, while the more mobile solutes (i.e., Na^+, NH_4^+, Cl^-, SO_4^{-2}) continued to migrate toward the South Ditch. Continual flushing of the aquifer matrix by cleaner groundwater (8) is hypothesized to result in the slow release of chromium and other metals from the aquifer matrix to groundwater (9) from the geochemical reaction zones and areas of residual soil contamination upgradient of the South Ditch surface water (10). Slow dissolution, desorption, and ion-exchange of aluminum, chromium, iron, and manganese from the reaction zone may continue to result in limited flocculent formation (11), however, eventual source depletion in the reaction zone should result in cessation of flocculent production as the quantities of aluminum, chromium, and iron in groundwater and surface water decrease to ambient background levels.

2.7.6.4 Summary of Surface Water and Flocculent Geochemistry

Elevated concentrations of several indicator analytes (specific conductivity, sulfate, chloride, ammonia, and nitrate) are present in South Ditch groundwater relative to the Off-Property West Ditch groundwater. Several of these analytes also occur at elevated concentrations in South Ditch surface water, raising the possibility that localized aquifer flushing of dissolved constituents from the site may be discharging to portions of the South Ditch. Attenuation of these analytes in the South Ditch is evident by lower concentrations at the downstream sample location (SW-15) relative to upstream locations (SW-16 and SW-17). In contrast to these indicator analytes, consistent detections of chromium in ditch groundwater are limited to one Off-Property West Ditch well (GW-77S), and one South Ditch well (GW-78S). A second South Ditch well, (GW-79S) has had only sporadic and low-level detections of chromium. Because chromium is virtually absent in ditch surface water samples, it can be concluded that dissolved chromium is not stable in the surface water and there is no current impact from dissolved chromium on ditch water

quality. The absence of chromium in surface water during 1995 and 1996 sampling, relative to the previous detections of chromium identified during the CSA (CRA 1993), may also indicate that the source of chromium to surface water may be undergoing depletion.

Geochemical modeling demonstrates that the formation of chromium-bearing flocculent in the ditch streambed can be simulated by the mixing of shallow ditch groundwater and surface water. The wells used in the modeling (GW-42S, GW-77S, and GW-78S) have sufficient dissolved aluminum, chromium, and/or iron to create a precipitate that is representative of the ditch flocculent composition. Electron microprobe analysis of flocculent samples demonstrate that chromium is bound (by coprecipitation or adsorption) to Fe- and Al- solid phases either as amorphous hydroxides or precipitate as a mineral form (e.g., redingtonite). Amorphous solid phases of Al-, Cr-, and Fe-sulfate in aquifer samples and flocculent samples indicate that sulfate is actively being sequestered from solution during formation of these phases. In surface water, these phases restrict chromium solubility to a level below laboratory detection limits, effectively eliminating dissolved chromium from the South Ditch System.

The apparent continued accumulation of flocculent in the Off-Property West Ditch streambed following flocculent removal in 1994, suggests that a source of chromium in the shallow groundwater may still be present although sporadic flushing of residual colloidal material near the streambed, scouring and redeposition of "old" flocculent, and the occurrence of non-chromium-bearing precipitates could also partly explain the reappearance of flocculent.

This source is hypothesized to result from the slow dissolution/desorption/ion-exchange from a (1) a geochemical reaction zone at the interface between the acidic groundwater and near-neutral surface water and (2) a residual soil contamination in the vicinity of the unlined pits and in places in the aquifer which accumulated inorganic consistuents and their reaction products (Figure 2-108). Potential additional sources of chromium to groundwater in the vicinity of the South Ditch may originate from the leaching of chromium-contaminated soil in the vicinity of the Central Pond. However, near-neutral pH conditions in ditch surface water (where chromium solubility is lowest) indicates that the geochemical environment in ditch surface water strongly favors chromium immobilization. The downstream transport of flocculent and particulate chromium is restricted due to the Weir IRA and the installation of hay bales as sediment transport barriers.

In summary, chromium migration from the Wilmington Facility to the South Ditch System is being controlled by a combination of flocculation, sediment barriers (hay bales), ambient site geochemical conditions, and probable source depletion. Chromium is present solely in the trivalent form, as particulate or solid phases of extremely low solubility. Migration of other analytes of concern to the Ditch System, such as Na, Cl, NH_4, and SO_4, appears to be limited to areas near GW-79S and the Central Pond. The concentration of these analytes is generally lower in surface water locations relative to corresponding groundwater locations, suggesting that natural attenuation and dilution is controlling solute concentrations discharging to the Ditch System.

3.0 PHYSICAL CHARACTERISTICS

MCP 40.0835(4)(d)

CHAPTER SUMMARY

The physical characteristics of the property and surrounding area affect the mobility, potential migration, and potential impact of the contamination. This section provides a description of the physical features in the vicinity of the Olin property that are relevant to understanding the contamination. The key findings of this section are:

- Surrounding land uses, regional surface water hydrology, site habitat characterization, and climate were all adequately defined in the Comprehensive Site Assessment (CSA), with minimal additional information added.

- Peat and organic muck have been identified in the Maple Meadow Brook wetland area with thicknesses to 33 feet.

- The overburden geology, deposited during glaciation, from the surface down to the bedrock, is comprised of 1) peat and organic muck (if present), 2) very fine to fine, clean sand, 3) less sorted coarse sands, gravels and cobbles with thick lenses of fine sand or silt, and 4) till at the base of the overburden material. The till is a poorly sorted, silty sand and gravel and can have (beneath Maple Meadow Brook) a dense gray-green to olive green clay matrix.

- The bedrock surface is irregular with a maximum relief of 120 feet. Less fractured, harder bedrock comprises the hills of the area. More highly weathered, fractured bedrock within fault zones has been more heavily eroded and is therefore not exposed and comprises the bedrock valleys. An example of an area underlain by weathered bedrock is the Maple Meadow Brook wetland and associated drainage system. Fractures, when present in the bedrock, are not significant water bearing zones (less than 1 gallon per minute [gpm] capacities).

- The dense layer present over 100 feet beneath Maple Meadow Brook wetlands at well GW-83D is contained within a closed bedrock depression.

- There is no connection between the Western Bedrock Valley and the slight bedrock low previously identified as the Southwestern Bedrock Valley.

- Groundwater from the Olin property flows to both the east and west. East of a groundwater divide, flow is to the east towards the South Ditch and East Ditch. West of the divide, flow is westward toward Altron, Maple Meadow Brook, and the surrounding wetlands.

- The sand and gravel deposits are heterogeneous with a high average permeability; they form a productive aquifer with the ability to transmit large quantities of water. In contrast, the underlying bedrock has a very low permeability and little storage capacity. It forms an insignificant part of the overall flow system.

- Movement of the dense layer is primarily controlled by the shape of the bedrock rather than by hydraulic heads and gradients. Liquid waste disposed into unlined impoundments on the property has more dense than natural groundwater and sank until it reached a less permeable barrier, the top of the bedrock. Once the dense liquid reached the bedrock, it continued to flow down slope along the top of the bedrock under the influence of gravity. Therefore, the direction of movement of this dense layer can be determined by examining the shape of the bedrock surface (Plate 2-1). Beneath the impoundments where the liquid was disposed (Lake Poly Liquid Waste Disposal Area [Lake Poly] and East and West Pits), the bedrock generally slopes to the west and southwest. Therefore, the dense material moved along the top of the bedrock towards the west and southwest. This process continued until the dense layer reached the bottom of the Western Bedrock Valley that is located beneath the center of the property and extends westward beneath Jewel Drive. It then continued to move down slope in the bottom of the Western Bedrock Valley toward the west, leading to the current location of the dense layer. A small amount of the dense material also moved from the center of the property towards the east within the Eastern Bedrock Valley by the same mechanism.

- Groundwater in this area is pumped from both the sand and gravel aquifer and the bedrock formation. The bedrock is a limited groundwater resource, but provides a domestic water supply to several residences. The sand and gravel aquifer was historically tapped by domestic water wells, but all these wells have been abandoned or are unused.

- The Town of Wilmington operates a municipal water supply wellfield surrounding Maple Meadow Brook. This wellfield consists of five wells (Butters Row No. 1, Butters Row No. 2, Chestnut Street No. 1, Chestnut Street No. 2, and Town Park) which pump an average of 1.4 million gallons per day. The wellhead protection area (DEP-Approved Zone II) surrounding these wells extends to Main Street and Eames Street, and crosses a small portion of the Olin property.

- The Altron facility located on Jewel Drive has two industrial supply wells used to provide process water. The most recent information from Altron indicates that these wells typically pump at a combined rate of approximately 180,000 gallons per day for 6 days a week.

- The interaction between groundwater and surface water along the Ditch System is complex and varies seasonally. Certain water bodies, such as the Central Wetland and the Ephemeral Drainage, are likely to receive groundwater discharge during wet periods; during dry periods, these features are dry. The On-Property West Ditch may provide recharge to groundwater throughout the year due to the water discharged at the National Pollutant Discharge Elimination System (NPDES)-permitted outfall on this ditch. The East Ditch appears to receive groundwater discharge throughout most of the year, as does the downstream portion of the South Ditch. There appears to be limited groundwater interaction with the upstream portion of the South Ditch or the Central Pond, due perhaps to low permeability of the underlying geologic materials or the streambed sediments. The Off-Property West Ditch appears to receive groundwater discharge during wet periods, but to provide recharge to groundwater during dry periods due to the action of the weir.

The information outlined above is integrated and presented in the sub-sections of this section as outlined below:

- Certain physical features were described in detail in the CSA. These previous characterizations are discussed in Section 3.1, including surrounding land uses, regional surface water hydrology, site habitat characterization, and climate.

- The geology of overburden materials is discussed in more detail in Section 3.2, with new information from the Supplemental Phase II Investigation presented.

- Regional and study area bedrock geology, including the bedrock topography is discussed in Section 3.3.

- A discussion expanding the CSA presentation of hydrogeology is presented in Section 3.4, including horizontal and vertical groundwater flow, flow of the dense layer, and groundwater management and usages.

The hydrology of the ditch system is discussed in Section 3.5.

3.1 PREVIOUS CHARACTERIZATIONS

Specific physical features that were adequately characterized in the CSA or other previous reports and for which little or no additional data have been collected include the following:

- Surrounding land use (Section 2.1 of the CSA)
- Regional surface water hydrology (Section 3.4 of the CSA)
- Local groundwater usage (Section 3.5 of the CSA; Section 7.0 of the Interim Update)
- Site Habitat Characterization (Site Habitat Characterization Report, March 1993)
- Climate (Section 3.6 of the CSA)
- Geology of the overburden materials (Section 5.3 of the CSA)

The CSA also discussed the topography of the area surrounding the property. The previous detailed topographic map (Plan 2 of the CSA) has been extended to cover a wider area. The extended topographic map is shown on Plate 1-1 of this report.

3.1.1 Surrounding Land Uses

The property is directly bounded on the east by Boston and Maine railroad tracks, on the south by the Woburn/Wilmington town line, on the west by a Boston and Maine railroad spur and on the north by Eames Street. The entire property is enclosed by an 8-foot high perimeter chain link fence, and access is restricted to three gates, which remain locked when the property is unattended.

Just beyond the property lines, the property is surrounded by light to heavy industry to the east, north, and west as shown on Plate 1-1. To the immediate south of the property is the old Woburn Town Dump. The area around the property becomes more residential further to the west, beyond the area of the industrial development. Areas along Main Street, Cook Avenue, and Border Avenue consist primarily of single family homes with some commercial and light industrial development.

As indicated above, much of the land surrounding the property has been developed for industry. Several commercial properties near the Olin property have reported independent environmental problems to the Massachusetts Department of Environmental Protection (MADEP). The MADEP reported trichloroethene (TCE) at Altron Industries to the west of the property along Jewel Drive in 1982. TCE concentrations have ranged from 12 to 50 parts per million (ppm) at one of the two Altron supply wells. In addition, east of the property, localized high levels of groundwater contamination have been observed in GW-80BR that unrelated to contamination

found in adjacent portions of the Olin property and are attributable to an off-site source. Other MADEP sites with known contamination or releases surrounding the property included Koch Membrane, United Tool and Die, Raffi and Swanson, Pepsi Cola, Jewel Drive Industrial Mall, the Getty Station, and the Rutland Corporation. Since the Olin property is bound in all directions by industrial properties and the City of Woburn Landfill, the potential sources of groundwater contamination are considered in discussions of groundwater analytical data in Section 2.0.

3.1.2 Surface Water Hydrology

The study area is situated at the headwaters of two watersheds. Most surface water from the Olin property drains through the on-property ditch system to the East Ditch, flows south to Halls Brook, and finally into the Aberjona River (Plate 1-1). Surface water west of the property drains into the Maple Meadow Brook wetlands, which eventually flows into the Ipswich River.

Approximately two thirds of the Olin property is situated in an area designated as "Zone C" by the Federal Emergency Management Agency (FEMA) under the National Flood Insurance Program (NFIP). "Zone C" is classified as an area outside both the 100 year and 500 year flood boundaries, which experiences minimal flooding. The remaining one-third of the Olin property, primarily the south-central portion of the property, is situated in an area designated as "Zone B" by FEMA under the NPIF. "Zone B" areas are those which lie between the 100 and 500 year floodplain or are areas subject to a 100 year flood with an average depth less than one foot or where the contributing drainage area is less than one square mile. Further flooding information is documented in the CSA, including a figure which depicts flood boundaries for the site and surrounding area (CRA, 1993).

Surface water features within the Olin property boundary include an extensive ditch system, a pond, and extensive wetlands in the southeastern section of the property. Additionally, surface water features outside the property include a ditch system and Maple Meadow Brook which runs through a significant wetland.

In general, the surface water in the ditch system originates primarily from groundwater discharge. The South Ditch primarily receives groundwater discharge all year while the Off-Property West Ditch shows seasonal recharge/discharge conditions. The Central Pond does not appear to have significant interaction with the groundwater system. The wetland areas are fed primarily by groundwater discharge and secondarily by surface water run-off. Surface water flow fluctuates in response to seasonal changes in evapotranspiration rates, precipitation, and groundwater conditions. In general, during the growing season, water levels and flow are relatively low due to higher rates of evapotranspiration and groundwater withdrawals.

3.1.3 Habitat Characterization

Wetlands Preservation performed a habitat characterization of the Olin property and some of the adjacent ditch systems (WPI, 1993). In general, the property was classified into uplands and wetlands areas. Within these areas, the habitat was classified based on vegetative cover, hydrology, soils, wildlife use, and other pertinent characteristics.

The upland area was broken into four different areas, primarily based on their land use. Three of the four areas were heavily maintained areas and include the industrial part of the property, the former lagoon area, and the Calcium-Sulfate Landfill. The fourth upland area has a mature forest which appears to have been left undisturbed for the past 60 years. These areas are depicted on Figure 3-1.

The southern section of the property is primarily wetlands and has been broken into five different areas, based primarily on the hydrology. The five areas represent the Off-Property and On-Property West Ditch system, the South Ditch system, the Ephemeral Drainage, the East Ditch, and an isolated wetland on the southwestern corner of the plant. In general, most areas have disturbed soils due to industrial development (Off-Property West Ditch) and channel improvement (On-Property West Ditch and South Ditch).

The manmade Central Pond is heavily disturbed and surrounded by soil berms and piles, but presently the area surrounding the pond is vegetated with upland trees, shrubs, and herbaceous species. The On-Property West Ditch and the South Ditch represent natural drainage areas which were modified to improve drainage during development of the property in about 1953 through 1954. The area around the South Ditch consists of upland trees, shrubs, and herbaceous species. It is currently in an early successional stage of revegetation. The East Ditch area lacks vegetation due to the high disturbance primarily associated with the adjacent railroad easement. The ditch is subject to cutting and spraying during maintenance of the railroad easement and the bank consists primarily of crushed stone which inhibits plant growth. The soils in the Off-Property West Ditch system are heavily disturbed by past excavation, regrading, and filling associated with the industrial park along Jewel Drive. This portion of the ditch system receives water from roof, parking lot, and surface run-off, in addition to groundwater discharge.

In general, the remaining wetlands on the property are vegetated with trees, scrub-shrub, or herbaceous plants. A majority of the wetland adjacent to the mature forest is in a natural condition.

3.1.4 Climate

The climate in this region is humid and temperate with fairly uniform monthly precipitation. As stated in the CSA, the average annual precipitation recorded at Reading, Massachusetts is 40.9 inches with October typically being the driest month and April being the wettest. Seventy-five percent of the precipitation occurs in the frost free season. The mean January temperature is

26.6° F (Fahrenheit) and the mean July temperature is 73.7° F with a mean annual temperature of 49.8° F. The prevailing wind direction is from the west-northwest during the winter months and from the southwest during the summer months with an average windspeed of 7 knots (CRA, 1993).

Daily precipitation data recorded at the Butters Row Treatment Plant were used to gauge precipitation within the study area. Monthly precipitation accumulations ranged from approximately 0.63 to over 6 inches and are summarized in Table 3-1.

3.2 OVERBURDEN GEOLOGY

In general, four types of overburden materials exist at various locations to varying degrees within the study area. Overburden materials include the following:

- Peat and organic muck
- Sands and silts
- Sand, gravel, cobbles and boulders
- Clayey to silty sand and gravel

At some locations within the study area, a layer of peat or organic muck occurred just below the ground surface. In general, the organic layer composition varied. A very dark brown, organic peat was encountered at the surface. The peat graded downward from dark brown, dense, highly fibrous and rich in organics to olive gray-green, plastic, organic silty muck. Peat and muck thickness varied throughout the site depending upon surface water features, and the majority of the peat/muck encountered occurred in the Maple Meadow Brook wetlands. Muck thickness increased towards the center of the wetlands and near Maple Meadow Brook and its tributaries within the wetland and ranged from one to 33 feet thick, in the Maple Meadow Brook wetlands. Sandy humus existed at locations where muck was not encountered.

Below the muck or surface soil, the overburden materials consisted of layers of a fine, clean sand and a mixture of sand, gravel, and cobbles. In general, a relatively thick and homogeneous layer of very fine to fine, clean, well sorted sand was encountered near the surface. The color of the sand ranged from yellow-orange, light brown, tan, and gray. This layer represents outwash materials deposited when glacial meltwaters drained into the valley during retreat of the glaciers. This layer is also characterized by well-sorted coarse-grained sand with little, rounded, pea-gravel.

Typically below the sand, the aquifer materials became less well-sorted and consisted of a mixture of light brown, sorted to poorly sorted, coarse sand, gravel, and cobbles. The gravel and cobbles were emplaced as high volumes of glacial meltwater deposited large loads of sediment at the immediate margin of the glacier. These deposits of gravel and cobbles (ice contact deposits) were transported as evidenced by the subrounded-subangular, mixed origin clasts. Thick lenses of fine sand or silt, ranging from less than one foot to several feet thick can exist within the ice contact deposits.

Till exists at the base of the overburden materials at various isolated locations throughout the study area. The till in areas on or near the Olin property are described as gray, poorly sorted, silty sand and gravel and are not always as compact as tills observed in the Maple Meadow Brook wetlands. The till observed in the borings completed in the Maple Meadow Brook wetlands is characterized by gravel in a dense, gray-green to olive green, clay matrix with little fine sand which occurred at various locations within the study area. The color change and the presence of fines are distinct characteristics of the basal till observed beneath the Maple Meadow Brook wetlands. The green color of the till is likely due to chlorite, epidote, and possibly iron minerals in a weathered regolith that was scoured and mixed with till during glaciation.

3.3 BEDROCK GEOLOGY

3.3.1 Regional Geologic Setting and History

The study area shown on Figure 2-2 lies within the Seaboard Lowland of the New England Physiographic Province. This area is characterized by a peneplaned upland bedrock surface which is partially covered by glacial (Pleistocene) and more recent (Holocene) sediments. Structurally, the bedrock in the subject area is within the northern part of the Composite Platform of Southeast New England (the Milford-Dedham zone on the State Map). The Milford-Dedham zone is used to describe the group of late Proterozoic and early Paleozoic rocks that lie between the Bloody Bluff Fault Zone and the Northern Boundary Fault of the Boston Basin 15 miles south of the site.

The Milford-Dedham and Nashoba zones, which lie north of the Bloody Bluff Fault Zone, contain intrusive rocks representing different tectonic events. The bedrock in the study area mainly consists of rocks within the Milford-Dedham zone which include late Proterozoic and early Paleozoic quartzitic sediments, volcaniclastics, and intrusive rocks. These rocks were formed over 600 million years ago and have a long and active history of tectonic deformation. These rocks were mainly altered by ductile deformation during tectonic stress induced from movement along the Bloody Bluff thrust fault over 500 million years ago. This period of deformation made fine-grained rocks from coarser-grained rocks by granulation during dislocation metamorphism, without significant chemical reconstitution of granulated minerals. Later deformation was brittle and was produced by lateral strike slip fault movement which is represented by the series of faults at an angle to the northeast-southwest trend of the Bloody Bluff Fault Zone.

The past history of deformation and the composition of rocks in the study area controlled how the ground surface looked prior to the last glaciation period, 11,000 to 14,000 years ago. This surface consisted of hills of more resistant bedrock and valleys eroded by the ancestral Ipswich River and its tributaries. During the last glacial advance, these ancestral drainage channels were scoured and further eroded into deep valleys in the bedrock surface, which in the vicinity of the site approximate the current surface drainage patterns. The ice sheet, moving approximately S40E, eroded the bedrock surface and filled-in valleys by deposition of ground moraine and outwash deposits to form a smooth and gently sloping surface. In the past few thousand years, a

rise of surface elevation of perhaps 10 meters has taken place and the upland areas have been undergoing active erosion, while younger Holocene fluvial sediments and organic deposits of peat and muck have covered the glacial deposits in the current stream valleys.

3.3.2 Study Area Geologic Setting

The composition and texture of the bedrock units in the study area reflects their original rock type and the history of tectonic deformation. These in turn control the relative hardness of the rock and the susceptibility to weathering and erosion. The present bedrock surface within the study area is irregular with a maximum relief of approximately 120 feet. The harder bedrock with less fracturing is well exposed as rounded and knobby hills, while the deeply weathered fractured rock within fault zones has been further eroded and covered by glacial deposits and is therefore poorly exposed.

The Milford-Dedham zone that makes up most of the study area, is bounded by the Bloody Bluff Fault Zone which trends northeast-southwest through the study area to the north (Figure 2-2). The Milford-Dedham zone is represented in the study area by three texturally different types of intrusive rock called gabbro-diorite which underlies the property and the areas south and east of the property. These rocks are generally moderately resistant to erosion and form the outcrops in the upland areas in the vicinity of the Calcium-Sulfate Landfill or the low-lying outcrops such as those along the northern side of the South Ditch. In areas of greater deformation or later faulting, the gabbro-diorite is irregularly eroded, although bedrock lows appear to coincide with fracture zones north and west of Eames Street and between Main Street and the east edge of the Maple Meadow Brook wetlands. In general, these areas form a topographically lower terrain with more gentle slopes.

The remainder of the study area lies within an area of very old faulting around the Bloody Bluff Fault Zone and somewhat younger faulting within the associated Burlington Mylonite Zone. The Bloody Bluff Fault Zone is a zone of several closely spaced individual thrust faults along which rock were crushed together (granulation) during movement along these thrust faults over 500 million years ago (early Proterozoic). The resulting fault zone is at least 600 feet wide in the study area and trends northeast-southwest roughly parallel to Maple Meadow Brook. The fault passes near the Chestnut Street Pumping Station to the south and passes beneath the Town Park area to the north.

The deformation along this fault zone has altered the original coarser-grained rocks to form mylonites which are very fine-grained rocks that weather and erode more easily. These mylonites which make up the Burlington Mylonite Zone extend as zones of more deformed rock (separated by less deformed rock) as far as 4 miles southwest away from the Bloody Bluff Fault Zone. In the study area, this zone of weakened and deeply weathered bedrock was eroded before and during glaciation to form the bedrock low currently occupied by the Maple Meadow Brook wetlands and its associated drainage system. Borings in the Maple Meadow Brook wetlands encountered a zone of severely weathered bedrock above the surface of competent bedrock or along suspected fracture zones within the bedrock. The weathered bedrock was a combination of silt, clay, and

rock fragments that maintained a remnant structure of the bedrock. The occurrence of the weathered bedrock was limited to areas within the Maple Meadow Brook wetlands which were underlain by the mylonite zone and the thickness of the weathered bedrock ranged from a few inches to several feet.

The extreme northwestern part of the study area lies within the Nashoba zone. This zone is north and west of the Bloody Bluff Fault Zone and is represented by the Andover Granite and granitic mylonites. In the low lying areas close to the Bloody Bluff Fault Zone such as near the old canal and Butters Row Water Treatment Plant, the Andover Granite is more eroded due to greater deformation and weathering. The hills to the northwest of the study area are underlain by the granitic rock which is less deformed.

3.3.3 Summary of Bedrock Characteristics

The rock types identified in the study area are shown on Figure 2-3 and are discussed in detail in Section 2.1. The characteristics of each bedrock type based on geologic mapping, rock cores, borehole geophysics, petrographic evaluation, slug tests, and downhole velocity surveys are summarized in Table 3-2. The evaluation of camera logs, caliper logs, and rock cores has shown that the bedrock contains many in-filled fractures and relatively few open fractures. Typically two to three open fractures were observed in each well, although the low yield of most of the wells (about 1 gpm) suggests that these fractures are not significant water bearing zones. Most of the fractures occurred in sets or zones which often corresponded with an increase in clay content or the boundary between different lithologic units as interpreted from the natural gamma and resistivity logs.

The regional fracture systems of the Bloody Bluff Fault Zone and the local subsets associated with the Burlington Mylonite Zone were generated from compressional and extentional states of stress. In this context, the zone of fracturing could penetrate to significant depth. However, low water yields, low transmissivities determined from slug tests, and observed in-filling of fractures with hydrothermally deposited or weathered minerals shows that the interconnection and water storage of these fractures is limited. In areas where less mineralogic alteration has occurred or where horizontal exfoliation or stress release fractures are present, fractures tend to be better connected and more open. Typically, these fractures are more prevalent in the upper portion of the bedrock and diminish in number with depth.

3.3.4 Bedrock Surface

The data obtained from borings drilled for well installation and soil sampling, the mapping of outcrops, and the seismic reflection surveys conducted in the study area have yielded a detailed picture of the morphology of the bedrock surface. The bedrock surface morphology has been shaped by lithologic differences in the bedrock, structural features such as ancient faults, and millions of years of erosion and glaciation. The bedrock surface is highly variable and has a significant amount of relief. A contour map of the bedrock surface which has been prepared using a collation of all data, is presented as Plate 2-1. The cross-sections shown on Figures 2-5 through

2-12 can be used to help interpret the bedrock contour map. These cross-sections are slices through overburden and the bedrock, and provide a vertical picture of the shape of the bedrock surface. The location of each of the cross-sections is shown on Plate 2-1.

As discussed in Section 2.1.3, the primary feature of the bedrock surface is the Western Bedrock Valley that extends from the property towards the west and reaches depths of over 120 feet beneath the Maple Meadow Brook wetlands at well GW-83D (Figure 2-12). As shown on Section H-H' (Figure 2-12) and on Plate 2-1, the dense layer present at GW-83D is restricted to the west by a bedrock saddle. North of GW-83D the bedrock rises towards a bedrock saddle which restricts the dense layer from flowing to a separate bedrock low at the GW-84 cluster. As shown in Section D-D' (Figure 2-8), the GW-84 and GW-85 well clusters are located in converging bedrock valleys and are separated by a bedrock high adjacent to these wells. Based on the current data, the Western Bedrock Valley terminates slightly north of where the former Middlesex Canal crosses the wetland (Figure 2-11).

An Eastern Bedrock Valley was identified in the CSA, and recent data has shown that this valley does not extend significantly beyond the eastern property boundary (Figure 2-5). The CSA also identified a possible Southwestern Bedrock Valley located along the southwestern property line and extending towards Breed Avenue. A seismic line through this area and the installation of additional wells (the GW-81 pair) has shown that bedrock elevations along the southwest property line are relatively high. Therefore, there is no connection between the Western Bedrock Valley with the slight bedrock low on Breed Avenue (Figure 2-6).

3.4 HYDROGEOLOGY

The hydrogeology has been described generally in the CSA. The discussion below includes information from the CSA, but is expanded to include additional data and characterizations obtained during the Supplemental Phase II investigation. Figure 3-2 is a schematic showing the major components of the groundwater flow system surrounding the Olin property.

3.4.1 Hydrostratigraphic Characterization

Groundwater in the area surrounding the Olin property generally occurs in two types of geologic formations: the unconsolidated sand and gravel deposits, and the crystalline bedrock. The geology of these units is described in more detail in Sections 3.2 and 3.3 above. The bedrock, in addition to underlying the sand and gravel deposits, also locally interrupts this aquifer in the form of minor outcrops and some major bedrock hills.

The unconsolidated sand and gravel deposits, where present, typically have a high porosity and permeability and thus are able to transmit large quantities of water, providing a highly productive aquifer. Locally, the unconsolidated aquifer exhibits a very high degree of heterogeneity. As evidenced from the detailed well logs generated during the installation of wells in the Western Bedrock Valley, relatively coarse and relatively finer deposits interfinger to produce a very

complex stratigraphic profile that does not correlate well horizontally. This is shown quantitatively in the slug test results which reveal hydraulic conductivities ranging between 2 and 4,000 feet/day. Because of this local variability, there will be preferential groundwater flow paths through the sand and gravel aquifer. However, given the complex nature of the glacial depositional environment, it is not realistic to map and correlate the existence of these localized zones. Outside the local scale, the aquifer has a high average hydraulic conductivity and transmissivity. Pumping test results indicate transmissivities between 2,000 and 13,000 square feet/day, with hydraulic conductivities between 20 and 250 feet/day. Based on annual reports submitted to MADEP by the Town of Wilmington, the combined usage at the five water supply wells making up the Maple Meadow Brook wellfield averages about 1.4 million gallons per day (MGD) on an annual basis (1992 through 1996 data); these wells are permitted to produce up to 3.5 million gallons per day.

In general, the aquifer is unconfined with the exception of the area beneath the Maple Meadow Brook wetlands. This area is overlain by the extensive, dense peat and muck layer which restricts groundwater flow and is likely to locally confine the sand and gravel aquifer.

In contrast to the high permeability of the overburden material, the underlying bedrock has a much lower permeability. Because of its low primary porosity, all groundwater flow takes place through fractures in the crystalline rock. These fractures occupy only a small part of the bedrock formation and so the bedrock typically contains a very small amount of water compared to the overlying sand and gravel. The flow system in the bedrock is anisotropic because of the structural controls on fracturing. The poorly yielding nature of these fractures is demonstrated by transmissivity values calculated from slug tests which range from 0.035 to 2.3 square feet/day. Calculated hydraulic conductivities would be very low; the CSA presented values of 0.03 to 0.06 feet/day. During purging of the bedrock wells prior to sampling, they are easily pumped dry. In contrast, wells in the sand and gravel show little drawdown even at significantly higher purging rates. In summary, the bedrock formation forms an insignificant part of the overall groundwater flow system in this area.

3.4.2 Horizontal Groundwater Flow Patterns

The Olin property is located on a groundwater divide, with flow from much of the property to the southeast within the Aberjona River watershed. Flow from a portion of the property and most of the area to the west is towards the west within the Ipswich River watershed. For reference, groundwater contour maps are provided on Plates 2-3 through 2-6, and water level data are presented in Tables 2-3 and 2-4. The location of the divide crosses the groundwater mound located on the northern portion of the Olin property and continues to the bedrock high beneath Cook Avenue. East of this divide, groundwater flow is locally towards the on-property ditch system, and to the East Ditch. To the west of this divide, flow is westward towards the Maple Meadow Brook wetlands and the water supply wells. The exact location of this divide varies

seasonally, sometimes significantly, based on local factors including the presence of the weir on the South Ditch and pumpage at the Altron industrial wells. Groundwater flow patterns observed in shallow and deep wells are similar.

Local groundwater flow patterns are slightly different than shown in the CSA due to the presence of the weir in the South Ditch. The weir has created localized seasonal mounding in this area with a component of flow to the northwest. East of this area, shallow groundwater flow is generally towards the east in the direction of the ditch systems as previously described. However, when groundwater levels are low (typically summer and fall), a distinct mound is formed in this area. As a result, the groundwater divide crosses through this mound allowing groundwater immediately west of the property to flow to the northwest towards the Maple Meadow Brook wetlands and the supply wells. When groundwater levels are high (winter and spring), this mounding is overcome, and local flow is towards the ditches and eventually to the southeast.

An additional seasonal difference in groundwater flow patterns is observed in the vicinity of the Maple Meadow Brook wellfield. During wet periods of the year, the wetland is submerged under several feet of standing water. It is likely that much groundwater is discharging to the wetland at this time, and that the supply wells are inducing localized groundwater recharge from this water body. Conversely, during dry periods of the year, the wetland often holds little standing water due to significant evapotranspiration.

Groundwater is recharged locally by rainfall on the ground surface. Because of the highly permeable nature of the surface soils and the predominant gentle slopes of the sand and gravel aquifer, a relatively high percentage of rainfall infiltrates to the subsurface, except where intercepted by wetlands or impermeable surfaces such as buildings and roads. In contrast, the rain that falls on bedrock exposures cannot infiltrate as readily because of the lower permeability and generally steeper slopes. Therefore, excess water runs off into the surrounding sand and gravel aquifer. This creates increased hydraulic heads in the aquifer adjacent to bedrock exposures. The lower permeability of the bedrock also supports steeper hydraulic gradients. Therefore, bedrock highs are also groundwater highs bounded by relatively steep gradients. Groundwater is also locally discharged to the on-property ditch system, to the Maple Meadow Brook wetlands, and to the miscellaneous groundwater users including the water supply wells. A more detailed discussion about the interaction between groundwater and the on-property ditch system is provided in Section 3.5 below.

3.4.3 Vertical Groundwater Flow

In general, vertical hydraulic gradients are not particularly strong, with head differences between shallow and deep wells typically on the order of tenths of a foot. Upward vertical gradients generally occur immediately adjacent to surface water bodies (Maple Meadow Brook and the ditches). Downward vertical gradients seem to dominate the remainder of the area although there are a number of wells with no significant head difference between the shallow and deep well.

Immediately adjacent to the water supply wells, vertical gradients in the shallow portions of the aquifer would be expected to be downward towards the intermediate pumping zone; gradients in the deep, upward.

Head differences between the overburden aquifer and the bedrock are relatively minor and do not indicate the presence of a significant confining unit between these formations. Vertical gradients in the GW-61 and GW-68 clusters are generally upwards, and downwards in the GW-62 and GW-65 clusters.

3.4.4 Flow of the Dense Layer

Movement of the dense layer is primarily controlled by the shape of the bedrock rather than by hydraulic heads and gradients. The waste material that was disposed into the unlined impoundments on the property was an acidic liquid with high concentrations of dissolved solids. Because these impoundments were unlined and the underlying soil was reasonably permeable, much of the liquid wastes would have infiltrated into the ground. This liquid was more dense than natural groundwater and thus it tended to sink until it reached a less permeable barrier, the top of the bedrock. It was this process that created the dense layer that is currently present at the base of the aquifer in the vicinity of the property. Once the dense liquid reached the bedrock, it continued to flow down slope along the top of the bedrock under the influence of gravity. Therefore, the direction of movement of this dense layer can be determined by examining the bedrock contour map which shows the shape of the bedrock surface (Plate 2-1). Beneath the unlined pits where the liquid was disposed (Lake Poly, East and West Pits, and acid pits), the bedrock generally slopes to the west and southwest. Therefore, the dense material moved along the top of the bedrock towards the west and southwest. This process continued until the dense layer reached the bottom of the Western Bedrock Valley that is located beneath the center of the property and passes underneath Jewel Drive. It then continued to move down slope in the bottom of the Western Bedrock Valley toward the west. This process led to the current location of the dense layer beneath the center of the property and within the Western Bedrock Valley beneath Jewel Drive, the Altron facility, the Koch facility and extending beneath the Maple Meadow Brook wetlands. A small amount of the dense material also moved from the center of the property towards the east within the Eastern Bedrock Valley by the same mechanism.

Potential future movement of the dense layer is discussed in Section 5.0, concerning fate and transport.

3.4.5 Rate of Groundwater Movement

Due to the local heterogeneities within the sand and gravel aquifer and the change in local gradients, groundwater velocities will be highly variable. The CSA presented average linear groundwater velocities of 100 to 325 feet/year beneath the property and 10 to 425 feet/year west of the property. These estimates are consistent with the results of the borehole flow monitoring that measured velocities up to 3.0 feet/day with an average of 1.7 feet/day. Velocities below about

70 feet/year (0.2 feet/day) are too slow for the instrument to detect. In addition to local variability associated with aquifer heterogeneity, there are likely to be seasonal variations in velocities associated with changes in the flow system.

3.4.6 Local Groundwater Management and Usage

The Town of Wilmington has formally defined a wellhead protection area around the Maple Meadow Brook wellfield (based on the study by IEP, 1990). This area has been accepted as a MADEP-Approved Wellhead Protection Area Zone II. This zone extends from the wells to the east to Main Street and north of Eames Street and includes a small piece of the northeastern portion of the Olin property. The outline of this zone is shown on Figure 3-3. Within this zone, groundwater is classed as GW-1 under the MCP; outside this zone is GW-2.

There are currently five commercial/industrial, 14 residential, and five public water supply groundwater withdrawals in the vicinity of the Olin property. The locations of these groundwater users relative to the property are shown on Figure 3-3. Specific information about these wells and the groundwater usage is summarized in Table 3-3. Water quality information has been obtained from all of these wells on several occasions. These data are discussed in detail in Section 4.

The five commercial/industrial groundwater withdrawals include the three shallow interceptor wells (IW-11, IW-12 and IW-13) at the Plant B Tank Farm on the Olin property and two industrial water supply wells on the Altron property. Olin's groundwater interceptor system is a groundwater/oil recovery and treatment system, and is discussed in more detail in Section 4.4.1 below. The system pumps an average of about 12,000 gallons per day (gpd). Altron's two wells (B1 and B3) are used to provide industrial process water and are permitted to pump a combined total of 190,000 gpd. The wells are set at intermediate depths in the sand and gravel aquifer. The CSA (CRA, 1993) reported pumping rates of approximately 50,000 gpd per well. More recent information from Altron (1997) indicates rates of 80,000 to 100,000 gpd per well.

Both the sand and gravel aquifer and the bedrock are utilized for domestic water supplies in this area. Ten homes along Cook Avenue and Border Avenue and two homes on Main Street have domestic wells set in the bedrock. According to the homeowners, the depths of these wells range from 90 to 705 feet, where known. There are also two homes along Main Street who have domestic wells set in the sand and gravel aquifer. The depths of these wells are expected to be shallow. In addition to these current users of groundwater, many of the homes in this area utilized domestic water supply wells prior to being placed on the municipal distribution system. The status of such wells as known to Olin is shown in Table 3-3. Olin is currently in the process of having certain domestic wells properly abandoned by the Massachusetts-licensed driller. The status of these abandonments as of June 13, 1997 is also provided on Table 3-3.

The five municipal water supply wells making up the Maple Meadow Brook wellfield include Butters Row No. 1, Butters Row No. 2, Chestnut Street No. 1, Chestnut Street No. 2, and Town Park. The Chestnut Street No. 2 well is also known as the Chestnut Street 1A well. This wellfield and the accompanying water treatment plant are owned and operated by the Town of

Wilmington. All five wells are set at intermediate depths within the sand and gravel aquifer and pump an average of 1.4 MGD. The combined maximum permitted yield of the five wells is 3.5 MGD. The wellfield was originally developed in the early 1960s, with the Chestnut Street No. 2 well added in 1992. Because of naturally high levels of iron and manganese in the groundwater as well as a later water quality problem associated with TCE, the Butters Row Treatment Plant was constructed and came on-line in 1981. Water from all five wells is routed through the plant which has a design capacity of 3 MGD with an emergency capacity of 4 MGD.

3.5 LOCAL HYDROLOGIC SYSTEM

The Supplemental Phase II Field Investigations included a more detailed study of the South Ditch. The specific results are provided in Section 2.4 above, and the major findings are summarized below. In addition, although limited additional data has been obtained concerning the other ditches, the current understanding of their hydrology is also briefly presented.

3.5.1 South and West Ditches

- Upstream of the weir in the Off-Property West Ditch, the interaction between surface water and groundwater is variable. In general, during periods of low water levels, the surface water, where present, recharges groundwater. During periods of high water levels, groundwater discharges to the ditch. Despite these generalizations, there is more local and seasonal variability within this section of the ditch.

- Within the South Ditch from the weir to the juncture with the East Ditch, hydraulic heads indicate that a regime of groundwater discharge to the South Ditch is typical in periods of both high and low water level. The flow study of the South Ditch suggests that there is little actual interaction between the groundwater and surface water in the vicinity of GW-78S, perhaps due to shallow bedrock in this area or to lower permeability of streambed sediments. From this area to just downstream of the weir, the South Ditch is sometimes gaining, sometimes losing, and sometimes mixed. However, the net change in flow within this reach is generally small. From the Central Pond to the eastern property line, the South Ditch receives groundwater discharge.

- Water levels in the Central Pond are higher than both groundwater and the nearby South Ditch. Therefore, neither the groundwater nor the ditches are a likely source for the water in the pond; instead, it most likely receives water from rainfall and nearby runoff. The water level mounding at the Pond suggests a poor hydrologic connection with either groundwater or the South Ditch.

- Because of the high permeability of surface soils and the small drainage areas for the West and South Ditches, peak stormwater runoff is dissipated very rapidly within the ditches; there is little potential for significant flow to take place from the ditches to the groundwater in the absence of the weir.

3.5.2 Other Surface Water Features

Little additional hydrologic data has been collected from the On-Property West Ditch, the Central Wetland, the Ephemeral Drainage, or the East Ditch. However, to provide a complete description of the entire ditch system, the current understanding of these features is summarized below.

- The On-Property West Ditch generally appears to recharge groundwater. This ditch receives a NPDES-permitted outfall from the Plant B groundwater treatment system averaging about 12,000 gpd, which is discharged in batches. Although there are no shallow wells in the immediate vicinity of this ditch, the elevation of the base of the ditch seems to generally be higher than nearby shallow groundwater. This may vary seasonally if winter and spring groundwater levels rise above the base of the ditch.

- The Ephemeral Drainage is expected to be a seasonal groundwater discharge point. When groundwater levels are high, groundwater flow is towards this low-lying area, which subsequent downstream flow to the South Ditch. Given the till and shallow bedrock located between the South Ditch and the Ephemeral Drainage, it is unlikely that significant underflow beneath the South Ditch reaches the Ephemeral Drainage, although this may take place locally. At the western end of the Ephemeral Drainage west of GW-55, there is overland flow to the South Ditch when water levels are particularly high.

- The Central Wetland is also expected to be a seasonal groundwater discharge point, receiving discharge when groundwater levels are high.

- Based on observations recorded in the CSA and additional observations during the Supplemental Phase II Field Investigations, the East Ditch is a perennial groundwater discharge point. Although this is a relatively small ditch, its drainage basin is significantly larger than that of the other ditches. All observations suggest that there is flowing water in this ditch throughout the year.

3.5.3 Site Hydrologic Balance

A qualitative water balance for the Olin property can be developed from our current understanding of site conditions. Average annual rainfall is approximately 41 inches (Baker, et al., 1964). Precipitation measurements at the Butters Row Treatment Plant for 1992 to 1995 are consistent with this average (see Table 3-1); 1996 was a particularly wet year.

The ground surface on the Olin property generally consists of impermeable cover (asphalt and buildings) within the former manufacturing areas or relatively permeable surface soils. Only limited areas to the southeast and northwest are underlain by bedrock or soils of low permeability. The topography of the ground surface (shown on Plate 1-1) shows that surface drainage from most of the undeveloped portions of the property is contained within the property or is directed to the ditches that immediately bound the property. Surface drainage from the developed area simply runs off into the surrounding soils. The topography also shows that there is little potential for significant runoff to enter the property from neighboring parcels except via the ditch system.

Based on the generally gentle slopes and relatively high permeability of surface soils, little actual runoff from the property is expected to take place. Instead, most of the precipitation is expected to infiltrate to the subsurface to recharge groundwater and/or be evapotranspirated. The southern half of the Olin property is heavily wooded; the northern and central portions are vegetated with grassy fields or woods outside the manufacturing areas. Regional studies (Baker, et al., 1964) indicate that annual losses to evapotranspiration are approximately 47 percent of precipitation. Therefore, approximately 53 percent of precipitation would be expected to enter the groundwater flow system. Using a drainage area of approximately 44 acres, about 3.5 million cubic feet of water per year recharges the groundwater over the Olin property.

Groundwater may enter the Olin property from neighboring parcels in only two areas: (1) from the old City of Woburn landfill to the south, and (2) from the area of Cook Avenue to the southwest. Both of these areas are expected to be groundwater highs with a component of flow towards the Olin property. Whether groundwater from these areas actually reaches the Olin property is not known due to lack of local data.

Once in the groundwater flow system, most of the water beneath the Olin property is potentially lost to the East Ditch, the South Ditch, the Central Wetland, the Ephemeral Drainage, and the Off-Property West Ditch. Measured hydraulic heads suggest that groundwater is unlikely to discharge to the On-Property West Ditch or the Central Pond. In addition, a small portion of the groundwater from the northern portion of the property would be expected to by-pass these features and flow further north and west, partially to the Altron supply wells. In the area of Plant B, groundwater is locally lost to the groundwater interceptor system, which may also be pumping a component of flow from the East Ditch. Flow measurements have been obtained from the South Ditch. Using an average discharge to the South Ditch of about 40 gallons per minute over a seven-month period when such flows can be measured suggests a minimum groundwater discharge to the South Ditch of approximately 1.6 million cubic feet per year. Discharge to both the East Ditch and the Off-Property West Ditch are also expected to be significant, but have not been measured.

4.0 NATURE AND EXTENT OF CONTAMINATION

MCP 40.0835(4)(f)

CHAPTER SUMMARY

This chapter presents the discussion on the nature and extent of contamination based on the results of the Comprehensive Site Assessment (CSA [CRA, 1993]) and the Supplemental Phase II Investigation identified in Section 2.0. The results of these two studies fully characterize risk at exposure points on-and off-property, provide a basis for evaluation of remedial alternatives (Phase III), and address Massachusetts Department of Environmental Protection (MADEP) comments by further characterizing the following:

- Background concentrations in selected media (soil, sediment, surface water, and groundwater on- and off-site),
- Surface soils and subsurface soils in former Lake Poly Liquid Waste Disposal Area (Lake Poly),
- Surface soil and sediment in areas with elevated concentrations of site chemicals,
- Surface water and groundwater geochemical interactions in the Ditch System leading to the formation of a chromium-bearing flocculent,
- Flocculent composition and occurrence in the Off-Property West Ditch and South Ditch,
- Extent, movement, and geochemistry and fate of contaminant in the deep groundwater and the dense layer, and
- Groundwater water quality and geochemistry in areas on and off the Olin property, and particularly the areas in the vicinity of groundwater users.

The additional analytical results for the site collected since the CSA (CRA, 1993) are used below to update the previous characterization of surface soil, subsurface soil, surface water, sediment, and groundwater on-property, and the surface water, sediment, and groundwater off-property. These additional analytical results include data collected as part of the Supplemental Phase II Investigations, South Ditch Weir Immediate Response Action (IRA), and the Drum Areas IRA. The Supplemental Phase II Investigations were initiated in 1994 mainly to provide additional information about:

- The dense layer,
- Chromium-bearing flocculent in the ditch system,
- The bedrock surface, and
- The hydrogeology and geochemistry of the Ditch System.

The results of the Supplemental Phase II Investigations confirmed the CSA finding that inorganic compounds are the most prevalent class of contaminants associated with the site. Ammonia, chloride, chromium, and sulfate are the main inorganic compounds that were detected at high concentrations in all media on-property and in the sediment and the dense layer off-property. The main conclusions regarding inorganics are:

- The greatest mass of inorganic contaminants resides in the dense layer.
- Slow diffusion of inorganics from the dense layer is the main mechanism for contaminant transport from the dense layer.
- Mobility of cationic metals, such as chromium, are limited by precipitation and adsorption reactions.
- The distribution of inorganics in the dense layer is controlled by the shape of the bedrock valleys and the resulting gravity-driven flow of the dense layer to its current position in the bedrock lows.
- In groundwater, the highest concentrations of inorganics are generally on-property and immediately west of the property.
- High concentrations of sulfate, and to a lesser degree ammonia, chloride, and sodium, were detected in the groundwater in the vicinity of, (or downgradient of), Lake Poly, the Lined Lagoons, the Acid Pits, and the Calcium-Sulfate Landfill. These inorganics appear to be mainly associated with residual soil contamination in these areas, although the dense layer, where present at higher elevations, may also be contributing some inorganics to the groundwater discharge areas along the Ditch System. In areas outside of the dense layer, the distribution of these inorganics in the groundwater is controlled by the location of residual sources, groundwater flow directions, and the location of shallow groundwater discharge areas along the Ditch System.

- The deep groundwater in the lower Western Bedrock Valley contains higher concentrations of inorganics, but does not contain chromium. The distribution of these inorganics in the deep groundwater appears to be mainly controlled by the shape of the Western Bedrock Valley, the presence of preferential flow units and barriers to flow, and deep groundwater flow directions.

The results of the Supplemental Phase II Investigations also confirmed the CSA finding that organic compounds are less prevalent off-property and are present at the highest concentrations in sediment, subsurface soil, and subsurface waste samples on-property. The main conclusions regarding organics are:

- The most prevalent semivolatile organic compounds (SVOCs); n-nitrodiphenylamine (NNDPA), phenol, and total phthalates. These SVOCs are detected at consistently higher concentrations in most media.

- Volatile organic compounds (VOCs) are generally detected at low concentrations in all media, with the exception of acetone and trimethylpentenes which were detected at elevated concentrations in most media on-property or immediately adjacent to the property.

- Pesticides were detected at low concentrations, and were consistently detected in the surface soils and sediments. Pesticides were sporadically detected in the subsurface soils, surface water, and shallow groundwater.

- Inside the dense layer, the distribution of organics are controlled by the shape of the bedrock valleys and the resulting gravity-driven flow of the dense layer.

- Outside of the dense layer, the distribution of organics in the groundwater is controlled by the location of residual sources, off-site sources, groundwater flow directions, and the location of shallow groundwater discharge areas along the Ditch System.

The results of the Supplemental Phase II Investigations and the CSA were used to identify the following the main impacted media associated with the major source areas. Olin has already acted to protect the public and on-property workers from potential risks in several of these areas including Plant B, the Off-Property West Ditch and South Ditch, and the Drum Areas. The major source areas, releases and the impacted media are listed below:

- Groundwater and soils in the former Plant B Production and Tank Farm Areas.

- Soils and Sediment in the area of former Lake Poly

- Soils and waste in Drum Area A and Drum Area B.

GW beneath Ephemeral Ditch GW-555

floodplain (lower S. Ditch) East Ditch

- Groundwater, surface water, and sediment near the South and West Ditches.
- Subsurface soils associated with the East and West Pits
- Subsurface soils associated with the Acid Pits and Lined Lagoons
- Shallow and Deep Groundwater at GW-54 well cluster
- Deep Groundwater and Dense Layer.

To address MADEP comments on potential future exposure to groundwater users west of the property, Olin has conducted intensive investigations to define the nature and extent of contaminants. The investigations have included installation of 14 new wells in remote areas within the Maple Meadow Brook wetlands, geochemical and groundwater modeling, and a monitoring program to protect the public health which includes 15 monitoring wells and the five public supply wells. The first ten months of data has shown that the concentrations of indicator parameters detected at the public supply wells are within the range of concentrations historically detected in samples from these wells.

The documentation in support of the key findings outlined above are integrated and presented in the sub-sections of this section as outlined below:

- Section 4.1 presents the results of extensive background sampling designed to address MADEP comments about characterization of background levels. This section summarizes site-specific background concentrations that were determined for major parameters and provides the MADEP background concentrations for the remaining inorganic parameters.

- The updated distribution of major chemicals in each media and the Supplemental Phase II Investigations are summarized in Section 4.2. In this section the additional analytical results collected as part of the Supplemental Phase II Investigations, South Ditch Weir IRA, and the Drum Areas IRA are compared to the CSA (CRA, 1993) results to highlight new information and to update the previous characterizations of surface soil, subsurface soil, surface water, sediment, and groundwater on-property and the surface water, sediment, and groundwater off-property.

- The nature and extent of the major parameters is presented in Section 4.3. This section identifies source areas for the parameters consistently detected at higher concentrations in all media, and evaluates transport pathways and impacts to receptor media. MADEP concerns for the characterization of deep groundwater pathways are addressed by examining the distribution of chemicals in deep groundwater and the dense layer together. MADEP concerns for shallow groundwater transport of contaminants to overburden wells used for water supply

and for potential vapor migration of VOCs into buildings are addressed by examining the distribution of VOC (including ammonia)s in the shallow groundwater separately.

- The major areas of potential environmental concern based on the presence of elevated levels of oil and hazardous materials (OHM) are discussed in Section 4.4., This section summarizes characterization of source areas or other areas of environmental concern based on the nature and extent of contamination in various areas of the site and on the following criteria: (1) the presence of separate-phase oils, (2) exceedence of Upper Concentration Limits (UCLs), (3) the presence of potential hazards, (4) the potential for transport (especially off-site) at levels of concern, and (5) contamination of adjacent media by the same parameters. The major areas of concern also represent the areas of the site with the highest concentrations of chemicals, but not necessarily with the greatest potential for exposure.

- An assessment of other miscellaneous areas, mainly the Solid Waste Management Units (SWMUs) identified in the CSA (CRA, 1993), are discussed in Section 4.5. These areas are impacted by low concentrations of site-related chemicals and do not represent significant source areas. Generally, these areas are subject to response actions by Olin under the MCP or other regulatory programs.

- The groundwater quality at the Maple Meadow Brook supply wells, and at the monitoring wells in and surrounding the Maple Meadow Brook wetlands is discussed in Section 4.6. This section briefly summarizes the monitoring programs from 1986 through the present, and the preliminary results of the ongoing Groundwater Monitoring Program for the Western Bedrock Valley (Smith, 1996) and the Sentinel Well Monitoring Program. The preliminary results include ten months of monitoring chemical conditions, groundwater levels, and pumping rates. In addition, this section brings together results of the Supplemental Phase II Investigations and summarizes the distribution of site-related chemicals in the capture zone of the wellfield and provides a basis for modeling and evaluating risk at the public supply wells and for other groundwater uses.

- The delineation of the Disposal Site boundary is discussed in Section 4.7. The extent of the Disposal Site depends on the background concentrations discussed in Section 4.1 and is mainly controlled by the concentration of indicator parameters in the shallow groundwater, deep groundwater, and dense layer. The boundary of the Disposal Site also reflects the presence of other site contaminants in the surface water and sediment on- and off-property and the presence of site contaminants in the surface and subsurface soils on-property.

4.1 BACKGROUND CONCENTRATIONS

Background parameter concentrations in soil, surface water, sediment, and groundwater in the area of the site have been characterized in accordance with the 'Scope of Work, Human Health Environmental Risk Assessment" work plan conditionally approved by the MADEP in May 1996. The additional sampling and analysis of site-specific background samples was conducted to address MADEP comments, to identify potential compounds of concern, and to provide a basis for evaluating risk. The use of background data in human health, safety, and public welfare risk characterizations is summarized in Section 6.1 and is presented in detail in Appendix R and the use of background data to characterize environmental risk is summarized in Section 6.2 and is presented in detail in Appendix S. The background sampling locations for soil, surface water, and sediment are shown on Figure 4-1. The background sampling locations for groundwater are shown on Figure 4-2.

4.1.1 Background Soil Samples

Two background soil samples were collected on November 2, 1992 during the installation of GW-67, located 1,400 feet west of the site (CRA, 1993). Samples BGS-01 (surface soil sample) and BH-41 (subsurface soil) were analyzed for inorganics and polynuclear aromatic hydrocarbons (PAHs).

Five additional soil background samples were collected by ABB Environmental Services (ABB-ES) on April 22, 1996. Samples SS015XXBKX, SS016XXBKX, SS017XXBKX, SS017XXBKD (duplicate), SS018XXBKX and SS019XXBKX were analyzed for PAHs and calcium, potassium, sodium, sulfate, and nitrogen-Ammonia as N. Soil background concentrations for other organic compounds are assumed to be below detection limits and the background concentrations for the remaining inorganic parameters (shown on Table 4-1) are assumed to be equal to the background concentrations presented in Table 2.1 of the MADEP's Guidance for Disposal Site Risk Characterization - In Support of the Massachusetts Contingency Plan (MCP), Interim Final Policy WSC/ORS-95-141 (MADEP, 1995). The analytical results for the seven background soil samples are presented in Appendix R. The soil background concentrations are summarized in Table 4-1. In that table, median and maximum concentrations are presented for the parameters for which site-specific background data were collected and the concentrations reported by the MADEP (90th percentile values) are presented for the remaining parameters.

4.1.2 Background Surface Water and Sediment Samples

One background surface water and one background sediment sample were collected on April 20, 1993 (CRA, 1993). These samples were collected at sampling location SW-30 in the East Ditch approximately 375 feet upstream of the northern boundary of the site. This sample was identified in the CSA (CRA, 1993) as a background sediment sample, since no contaminants were detected in this sample which were indicative of a release from the site.

ABB-ES collected 15 surface water and 15 sediment background samples between April 1 and April 4, 1996. The additional samples were collected to address MADEP comments that the original surface water and sediment samples collected at location SW-30 did not meet the MCP definition of background because it appears the location of the background sample is being impacted by an "upstream" source. Consequently, these two samples are no longer considered background samples, although they may represent local conditions with respect to environmental receptors in the East Ditch area.

Five surface water samples (SW001XXBKX through SW004XXBKX and SW014XXBKX and its duplicate, SW014XXBKD), and five sediment samples (SD001XXBKX through SD004XXBKX and SD014XXBKX and its duplicate, SD014XXBKD) were analyzed for inorganics (method 6010), total solids (sediment only), total organic carbon (sediment only), SVOCs (method 8270B), VOCs, trimethylpentenes (method 8240), and target compound list (TCL) pesticides (method 8080). The remaining surface water samples (SW005XXBKX through SW013XXBKX) and sediment samples (SD005XXBKX through SD013XXBKX) were analyzed for TCL pesticides (method 8080). The analytical results for the 15 background surface water and sediment samples are presented in Appendix R. The surface water background concentrations are summarized in Table 4-2 and sediment background concentrations are summarized in Table 4-3. In these tables, median and maximum concentrations are presented for the parameters for which site-specific background data were collected.

4.1.3 Background Groundwater Samples

Groundwater monitoring wells were not installed specifically to collect background water quality data. It is important to note that the Olin property lies on a groundwater divide, that groundwater flows away from the center of the facility, and that an upgradient background groundwater locations does not exist and is not relevant to determining background groundwater levels at the site. Therefore, ABB-ES examined the levels of inorganics in over 2,000 groundwater samples to determine which wells lie in the Disposal Site and which are representative of background conditions at the site. The list of samples and analytical results for the background groundwater sampling locations are presented in Appendix R. The background groundwater levels were used to define the Disposal Site boundaries (Section 4.7) which allowed the site data to be grouped based on groundwater categories and potential future use of groundwater. Section 6 summarizes the groundwater classifications and calculation of exposure point concentrations. The groundwater background concentrations are summarized in Table 4-4. In this table, median and maximum concentrations are presented for the parameters for which site-specific background data were collected.

4.2 MEDIA CHARACTERIZATION

The additional analytical results for the site collected since the CSA (CRA, 1993) are used below to update the previous characterization of surface soil, subsurface soil, surface water, sediment, and groundwater on-property and the surface water, sediment, and groundwater off-property. These additional analytical results include data collected as part of the Supplemental Phase II Investigations, South Ditch Weir Immediate Response Action (IRA), and the Drum Areas IRA.

4.2.1 Waste Material Characterization

4.2.1.1 Drum Areas Waste Summary

The investigations in the drum disposal areas are discussed in detail in Section 1.3.5 and in the "Test Pit Excavation Summary Report" (CRA, 1992a). This section discusses the results of chemical analyses of the waste material removed from these areas in addition to the CSA (CRA, 1993) analytical results for waste material samples.

The CSA (CRA, 1993) identified three areas with buried drums or debris: Drum Area A, Drum Area B, and the Buried Debris Area. Plate 1-2 shows the location of the Drum Areas. As discussed below, the parameters and concentrations detected in Buried Debris Area soils are much lower than found in samples from Drum Area A and Drum Area B. Therefore, the two buried drum areas of concern are Drum Area A and Drum Area B. Table 4-5 presents a summary of the Drum Area analytical results for waste material and subsurface soil samples and includes the results of four additional samples from waste material excavated from the test pits in both Drum Areas. The shaded sample locations indicate the additional samples with maximum parameter concentrations greater than those detected in the CSA (CRA, 1993) or the locations with detections of parameters that were below detection limits previously in the CSA (CRA, 1993).

The four additional waste samples (OW-249A through OW-249D) were collected to characterize drums and wastes that were excavated for disposal purposes. Since these samples were collected from drum overpacks and wastes that had been segregated for disposal purposes, the record of which samples are associated with which Drum Area is somewhat unclear. Since the results for these samples show higher concentrations than previous waste samples, it was important to include them to fully characterize the Drum Areas. As listed on Table 4-5 and discussed below, the samples were assigned to each Drum Area based on (1) the descriptions of this waste material and of material sampled from test pits, (2) the similarities in chemical results, and (3) the waste disposal records.

Test pits in Drum Area A uncovered extensive buried drums and miscellaneous waste. CRA collected three drum samples and a duplicate sample from the Drum Area A. Three additional waste characterization samples, OW-249A through OW-249C, were used to update the CSA (CRA, 1993) characterization. Most of the VOCs were detected at concentrations of less than 1

milligram per kilogram (mg/kg) and the maximum detected concentrations were 360 mg/kg for 2,4,4-trimethyl-1-pentene and 93 mg/kg for 2,4,4-trimethyl-2-pentene. The main SVOCs detected are bis(2-ethylhexyl)phthalate at a maximum concentration of 87 mg/kg and n- NNDPA at a maximum concentration of 21,000 mg/kg. No PCBs or pesticides were detected in the waste samples. Inorganics detected at elevated concentrations include ammonia, calcium, chloride, chromium, iron, potassium, sodium, and sulfate.

The test pit in Drum Area B found scattered buried drums, laboratory bottles, and miscellaneous wastes. Conestoga-Rover & Associates (CRA) collected one drum sample and one soil sample from the Drum Area B. The analytical results of one waste characterization sample OW-249D was used to update the CSA (CRA, 1993) characterization of Drum Area B. The main VOCs detected in the waste samples included toluene and acetone. The main detected SVOCs included chlorobenzenes, bis(2-ethylhexyl)phthalate, indeno(1,2,3-cd)pyrene, NNDPA, and phenolic compounds. The concentration of inorganics detected in Drum Area B waste samples were relatively lower than Drum Area A, although elevated concentrations calcium, chromium, and sulfate were detected in subsurface soil samples from Drum Area B.

Test pits in the Buried Debris Area excavated crushed drums, drum parts, rubbish, and contaminated soil. Four VOCs were detected in the soil sample at concentrations of less than 0.002 mg/kg. Two SVOCs were detected in the soil sample; phenol at a concentration of 5.3 J mg/kg and NNDPA at a concentration of 1.6 J mg/kg. No PCBs or pesticides were detected in the soil sample. Ammonia was the main inorganic detected in subsurface soil samples at a concentration of 490 mg/kg. In conclusion, generally fewer chemicals at lower concentrations were detected at the Buried Debris Area as compared to the results of soil and waste samples at Drum Area A and B.

4.2.1.2 Floating Oil at Plant B

As discussed in the CSA (CRA, 1993), the layer of floating oil present around the interceptor wells located at the former tank farm at Plant B is attributable to spills, leaks, and releases of oily compounds associated with the former manufacturing processes and storage tanks in this area. The oil thickness on wells at Plant B is typically less than one-half inch, and the interceptor well with the largest amount of oil, IW-11, typically recovers about one liter of oil per week (personal communications with Don Cameron of Olin, 1997). Olin periodically pumps out the accumulated oil from the interceptor well into drums and disposes of it at an appropriate off-site facility. No oil was detected in the wells closest to the East Ditch. As discussed in Section 1.3, three interceptor wells have been used to prevent the discharge of oil and hazardous material into the East Ditch since 1982. Figure 4-3 shows a recent groundwater contour map which demonstrates that an inward gradient is maintained around Plant B even in the wet season during a period of higher groundwater elevations.

The parameters detected in the floating oil and groundwater are presented on Table 4-6. As reported in the CSA (CRA, 1993) the analytical results of an oil sample collected from IW-11 on August 12, 1992 show that the oil contains high concentrations of bis(2-ethylhexyl)phthalate,

trimethylpentenes (TMPs), and NNDPA. In contrast, the concentrations of these SVOCs and other parameters in the groundwater at IW-11 is over 100,000 times less than detected in the oil. On this basis the analytical results from the three interceptor wells, IW-11, IW-12, and IW-13, were re-evaluated and classified as either oil, a mixture of oil/groundwater, or groundwater as presented on Table 4-6. On this table the analytical results show that organics are found at low concentrations in the groundwater, while the inorganics are detected at about the same concentration in the groundwater, oil/groundwater, and oil. This demonstrates that the organics in the oil are not partitioning into the groundwater and that the inorganics are probably derived from a source other than the oil.

The main VOCs detected in the groundwater, at low concentrations, are TMPs, acetone, and xylenes. The main SVOCs detected in the groundwater, also at low concentrations, include phthalates and NNDPA. The inorganic compounds detected, ammonia, chloride, chromium, manganese, and sulfate, have similar concentrations in the oil, groundwater, and oil/groundwater mixture. As discussed above, the groundwater at Plant B is contained on-property by the interceptor wells.

4.2.2 Surface Soil Characterization

The CSA (CRA, 1993) included ten composite samples (grid Areas 1-10), four hand-auger grab samples (SWMUs 25, 27, 30, and 33), and one field duplicate sample. The Supplemental Phase II Investigations collected 54 additional surface soils samples in December 1996 and the results are discussed in Section 2.3. In 1997, ABB-ES collected seven additional soil samples in support of toxicity evaluations for the Ecological Risk Assessment. The toxicity sample results are consistent with the results of Supplemental Phase II Investigations and are discussed in Appendix S. The locations of the surface soil samples collected in the CSA (CRA, 1993) and the Supplemental Phase II Investigations are shown on Plate 2-8.

A summary of maximum concentrations for parameters frequently detected at elevated concentrations in the Supplemental Phase II Investigations and CSA (CRA, 1993) surface soil samples is presented in Table 4-7. On this table, the shaded sample locations indicate Supplemental Phase II Investigations sampling locations with maximum parameter concentrations greater than those detected in the CSA (CRA, 1993) or the locations with detections of parameters that were below detection limits previously in the CSA (CRA, 1993). Detailed comparisons of the Supplemental Phase II Investigations and CSA (CRA, 1993) surface soil results are discussed in Section 2.3. In general, the results of the Supplemental Phase II Investigations are similar to the results of the CSA (CRA, 1993) with the following exceptions:

- The resampling of CSA (CRA, 1993) grid Area 1 shows generally lower concentrations of most inorganics, including chromium.

- High concentrations of NNDPA were detected in surface soil samples from the Lake Poly area.

- Higher concentrations of phenols and pesticides were detected in the surface soil samples from Drum Area B.

- Higher concentrations of PAHs were detected in grid Area 8. As discussed in Section 2.3, all of the maximum concentrations for PAHs listed on Table 4-7 are attributed to one sampling location (A8CW-1) which is not considered representative since the PAH concentrations of other 19 samples in grid Area 8 were 100 to 1,000 times lower.

- Additional surface soil samples from the Central Pond drainage areas suggest that releases from the former Acid Pits and former Lined Lagoons may have flowed through ditches and over ground to the Central Pond. This is supported by generally higher concentrations of chromium in samples from the northern portion of the Central Pond drainage areas and higher concentrations of sulfate in samples from the western portion of the Central Pond drainage areas.

- A higher concentration of arsenic was detected in the surface soil sample from Drum Area A.

The analytical data collected during the CSA (CRA, 1993) and the Supplemental Phase II Investigations indicate that the following major compounds were most frequently detected at elevated concentrations in the surface soils.

Volatile Organic Compounds	**Semivolatile Organic Compounds**		**Inorganics**	
Acetone	Benzoic Acid	N-nitrosodiphenylamine	Ammonia	Potassium
Methylene Chloride	Bis(2-ethylhexyl)phthalate	Phenanthrene	Calcium	Sodium
Toluene	Di-n-butylphthalate	Pyrene	Chromium	Sulfate
1,1,1-Trichloroethane	Indeno(1,2,3-cd)pyrene		Manganese	
Trimethylpentenes				

The analytical data collected in the CSA (CRA, 1993) and the Supplemental Phase II Investigations show that bis(2-ethylhexyl)phthalate and the inorganic compounds, ammonia, chromium, and sulfate, were the major parameters detected in the surface soils. Bis(2-ethylhexyl)phthalate and the other phthalates are detected across the facility and were detected at the highest concentrations in the surface soils adjacent to Lake Poly along the banks of the upper On-Property West Ditch (SWMU 27), the upper South Ditch (SWMU 30), and the Ephemeral Drainage (SWMU 33). As discussed in the CSA (CRA, 1993), the high concentrations of organics detected in surface soils along the ditch banks are typically accompanied by the presence of residual waste oils which were released into the On-Property West Ditch. The major inorganic compounds detected, ammonia, chromium, and sulfate, were also detected in samples from these locations at high concentrations, and in samples from the area east of Plant D (SWMU 26) and in grid Areas 1, 8, and 9.

4.2.3 Subsurface Soil Characterization

The CSA (CRA, 1993) included collection of 40 subsurface soil samples and one background soil sample. As discussed in Section 2.3.2, the Supplemental Phase II Investigations included additional subsurface samples at borings LPB-1 and LPB-2 in the former Lake Poly to further characterize subsurface soils and to evaluate the vertical extent of contamination.

A summary of maximum concentrations for parameters frequently detected at elevated concentrations in the Supplemental Phase II Investigations and CSA (CRA, 1993) subsurface soil samples is presented in Table 4-8. On this table, the shaded sample locations indicate Supplemental Phase II Investigation sampling locations with maximum parameter concentrations greater than those detected in the CSA (CRA, 1993) or the locations with the detections of parameters that were below detection limits previously in the CSA (CRA, 1993). Detailed discussions of the Supplemental Phase II Investigations Lake Poly subsurface soil results are provided in Section 2.3. The main differences between the analytical results of the Supplemental Phase II Investigations and the CSA (CRA, 1993) are:

- Higher concentrations of TMPs, ethylbenzene, toluene, NNDPA, ammonia, and chromium were detected in subsurface soil samples from the former Lake Poly.

- New detections of 4-bromophenyl-phenylether, 4-chlorophenyl-phenylether, PAHs, and dibenzofuran were found in one subsurface soil sample from the former Lake Poly.

The analytical data collected during the CSA (CRA, 1993) and the Supplemental Phase II Investigations indicate that the following major organic and inorganic compounds were most frequently detected at significant concentrations in the subsurface soils. In general, the highest concentrations of these organic and inorganic compounds were found in the vicinity of the former Lake Poly.

Volatile Organic Compounds	Semivolatile Organic Compounds	Inorganics	
Acetone	Bis(2-ethylhexyl)phthalate	Ammonia	Potassium
2-Butanone (MEK)	Butyl Benzylphthalate	Calcium	Sodium
2-Hexanone	Di-n-octylphthalate	Chromium	Sulfate
Toluene	N-nitrosodiphenylamine		
Trimethylpentenes	Phenols		

The VOC data presented in the CSA (CRA, 1993) and the results of the Supplemental Phase II Investigations show that all of the VOCs listed above were detected in subsurface soils across the facility. The highest levels of TMPs were detected in former Lake Poly, the southeast corner of the West Warehouse, the area east of Plant D, and in the Plant B Area. The maximum concentrations of 2-butanone and acetone were detected in the area of the former Lined Lagoons and Acid Pits. Most of the other VOCs were sporadically and infrequently detected in subsurface soil samples at concentrations less than 0.035 mg/kg.

The most prevalent SVOCs detected in subsurface soils are NNDPA and bis(2 ethylhexyl)phthalate. NNDPA was detected in subsurface soils across the facility, although the highest concentration (3,400 mg/kg) was detected in a sample from Lake Poly. Similar to NNDPA, bis(2 ethylhexyl)phthalate was also detected also in subsurface soils across the facility and the highest concentration (6,700 mg/kg) was also detected in a sample from the vicinity of Lake Poly.

As discussed in the CSA (CRA, 1993), pesticides were detected in two boring locations at concentrations less than 0.15 mg/kg. Five pesticides were detected in a subsurface soil sample from Lake Poly and two pesticides were detected in a subsurface soil sample from the northeast corner of Lined Lagoon I. PCBs were not detected in any subsurface soil samples.

Elevated inorganics, including ammonia, calcium, chromium, potassium, sodium, and sulfate, were detected in subsurface soils across the facility. The highest concentrations of ammonia, chromium, and sodium were detected in the vicinity of Lake Poly. The highest concentrations of calcium, potassium, and sulfate were detected in the vicinity of the Acid pits and Lined Lagoons.

4.2.4 Sediment Characterization

The CSA (CRA, 1993) included collection of 30 sediment samples from the Ditch System and from Central Pond. As discussed in Section 2.7 above, additional sediment samples were collected to characterize inorganics in the flocculent in the Off-Property West Ditch and South Ditch. The Supplemental Phase II Investigations also included an additional sediment sample from the Central Pond and seven additional sediment samples for toxicity testing to support the Ecological Risk Assessment. The toxicity sample results are consistent with the results of CSA and Supplemental Phase II Investigations and are discussed in Appendix S.

A summary of maximum concentrations detected for parameters frequently detected at elevated concentrations in the Supplemental Phase II Investigations and CSA (CRA, 1993) sediment samples is presented in Table 4-9. On this table, the shaded sample locations indicate Supplemental Phase II Investigations sampling locations with maximum parameter concentrations greater than those detected in the CSA (CRA, 1993) or the locations with detections of parameters that were below detection limits previously in the CSA (CRA, 1993). Detailed discussions of the Supplemental Phase II Investigations Lake Poly sediment and flocculent results are provided in Section 2.7. The main differences between the analytical results of the Supplemental Phase II Investigations and the CSA (CRA, 1993) are:

- Slightly higher concentrations of tetrachloroethane were detected in the sediment sample from the Central Pond.

- A higher concentration of 4-bromophenyl-phenylether was detected in the sediment sample from the Central Pond. The results of subsurface soil and sediment results show that 4-bromophenyl-phenylether is only detected in this

sediment sample from the Central Pond, in the sediment sample locations in the South Ditch immediately adjacent to the Central Pond, and in subsurface soil samples from Lake Poly. These results suggest that some of the contamination in the Central Pond may have resulted from releases from Lake Poly that were transported via the ditches or by other means to the Central Pond.

The analytical data collected during the CSA (CRA, 1993) and the Supplemental Phase II Investigations indicate that the following major organic and inorganic compounds were most frequently detected at elevated concentrations in sediments at the site.

Volatile Organic Compounds	Semivolatile Organic Compounds		Inorganics	
Acetone	Benzoic Acid	Indeno(1,2,3-cd)pyrene	Aluminum	Manganese
Toluene	Bis(2-ethylhexyl)phthalate	N-nitrosodiphenylamine	Ammonia	Potassium
Trimethylpentenes	Butyl Benzylphthalate	Phenanthrene	Calcium	Sodium
	Di-n-butylphthalate	Phenol	Chloride	Sulfate
	Di-n-octylphthalate	Pyrene	Chromium	

The highest concentrations of all the above compounds were generally detected in the On-Property West Ditch and the concentrations decrease downstream. As discussed in the CSA (CRA, 1993), the high concentrations of organics detected in sediments along the ditch are typically accompanied by the presence of residual waste oils which were released into the On-Property West Ditch. Historically, Lake Poly discharged or overflowed into the On-Property West Ditch. The elevated concentrations of inorganics in the sediments are also believed to be associated with the direct releases from former liquid waste disposal areas, including Lake Poly, into the Ditch System and the discharge of contaminated groundwater into the Ditch System during periods of high groundwater (or low surface water) levels. The electron microprobe analysis of the flocculent conducted in the Supplemental Phase II Investigations shows that chromium is bound to iron- and aluminum-solid phases as either amorphous hydroxides or precipitate in mineral form. These phases of chromium are very insoluble in the surface water in the Ditch System.

As discussed in the CSA (CRA, 1993), the highest concentrations of other VOCs (trichloroethene, 1,2-dichloroethene (total), chlorobenzene, tetrachloroethene, and vinyl chloride) detected in the sediment were found at sampling locations SW-23 in the East Ditch downstream of the facility and are attributed to an off-site source. This is supported by the detection of the highest concentrations of these VOCs in groundwater samples from off-property well GW-49D and the GW-80 well cluster, which lie immediately east of the East Ditch.

4.2.5 Surface Water Characterization

The CSA (CRA, 1993) included collection of two rounds of surface water samples. As discussed in Sections 2.5 and 2.7, additional rounds of surface water sampling have been performed as part of the Supplemental Phase II Investigations mainly to characterize shallow groundwater and surface water interactions and the formation of flocculent both on and off the Olin Property.

A summary of maximum concentrations for the parameters most frequently detected at elevated concentrations in the Supplemental Phase II Investigations and CSA (CRA, 1993) surface water samples is presented in Table 4-10. This table presents the maximum concentrations detected in surface water samples in both Supplemental Phase II and the CSA (CRA, 1993). As shown on this table, none of the Supplemental Phase II Investigations surface water results exceeded the maximum parameter concentrations detected in the CSA (CRA, 1993) surface water samples. Detailed discussions of surface water results and geochemical evaluation are provided in Section 2.7.

The analytical data collected during the CSA (CRA, 1993) and the Supplemental Phase II Investigations indicate that the following organic and inorganic compounds were frequently detected at elevated concentrations in the surface water.

Volatile Organic Compounds	Semivolatile Organic Compounds	Inorganics	
*Acetone	*Bis(2-ethylhexyl)phthalate	*Aluminum	Iron
*Toluene	*N-nitrosodiphenylamine	*Ammonia	*Managnese
2-Hexanone	Phenol	*Calcium	*Sodium
*Trimethylpentenes		Chloride	*Sulfate
		*Chromium	Zinc
		Copper	

Note: Compounds with asterisk (*) also frequently detected in sediment.

As discussed in the CSA (CRA, 1993), the highest concentrations of acetone, TMPs, NNDPA, phenol and all of the above inorganics except for iron and sodium were detected in surface water samples from the Off-Property West Ditch. The Off-Property West Ditch receives groundwater discharge from the facility and surface water runoff from the surrounding industrial and commercial plants. The concentrations of the above parameters were also elevated in the surface water samples collected from the On-Property West Ditch. The On-Property West Ditch is considered a groundwater recharge area and the elevated concentrations in the surface water samples may be associated with the elevated concentrations of the same parameters in the sediments. Based on aerial photographs and the distribution of chemicals detected in sediment samples, Lake Poly discharged historically into the On-Property West Ditch.

As discussed in the CSA (CRA, 1993), the other VOCs (1,1,1-trichloroethane, 1,1-dichloroethane, chloroethane, methlyene chloride, and trichloroethene) detected in the surface water in the upstream portions of the East Ditch are probably attributable to an off-site source. In addition, the concentrations of toluene detected at the furthest upstream samples in the East Ditch are also probably attributable to an off-site source.

4.2.6 Groundwater Characterization

The CSA (CRA, 1993) included collection of a large number of groundwater samples. As discussed in Section 2.5, additional rounds of groundwater sampling have been performed as part of the Supplemental Phase II Investigations to update the CSA (CRA, 1993) results and further characterize the groundwater system both on and off the Olin Property.

Summaries of maximum concentrations of the parameters most frequently detected at elevated concentrations in the Supplemental Phase II Investigations and CSA (CRA, 1993) for shallow groundwater, deep groundwater and the dense layer, and bedrock groundwater samples are presented in Tables 4-11, 4-12, and 4-13, respectively. On these tables, the shaded sample locations indicate Supplemental Phase II Investigation sampling locations with maximum parameter concentrations greater than those detected in the CSA (CRA, 1993) or locations with detections of parameters that were not detected previously in the CSA (CRA, 1993). The groundwater results of the Supplemental Phase II Investigations are generally consistent with the CSA (CRA, 1993) with the following exceptions or new findings:

- Groundwater results from new wells, GW-80S, GW-80D, and GW-80BR, located upgradient and southeast of the property, confirm the CSA (CRA, 1993) conclusion that groundwater quality in this area is degraded by an off-site source. The results show a different suite of VOCs (ethylbenzene, toluene, and xylenes) with higher concentrations than detected in samples from wells on-property (Tables 4-11 though 4-13). The highest concentrations were detected in the bedrock well (GW-80BR).

- Lower concentrations of inorganics were detected in samples from most shallow groundwater wells along the Off-Property West Ditch. Concentrations of ammonia and sulfate have increased at GW-43S. As discussed in Section 2.4.1, the general improvement in shallow groundwater water quality is due to the clean ponded water entering the groundwater flow system, while the increase in the concentrations of some parameters is due to localized changes in groundwater flow directions.

- Higher concentrations of ethylbenzene, toluene, and 2-hexanone were detected in shallow groundwater samples from Plant B interceptor well IW-12 in the Supplemental Phase II Investigations.

- Higher concentrations of chromium were detected in a shallow groundwater sample from a new well, GW-78S, located along the upper South Ditch, than previously detected in CSA shallow groundwater samples from wells on-property. This change may also be a result of localized changes in groundwater flow directions resulting from construction of the weir (although a comparable monitoring well did not exist during the CSA).

The analytical data collected during the CSA (CRA, 1993) and the Supplemental Phase II Investigations indicate that the following major organic and inorganic compounds were frequently detected at significant concentrations in samples from the shallow and deep groundwater (excluding the dense layer).

Volatile Organic Compounds	Semivolatile Organic Compounds	Inorganics	
Acetone	Bis(2-ethylhexyl)phthalate	Ammonia	Potassium
2-Butanone (MEK)	Butyl Benzylphthalate	Calcium	Sodium
2-Hexanone	Di-n-octylphthalate	Chromium	Sulfate
Toluene	N-nitrosodiphenylamine		
Trimethylpentenes			

4.2.7 Dense Layer Characterization

The CSA (CRA, 1993) determined that a dense layer with high concentrations of inorganic constituents occupies bedrock lows in the deep portion of the sand and gravel aquifer. The additional investigations to determine the topography of the bedrock surface that controls dense layer migration are discussed in Section 2 and additional investigations to determine the lateral and vertical extent of the dense layer are discussed in Section 2.6. The additional rounds of dense layer sampling and geochemical evaluations performed as part of the Supplemental Phase II Investigations are discussed in Section 2.7.

The primary sources of the inorganic chemicals in the groundwater are generally considered to be the unlined pits: Lake Poly, East and West Pits, and Acid Pits which were located near the center of the facility. The liquid wastes disposed in these areas were acidic, and contained high concentrations of ammonia, calcium, chloride, chromium, sodium, and sulfate. Because of the high specific gravity (up to 1.11 grams per cubic centimeter [g/cm^3]), these liquid wastes sank through the shallow aquifer. The subsequent movement of the dense layer was primarily controlled by the shape of the bedrock (Plate 2-1).

The dense layer moved along the top of the bedrock towards the west and southwest within the Western Bedrock Valley that is located beneath the center of the property and has passed underneath Jewel Drive and the Altron facility. As liquid waste discharges continued at the property, the dense layer filled the bedrock lows beneath the Koch facility and near the GW-59 well cluster and some of the upper dense layer eventually spilled over the bedrock saddle north of the GW-62 well cluster. This portion of the dense layer continued to move west under the influence of gravity down slope along the bottom of the Western Bedrock Valley and accumulated in the bedrock low at the GW-83 well cluster. The Supplemental Phase II Investigations have shown the dense layer beneath the Maple Meadow Brook wetland is confined to the bedrock low at the GW-83 well cluster and does not extend to the north or west. As shown on Plate 2-10, the main body of the dense layer resides in the upper portion of the Western Bedrock Valley (east of Main Street) and is separate from the small, isolated pool of dense layer residing in the bedrock low at GW-83D.

Further characterization of the dense layer and the overlying shallow groundwater has shown that the contact is sharp and that the overlying shallow groundwater contains inorganic concentrations that are generally 100 to 1,000 times less than that found in the dense layer. The depth-discrete analytical data from the multilevel piezometers and the induction logs has shown that the dense layer itself is stratified. As would be expected, the highest concentrations of inorganics, particularly cationic metals such as chromium, are found at the bottom of bedrock in the lowest pH regions of the dense layer. Inorganic concentrations decrease toward the interface and are 10 to 1,000 times lower in the transition layer above the dense layer and in distal portions of the dense layer.

The concentration range of major inorganic constituents in the dense layer summarized below shows that the lowest concentration of chromium was detected in dense layer samples from the lower Western Bedrock Valley. The Supplemental Phase II Investigations described in Section 2.7 have shown that differences in reactivity of different inorganics with aquifer material has resulted in a separation of the mobile anions from less mobile cationic metals in areas downslope from the source areas. The concentrations of metal solutes including chromium, aluminum, iron, and manganese are limited by pH-dependent solubilities of metal hydroxide solids, and are therefore, largely confined to the low pH regions of the dense layer. The concentrations of anions including chloride and sulfate are not appreciably limited by solubility controls, hence their concentrations are affected primarily by dilution.

		Concentration Range		**Maximum Concentration**
Dense Layer		**Upper Western Bedrock Valley**		**Lower Western Bedrock Valley**
Major Inorganic Parameter	**Unit**	**Minimum**	**Maximum**	**at GW-83D**
pH	S.U.	3.4	5.6	5.02 (minimum)
Specific gravity	g/cm^3	1.01	1.11	Not available
Specific Conductivity	umhos/cm	15,410	102,600	38,600
Aluminum	mg/l	1.2	2,500	27.8
Ammonia	mg/l	2.4	12,000	1,650
Chloride	mg/l	1,100	17,000	4,690
Chromium	mg/l	4.4	4,700	0.087/0.131*
Iron	mg/l	5.5	3,400	1280/
Manganese	mg/l	9.3	620	46.2/
Sodium	mg/l	71.1	25,000	6,120/
Sulfate	mg/l	6,900	77,000	17,300
Total Dissolved Solids	mg/l	430	98,000	35,500

* Filtered/unfiltered

Organics are also detected in the dense layer at low concentrations. Based on historical records and the presence of higher concentrations of these organics in subsurface soil samples from the liquid waste disposal areas, it appears that these organics were components of the liquid wastes or were entrained in the dense layer as it flowed through areas with organics in the deep groundwater. The most frequently detected organics at consistently elevated concentrations are summarized below. Phenol and acetone were detected at relatively higher concentrations

(approximately 1 milligram per liter [mg/l]) in most dense layer wells which may be due to their greater tendency to dissolve in water and/or higher initial concentrations in disposal areas. Inside the dense layer, the distribution both of inorganics and organics are controlled by the shape of the bedrock valleys and the resulting gravity-driven flow of dense layer from source areas to its current resting place in bedrock lows.

Major Dense Layer VOCs Volatile Organic Compounds		Major Dense Layer SVOCs Semivolatile Organic Compounds	
Parameter	**Maximum Concentration (ppm)**	**Parameter**	**Maximum Concentration (ppm)**
Acetone	0.92	Benzoic Acid	0.061
Bromoform	0.75	Bis(2-ethylhexyl)phthalate	0.18
2-Butanone (MEK)	0.645	4-bromophenyl-phenylether	0.034
2-Hexanone	0.92	Napthalene	0.088
Toluene	0.17	Phenol	1.6
Trimethylpentenes	0.168	N-Nitrosodiphenylamine	0.008

4.3 NATURE AND EXTENT OF MAJOR CHEMICALS

4.3.1 Identification of Major Chemicals

Under the MADEP Risk Assessment Guidelines, over 140 organic and inorganic compounds could be considered chemicals of concern based on frequency of detections and background concentrations. In order to provide a more meaningful context of identifying areas of concern, the results of the CSA (CRA, 1993) and the Supplemental Phase II Investigations were reviewed to identify the frequently detected parameters in each media and the parameters that are consistently detected in most media. The parameters selected to define the major areas of concern are listed below. The occurrence and distribution of other parameters are presented in detail in the CSA (CRA, 1993) and are only briefly summarized in the following sections. The Phase II Human Health Risk Assessment presented in Appendix R considered all chemicals detected through laboratory analyses.

Selected Major Chemicals

Inorganics	**Volatiles**
Ammonia	Acetone
Chloride	Total Trimethylpentenes
Chromium	Total Volatiles
Sulfate	
Semivolatiles	**Pesticides**
N-nitrosodiphenylamine	Total Pesticides
Phenols	
Total Phthalates	

The major inorganic compounds listed above were selected because they are frequently detected at high concentrations in all media and they have served as indicator parameters for the dense layer. Likewise, the SVOCs selected; NNDPA, phenol, and total phthalates, were also detected at consistently higher concentrations in most media. Bis(2-ethylhexyl)phthalate is the predominate phthalate detected in most media. However, to show the distribution of all phthalates, the phthalates were summed for each sample. Phenols were selected to represent the acid extractable compounds that generally have a higher mobility in the environment than the other SVOCs.

VOCs are generally detected at low concentrations in all media, with the exception of acetone and trimethylpentenes (TMPs) which were selected because they were detected at elevated concentrations in most media. The other VOCs detected were summed for each sample (excluding acetone and TMPs) to show their distribution.

Pesticides were frequently detected in surface soils and sediments and were generally present at low to very low concentrations in all media. Pesticides including aldrin, alpha-BHC, gamma-BHC (lindane), 4,4'-DDT, endosulfan II, and toxaphene were detected most consistently in the surface soils and sediments and are sporadically detected in the subsurface soils, surface water, and shallow groundwater. Since no single pesticide predominated in all media the pesticides were summed for each sample to show the distribution of all pesticides.

4.3.2 Presentation of Major Chemicals

The distribution of these selected parameters in all media, based on the results of the CSA (CRA, 1993) and the Supplemental Phase II Investigations, is presented on the maps in Plates 4-1 through 4-15. The following sections discuss the distribution of the major chemicals in all media. For each major chemical, potential source areas are first identified by reviewing the soil and waste sample chemical results for locations with maximum contaminant concentrations. Then each section evaluates the extent and distribution of each chemical in the other media (dense layer, groundwater, surface water and sediments) and assesses whether these media serve as potential pathways or receptors to the chemical.

Since there were multiple rounds of sampling and analysis at the same location, the following general guidelines were used to depict the distribution of the selected parameters in each media. For composite surface soil samples, the most recent analytical results were used to re-characterize the CSA (CRA, 1993) grid areas or to characterize specific subareas within the CSA (CRA, 1993) grid areas. For grab surface soil samples, subsurface soil, and sediment samples, the maximum analytical result was used at each sampling location.

The deep groundwater isoconcentration contours are also based on the maximum concentration that was consistently detected at wells that monitor the base of the aquifer. Since the highest concentrations of chemicals in the deep groundwater are mainly associated with the dense layer which is restricted to the bedrock lows, the deep groundwater isoconcentration contours considered the shape of the bedrock surface (Plate 2-1). The MADEP concerns for the

characterization of deep groundwater pathways are addressed by examining the distribution of chemicals in deep groundwater and the dense layer together, although the reader should note that different mechanisms control the potential transport of contaminants. In the dense layer, gravity flow is the main potential transport mechanism. In the deep groundwater (not the dense layer), advective flow is the main potential transport mechanism, although some component of gravity flow may exist to varying degrees in the deep groundwater with high TDS concentrations.

The shallow groundwater isoconcentration contours and surface water concentrations presented on Plates 4-1 through 4-15 are generally based on maximum concentration detected, but also reflect the most recent analytical results at each sampling location and the flow direction of shallow groundwater determined from the groundwater contour maps (Plates 2-3 and 2-5). As discussed in Section 2.4.1, the shallow groundwater and surface water quality in the vicinity of the Off-Property West Ditch and upper South Ditch has generally improved since the installation of the weir due to the influx of clean ponded water from the Off-Property West Ditch into the groundwater flow system. Because the weir is the result of an IRA and the changes in surface water quality can be viewed as a temporary condition, the surface water concentrations presented on Plates 4-5 through 4-15 show the maximum concentration detected at both CSA (CRA, 1993) and Supplemental Phase II Investigation sampling locations. For the shallow groundwater isoconcentration contours, the CSA (CRA, 1993) maximum concentrations were used at most of the wells, while the most recent analytical results for wells along the Off-Property West Ditch, upper South Ditch, and downgradient areas were used to depict the extent of chemicals that may have changed due to the influx of clean ponded water from the portion of the Ditch System affected by the weir.

4.3.3 Inorganics

Inorganic compounds are the main class of contaminants associated with the site. Ammonia, chloride, chromium, and sulfate are found at elevated concentrations in most media on-property and in the sediment, surface water, and groundwater off-property. The distribution of other inorganics such as aluminum, calcium, iron, potassium, and sodium are similar to the selected inorganics and are discussed in the CSA (CRA, 1993). The geochemistry of the ammonia, chloride, chromium, sulfate, and other major inorganics are discussed in Section 2.7.

The processes and the waste by-products which contaminated soils and produced the inorganic contamination in groundwater are summarized on Table 4-14. Aluminum, ammonia, calcium, chloride, sodium, and sulfate were production byproducts throughout the active life of the facility, although a waste water treatment system completed in 1972 neutralized acidic sulfate-bearing effluent with lime. The resulting calcium sulfate slurry was impounded in the Lined Lagoons and later disposed of in the Calcium Sulfate Landfill. The chromium originated from sodium dichromate which was used until 1967 in the Kempore® process. Based on historical documents and analytical results, the acidic waste containing chromium sulfate was discharged to Lake Poly and to the three former acid pits north of the South Ditch. The historical records and the

distribution of chromium and other parameters discussed below also suggests that the two former unlined pits, East and West Pits, located beneath the present East and West Warehouses, also contributed to the inorganic contamination through liquid waste disposal.

Based on the available knowledge of chemicals which were discharged into the unlined pits , these disposal areas are considered to be the primary source of the inorganic contaminants in the shallow and deep groundwater, and the inorganics found in the sediments and surface water on- and off-property. In general, the on-property areas with high concentrations of inorganics in surface soils, subsurface soils, sediments, surface water, and the shallow groundwater coincide with the former location of Lake Poly, other unlined pits, and lagoons. The shallow groundwater sample results from wells around the former lagoons show a disproportionate enrichment of calcium and sulfate and may indicate that as rainwater infiltrates site soils, it dissolves calcium, sulfate, and other inorganics (not chromium) from residual lagoon material.

Plates 4-1 through 4-4 show the isoconcentration contours for ammonia, chloride, chromium, and sulfate in the shallow and deep groundwater at the site. The highest concentrations of ammonia, chloride, chromium, and sulfate shown on these figures are consistently observed in the dense layer which occupies the bedrock lows, both on-property and off-property. The dense layer comprises the same inorganic compounds discharged into the unlined pits which have flowed along bedrock surface to its current position. The dense layer begins in the center of the facility and extends to just past the property boundary to the east, and down the Western Bedrock Valley beneath Maple Meadow Brook wetlands to the west.

In the shallow groundwater the highest concentrations of inorganics are on-property and immediately west of the property. Further characterization of the shallow groundwater and surface water has shown that the desorption and flushing of residual sources contributes inorganics to these media in these areas. The data suggest that these residual sources include (1) a geochemical reaction zone at the interface between the acidic groundwater and near-neutral surface water and (2) a residual soil contamination in the vicinity of the unlined pits and in places in the aquifer that accumulated inorganic constituents and their reaction products. The higher concentrations of sulfate, and to a lesser degree ammonia, chloride, and sodium, are detected in the shallow groundwater in the vicinity of or downgradient of Lake Poly, the Lined Lagoons, the Acid Pits, and the Calcium-Sulfate Landfill. These inorganics appear to be mainly associated with residual soil contamination in these areas, although the dense layer, where present, may also be contributing some inorganics to the shallow groundwater discharge areas along the Ditch System. The shape of the shallow groundwater isoconcentration contours generally corresponds to the shallow groundwater flow directions and the areas with higher concentrations of inorganics coincide with the location of former liquid waste disposal areas and the shallow groundwater discharge areas along the Ditch System. The distribution of the major inorganic chemicals in all media is discussed below.

4.3.3.1 Ammonia

Potential Sources of Ammonia

Ammonia was used as a raw material during the period of active production from 1953 through 1986. Waste byproducts such as ammonium chloride and ammonium sulfate were discharged to Lake Poly and other unlined pits from 1953 until 1972 when the waste water treatment plant and Lined Lagoons were constructed. The dense layer formed from these discharges and now contains the highest concentrations of ammonia detected in the subsurface. The results of geochemical investigations show that the pH conditions in the dense layer, groundwater, and surface water cause most ammonia to occur as a dissolved ammonium ion and not the volatile form of ammonia. Ammonia is also a naturally occurring compound and is present as chemical and biological products of the nitrogen cycle in the aquifer and the associated recharge/discharge areas. In wetlands subject to periodic flooding, ammonia concentrations in soils increase after flooding due to oxygen depletion and nitrate reduction (Mitsch and Gosselink, 1993). The maximum background concentration of ammonia in the groundwater is 4 mg/l. Ammonia was not detected in the background soil, sediment, and surface water samples.

Distribution of Ammonia

As discussed in Section 2, additional ammonia analyses since the CSA (CRA, 1993) mainly included the surface water, shallow groundwater, dense layer, and surface soil samples, although additional subsurface soil samples from Lake Poly and one additional flocculent sample were also analyzed for ammonia. Based on the data from the CSA (CRA, 1993) and the Supplemental Phase II Investigations, the distribution of ammonia in the shallow and deep groundwater at the site is presented on Plate 4-1. The distribution of ammonia in surface soil, sediment, surface water and groundwater on-property and in adjacent areas is presented on Plate 4-5. The distribution of ammonia in surface soils, subsurface soils, wastes, and the dense layer are presented below first to identify the major source areas, then the distribution of ammonia in the groundwater, surface water, and sediments are discussed to show the potential impacts to these receptors or pathways.

The ammonia concentration in additional surface soil samples collected in the Supplemental Phase II Investigations shown on Plate 4-5 are consistent with the results of the CSA (CRA, 1993). The surface soil results show that the highest concentrations of ammonia were detected in samples from production areas, disposal areas, and the adjacent low lying areas. The highest concentrations of ammonia were detected in surface soil samples from:

- an area with stained-soils near the upper On-Property West Ditch (SWMU 27),
- an area with stained-soils east of Plant D (SWMU 26),
- an area along the banks of the upper Ephemeral Ditch (SWMU 33),

- an area with stained-soils along the banks of the upper South Ditch (SWMU 30), and

- the vicinity of the South Ditch and Central Pond (grid Areas 1, 8, and 9).

The elevated ammonia concentration in these areas is believed to be related to releases from manufacturing and disposal activities. The elevated concentrations of ammonia detected in samples from the upper On-Property West Ditch (SWMU 27), along the banks of the upper Ephemeral Ditch (SWMU 33), and along the banks of the upper South Ditch (SWMU 30) may be related to contaminated sediment which was deposited along the banks of the Ditch System at these locations.

In wastes and subsurface soils the highest concentrations of ammonia were previously detected in CSA samples from the Drum Area A (TP-6, 2,100 mg/kg), the Calcium-Sulfate Landfill, the Buried Debris Area, Plant D and the black area east of Plant D (SWMU 26), the Acid Pits and Lined Lagoons, and Drum Area A. As shown below, the Supplemental Phase II Investigations at Lake Poly detected the highest concentrations of ammonia in subsurface soils. As shown on Plate 4-5, the shallow groundwater concentrations are also elevated around Lake Poly, Drum Area A, the Calcium-Sulfate Landfill, Plant D and the black area east of Plant D (SWMU 26), and Plant B. These data suggest that subsurface soils in these areas may be contributing ammonia to the groundwater in these areas.

Media	Location	Maximum Concentration (ppm)
Subsurface Soils	Lake Poly (LPB-1)	10,000
	Calcium-Sulfate Landfill (TP1-SN1)	640 J
	Buried Debris Area (TP19-SN6)	490
	Plant D (BH-31)	400 J
	Lake Poly (LPB-2)	390
	Lake Poly (BH-17)	360
	Lined Lagoon/Acid Pits (BH-1)	210 J
	Drum Area B (TP21-SN8)	0.047

As shown on Plates 4-1 and 4-5, the greatest current potential source of ammonia at the site is deep groundwater and dense layer. The distribution of ammonia concentrations in the deep groundwater shown on these figures is mainly controlled by the position of the dense layer which occupies the bedrock lows on-property and off-property. The ammonia concentrations detected in dense layer samples collected during the Supplemental Phase II Investigations are similar to those reported in the CSA (CRA, 1993). However, as discussed in Section 2.7.4, the multilevel piezometer results show that ammonia concentrations vary and do not show a consistent increase with depth within the dense layer, unlike other dense layer inorganic constituents such as chloride, chromium, sulfate, and sodium. Variable ammonia results were also observed in the CSA sampling events and may be due to matrix interferences from high solute concentrations in the

dense layer. As shown on Table 4-14, these variations of ammonia concentrations in the dense layer may also be a result of the variations in: the concentrations of ammonia in waste byproducts generated over time, the different chemical processes, and the timing and volume of discharge into the unlined pits.

Outside of the dense layer, the ammonia concentrations in the deep groundwater are elevated near Plant B, the area east of Plant D (SWMU 26), the Calcium-Sulfate Landfill, at GW-50D, and within the lower Western Bedrock Valley (GW-84D-GW-87D). The elevated ammonia concentrations in the deep groundwater near Plant B and Plant D and the Calcium-Sulfate Landfill coincide with elevated ammonia concentrations detected in subsurface soil samples in these areas. The elevated concentrations at GW-50D may be related to the flow of deeper groundwater (with higher ammonia concentrations) over the bedrock saddle which separates the Western Bedrock Valley from the Eastern Bedrock Valley. The elevated ammonia concentrations in the lower Western Bedrock Valley are also attributed to the flow of deeper groundwater (with higher ammonia concentrations) over a bedrock saddle which separates the bedrock low at GW-83D from the lower Western Bedrock Valley.

The areas with high concentrations of ammonia in the shallow groundwater closely coincide with the bedrock lows occupied by the dense layer, extending to the west just past the Olin property and dropping off sharply at the East Ditch. These areas also generally correspond to the groundwater discharge areas. As discussed in Section 2.7, the groundwater results suggest that ammonia diffuses out of the dense layer faster than other dense layer constituents. The shallow groundwater quality results on-property show detections of ammonia at 100 mg/l to 1,000 mg/l above the dense layer and detections of 10 mg/l to 100 mg/l near other source areas outside of the dense layer.

As shown below, ammonia concentrations have generally decreased over time in the shallow and deep groundwater on-property and appear to have increased over time in the shallow groundwater immediately west of the property. As discussed in Section 2.4.1, additional groundwater data collected since the CSA (CRA, 1993) has shown that the shallow groundwater water quality in the Off-Property West Ditch has generally improved since the installation of the weir, due to the clean ponded water entering the groundwater flow system and continued flushing of the aquifer from infiltration of precipitation. The increased heads upstream of the weir and the resulting changes in shallow groundwater flow directions may be increasing ammonia concentrations in the shallow groundwater in a localized area northwest of the Off-Property West Ditch.

		Ammonia Maximum Concentration (ppm)		
Media	**Selected Wells**	**Pre-CSA**	**CSA**	**Post-CSA**
Shallow	Western portion of property (GW-10S)	140	5.5	7
Groundwater	Western portion of property (GW-11)	60,000	180	170
On-property	Central portion of property (GW-35S)	160	180	91
On-property	Easter portion of property (GW-50S)	66	150	140
Shallow	Altron facility (well B1)	No data	40	60.7
Groundwater	North off-property W. Ditch (GW-43S)	20	22	120
Off-property	Central off-property W. Ditch (GW-42S)	8.6	6.1	8.7

Similar to the groundwater, the historical results for ammonia concentrations in the surface water have also decreased by a factor of 10 to 100 times. As discussed in Section 2.4.1, additional surface water data collected since the CSA (CRA, 1993) has shown that the surface water quality in the Off-Property West Ditch has generally improved since the installation of the weir, due to the effect of clean ponded water entering the groundwater flow system and the resulting localized changes in shallow groundwater discharge points. As shown below, ammonia concentrations have decreased by an order of magnitude since the CSA (CRA, 1993) along the Off-Property West Ditch and lower South Ditch, but have increased along the upper South Ditch below the weir. The decreases reflect the influx of cleaner surface water and ponding behind the weir, while the increases are probably related to localized changes in groundwater flow directions brought about by the weir.

Media	**Location**	**Ammonia Maximum Concentration (ppm)**	
		CSA/Pre-Weir	Post-Weir
Surface Water	Off-Property West Ditch	380 (SW-16)	35 (SW-11-95)
	Upper South Ditch	50 (SW-11)	170 j (SW-17-95)
	Lower South Ditch	160 (SW-6)	28 (SW-15-95)

Some of the variations in ammonia concentrations may also be explained by temporal changes in shallow groundwater discharge points. As discussed in Section 2.7.6 and shown on Figure 2-95 the ammonia concentrations in surface water (SW-16-95 and SW-17-95) and groundwater (GW-78S and GW-79S) between the weir and the Central Pond increase in the drier months (July and October). Ammonia concentrations in surface water (SW-15-95) and groundwater (GW-50S) samples collected from the lower South Ditch show higher concentrations in the wetter months (February and May). These temporal changes in ammonia concentration may indicate that deeper groundwater discharges in the vicinity of the lower South Ditch, particularly during drier months. There is no clear seasonal pattern in ammonia concentrations in surface water samples from the Off-Property West Ditch, however generally higher concentrations of ammonia were detected in surface water and shallow groundwater samples (SW-11-95 and GW-79S) collected from the north Off-Property West Ditch in October 1995.

The ammonia concentration in additional sediment samples collected in the Supplemental Phase II Investigations shown on Plate 4-5 are generally consistent with the results of the CSA (CRA, 1993). The results of one additional sediment sample, SED-17/11 (composited from the flocculent collected from the Off-Property West Ditch and upper South Ditch) contained higher ammonia concentrations (2,300 mg/kg) than detected previously in the sediment. The other sediment results show that elevated ammonia concentrations are detected downstream of the former production and disposal areas, the East Ditch adjacent to Plant B and Plant D, the upper On-Property West Ditch, and the South Ditch. The elevated ammonia concentrations in sediment samples from the Off-Property West Ditch are not adjacent to areas with elevated ammonia in surface soils, but do correspond to sample locations with elevated ammonia concentrations in the surface water. This suggests that some of the ammonia in sediments may be due to the discharge of shallow groundwater. The ammonia concentrations in samples from the downstream sections of the East Ditch are relatively higher than concentrations in samples from the upstream East Ditch and lower South Ditch. This suggests that the downstream portions of the East Ditch may have accumulated ammonia-bearing sediment from historical releases.

4.3.3.2 Chloride

Potential Sources of Chloride

Chloride in the form of aluminum chloride, ammonium chloride, hydrochloric acid, sodium chlorate, sodium chloride, and chlorine was used as a raw material in the various chemical manufacturing processes during the period of active production from 1953 through 1986. Waste byproducts containing ammonium chloride, hydrochloric acid, and sodium chloride were discharged to Lake Poly and/or other unlined pits until 1972 when the waste water treatment plant and Lined Lagoons were constructed. Other known sources of chloride in the study area include the National Pollutant Discharge Elimination System (NPDES) surface water discharge (typically chloride concentration of approximately 200 mg/l) into the On-Property West Ditch from the Plant B groundwater treatment system (which uses sodium hypochlorite), and non-point source contributions from road salt application. Background chloride concentrations in surface water and groundwater are 110 mg/kg and 235 mg/kg, respectively. Chloride was not detected in the background soil and sediment samples.

Distribution of Chloride

As discussed in Section 2, additional data on chloride obtained since the CSA (CRA, 1993) was collected from the surface water, shallow groundwater, and dense layer media, although additional subsurface soil samples from Lake Poly and one flocculent sample were also analyzed for chloride. Based on the data from the CSA (CRA, 1993) and the Supplemental Phase II Investigations, the distribution of chloride in the shallow and deep groundwater at the site is presented on Plate 4-2. The distribution of chloride in surface soil, sediment, surface water and groundwater on-property and in adjacent areas is presented on Plate 4-6. The distribution of

chloride in surface soils, subsurface soils, wastes, and the dense layer is discussed below to first identify the major source areas, then the distribution of chloride in the groundwater, surface water, and sediments are discussed to show the potential impacts to these receptors or pathways.

In subsurface soils and waste media, the CSA (CRA, 1993) sampling detected the highest concentrations of chloride in waste material samples from Drum Area A and in subsurface soil samples from the Calcium-Sulfate Landfill, the Buried Debris Area, Drum Area B, the area east of Plant D (SWMU 26), Lake Poly, and from near the Acid Pits and Lined Lagoons. The subsurface soil samples collected from Lake Poly in the Supplemental Phase II Investigations show similar chloride concentrations to those detected in the CSA (CRA, 1993).

No additional surface soil samples were analyzed for chloride as part of the Supplemental Phase II Investigations. The CSA (CRA, 1993) surface soil results show that the highest concentrations of chloride were detected in samples from along the banks of the upper Ephemeral Ditch (SWMU 33), from the black area along the banks of the upper South Ditch (SWMU 30), and in grid Area 8. The elevated concentrations of chloride along the banks of the upper Ephemeral Ditch (SWMU 33) and from the area along the banks of the upper South Ditch (SWMU 30) are believed to be related to contaminated sediment which was deposited along the banks of the Ditch System at these locations.

As shown on Plate 4-2 and Plate 4-6, the deep groundwater and dense layer are the greatest current source of chloride at the site. The distribution of chloride concentrations in the deep groundwater is mainly controlled by the position of the dense layer which occupies the bedrock lows both on and off the Olin property. As shown below, the maximum chloride concentrations detected in dense layer samples collected from the multilevel piezometers in the Supplemental Phase II Investigations are similar to those reported in the CSA (CRA, 1993). As discussed in Section 2.7.4, the improved vertical resolution afforded by the multilevel piezometers shows that chloride concentrations increase with depth within the dense layer. In addition, the results of the Supplemental Phase II Investigations show that chloride concentrations in the dense layer decrease with distance from the source areas due to dilution and longitudinal dispersion (Section 5.1.2).

Media	Location	Chloride Maximum Concentration (ppm)	
		CSA (CRA, 1993)	Supp. Phase II
Dense Layer	Multilevel Piezometer MP-1, Port 1	not installed	17,000
	Multilevel Piezometer MP-2, Port 1	not installed	17,000
	Multilevel Piezometer MP-3, Port 1	not installed	12,000
	GW-42D (near MP-2)	19,000	24,000
	GW-44D (near MP-3)	12,000	13,000
	GW-59D	9,600	not tested
	GW-83D	not installed	4,690

Outside of the dense layer, the chloride concentrations in the deep groundwater are elevated near the former Lined Lagoons, Acid Pits, and the Calcium-Sulfate Landfill, near the lower Ephemeral Drainage at GW-50D, and within the lower Western Bedrock Valley (GW-84D - GW-87D). The elevated chloride concentrations in the deep groundwater at the Lined Lagoons and Acid Pits and Calcium-Sulfate Landfill coincide with elevated chloride concentrations detected in subsurface soil samples. As discussed previously, the elevated concentrations at the GW-50D may be related to the flow of deeper groundwater (with higher chloride concentrations) over the bedrock ridge which separates the Western Bedrock Valley from the Eastern Bedrock Valley. The elevated chloride concentrations in the lower Western Bedrock Valley are also attributed to the flow of deeper groundwater (with higher chloride concentrations) over a bedrock ridge which separates the dense layer in the bedrock low at GW-83D from the lower Western Bedrock Valley.

In the shallow groundwater the areas with high concentrations of chloride closely coincide with the shallow groundwater discharge areas along the Ditch System. In general, the chloride concentrations in the shallow groundwater wells adjacent to the Off-Property West Ditch are lower than those detected in wells adjacent to the lower South Ditch. As shown on Plate 4-6, the areas with shallow groundwater concentrations exceeding the secondary maximum contaminant limit (MCL) are largely on-property, with the exception of a small area around the Off-Property West Ditch. In this area at MP-2, the chloride results for the upper ports (ports 14-17) are higher than the intermediate ports (ports 7-13) which may reflect another shallow source of chloride. The other multilevel piezometers show that the concentration of chloride decreases with depth to the top of the dense layer and then increases with depth through the dense layer. In addition, the results of the surface water studies (Section 2.7.6) show that the concentrations of chloride in surface water do not correlate well with the concentrations of sulfate. These results also suggest that the chloride observed in the shallow groundwater is not derived from the dense layer and is from some other source.

In general, the surface water results show that the consistently highest chloride concentrations have been detected in samples collected from the On-Property West Ditch and the Off-Property West Ditch at locations SW-15, SW-16, and SW-17. The chloride concentrations in the surface water samples collected from the On-Property West Ditch and sampling locations downstream may be attributed to the NPDES discharge upstream. The highest chloride concentration of 590 mg/l was detected in the surface water sample collected at SW-16 in September 1992; however the chloride concentration at this location dropped to 200 mg/l for the sample collected in December 1992. As shown below, the installation of the weir has not appreciably changed chloride concentrations in the Off-Property West Ditch and South Ditch, although the chloride concentrations in shallow groundwater wells around the Ditch System have decreased considerably. This also suggests that there is another source of chloride controlling surface water chloride concentrations.

Media	Location	Chloride Maximum Concentration (ppm)	
		CSA/Pre-Weir	Post-Weir
Surface Water	Off-Property West Ditch	590/200 (SW-16)	190 (SW-12-95)
	Upper South Ditch	190 (SW-11)	160 (SW-17-95)
	Lower South Ditch	170 (SW-6)	130 (SW-15-95)

The surface water results for sampling locations along the East Ditch and South Ditch also show elevated chloride at some locations, however the results of multiple rounds show a good deal of variation in chloride concentrations at the same sampling location. The variation of chloride concentrations may be due to road salt application or temporal differences in groundwater discharge. For example, the results for sampling locations SW-1 and SW-2 along the East Ditch in August and November 1992 were approximately 50 mg/l, while the results for December 1992 were approximately 200 mg/l.

The sediment results show that the highest concentrations of chloride were detected in samples from the Off-Property West Ditch (1,400 mg/kg at SW-16). The flocculent sample, Floc RP-2, collected adjacent to SW-16 as part of the Supplemental Phase II Investigations, shows lower chloride concentrations (210 mg/kg) than detected in the CSA (CRA, 1993) and is attributed to variability in chloride concentrations in the sediment. The next highest concentrations of chloride between 100 mg/kg and 300 mg/kg were detected in sediment samples from the On-Property West Ditch, the South Ditch, and the downstream sample locations along the East Ditch.

4.3.3.3 Chromium

Potential Sources and Types of Chromium

Chromium in the form of sodium dichromate was used as a raw material in the Kempore® process during the period of active production from 1956 through 1967 (CRA, 1993). During this time period, waste byproducts containing chromium sulfate were discharged to Lake Poly and other unlined pits up until 1967 when sodium chlorate was substituted for sodium dichromate in the chemical manufacturing processes. Chromium is a naturally occurring element and is present in common accessory minerals in the overburden and bedrock in the study area. Background chromium concentrations for soils and sediment are 16 mg/kg and 1.2 mg/kg, respectively. Chromium was not detected in background surface water and groundwater samples.

Chromium can be present in the environment in two valence states, as chromium(III) [trivalent chromium] and chromium(VI) [hexavalent chromium]. As discussed in Section 2.7.2.1.2, previous groundwater analytical results indicating the presence of chromium(VI) were probably in error because of matrix interferences in the laboratory method (Method 6010). Laboratory analyses for chromium(VI) performed using a different analytical method (Method 7196) in the Supplemental Phase II Investigations and geochemical modeling have shown that chromium(III), not chromium(VI), is present in water samples at the site.

Because chromium(VI) in soils poses a greater potential risk to humans than does chromium (III), such risk could be more accurately assessed by quantifying the proportion of total chromium in site soils that occurs as chromium(VI). Unfortunately, no analytical method is available to reliably measure chromium(VI) in soils. The Environmental Protection Agency recommends digesting soil samples with a sodium carbonate–sodium hydroxide extraction (EPA Method 3060). However, James et al. (1995) reported that chromium(VI) spikes added to anoxic sediments using this method were consistently unrecoverable (i.e., 0 percent spike recovery), and spike recoveries from other soil substrates depended on soil type and the mineralogy of the hexavalent chromium spike. This study also suggested that soils containing high concentrations of manganese oxide may form chromium(VI) during the extraction procedure, potentially causing false detection of chromium(VI). In addition, the former chemical manufacturing processes were likely to have oxidized the chromium(VI) in raw materials to chromium(III).

Several sediment and site soil samples were analyzed for chromium(VI) using EPA Method 3060 during the CSA. For the reasons described above, analytical results obtained by this method cannot be considered reliable. Furthermore, no acceptable alternative analytical method was identified during this investigation. Therefore, no soil or sediment samples were analyzed for chromium(VI) during the Supplemental Phase II Investigations. Due to the lack of confirming data, the risk characterization presented in Appendices R and S take a conservative approach and use the available hexavalent chromium results, but these results are not discussed below because they are not considered to be representative of site conditions.

Distribution of Chromium

As discussed in Section 2, one of the main focuses of the Supplemental Phase II Investigations was to determine the geochemistry and fate of chromium in the surface water, flocculent, and ground water. The Supplemental Phase II Investigations also further characterized the distribution of chromium in subsurface and surface soils in selected areas. Based on the data from the CSA (CRA, 1993) and the Supplemental Phase II Investigations, the distribution of chromium in the shallow and deep groundwater at the site is presented on Plate 4-3. The updated distribution of chromium in surface soil, sediment, surface water and groundwater on-property and in adjacent areas is presented on Plate 4-7. The distribution of chromium in surface soils, subsurface soils, wastes, and the dense layer are discussed below to first identify the major source areas, then the distribution of chromium in the groundwater, surface water, and sediments are discussed to show the potential impacts to these receptors or pathways.

The surface soil results show that the highest concentrations of chromium were detected in samples adjacent to the On-Property West Ditch and upper South Ditch. The elevated chromium concentrations in surface soils adjacent to the upper On-Property West Ditch (SWMU 27) are attributed to historical overflows from Lake Poly. The elevated concentrations of chloride along the banks of the upper Ephemeral Ditch (SWMU 33) and the upper South Ditch (SWMU 30) are believed to be related to contaminated sediment which was deposited along the banks of the Ditch System at these locations. The Supplemental Phase II surface soil results show that the low-lying

drainage areas around the Central Pond and the Central Wetlands also have elevated chromium concentrations. The highest concentrations of chromium were detected in samples collected from the northern portion of the Central Pond Drainage Areas, and these results suggest that these areas received chromium-bearing overflows from the former Acid Pits via a former drainage ditch and/or via overflows along the ground surface.

In subsurface soil and waste media, the highest concentration of chromium, 17,000 mg/kg, was detected in a soil sample collected from a depth of 6 to 8 feet in the former Lake Poly. The nine subsurface soils with the highest levels of chromium were collected in the area of Lake Poly. The next highest chromium concentrations are found in subsurface soils downgradient of the East and West Pits and former Acid Pits. Elevated chromium concentrations were also detected in subsurface soils from Drum Area B and the Buried Debris Area and were detected in waste samples from both Drum Areas A and B.

As shown on Plate 4-3 and Plate 4-7, the dense layer is the greatest potential source of chromium at the site. In the deep groundwater, high concentrations of chromium are only found in samples from the dense layer. As discussed in Section 2.7, the concentration of chromium in the groundwater is limited by pH-dependent solubilities of metal oxyhydroxide solids, and is therefore largely confined to the low pH regions of the dense layer. The deep groundwater isoconcentration contours on Plate 4-3 show that in the deep groundwater, the highest chromium concentrations occur within the lowest pH portions of the dense layer at MP-2 and then decrease down the bedrock valley. As shown below, the maximum chromium concentrations detected in dense layer samples collected from the multilevel piezometers in the Supplemental Phase II Investigations are similar to those reported in the CSA (CRA, 1993). The improved vertical resolution afforded by the multilevel piezometers shows that chromium concentrations increase with depth and with lower pH within the dense layer. In addition, the results of the Supplemental Phase II Investigations show that as the dense layer migrates down the bedrock valleys, chromium concentrations decrease due to attenuation by precipitation and adsorption reactions (Section 2.7).

Media	Location	Dissolved Chromium Maximum Concentration (ppm)	
		CSA (CRA, 1993)	Supp. Phase II
Dense Layer	Multilevel Piezometer MP-1, Port 1	not installed	2,200
	Multilevel Piezometer MP-2, Port 1	not installed	4,700
	Multilevel Piezometer MP-3, Port 1	not installed	1,100
	GW-42D (near MP-2)	3,600	5,200
	GW-44D (near MP-3)	970	900
	GW-59D	32	not tested
	GW-83D	not installed	0.087

Concentrations of chromium in the shallow groundwater are 1,000 times less than in the dense layer. The areas with slightly elevated (< 1 mg/l) concentrations of chromium coincide with the shallow groundwater discharge areas along the Ditch System, the vicinity of Lake Poly, and well GW-54D. The elevated chromium concentrations in the shallow groundwater near Lake Poly coincide with elevated chromium concentrations detected in subsurface soil samples (Plate 4-7). The elevated chromium concentrations in the sample from GW-54D may be remnants of past discharges of chromium wastes to Lake Poly or may be related to leakage of former waste water lines in this area which discharged into Lake Poly. As shown on Figure 2-103, the chromium concentrations in shallow groundwater samples around the Off-Property West Ditch have dramatically decreased due to the effect of the ponded water from the weir on the shallow groundwater system.

As discussed in Section 2.7, the current distribution of chromium in the shallow groundwater and surface water in the vicinity of the Off-Property West Ditch and upper South Ditch is explained by the geochemical and discharge conditions. Geochemical modeling, aquifer material testing, and the multilevel piezometer results show that diffusion of chromium in the vertical direction is limited by precipitation and adsorption reactions, although low concentrations of chromium can enter the shallow groundwater system and then are seasonally discharged to the Ditch System. The higher concentration of chromium in the vicinity of the Central Pond (GW-7) is related to the lower pH (4.71) of shallow groundwater in this area which enables higher concentrations of chromium to remain in solution. In general, the increase in hydraulic heads and influx of clean water due to the weir have minimized the discharge of chromium-bearing shallow groundwater into the Ditch System above the weir, but may have moved the location of shallow groundwater discharge of chromium in the Ditch System to the upper South Ditch below the weir.

As discussed in Section 2.7, slight increases in pH (4.5 to 5.8) reduce the solubility of chromium and cause it to precipitate or partition to solid phase oxyhydroxides. Seasonal variations in pH at GW-42S shown on Figure 2-104 show that lower pH conditions are present in the wetter months (October through May) in the shallow groundwater at this location and that the lowest pH was detected in samples collected in February 1995 and 1996. The data from the shallow groundwater wells around the Ditch System generally do not show that decreases in pH result in increases in chromium concentrations. However, multilevel piezometer results show that pH and chromium concentrations are related in the shallow groundwater. These results suggest that the shallow groundwater seasonally discharges low concentrations of chromium to the surface water when increased hydraulic head and low pH conditions coexist.

In general, prior to weir installation, the surface water results showed that the consistently highest chromium concentrations were detected in samples collected from the Off-Property West Ditch and upper South Ditch. Generally lower concentrations were detected in the lower South Ditch and downstream sampling points along the East Ditch. Elevated chromium concentrations were also detected in the upper East Ditch at SW-1 and SW-2 during the December 1991 sampling event, but chromium was not detected at these locations in the two subsequent sampling events in August and November 1992. Additional surface water data collected since the CSA (CRA, 1993)

listed below shows that the chromium concentrations in the Off-Property West Ditch and South Ditch have greatly decreased since the installation of the weir. This reduction in chromium concentrations is attributed to continued flushing of the aquifer from infiltration of precipitation and to the near neutral pH, clean ponded water entering the groundwater flow system, suppressing shallow groundwater discharge, and driving attenuation reactions.

Media	Location (see Plate 4-7 for sample locations)	Chromium Maximum Concentration (ppm)	
		CSA/Pre-Weir	Post Weir
Surface Water	North Off-Property West Ditch	0.55	- -
	Central Off-Property West Ditch	12	- -
	South Off-Property West Ditch	11	- -
	Upper South Ditch above Weir	1.2	* *
	Upper South Ditch below Weir	0.52	0.016
	Central South Ditch at Pond	0.72	* *
	Lower South Ditch	0.39	0.023
	Lower Ephemeral Drainage	0.13	- -
	- - Not detected * * Not tested		

The resulting localized changes in shallow groundwater discharge points, temporal changes in pH conditions in shallow groundwater and surface water, and the configuration of the bedrock surface may be responsible for continued detections of chromium in the South Ditch below the weir at SW-17-95 in February 1995 and at SW-15-95 in May 1995. As shown on Figures 2-97 and 2-102, consistently lower pH and higher chromium concentrations were detected in shallow groundwater samples where the bedrock approaches the surface at GW-77S and GW-78S. The apparent continued periodic discharge of chromium-bearing shallow groundwater at these locations may be related to the shape of the bedrock surface which rises east and west of this location (Plate 2-1) which would tend to create upward groundwater gradients in this area.

The distribution of chromium in sediments presented on Plate 4-7 suggests that the source of chromium is the discharge of chromium-bearing wastes into the On-Property West Ditch and South Ditch, and the discharge of chromium contaminated groundwater and precipitation of a chromium-bearing flocculent. The laboratory analysis of flocculent (Tables 2-50 and 2-51) from the Off-Property West Ditch shows that it is composed of many of the same parameters that are found in the shallow groundwater and dense layer, which suggests that the origin of flocculent is the result of groundwater discharge to the Ditch System. The total chromium concentrations detected in the flocculent in the Supplemental Phase II Investigations are shown on Figure 2-107. These results show that the highest concentrations of chromium occur in the On-Property West Ditch and upper South Ditch and generally decrease downstream. The higher chromium concentrations along the lower South Ditch (FLOC-3 and FLOC-5) are attributed to accumulation of chromium-bearing sediments in areas of lower gradient and slower surface water velocity.

The distribution of chromium concentrations in the sediments appears to be controlled by the geochemical interactions between shallow groundwater and surface water and by the transport and re-deposition of chromium-bearing sediments. The elevated concentrations of chromium in samples from the downstream sections of the East Ditch compared to the lower concentrations in samples from the upstream East Ditch and lower South Ditch suggests that the downstream portions of the East Ditch have accumulated sediment from past discharges of chromium-bearing sediment.

4.3.3.4 Sulfate

Potential Sources of Sulfate

Sulfuric acid was used as a raw material during the period of active production from 1953 through 1986. Waste byproducts such as sulfuric acid, chromium sulfate, sodium sulfate, and ammonium sulfate were discharged to Lake Poly and other unlined pits from 1953 until 1972 when the waste water treatment plant began neutralizing the acidic sulfate-bearing effluent with lime. The resulting calcium sulfate or gypsum slurry was impounded in the Lined Lagoons and later disposed of in the Calcium-Sulfate Landfill. The solubility of residual calcium sulfate is largely controlled by the amount and solubilities of the sulfate salts present, although in acidic soil conditions, sulfate concentrations are also controlled by adsorption to particle surfaces. Sulfate is a naturally occurring compound and is also present in the environment from acidic precipitation. In wetlands subjected to periodic flooding, sulfate concentrations in soils increase during dry periods due to oxidation of sulfides and other forms of sulfur (Mitsch and Gosselink, 1993). The background sulfate concentration for soils is 30 mg/kg, and the background concentrations in surface water and groundwater are 24 mg/l and 150 mg/l, respectively.

Distribution of Sulfate

As discussed in Section 2, additional sulfate analyses since the CSA (CRA, 1993) mainly included the surface water, shallow groundwater, dense layer, and surface soil samples. In addition, subsurface soil samples from Lake Poly and additional flocculent and sediment samples were also analyzed for sulfate. Based on the results of the CSA (CRA, 1993) and the Supplemental Phase II Investigations, the updated distribution of sulfate in the shallow and deep groundwater at the site is presented on Plate 4-4. The distribution of sulfate in surface soil, sediment, surface water and groundwater on-property and in adjacent areas is presented on Plate 4-8. The distribution of sulfate in surface soils, subsurface soils, wastes, and the dense layer is discussed below to first identify the major source areas, then the distribution of sulfate in the groundwater, surface water, and sediments is discussed to show the potential impacts to these receptors or pathways.

In general, the Supplemental Phase II Investigations show that the highest concentrations of sulfate are found in subsurface soils in the vicinity of the Acid Pits/Lined Lagoons, Drum Area B, the Calcium-Sulfate Landfill, the Buried Debris Area, and the area east of Plant D (SWMU 26). The elevated sulfate concentrations in these areas are attributed to historical releases from the former manufacturing and disposal areas. The Supplemental Phase II surface soil results show

that the low-lying drainage areas around the Central Pond and the Central Wetlands also have elevated sulfate concentrations. The highest sulfate concentrations are found in samples collected from residual calcium sulfate material (see Section 2.3) along the drainage area west of the Central Pond.

In the subsurface soils the highest concentrations of sulfate (from 27,000 mg/kg to 33,000 mg/kg) are found in the vicinity of the Acid Pits/Lined Lagoons, Drum Area B, Calcium-Sulfate Landfill and the Buried Debris Area (Plate 4-8). Significant detections of sulfate are also found in subsurface soils at Lake Poly, the vicinity of Plant D, Drum Areas A and B, and the south side of the Warehouses. The sulfate results for the Supplemental Phase II Investigations in Lake Poly are consistent with the findings of the CSA (CRA, 1993).

As shown on Plate 4-4 and Plate 4-8, the dense layer is the greatest potential source of sulfate at the site. The distribution of sulfate concentrations in the deep groundwater is mainly controlled by the position of the dense layer which occupies the bedrock lows on-property and off-property. The maximum concentrations of sulfate detected in dense layer samples collected from the multilevel piezometers in the Supplemental Phase II Investigations are generally similar to those detected in the CSA (CRA, 1993). As shown on Figure 2-79, the concentration of sulfate increases with depth within the dense layer. In addition, the results of the Supplemental Phase II Investigations show that sulfate concentrations in the dense layer decrease with distance from the source areas due to dilution and longitudinal dispersion (Section 5.1.2).

Media	Location	Sulfate Maximum Concentration (ppm)	
		CSA (CRA, 1993)	Supp. Phase II
Dense Layer	Multilevel Piezometer MP-1, Port 1	not installed	74,000
	Multilevel Piezometer MP-2, Port 1	not installed	77,000
	Multilevel Piezometer MP-3, Port 1	not installed	48,000
	GW-42D (near MP-2)	81,000	74,000
	GW-44D (near MP-3)	40,000	28,000
	GW-59D	34,000	not tested
	GW-83D	not installed	17,300

Outside of the dense layer, the sulfate concentrations in the deep groundwater are elevated near the former Acid Pits and Lined Lagoons, the Calcium-Sulfate Landfill, in the area east of Plant D (SWMU 26), at GW-50D, and within the lower Western Bedrock Valley (GW-84D - GW-87D). The elevated sulfate concentrations in the deep groundwater near the Acid Pits/Lined Lagoons and the Calcium-Sulfate Landfill coincide with elevated sulfate concentrations detected in subsurface soil samples. As discussed with the other indicator parameters, the elevated concentrations at the GW-50D and in the lower Western Bedrock Valley may be related to the flow of deeper groundwater (with higher sulfate concentrations) over the bedrock saddles.

The areas with high concentrations of sulfate in the shallow groundwater on-property closely coincide with the areas with the highest sulfate concentrations in soils (Plate 4-8) and show a distribution of concentrations partly controlled by shallow groundwater flow directions toward discharge points along the Ditch System. The shallow groundwater quality results show detections of sulfate at 100 mg/l to 1,000 mg/l in areas above the dense layer and outside of the dense layer. On-property and immediately west of the property (at MP-1, MP-2, and MP-3) sulfate concentrations drop from over 10,000 mg/l in the dense layer to less than 100 mg/l a few feet above the interface and then increase in the shallow groundwater toward the ground surface. This suggests that the subsurface soils in the Acid Pits and Lined Lagoons, the Calcium-Sulfate Landfill, the area near Plant D, and the Drum Areas are the primary sources of low to moderate concentrations of sulfate to the shallow groundwater. Off-property, the dense layer is probably the dominant contributor of sulfate to the shallow groundwater, since sulfate is not appreciably limited by solubility controls and is primarily affected by dilution.

Historical results from 1977 and 1978 show that sulfate concentrations in both the surface water and shallow groundwater have decreased by a factor of 10 to 100. Prior to weir installation, the surface water results show that the consistently highest sulfate concentrations were detected in samples collected from the Off-Property West Ditch and upper South Ditch. Generally lower concentrations were detected in the lower South Ditch and downstream sampling points along the East Ditch. Elevated sulfate concentrations were also detected in the upper East Ditch at SW-1 and SW-2 during the December 1991 sampling event, but concentrations of sulfate were 20 times lower at these locations in the two subsequent sampling events in August and November 1992. Additional surface water data collected since the CSA listed below shows that the sulfate concentrations in the Off-Property West Ditch and South Ditch have greatly decreased since the installation of the weir. This reduction in sulfate concentrations is attributed to continued flushing of the aquifer from infiltration of precipitation and to the effect of the weir.

Media	Location (see Plate 4-8 for sample locations)	Sulfate Maximum Concentration (ppm)	
		CSA / Pre-Weir	Post Weir
Surface Water	North Off-Property West Ditch	380	190
	Central Off-Property West Ditch	3,000	51
	South Off-Property West Ditch	760	54
	Upper South Ditch above Weir	600	* *
	Upper South Ditch below Weir	380	660
	Central South Ditch at Pond	530	* *
	Lower South Ditch	410	280
	Lower Ephemeral Drainage	290	130
	- - Not detected * * Not tested		

The resulting localized changes in shallow groundwater discharge points and the configuration of bedrock surface may be responsible for higher detections of sulfate in the South Ditch below the weir at SW-17-95 in July and October 1995. As shown on Figure 2-92, the consistently highest sulfate concentrations in shallow groundwater samples were also detected where the bedrock

approaches the surface at GW-79S. In addition, the sulfate results for surface water samples below the weir show an increase in sulfate concentrations. It appears the residual sulfate in soils in the vicinity of the former Acid Pits, Lined Lagoons, and Central Pond contributes sulfate to the shallow groundwater which discharges to the surface water in areas where the bedrock surface approaches the ground surface.

The sediment results show that there are generally elevated concentrations of sulfate throughout the Ditch System. The highest concentrations of sulfate occur in the downstream portion of the East Ditch and the Off-Property West Ditch. Consistently elevated sulfate concentrations are also observed in the upper South Ditch, and sporadically high concentrations are observed in the lower half of the South Ditch and portion of the East Ditch adjacent to the facility. The EMPA analysis of flocculent shows that it contains several solid phases of sulfate along with amorphous hydroxide mixtures of aluminum, chromium, and iron. The analyses and geochemical modeling suggest that some of the sulfate in sediments is the result of groundwater discharge to the Ditch System. The distribution of sulfate concentrations in the sediments appears to be controlled by these geochemical interactions between shallow groundwater and surface water and by transport and re-deposition of sulfate-bearing sediments. The higher sulfate concentrations along the lower East Ditch (SW-23 to SW-27) are attributed to accumulation of sulfate-bearing sediments in areas of lower gradient and slower surface water velocity.

4.3.4 Semivolatiles

The most prevalent SVOCs identified in the CSA (CRA, 1993) are phthalates, NNDPA, and phenols. PAHs are typically only present at high concentrations in waste material, sediments, and subsurface soils in the former disposal areas, but are present at low concentrations in the groundwater. The distribution of PAHs and other SVOCs are discussed in the CSA (CRA, 1993). Of the SVOCs detected, bis(2-ethylhexyl)phthalate is the most prevalent and is present at the highest concentrations. As discussed in Section 4.1, the maximum background concentrations of total phthalates are 2,064 micrograms per kilogram (ug/kg) in sediment and 73 micrograms per liter (ug/l) in groundwater. For total PAHs, the maximum background concentrations are 266 ug/kg in soils, 3,290 ug/kg in sediment, and 1.2 ug/l in groundwater. The SVOC background concentrations for other media and other parameters are below detection limits.

On the Olin property, low concentrations of phthalates, NNDPA, and phenols are detected in the shallow groundwater adjacent to or downgradient of source areas. As discussed in Section 4.2.1.2 above, moderate concentrations of SVOCs are associated with the oil that is being contained and recovered at Plant B. The dense layer in the vicinity of Drum Area B and adjacent to Drum Area A and Lake Poly generally has higher concentrations of SVOCs than other areas. Within the dense layer, the distributions of organics are controlled by the shape of the bedrock valleys and the resulting gravity-driven flow of dense layer from source areas to its current resting place in bedrock lows. Outside of the dense layer, the distribution of these organics in the groundwater is controlled by the location of residual sources, groundwater flow directions, and the location of shallow groundwater discharge areas along the Ditch System.

As discussed in the CSA (CRA, 1993), NNDPA and phthalates have very high adsorption potential and thus minimal mobility in the surface water and groundwater and a greater tendency to adhere to the soil matrix and sediments. Bis(2-ethylhexyl)phthalate and di-n-octylphthalate are the main phthalates detected in groundwater at Plant B, the central portion of the property, and in the vicinity of the Off-Property West Ditch. Compared to phthalates, phenols have lower adsorption potentials and therefore have a greater tendency to partition to water. The distribution of phenols shows generally lower concentrations in soils and sediments and a much wider distribution in the deep groundwater where their mobility may have been enhanced by the acidic dense layer. NNDPA, phenols, and phthalates appear to be discharging from the shallow groundwater to portions of the Ditch system at trace concentrations. Based on the results of the CSA (CRA, 1993) and the Supplemental Phase II investigations, the distributions of NNDPA, phenols, and phthalates in surface soil, subsurface soil, sediment, surface waters and groundwater on-property and in adjacent areas are presented on Plates 4-9 through 4-12 and are discussed below.

4.3.4.1 Phthalates

Sources and Types of Phthalates

Phthalates were mainly used in the manufacture of liquid plasticizers from the mid-1950s to the early 1960s. Several other products including Kempore® dispersions and Actafoam R-3 contained or used di-n-octylphthalate or other phthalates as raw materials from the early 1960s until the cessation of manufacturing activities in 1986. The products containing or derived from phthalates were mainly made in Plant B and Plant C, and the raw materials were stored at the former Plant B tank farm. Given the time period of production and the distribution of phthalates in the surface, subsurface soils, and sediments, it is believed that phthalates were disposed of in Lake Poly, Drum Areas A and B, and possibly the former Acid Pits.

The phthalates analyzed for and detected in samples include bis(2-ethylhexyl)phthalate, butylbenzylphthalate, di-n-butylphthalate, di-n-octylphthalate, diethylphthalate, and dimethylphthalate. The maximum concentrations and number of locations where phthalates were detected in each media are summarized on Tables 4-5 through 4-13. As these results show, bis(2-ethylhexyl)phthalate is the most frequently detected phthalate and is typically present at the highest concentrations in all media. Di-n-butylphthalate is frequently detected in soils, sediment, and groundwater, but is only present at elevated concentrations in samples from surface soil, subsurface soil, and sediment. Butylbenzylphthalate and di-n-octylphthalate are detected at fewer locations in each media and are also present at elevated concentrations in only surface soil, subsurface soil, and sediment samples. As shown on Tables 4-5 through 4-13, diethylphthalate and dimethylphthalate are infrequently detected at low concentrations.

Distribution of Phthalates

The distribution of total phthalates, consisting of the sum of all detected phthalates in surface soil, subsurface soil, sediment, surface water and groundwater samples on-property and in adjacent areas is presented on Plate 4-9. The distribution of phthalates in surface soils, subsurface soils, and wastes is discussed below to first identify the major source areas, and then the distributions of phthalates in the groundwater, dense layer surface water, and sediments are discussed to show the potential impacts to these receptors or pathways.

The highest concentration of phthalates, primarily bis(2-ethylhexyl)phthalate and di-n-octylphthalate, was detected in an oil sample collected from recovery well IW-11 at Plant B. As discussed in Section 4.2.1.2 above, the thin layer of floating oil is contained by three recovery wells, and there is relatively little partitioning of phthalates from the oil into the groundwater. As shown on Table 4-6, the concentration of total phthalates in the groundwater at Plant B is over 100,000 times less than in the oil.

The highest concentrations of phthalates in subsurface soils and surface soils were detected in samples from former disposal areas, former production areas, or from associated adjacent low lying areas. As summarized below, the highest total phthalate concentrations in subsurface soils were detected at Lake Poly, Drum Area B, Plant C, and Plant B. Elevated phthalate concentrations were also detected in subsurface soils in the vicinity of the former Acid Pits. In surface soils, high concentrations of phthalates were detected in samples from along the banks of the On-Property West Ditch (SWMU 27) and South Ditch (SWMU 30). Elevated concentrations of phthalates were also detected in surface soil samples from the low lying area east of Plant D (SWMU 26), from the areas where phthalates were stored or spilled (grid Area 6 and 7), from along the banks of the Ephemeral Ditch (SWMU 33), and from the area around the Central Pond (grid Area 8).

		Total Phthalates
Media	**Location**	**Maximum Concentration (ppm)**
Waste	Plant B - oil sample IW-11	670,000
	Drum Areas A and B - drum material	87 and 44
Subsurface Soil	Lake Poly (BH-38, LPB-1, and BH-12)	6,700
	Plant C (BH-26)	1,200
	Drum Area B (TP-21)	1,100
	Plant B (BH-34)	950
	Acid Pits (BH-4)	27
Surface Soil	SWMU 30 - South Ditch	32,119
	SWMU 27 - West Ditch	5,517
	SWMU 33 - Ephemeral Ditch	34.3
	SWMU 26 - Plant B Area	5.2

The sample results for both shallow and deep groundwater and dense layer wells show low concentrations of phthalates. The historical results show that the concentrations of phthalates have decreased since 1988. The deep groundwater outside of the dense layer shows more

elevated concentrations of total phthalates in the vicinity of Drum Area B, Plant B, and downgradient of Lake Poly. The shallow groundwater quality results also show the highest detections of total phthalates in these areas. These areas have corresponding high concentrations of phthalates in subsurface soils which suggests that these areas may be sources of low concentrations of phthalates to the shallow groundwater.

In the dense layer, distribution of total phthalates demonstrates a tendency to adsorb to aquifer materials. The concentration of total phthalates decreases dramatically away from the source areas and is highest near suspected source areas (0.16 mg/l at GW-22D to 0.18 mg/l at GW-27D). The concentrations continue to decrease downslope to concentrations of less than 0.01 mg/l just west of the facility boundary.

Media	Location	Total Phthalates Maximum Concentration (ppm)
Shallow Groundwater	Plant B- Interceptor Wells (IW-11 to 13)	0.047
	Lake Poly (GW-29S)	0.029
	Drum Area A (GW-22S)	0.006
	NW Corner of Property (GW-21S)	0.002
	Off-Property West Ditch (GW-79S)	0.002
Deep Groundwater	Off-property (GW-81D)	0.021
	Lake Poly (GW-29D)	0.006
	NW Corner of Property (GW-21D)	0.003
	Plant C (GW-53D)	0.001
Dense Layer	Drum Area B (GW-27D)	0.18*
	Drum Area A (GW-22D)	0.006
		*Phthalates ND at GW-27D on 11/92

Phthalates are commonly detected at low concentrations in surface water samples from the East Ditch, Off-Property West Ditch, lower South Ditch, and lower Ephemeral Drainage. These areas appear to coincide with the distribution of total phthalates in both the shallow and deep groundwater. As shown below, the concentrations detected in the surface water samples are generally higher than detected in shallow groundwater samples. This may indicate that deeper groundwater (with higher phthalate concentrations) or sediments, particularly in the area of the lower Ephemeral Drainage and lower South Ditch, may be contributing phthalates to the surface water.

Media	Location	N-Nitrosodiphenylamine Concentration Range (ppm)
Surface Water	Lower South Ditch	0.024 to 0.048
	Lower Ephemeral Drainage	ND to 0.013
	Downstream East Ditch	0.011 to 0.074

The distribution of phthalates in sediment shown on Plate 4-11 suggests that the phthalates originated from disposal or production areas and were then transported down the ditch system. Generally higher concentrations of total phthalates were detected in sediments than were detected in soils. This suggests that sediment contamination originated from releases into the On-Property Ditch System rather than from erosion and transport of contaminated surface soils. The highest total phthalate concentration (152,000 mg/kg) was detected in samples from the On-Property West Ditch, downstream of Lake Poly, where high concentrations of total phthalates were detected in subsurface soils. As shown below, the concentration of total phthalates generally decreases downstream along the West Ditch and South Ditch, although more elevated total phthalate concentrations are observed adjacent to the Central Pond and the associated drainage areas. Elevated concentrations of total phthalates in sediments were also detected in samples from the East Ditch adjacent to Plant B and relatively low concentration were detected in samples at the downstream sampling points SW-24 to SW-28, which appear to be accumulation points of historically transported sediments.

		Total Phthalates
Media	**Location**	**Maximum Concentration (ppm)**
Sediment	On-Property West Ditch	152,000
	Upper South Ditch, below W. Ditch	3,500
	South Ditch near Central Pond	57,000
	Central Pond	851
	Lower South Ditch, below C. Pond	3,500
	East Ditch near Plant B	442
	East Ditch Downstream (SW-24 to -27)	0.074

4.3.4.2 N-Nitrosodiphenylamine

Sources of N-Nitrosodiphenylamine

A product called Witrol N, also known as NNDPA, was produced at the facility in the mid-1960s. NNDPA was produced from raw materials mainly consisting of diphenylamine, sodium nitrate, and sulfuric acid and the waste products included sodium nitrite, sodium sulfate, and NNDPA. Given the time period of production and the distribution of NNDPA in the surface and subsurface soils, it is believed that NNDPA was produced at Plant B and that the waste byproducts were disposed of in Lake Poly, Drum Areas A and B, and the former Acid Pits.

Distribution of N-Nitrosodiphenylamine

The distribution of NNDPA in surface soils, subsurface soils, and wastes are discussed below to first identify the major source areas, and then the distribution of NNDPA in the groundwater, dense layer, surface water, and sediments are discussed to show the potential impacts to these receptors or pathways. Based on the results from the CSA (CRA, 1993), the distribution of NNDPA in subsurface soils is shown on Plate 4-9. The highest concentrations of NNDPA were

detected in an oil sample from Plant B and in drummed material from Drum Area A. As shown below, much lower concentrations of NNDPA were detected in waste material samples from Drum Area B.

Media	Location	N-Nitrosodiphenylamine Maximum Concentration (ppm)
Waste	Plant B - oil sample from IW-11	34,000
	Drum Area A - drum material	2,100
	Drum Area B - drum material	1.5

As shown on Plate 4-9, high concentrations of NNDPA were detected in subsurface samples collected from Lake Poly, and elevated concentrations were detected in samples from the immediate vicinity of the former Acid Pits. In surface soils, elevated concentrations of NNDPA were detected in samples from former disposal areas (Lake Poly and Drum Areas) and in samples from former production areas (grid Areas 5 and 6). In addition, elevated concentrations of NNDPA were also detected in surface soils samples from low lying areas adjacent to Lake Poly (SWMU 27) and along the banks of the Ditch System downstream from the former Lake Poly (SWMUs 30 and 33).

The sediment results show higher concentrations of NNDPA than in surface soils, with the highest concentrations (4,800 to 6,200 mg/kg) occurring in the On-Property West Ditch downstream of the two source areas with high concentrations of NNDPA, Drum Area A and Lake Poly. As with the phthalates, the comparison of NNDPA concentrations of samples from surface soils, subsurface soils, and sediment (presented below) shows high concentrations of NNDPA in subsurface soils in a disposal area (Lake Poly) and in the sediments downstream. This suggests that NNDPA-wastes were released directly into the on-property Ditch System. The concentration of NNDPA generally decreases downstream along the South Ditch, although more elevated NNDPA concentrations were detected in sediment samples adjacent to the Central Pond and the associated drainage areas. Low concentrations of NNDPA were also detected in sediment samples from the East Ditch adjacent to Plant B and at the downstream sampling points.

Media	Location	N-Nitrosodiphenylamine Maximum Concentration (ppm)
Subsurface Soil	Lake Poly (BH-12)	3,400
	Plant B (BH-34)	0.36
Surface Soil	Lake Poly (Lake Poly-1)	210
	Upper On-Property West Ditch (SWMU-27)	32
	Upper South Ditch Banks (BS-015)	2.8
	Drum Area A	2.8
	Drum Area B	1.0
	Upper Ephemeral Ditch (SWMU-33, 4 33-BS-016)	0.55
	Plant Production Area (Area 6)	0.48
	Central Pond Area (Area 8-4, BS-017)	0.30
	Plant B (SWMU 26)	0.26
Sediment	On-Property West Ditch	6,200
	Upper S. Ditch Above Central Pond (SW-9)	720
	Upper S. Ditch Below W. Ditch (SW-10)	52
	Upper S. Ditch Below Central Pond (SW-8)	30
	Upper S. Ditch (SW-11)	22
	Upper On-Property W. Ditch (SW-12)	20
	Lower East Ditch (SW-23)	2.1
	East Ditch Below S. Ditch (SW-5)	1.8
	East Ditch Near Plant B (SW-3)	0.82

The results of the CSA show that low concentrations of NNDPA were detected in shallow and deep groundwater, and trace concentrations of NNDPA were detected in the dense layer. The analytical results show the highest detections of NNDPA in wells at Plant B and in wells downgradient of Lake Poly, the East and West Pits, and Drum Area A. Elevated concentrations of NNDPA have been detected in the shallow groundwater in the western portion of the site with the lower concentrations detected downgradient of the Acid Pits. As discussed below, the areas with high concentrations of NNDPA detected during the CSA in the shallow groundwater at the western portion of the property (GW-11 and GW-33S) also coincide with historically elevated detections in these wells and nearby dense layer wells. In the groundwater, NNDPA shows a narrower distribution than phthalates and appears to be discharging from the shallow groundwater into the surface water of the Ditch System at trace concentrations. It is also possible that the sediment may contribute NNDPA to the surface water or that shallow groundwater contributes NNDPA to the sediments. The sediment concentrations in the Off-Property West Ditch are generally less than 1 mg/kg NNDPA and lower concentrations are detected in shallow groundwater (0.012 mg/l) and surface water (0.031 mg/l).

The surface water results from the CSA (CRA, 1993) show that low concentrations of NNDPA were detected in the Off-Property West Ditch, South Ditch, and East Ditch near Plant B. These portions of the Ditch System also coincide with areas with NNDPA detections in sediment and shallow groundwater samples. The highest surface water concentration (0.031 mg/l) was detected in the Off-Property West Ditch and is consistent with hydrogeologic data that shows that

this portion of the Ditch System is a groundwater discharge point. Although analytical results are not available, NNDPA concentrations have probably decreased in the Off-Property West Ditch and South Ditch, based on the weir effects on the concentrations of indicator parameters.

4.3.4.3 Phenols

Sources of Phenols

Phenolic resins were produced at the facility between the early- to mid-1960s. Phenols are present in all of the media at the Olin property and are found in the groundwater, dense layer, surface water, and sediment off-property. The frequently detected phenolic compounds include phenol, 2-chlorophenol, 2,4-dichlorophenol, 2,4-dimethylphenol, 2-nitrophenol, and 4-nitrophenol. Given the time period of production and the distribution of phenols in the surface soils, subsurface soils, and sediments it is believed that phenols were disposed of in Lake Poly, Drum Area B, and possibly the East and West Pits.

Distribution of Phenol

The distributions of the above phenolic compounds are generally similar. Phenol results from the CSA (CRA, 1993) in all media are shown on Plate 4-10. The distribution of phenol in surface soils, subsurface soils, and wastes are discussed below to first identify the major source areas, and then the distribution of phenol in the groundwater, dense layer surface water, and sediments are discussed to show the potential impacts to these receptors or pathways.

The highest concentrations of total phenols were observed in a waste characterization sample of drum material removed from Drum Area B, surface soils at Drum Area B, a subsurface soil sample from the Buried Debris Area, a sediment sample from the On-Property West Ditch adjacent to Lake Poly, and the dense layer. As summarized below, the waste material and soil analytical results from Drum Area A did not show high concentrations of phenolics, however the drums removed from Drum Area A contained material tentatively identified as hard phenolic resins by the former plant manager (CRA, 1992).

Low concentrations of phenols were also detected in subsurface soils in the Buried Debris Area, in the vicinity of Lake Poly, the Acid Pits, and Lined Lagoon II. The highest phenol concentration was detected in the surface soil sample from Drum Area B, and low concentrations were detected in the surface soil samples from the former production areas.

Media	Location	Phenol Maximum Concentration (ppm)
Waste	Plant B - oil sample from IW-11	ND
	Drum Area A - drum material	ND
	Drum Area B - drum material (OW-249D)	510
Subsurface Soil	Buried Debris Area (TP-19)	5.3 J
	Lake Poly/Drum Storage Area (BH-36)	0.25 J
	Lined Lagoons/Acid Pits (BH-8)	0.1 J
Surface Soil	Drum Area B (Drum B Comp)	2.4
	Main Plant (Area 5)	0.069 J
	Main Plant Area (Area 6)	0.047 J

Trace to low phenol concentrations were detected in a few shallow and deep groundwater samples, and low concentrations of phenol were detected in most of the dense layer wells. As shown on Plate 4-10, the highest concentration of phenol in shallow groundwater is centered around well GW-35S, and elevated (but lower) concentrations were detected in wells to the north in the direction of the East and West Pits. Trace concentrations (1 J ug/l to 3 J ug/l) of phenol were detected in shallow groundwater samples from wells downgradient of the above source areas, GW-27S, GW-33S, GW-52S, and GW-55S. In the deep groundwater, trace concentrations of phenol were detected in well samples near Plant B and west of Plant C-1. These results may suggest that adjacent source areas are potentially contributing to trace concentrations of phenols detected in the groundwater.

As discussed previously, phenol has lower adsorption potentials than phthalates and NNDPA and therefore has a greater tendency to partition to water. This is demonstrated by the low concentrations in soils and sediments and the wider distribution of phenol in the dense layer where the mobility of phenol may have been enhanced by the acidic conditions in the dense layer. The phenol concentrations in the dense layer range from 0.15 mg/l to a maximum of 1.0 mg/l at bedrock lows beneath the property, from 0.6 mg/l to at maximum of 1.6 mg/l (GW-59D) in the upper Western Bedrock Valley between the property and Main Street. Phenols were also detected at a concentration of 0.063 mg/l at the bedrock low at GW-83D in the lower Western Bedrock Valley.

The surface water quality results for the Off-Property West Ditch, upstream portions of the South Ditch, and East Ditch at sample location SW-3 may suggest that the shallow groundwater and/or sediment is contributing trace concentrations of phenols to these portions of the Ditch System. The highest phenol concentrations detected in surface water samples are listed below.

Media	Location	Phenol Maximum Concentration (ppm)
Surface Water	N. Off- Property West Ditch (SW-15, SW-16)	0.003J
	Upper South Ditch (SW-14)	0.002J
	South Off- Property West Ditch (SW-17)	0.002
	East Ditch Near Plant B (SW-3)	0.001J

The sediment results listed below show that the highest concentrations of phenols occur in the On-Property West Ditch and that concentrations generally decrease downstream along the South Ditch. Four out of the seven highest phenol concentrations were detected in sediment samples from the East Ditch over 900 feet downstream of the confluence with the South Ditch. This data suggests that these portions of the East Ditch have accumulated sediment impacted by past discharges or that there are off-property sources of phenol.

Media	Location	Phenol Maximum Concentration (ppm)
Sediment	Upper On-Property West Ditch (SW-12)	2.3
	South Ditch Near Central Pond (SW-9)	0.58
	Lower East Ditch (SW-27)	0.36J
	Upper S. Ditch Below West Ditch (SW-11)	0.14J
	Lower South Ditch (SW-6)	0.076J

4.3.5 Volatile Organic Compounds

Sources and Types of Volatile Organic Compounds

As discussed in the CSA (CRA, 1993), volatile organic compounds (VOCs) were detected at low to moderate concentrations on-property and at low concentrations off-property. The Supplemental Phase II Investigations have confirmed that the most frequently detected VOCs in several media were acetone and trimethylpentenes. The CSA (CRA, 1993) and Supplemental Phase II Investigations show that the other VOCs mainly detected in several media at the site were 2-butanone, ethylbenzene, 2-hexanone, methylene chloride, 1,1,1-trichloroethane, and toluene.

Former employees at the facility have stated that acetone was used for cleaning machinery and equipment. Trimethypentenes (TMPs) detected in samples consist mostly of 2,4,4-trimethyl-1-pentene which is typically accompanied by lower concentrations of 2,4,4-trimethyl-2-pentene. TMPs, also called di-isobutylene, were used as a raw material for the Wytox ADP process at Plant B beginning in the early 1960's. The material balance calculations performed for the 1969 pollution control study (Badger, 1969) show that the Wytox ADP process generated 19,000 pounds per month of TMPs which were discharged on-property. This process operated 3 to 5 days per week at Plant B, and prior to 1972 the waste water was discharged to Lake Poly, the Acid Pits, and possibly the unlined East and West Pits.

The distribution of acetone concentrations in the subsurface soils and groundwater suggests that acetone was released at Lake Poly, the Drum Areas, Plant B, and the Acid Pits and that TMPs were released at Plant B, Plant C-1, Lake Poly, Drum Area A, Plant D, and the area east of Plant D (SWMU 27).

Distribution of Volatile Organic Compounds

Based on the results of the CSA (CRA, 1993) and the Supplemental Phase II Investigations the distribution of acetone, total TMPs, and other volatile organic compounds (VOCs excluding acetone and TMPs), in surface soil, sediment, surface water and groundwater on-property and in adjacent areas is presented on Plates 4-12, 4-13, and 4-14, respectively. The distribution of VOCs in surface soils, subsurface soils, and wastes are discussed below to first identify the major source areas, and then the distribution of VOCs in the groundwater, dense layer surface water, and sediments are discussed to show the potential impacts to these receptors or pathways.

The total TMPs were calculated by summing the results for 2,4,4-trimethyl-1-pentene and 2,4,4-trimethyl-2-pentene at each sampling location. The total of other VOCs, excluding acetone and TMPs, were calculated by summing all other VOCs detected at a sampling location. The other VOCs frequently detected at low concentrations in several media at the site include 2-butanone, ethylbenzene, 2-hexanone, methylene chloride, 1,1,1-trichloroethane, and toluene. Bromoform and chloroform are mainly detected at low concentrations in the groundwater and dense layer, while teterachloroethane has been detected at low concentrations in the surface soils. The CSA (CRA, 1993) discussed the analytical results and distribution of the VOCs detected at the site and the distribution of acetone, TMPs, and the other VOCs is summarized below.

The highest concentrations of acetone, TMPs, and other VOCs in waste material and subsurface soils are summarized below. As shown, the highest concentration of VOCs in all of the sampled media were detected in samples from the floating oil at Plant B, waste material in Drum Areas A and B, and in subsurface soil samples from Lake Poly, the Acid Pits and/or Lined Lagoons, and former production areas of the facility, including Plant B and Plant C and the black area east of Plant D (SWMU 26). The main VOCs in the samples from these areas were TMPs, with the exception of the samples from Drum Area B and from near the Acid Pits/Lined Lagoons (BH-1), which mainly contained acetone. Other VOCs, besides acetone and TMPs, that were also frequently detected at low concentrations in these media include 2-hexanone and toluene in most areas and also ethylbenzene and styrene in the oil sample from Plant B and the subsurface soil sample from BH-15.

		Maximum Concentration (ppm)		
Media	**Location**	Acetone	TMPs	Other VOCs
Waste	Plant B - oil sample from IW-11	- -	20,600	18.15
	Drum Area A - drum material (OW-249A-C)	11	453	11.20
	Drum Area B - drum material (OW-249D)	690	- -	21.87
Subsurface Soil	Lake Poly (LPB-1)	0.12	550	2.6
	Acid Pits/Lined Lagoons (BH-1)	17	- -	2.8
	Lake Poly/SW of Warehouses (BH-15)	0.12	6	14.25
	Area east of Plant D (SWMU 26 @ BH-23)	- -	12.1	0.012
	Near Drum Area A (BH-9)	- -	4.21	0.065
	Plant B Tank Farm (BH-34)	- -	2.42	1.7
	Plant C (BH-26)	0.09	1.36	0.33
	Plant B Production Area (BH-28)	- -	0.46	0.082
- - : Not detected	Other VOCs: total of all VOCs except acetone and TMPs			

As expected, low concentrations of VOCs were detected in surface soils. As discussed in Section 2.3, the surface soil results of the Supplemental Phase II Investigations showed more detections of 1,1,1-trichloroethane in grid Areas 1, 8, and 9 than previously detected in the CSA (CRA, 1993). The highest concentrations of acetone, trimethylpentenes, and 1,1,1-trichloroethane (1,1,1-TCA) in surface soils are summarized below. The distribution of other VOCs, which include 1,1,1-TCA (Plate 4-14), showed that the highest concentrations of VOCs in surface soils were detected in samples from the former production areas, disposal areas, and the adjacent low-lying areas.

		Maximum Concentration (ppm)		
Media	**Location**	**Acetone**	**TMPs**	**1,1,1-TCA**
Surface Soil	Near On-Property W. Ditch (SWMU 27)	0.093 J	0.339 J	- -
	Area 1 (Area 1-4)	0.036	- -	0.230
	Area 9 Central Wetland (A9CW-3)	- -	- -	0.071
	Area 8 (Area 8)	- -	0.019 J	- -
	Lake Poly (Lake Poly-2)	0.031 B	- -	0.009
	Drum Area B (G2-DRMB)	0.022 B	0.0008 J	0.016
	Drum Area A (GB4-DRMA)	0.019 B	- -	0.018
	- - : Not detected			

As shown on Plates 4-12 through 4-14, low concentrations of acetone, TMPs, and other VOCs were detected in samples from the shallow groundwater, deep groundwater, and dense layer. The highest concentrations of acetone were detected in the groundwater and dense layer samples from the central portion of the property. These areas generally correspond to areas where elevated acetone concentrations were detected in subsurface soil or waste samples. It should be noted that the concentration of acetone in the groundwater and dense layer samples from most wells varied considerably between sampling events (CSA Rounds 1 and 2) and that the acetone

isoconcentration contours shown on Plate 4-12 reflect the maximum concentrations where more recent (1996- and 1997) data was unavailable. For instance, acetone was detected at GW-6D at a concentration of 13 mg/l in the sample collected on the November 3, 1992 but was not detected (<0.03 mg/l) in the sample collected on August 4, 1997. Data validation (Appendix M, CSA, 1993) of the Round 1 and Round 2 sampling events did not indicate that the acetone results presented on Plate 4-12 were due to laboratory contamination. The groundwater and dense layer sample results showed that the acetone was consistently detected at low concentrations, although the underlying reason for the apparent variability in acetone concentrations is unknown.

As shown on Plate 4-13, the highest concentrations of TMPs occurred in shallow groundwater samples from the interceptor wells at Plant B and in shallow and deep groundwater samples from the central-western portion of property. Shallow groundwater quality results for on-property wells GW-11, GW-27S, GW-34S, and GW-54S show detections of TMPs ranging from 0.05 mg/l to 2.62 mg/l. The areas with elevated TMPs in the shallow groundwater generally coincide with the areas with the highest concentrations of these parameters detected in subsurface soil or waste samples. In the deep groundwater, TMPs were mainly detected at low concentrations in the central portion of the property. TMPs were infrequently detected at low concentrations in dense layer samples and the highest concentration (0.191 mg/l) was detected in the sample from GW-38.

The other VOCs (besides acetone and TMPs) related to the facility were mainly detected at low concentrations in the shallow groundwater and dense layer (Plate 4-14). As discussed in the CSA (CRA, 1993), the other VOCs detected in samples from the shallow and deep groundwater wells east of the property (GW-49D, GW-80S/D) are attributed to an off-site source. In addition, the occurrence of trichloroethene, 1,2-dichloroethene, toluene and other VOCs that are not related to the site were also detected in samples from the new wells in Maple Meadow Brook wetlands and in the public supply wells. On-property, the highest concentrations of other VOCs were detected in shallow groundwater samples from near Drum Area A, the central portion of the property, and the interceptor wells at Plant B. In general, the most frequently detected VOCs in shallow groundwater samples from the central portion of the property include toluene and 2-hexanone. Exceptions to this include detections of 2-hexanone, xylenes, and ethylbenzene in the sample from interceptor well IW-12 at Plant B and detections of 4-methyl-2-pentanone in the sample from GW-22S near Drum Area A. The deep groundwater isoconcentration contours for other VOCs presented on Plate 4-14 mainly reflect the concentrations of other VOCs in the samples from the dense layer occupying the Western Bedrock Valley. The most frequently detected VOCs in the dense layer samples (besides acetone) were bromoform, chloroform, and toluene.

As discussed in the CSA (CRA, 1993) and as shown below, TMPs are the main VOCs related to the facility that are detected in the surface water. TMPs were detected at the highest concentrations in samples from the Off-Property West Ditch and are sporadically detected in samples from the upper South Ditch. Acetone was sporadically detected in surface water samples from the Off-Property West Ditch. The surface water results for multiple sampling events presented below shows that the concentrations of acetone, trimethylpentenes, and other VOCs in surface water may fluctuate dependent on temporal variations in groundwater discharge to the

Ditch System. The distribution of TMPs in the shallow groundwater, surface water, and sediments shown on Plate 4-13 suggests that the shallow groundwater (and possibly sediments) may be contributing low concentrations of TMPs to the Off-Property West Ditch and upper South Ditch.

Surface Water	Concentration (ppm) 8/92 & 9/92 Sampling Event			Concentration (ppm) 11/92 - 3/93 Sampling Event		
Sampling Locations	**Acetone**	**TMPs**	**Other VOCs**	**Acetone**	**TMPs**	**Other VOCs**
Off-Property W. Ditch						
SW-15	0.015	0.05	0.023	- -	0.281	0.001
SW-16	0.021	- -	0.019	- -	- -	- -
SW-18	0.008	- -	0.001	- -	- -	- -
SW-17	- -	0.003	- -	0.093	0.008	0.022
South Ditch						
SW-14	- -	0.004	- -	- -	0.038	0.003
SW-11	- -	0.004	- -	- -	0.016	0.001
SW-10	- -	- -	- -	- -	0.016	0.001
SW-9	- -	- -	- -	- -	0.008	- -
SW-8	- -	- -	- -	- -	- -	- -
SW-7	- -	- -	- -	- -	0.007	- -
SW-6	- -	- -	- -	- -	0.005	- -

- - : Not detected
Other VOCs: total of all VOCs except acetone and TMPs

As shown below and on Plates 4-12 through 4-14, TMPs were the main VOC related to the facility detected in the sediment samples. TMPs were detected at the highest concentrations in samples from the On-Property West Ditch, the Central Pond, and the South Ditch. Acetone was sporadically detected at low concentrations mainly in the sediment samples from the South Ditch and East Ditch. The highest concentrations of other VOCs were detected in sediment samples from the On-Property West Ditch and lower East Ditch. The primary VOC detected at the sample from the On-Property West Ditch was toluene (SW-13, 1.1 mg/kg). The main VOCs detected in the sample from the lower East Ditch were trichloroethene and 1,2-dichloroethene (SW-23, 0.12 mg/kg). As discussed in the CSA, (CRA, 1993) these VOCs detected in sediment sample SW-23 from the lower East Ditch indicate an off-site source.

		Maximum Concentration (ppm)		
Media	Location	Acetone	TMPs	Other VOCs
Sediment	On-Property West Ditch (SW-13)	0.055	37.4	1.253
	Central Pond (Pond)	0.055	13.8	0.101
	North Off-Property West Ditch (SW-15)	0.038	8.3	0.011
	South Ditch Near Central Pond (SW-08)	1.7	6.2	0.108
	Upper S. Ditch Below West Ditch (SW-10)	0.092	1.73	0.007
	Lower East Ditch (SW-25)	1.4	1.3	0.333
	Lower South Ditch (SW-6)	0.12	0.209	0.059
	East Ditch Above South Ditch (SW-4)	0.14	0.037	0.008
	Upper East Ditch (SW-01)	0.027	0.034	0.007
	Lower Ephemeral Ditch (SW-20)	0.007	0.004	0.008
	Other VOCs: total of all VOCs except acetone and TMPs			

4.3.6 Pesticides and PCBs

Pesticides were generally detected at very low concentration at the site. The most frequently detected pesticides include aldrin, alpha-BHC, gamma-BHC (lindane), 4,4'-DDT, endosulfan II, and toxaphene. Pesticides were most consistently detected in the surface soil and sediment samples and were sporadically detected in the subsurface soils, surface water, and shallow groundwater. PCBs were only detected in one surface soil sample from the area with black oily material (SWMU 30) on the banks of the upper South Ditch and in one shallow groundwater sample from GW-54S. Since no single pesticide predominated in all media, the pesticides detected were summed for each sample to show the distribution of all pesticides/PCBs. The distribution of total pesticides and PCBs in surface soil, sediment, surface water and groundwater on-property and in adjacent areas is presented on Plate 4-15.

As shown below, pesticides are detected at low concentrations and were detected at the highest concentrations in the surface soil and sediment samples, and in waste samples from Drum Area A. Subsurface soil samples show detections in only two samples, one from Lake Poly and one in the vicinity of Drum Area A and the Acid Pits. The distribution of pesticides in surface soils shown on Plate 4-15 shows that the highest concentrations were generally detected in the western portion of the property and in low-lying areas including the areas east of Plant D (SWMU 26), the Central Wetland, and around the Central Pond. The distribution of pesticides in sediments shows that pesticide concentrations generally decrease downstream of the On-Property West Ditch.

		Total Pesticides/PCBs
Media	**Location**	**Maximum Concentration (ppm)**
Waste	Drum Area A - drum material (OW-249C)	8
Subsurface Soil	Lake Poly (BH-12)	0.335
	Acid Pits/Lined Lagoons (BH-3)	0.093
Surface Soil	Black oily area U. South Ditch (SWMU-30)	2.04
	Northwest corner of property (Area 4)	1.448
	Upper Ephemeral Drainage (SWMU-33)	0.98 (PCBs)
	Drum Area B (DRMB-COMP)	0.368
	SWMU-27	0.312
	SWMU-26 (SWMU-26 COMP)	0.164
	Area 8 (Area 8-4)	
Sediment	East Ditch Near Plant B (SW-3)	6.5
	On-Property West Ditch (SW-13)	5.56
	Lower East Ditch (SW-25)	3.81
	Off-Property West Ditch (SW-15)	1.07
	South Ditch Near Central Pond (SW-07)	0.618
	Lower South Ditch (SW-6)	0.31
	Central Pond (Pond)	0.26
	Upper South Ditch (SW-11)	0.098
	Ephemeral Ditch (SW-21)	0.0072
	Upper East Ditch (SW-30)	0.0025

The surface water and ground water results show sporadic and infrequent detections of pesticides, with total pesticide concentrations generally less than 0.0005 mg/l. The highest concentration of total pesticides (0.001 mg/l) was detected in the shallow groundwater in a sample collected from well GW-35S, which is next to Drum Area A. In the deep groundwater, the highest concentration of total pesticides (0.0027 mg/l) was detected at GW-55D. In the surface water, pesticides were detected at low concentrations at two locations, SW-5 and SW-15. The highest concentration of total pesticides (0.00113 mg/l for five pesticides) was detected at SW-15 during the September 1992 sampling event. Only one pesticide, heptachlor epoxide, was detected at SW-15, at a concentration of 0.00018 mg/l in the subsequent sampling event in December 1992.

4.4 MAJOR AREAS OF ENVIRONMENTAL CONCERN

This section summarizes the major source areas of environmental concern and the impacted media based on the nature and extent of contamination. Much of the information was previously presented in the CSA (CRA, 1993). The CSA (CRA, 1993) data was used with the results of the Phase II Supplemental Investigations described in Section 2 to further characterize the nature and extent of chemicals of concern and the hydrogeologic conditions in these areas. These areas also

represent the areas of the site with the highest concentrations of chemicals, but do not necessarily represent the areas with the greatest potential for exposure. The major areas of concern were determined based on the following potential environmental concerns:

- the presence of separate-phase oils
- exceedance of Upper Concentration Limits (UCLs, 310 Code of Massachusetts Regulation [CMR] 40.0996(5), 4/5/96)
- the potential for transport (especially off-site) at levels of concern
- condition of substantial release migration
- presence of elevated concentrations of the same parameters in adjacent and related media (e.g. surface water and sediment)
- potential hazard

Here, the site and impacted media have been grouped according to the way they are expected to be addressed during the Remedial Action Plan (Phase III). The risk associated with each of the impacted media is summarized in Section 6.0. Impacted surface and subsurface soils are generally grouped with a known source area. Surface water bodies and associated sediments are likewise grouped together as receptors to source areas, or as source areas themselves for a limited number of chemicals. The major areas of concern and impacted media listed below are described in the following sections.

Major Areas of Potential Environmental Concern

On-Property

- Groundwater and soils in the former Plant B Production and Tank Farm Areas
- Soils and sediment in the area of former Lake Poly
- Soils and waste in Drum Area A and Drum Area B
- Subsurface soils associated with the East and West Pits
- Subsurface soils associated with the Acid Pits and Lined Lagoons
- Shallow and Deep Groundwater at GW-54 well cluster

Off-Property

- Groundwater, surface water, and sediment near the South and West Ditches
- Deep Groundwater and Dense Layer

4.4.1 Plant B and Surrounding Area

The Plant B Area encompasses four Solid Waste Management Units (SWMUs) identified in the CSA (CRA, 1993): the drum storage unit (SWMU 2), the Plant B Pit (SWMU 11), the interceptor well system (SWMU 13), and the production area and tank farm (SWMU 23). Plant B is the former location of six 15,000 gallon storage tanks and manufacturing activities which included oils and plasticizers such as bis(2-ethylhexyl)phthalate, NNDPA, and "Process Oils". Prior to Olin's ownership in 1980, the Plant B Tank Farm sat on grade with no dike or spill containment. Olin installed a pad and curb around the storage tank area in 1981. When construction was in progress, contaminated soils were encountered and removed from under and around the storage tanks and disposed of off-site in a secure landfill. The production area buildings, drum storage unit, and pit were closed in 1986 in accordance with the closure plan submitted to the MADEP.

As discussed previously, the separate-phase oil and groundwater are contained by the three interceptor wells that prevent the discharge of oil and hazardous material into the East Ditch. Groundwater extraction and treatment has been ongoing at Plant B since 1982. The results of subsurface soil investigations show that soils are impacted by low levels of VOCs, moderate levels of inorganics, and high concentrations of SVOCs, including bis(2-ethylhexyl)phthalate, di-n-octylphthalate, and NNDPA. The oil in the area around Plant B contains organics including high concentrations of TMPs and bis(2-ethylhexyl)phthalate, and lesser amounts of NNDPA and di-n-octylphthalate. The groundwater results show much lower concentrations of these organics and moderate concentrations of inorganics including ammonia, chloride, chromium, manganese, and sulfate which do not appear to be related to the floating oil. The following summarizes the potential environmental concerns associated with Plant B:

- The presence of separate-phase, floating oil contained by the interceptor wells and exceedance of UCLs for bis(2-ethylhexyl)phthalate in oil, but not in the groundwater.

- Potential contamination of adjacent media by the same parameters including subsurface soils contributions to groundwater and, where oil is present, potential contamination of subsurface soils by oils and the associated SVOCs in the oils.

- Subsurface soils where oil is not present may also be acting as a source of low concentrations of SVOCs and TMPs to the groundwater at concentrations above the GW-1 level of concern.

4.4.2 Lake Poly Liquid Waste Disposal Area

The history of Lake Poly is discussed in detail in Section 1.3.1. The liquid waste disposed in Lake Poly contained chromium (through 1967), sulfuric acid, ammonium sulfate, sodium chloride, and other inorganic and organic constituents. The Lake Poly area encompasses two areas of concern: Lake Poly (SWMU 14) and a former drum storage area west of Lake Poly. The drum storage

area was formerly located at the paved area west of Lake Poly and was used for drum storage from approximately 1954 through 1971 based on aerial photographs. This area was not identified as a separate SWMU in the CSA (CRA, 1993), but was included in subsurface investigations. Two other adjacent areas which are also believed to be impacted by disposal activities at Lake Poly include the black area near the West Ditch (SWMU 27) and the oily sediment beneath the West Ditch (SWMU 28). These areas are discussed within the context of the On-Property West Ditch in Section 4.4.4.3.

The CSA (CRA, 1993) and Supplemental Phase II Investigations have shown that the subsurface and surface soils at Lake Poly contain high concentrations of SVOCs, VOCs, and inorganics. The most prevalent compounds at the highest concentrations in the subsurface and surface soils are bis(2-ethylhexyl)phthalate, NNDPA, TMPs, ammonia, and chromium. In the groundwater, phthalates, TMPs, acetone, ammonia, and chromium are also detected in the shallow groundwater downgradient of Lake Poly. The following summarizes the potential environmental concerns associated with Lake Poly:

- Surface soils may act as a potential source due to potential erosion and transport of contaminants to the On-Property West Ditch.

- Subsurface soils may be a source to the groundwater based on the presence of elevated concentrations of the same parameters in these related media.

- Sediments in the adjacent On-Property West Ditch may be a source of contaminants to points downstream due to erosion, scouring, and potential transport of sediments down the Ditch System.

4.4.3 Drum Disposal Areas

The investigations in the drum disposal areas are discussed in detail in the CSA (CRA, 1993) and Section 1.3.5. The test pit program conducted as part of the CSA (CRA, 1993) confirmed the presence of buried drums or drum parts in three areas, Drum Area A, Drum Area B, and the Buried Debris Area. In general, the analytical results show that the highest concentrations of inorganics and organics are detected in Drum Area A and Drum Area B. The parameters and concentrations in the Buried Debris Area are much lower and appear to be associated with overflows from Lake Poly. Therefore, the two buried drum areas of concern are Drum Area A and Drum Area B.

An Imminent Hazard Evaluation for Drum Areas A and B was prepared in accordance with 310 CMR 40.0950 and submitted to MADEP in November, 1994 (ABB-ES, 1994). The evaluation concluded that the buried drum areas could pose an imminent hazard due to safety because the presence of Opex® (dinitrosopenta-methylenetetramine) and Kempore® (azodicarbonamide) may represent a threat of explosion or fire if subjected to heat, flame, or friction. The IRA Completion Report summarizing all the actions completed by Olin was submitted to MADEP on June 28, 1996 (ABB-ES, 1996). This report concluded that Olin has completed all components of the IRA

pursuant to 310 CMR 40.0424 of the MCP, that a condition of Substantial Release Migration does not exist for Opex® and Kempore®, and that No Imminent Hazard currently exists due to the buried drum areas on the Olin facility. Olin maintains the fencing surrounding the two buried drum areas and also the facility fencing to prevent access to these areas, thereby eliminating the need to continue the IRAs.

The results of subsurface investigations in Drum Area A show that waste or subsurface soils contain high concentrations of TMPs, NNDPA, and inorganics including ammonia, chloride, chromium, sodium, and sulfate. The surface soil results for Drum Area A show elevated concentrations of phthalates, NNDPA, and several inorganics, including aluminum, chromium, and potassium. In addition, the analytical results for well GW-35D, adjacent to the Drum Area A, show Kempore® in the groundwater at a concentration of 3.8 mg/l, which suggests that Kempore® is also present in the soils or waste material in this area. The following summarizes the potential environmental concerns associated with Drum Area A:

- Surface soils are a potential source due to potential erosion and transport of contaminates to the On-Property West Ditch.

- Subsurface soils may be a source to the groundwater based on the presence of elevated concentrations of the same parameters in these related media, although other nearby sources may also be responsible for the organics and inorganics detected in groundwater.

- Subsurface wastes may present a continuing hazard due to flammability, if disturbed.

The results of subsurface investigations in Drum Area B show that waste or subsurface soils contain elevated concentrations of acetone, 2-hexanone, 4-methyl-2-pentanone, bis(2-ethylhexyl)phthalate, phenol, and inorganics including chromium and sulfate. The surface soil results for Drum Area B show elevated concentrations of phthalates, phenols, NNDPA, and several inorganics including aluminum, ammonia, and chromium. The following summarizes the potential environmental concerns associated with Drum Area B:

- Surface soils are a potential source due to potential erosion and transport of contaminants to the Central Wetland, Central Pond, and South Ditch.

- Subsurface soils may be a source to the groundwater based on the presence of elevated concentrations of the same parameters in these related media, although other nearby sources may be responsible for the organics and inorganics detected in groundwater.

- Subsurface wastes may present a continuing hazard due to flammability, if disturbed.

4.4.4 Ditch System

4.4.4.1 Identification of Areas of Concern

As discussed in Section 1.3.3, the Ditch System is an interconnected drainage system that borders three sides of the facility and has a small fenced-in section offsite, to the west (Plate 2-9). These surface water bodies are named the South Ditch, On-Property West Ditch, Off-Property West Ditch, East Ditch, and Ephemeral Drainage.

The South Ditch and Off-Property West Ditch segments of the ditch system are distinct in that shallow groundwater flowing from the former surface impoundments has consistently discharged west and south to these segments of the ditch system. The South Ditch and Off-Property West Ditch are also discussed together in Section 4.4.4.2 due to the flocculent formation in these segments of the Ditch System and the corresponding elevated chromium concentrations.

Different transport mechanisms and source areas dominate the remaining segments of the Ditch System. The on-property portion of the West Ditch is mostly a groundwater recharge area and is thought to have been a receptor of prior releases or disposal activities at Lake Poly and is discussed in Section 4.4.4.3. The Central Pond, Central Wetland, and the surrounding area (grid Area 8) are not directly associated with the ditch system, but lie in close proximity to the South Ditch. These areas may be periodic contributors of contaminants to the South Ditch via surface pathways, but are hydrogeologically separate and are discussed in Section 4.4.4.4. The main area of concern associated with the Ephemeral Drainage is the surface soils near the GW-55 well nest (SWMU 33) and is discussed in Section 4.4.4.5. Surface water also leaves the facility in the East Ditch, and the main media of concern are sediments in the downstream portion of the East Ditch, which is discussed in Section 4.4.4.6.

4.4.4.2 Off-Property West Ditch and South Ditch

Olin has acted to contain contaminants and reduce risks associated with the Off-Property West Ditch and South Ditch. Potential public exposure to surface water and sediment in the Off-Property West Ditch were evaluated and a fence was erected to eliminate exposure. To prevent downstream migration of the chromium-bearing precipitate, Olin implemented an IRA as approved by the MADEP. A temporary weir has been placed across the South Ditch to increase the water level in the Off-Property West Ditch. The damming and ponding of water behind the weir has allowed the flocculent to settle out of the surface water. In addition, the increased surface water head minimizes the discharge of shallow groundwater to the ditches, and reduces the creation of additional flocculent. A hay bale filter dam is also in place further downstream across the South Ditch to prevent downstream migration of flocculent. The precipitate was vacuumed from the ditches upstream of the weir in June 1994 after the weir was built. The weir plate was installed in July 1994. The associated monitoring program was begun in April 1994, prior to installation of the weir. The location of the weir and ditch system are shown on Plate 2-9.

As discussed in Section 4.3, additional surface water and shallow groundwater data collected since the CSA (CRA, 1993), has shown that the inorganic concentrations in the Off-Property West Ditch and South Ditch have greatly decreased since the installation of the weir. This reduction in inorganics concentrations is attributed to clean ponded water entering the groundwater flow system, suppression of shallow groundwater discharge, and geochemical reactions that attenuate cationic metals such as chromium. The Supplemental Phase II Investigations have shown that the geochemical environment in the Ditch System surface water strongly favors the immobilization of dissolved chromium through precipitation and adsorption reactions. The Supplemental Phase II Investigations have also shown that the increased heads created by the weir have generally improved water quality. However, the increased heads have moved some inorganics and VOCs in the shallow groundwater further west and may have changed the location of the dominant groundwater discharge points along the South Ditch.

The South Ditch and Off-Property West Ditch are areas of environmental concern for shallow groundwater, surface water, and sediment. A related area of concern is the black oily material along the banks of the upper South Ditch (SWMU 30). Sediment results in both branches of the Ditch System and surface soils (SWMU 30) show elevated concentrations of chromium, sulfate, phthalates, and NNDPA, while 4-bromophenyl-phenylether and 4-chlorophenyl-phenylether were only detected in the South Ditch sediments. The CSA (CRA, 1993) identified elevated concentrations of ammonia, chromium, and sulfate in the surface water.

The following summarizes the potential environmental concerns associated with the Off-Property West Ditch and South Ditch:

- Shallow Groundwater in the vicinity of the Off-Property West Ditch and entire South Ditch appears to be a pathway for contaminant transport to surface water.

- Sediments in the Off-Property West Ditch and in the South Ditch are a potential source to downstream locations due to potential erosion and transport of contaminants downstream.

- Surface soils along the banks of the upper South Ditch (SWMU 30) are potential sources due to potential erosion and transport of contaminants into the Ditch System.

4.4.4.3 On-Property West Ditch

The On-Property West Ditch includes two areas of concern: the black area near the northern end of the On-Property West Ditch (SWMU 27) and the oily sediment within the On-Property West Ditch (SWMU 28). These areas are believed to be associated with Lake Poly based on the parameters detected, and on 1955 and 1967 aerial photographs which show that Lake Poly drained into the On-Property West Ditch and flooded the adjacent low lying area (SWMU 27). In general, the parameters detected in the SWMU 27 samples are similar to those found in Lake Poly, although VOCs and inorganics are detected at relatively lower concentrations.

The sediments in the On-Property West Ditch and adjacent surface soils (SWMU 27) have high concentrations of TMPs, phthalates, NNDPA, PAHs, and inorganics including ammonia, chloride, chromium, and sulfate. Pesticides were detected at low concentrations in sediments and surface soils in these areas. In the surface water, ammonia, phthalates, and VOCs are detected at low concentrations and sulfate is detected at moderate concentrations. The Supplemental Phase II Investigations have shown that the On-Property West Ditch is not a groundwater discharge area, and that NNDPA, phthalates, phenols, and pesticides are not detected in the surface water. Overall, the concentrations of inorganics and organics present in the On-Property West Ditch surface water are lower than the concentrations found in the South Ditch or Off-Property West Ditch surface water. The following summarizes the potential environmental concerns associated with the On-Property West Ditch:

- Sediments in the On-Property West Ditch are a potential source of contamination to downstream locations due to potential erosion and transport of contaminants downstream.

- Surface soils along the banks of the upper On-Property West Ditch (SWMU 27) are a potential source due to potential erosion and transport of contaminants into the Ditch System.

4.4.4.4 Central Pond, Central Wetland Area, and Surrounding Areas

The Central Pond and surrounding drainage areas lie within CSA (CRA, 1993) grid Area 8 and the Central Wetland area extends across CSA (CRA, 1993) grid Areas 8 and 9. The Central Wetland area is a distinct area of surface water drainage located in the central portion of the site that extends from the former location of the Acid Pits and Lined Lagoons to the eastern property line (Plate 2-8). The Central Wetland area coincides with an area of stressed vegetation present when Olin purchased the site. These areas appear to have received releases from the Acid Pits and Lined Lagoons. Site plans suggest that the Central Pond was connected to the South Ditch through a small drainage channel, and the analytical results for adjacent sediment samples from the South Ditch show elevated concentrations of the same chemicals detected in samples from the Central Pond and surrounding drainage areas.

The Central Pond sediment contains oils and has elevated concentrations of TMPs, NNDPA, bis(2-ethylhexyl)phthalate, 4-bromophenyl-phenylether, pesticides, and inorganics including aluminum, chromium, sodium, and sulfate. The Central Pond surface water has elevated levels of chromium and sulfate. The surface soils in the surrounding area (grid Area 8) have elevated concentrations of bis(2-ethylhexyl)phthalate, PAHs, pesticides, and inorganics, including chromium and sulfate. The highest concentrations of these parameters were observed in pockets of residual wastes that were typically found in the drainage areas that lead to the Central Pond. The Supplemental Phase II Investigations have shown that the surface soils in the Central Wetland

within grid Area 9 have elevated concentrations of 1,1,1-trichloroethane, bis(2-ethylhexyl)phthalate, and ammonia. The following summarizes the potential environmental concerns associated with the Central Pond, CSA (CRA, 1993) grid Area 8, and the Central Wetland:

- Sediments in the Central Pond and Central Wetland may be a potential source of contamination to downstream locations due to potential erosion and transport to the Ditch System.

- Surface soils associated with the Central Pond, CSA (CRA, 1993) grid Area 8, and the Central Wetland may be a potential source due to potential erosion and transport to the Ditch System.

4.4.4.5 Ephemeral Drainage

The media of concern at the Ephemeral Drainage area include surface water, sediment, and surface soils in the area near the GW-55 well nest (SWMU 33). The Ephemeral Drainage sediment and surface water have elevated concentrations of sulfate. The surface soils in the area near the GW-55 well nest (SWMU 33) have elevated concentrations of bis(2-ethylhexyl)phthalate, pesticides, chromium, and sulfate. The shallow groundwater samples from wells in the western and eastern portions of the Ephemeral Drainage show elevated concentrations of ammonia, chloride, and sulfate. The deep groundwater beneath these portions of the Ephemeral Drainage show elevated concentrations of VOCs, phthalates, ammonia, chloride, and sulfate, while the area near the GW-55 well nest also show elevated concentrations of chromium and pesticides. The parameters detected at elevated concentrations in these areas are believed to be unrelated to the sediments, although the shallow groundwater may contribute chemicals to the surface water. The following summarizes the potential environmental concerns associated with the Ephemeral Drainage:

- Sediments in the Ephemeral Drainage may be a potential source of contamination to downstream locations due to potential erosion and transport to the Ditch System.

- Surface soils along the banks of the upper Ephemeral Drainage (SWMU 33) may be a potential source due to potential erosion and transport to the Ditch System.

- Shallow and deep groundwater in the uppermost and lowermost Ephemeral Drainage may be a potential pathway or source of contamination to the surface water.

4.4.4.6 East Ditch

The East Ditch sediments have elevated concentrations of TMPs, NNDPA, phthalates, PAHs, and inorganics including ammonia, chloride, chromium, sodium, and sulfate. The East Ditch surface water has elevated concentrations of TMPs, chromium, and sulfate. The CSA (CRA, 1993) concluded that the presence of toluene in the surface water and trichloroethene, 1,2-dichloroethene, methylene chloride, and chlororethene in sediments in the East Ditch are not associated with the facility and are from an off-site source. The East Ditch is channelized or culverted along most of its length and therefore, as discussed in the Ecological Risk Characterization presented in Appendix S, the habitat value is low and no adverse risk exists. The following potential environmental concerns are associated with the East Ditch:

- Sediments in the East Ditch may be a potential source of contaminants to downstream locations due to potential transport and erosion of contaminated sediments

4.4.5 East and West Pits

The history of the East and West Pits is discussed in detail in Section 1.3.1. The East and West Pits at the facility were formerly located in the southern portion of the manufacturing area and are beneath the East and West Warehouses. Historical aerial photographs show that these unlined pits were constructed after 1955, and site plans and building permits from 1964 and 1965 indicate that the East and West Pits were probably backfilled in about 1964 to 1965, coinciding with construction of the East and West Warehouses. It is believed that liquid acidic wastes containing chromium sulfate were discharged into the East and West Pits.

The area of the East and West Pits also encompasses four Solid Waste Management Units (SWMUs) identified in the CSA (CRA, 1993): the sodium hydroxide tank unit (SWMU 5), the former wastewater treatment plant (SWMU 8) and sump (SWMU 31), and the Wytox spill area (SWMU 24). These SWMUs are considered separately in Section 4.5. Due to cover of the East and West Pits by the warehouses and other structures, investigations in the area of the East and West Pits have been limited to a soil vapor survey and test pit program to locate buried drums beneath the Warehouses. No visibly contaminated soils were observed in the test pits and no soil samples were collected, although the soil vapor survey for total volatile organics suggests that VOC contamination is present in the soils. The analytical results from subsurface soil borings and groundwater wells around the perimeter of the warehouses discussed below provide a good indication of the parameters and concentrations that may be present beneath the warehouses.

Subsurface soil samples from two CSA (CRA, 1993) soil borings (BH13 and BH20) immediately south of the East and West Pits detected elevated chromium concentrations. As reported in the CSA (CRA, 1993), the groundwater in the area around the East and West Pits is mainly impacted by inorganics, although higher levels of organics are observed near Lake Poly and Drum Area A. The shallow groundwater in this area contains elevated concentrations of trimethylpentenes,

acetone, ammonia, chloride, chromium, manganese, and sulfate. It is unknown whether the East and West Pits are also a source of these chemicals. The following summarizes the potential environmental concerns that may be associated with the East and West Pits:

- Surface soils are not a potential source due to coverage of this area with buildings and other impervious surfaces.

- Subsurface soils may be a source due to the presence of elevated concentrations of the same parameters in subsurface soils and the groundwater in the vicinity of the East and West Pits.

4.4.6 Acid Pits and Lined Lagoons

4.4.6.1 Acid Pits

The history of the three unlined Acid Pits is discussed in detail in Section 1.3.1. The Acid Pits (SWMU 15) at the facility were formerly located in the southern portion of the manufacturing area and the locations of this series of three pits coincided with and pre-dated the Lined Lagoons. It is believed that these acid pits were constructed in 1964 and 1965 and were replaced by the Lined Lagoons in about 1972. It is also believed that liquid acidic wastes containing chromium sulfate were discharged into the Second Acid Pits until 1967, when the use of sodium dichromate in the Kempore® process ceased.

The results of subsurface soil investigations show that elevated concentrations of TMPs, acetone, phthalates, NNDPA, ammonia, and chromium are detected in the subsurface soils in the vicinity of the Acid Pits. It is difficult to associate specific parameters detected in the subsurface soils in the vicinity of the Acid Pits with groundwater results for surrounding wells due to the proximity of other potential source areas and the presence of a groundwater divide in this area. The Acid Pits are downgradient from the East and West Pits, and are co-located with the Lined Lagoons and Drum Areas. However, the higher concentrations of TMPs, acetone, phthalates, ammonia, and chromium are detected downgradient of the Acid Pits. The following summarizes the potential environmental concerns that are associated with the Acid Pits:

- Subsurface soils may be a potential source due to the presence of the same parameters in both the subsurface soils and groundwater.

4.4.6.2 Lined Lagoons

The history of the Lined Lagoons is discussed in detail in Section 1.3.1. Lagoon I (SWMU 9) and Lagoon II (SWMU 10) were constructed in 1972 and were originally lined with a polyvinyl chloride (PVC) liner. The acid waste streams in the plant were neutralized with lime hydroxide and the resulting slurry was sent to the Lined Lagoons. Periodically, this material was dredged and the calcium sulfate sludge was sent to the Calcium-Sulfate Landfill. Olin relined Lagoon I in around 1981 and Lagoon II in about 1983. At this time, the lagoons were lined with a 36-

millimeter Hypalon liner resting on 1-foot of compacted sand. Because the lined lagoons received mainly sulfate-rich inorganic (with no appreciable chromium) effluent neutralized with lime, only inorganics, mainly calcium and sulfate, are the main parameters associated with the lined lagoons. The lagoons and the Calcium Sulfate Landfill were closed after 1986, when operations at the facility were ceased.

The results of subsurface soil investigations show that soils are impacted by high concentrations of calcium and sulfate. The shallow groundwater results show elevated ammonia, chloride, and sulfate concentrations downgradient of the Lined Lagoons, although concentrations have decreased considerably since closure of the Lined Lagoons. This data suggests that the lagoons were a source of some inorganics such as ammonia, calcium, chloride, sodium, and sulfate and that closure of Lined Lagoons removed most of this source. Recent shallow groundwater monitoring data from wells downgradient of the Lined Lagoons and Acid Pits continue to show elevated concentrations of inorganics which suggests that aquifer flushing is incomplete and a residual soil source may still be present. As discussed in Section 4.4.4.4, the elevated concentrations of sulfate in surface soils in the drainage areas northwest of the Central Pond are probably also associated with releases from the Lined Lagoons. The following summarizes the potential environmental concerns that are associated with the Lined Lagoons:

- Subsurface soils were largely excavated during lagoon closure, residual calcium sulfate in soils may be a source of inorganics based on the presence of the same parameters in both the subsurface soils and shallow groundwater.

4.4.7 Calcium-Sulfate Landfill

The history of the Calcium-Sulfate Landfill is discussed in detail in Section 1.3.1. The disposal of calcium sulfate in the Calcium-Sulfate Landfill began in January 1975 in accordance with the design approved by the MADEP in October 1974. When Olin shut down operations in 1986, part of the shutdown included removal of the two Lined Lagoons. All calcium sulfate sludge and liners were removed to the Calcium-Sulfate Landfill and waste placement ceased in December 1986. The closure was completed by capping with the placement of a minimum total cover thickness of one foot including a minimum of three inches of topsoil over a minimum of nine inches of low permeability (1×10^{-6} centimeters per second [cm/sec]) material.

The CSA (CRA, 1993) soil analytical data show that surface soils in the area have low concentrations of phthalates and inorganics, and that subsurface soils/wastes, as expected, have high concentrations of ammonia, calcium, and sulfate. The surface water and sediment of the nearby Ephemeral Drainage have low to moderate concentrations of sulfate and phthalates. The groundwater analytical results for wells surrounding the Calcium Sulfate Landfill show that only inorganic compounds are consistently detected in the groundwater. Elevated concentrations of sulfate have been detected in well SL-03, which monitors the southern side of the landfill and elevated concentrations of ammonia, chloride, sulfate have been detected in wells SL-05 and SL-06 which also monitor the southerr side of the landfill. The following summarizes potential environmental concerns that may be associated with Calcium-Sulfate Landfill:

- Surface soils are not acting as a source of contaminants due to the soil cap on the landfill.

- Subsurface soils/wastes may acting as a source to shallow and deep groundwater due to presence of similar analytes in these media.

4.4.8 Plant D and Surrounding Area

The CSA (CRA, 1993) identified subsurface soils contaminated with SVOCs in the vicinity of the former Plant D and the associated SWMUs, the Plant D Drum Storage Area (SWMU 1), the Plant D Byproduct Tank of ammonia hydroxide (SWMU 4), and the low-lying black area where soils were removed near Plant D (SWMU 26). Surface soil results for these areas are represented by the CSA (CRA, 1993) grid Areas 6, 7, and 8 composite samples and samples collected from SWMU 26 which show elevated concentrations of phthalates, PAHs, gamma-BHC (lindane), and sulfate. The results of subsurface soil investigations detected elevated concentrations of TMPs and sulfate at SWMU 26 and elevated concentrations of ammonia in subsurface soils at the former Plant D storage tank area. The surface soil results show that Plant D soils are impacted by low levels of chromium, moderate concentrations of ammonia, VOCs, and phthalates. The following summarizes the potential environmental concerns that may be associated with the Plant D and the area to the east (SWMU 26):

- Surface soils are not likely acting as a source due to the lack of transport pathways from the low lying area at SWMU 26 to receptors such as the East Ditch.

- Subsurface soils may be releasing ammonia to groundwater which is then transported to the East Ditch.

4.4.9 Groundwater at GW-54 Well Cluster

The shallow and deep groundwater results for the GW-54 well cluster show elevated concentrations of TMPs, NNDPA, chromium, and sulfate. The concentration of these parameters are generally higher in the deep groundwater. Other indicator parameters, including chloride and ammonia, were detected at relatively low concentrations. Trace levels (<0.002 ug/l) of phthalates, PCBs, and pesticides were sporadically detected in groundwater samples from GW-54S and GW-54D. No subsurface soil samples were collected in the immediate vicinity of the GW-54 well cluster, however subsurface soil samples from BH-26 adjacent to Plant C-1 also show elevated concentrations of TMPs, NNDPA, chromium, and sulfate. Other potential sources include Lake Poly and leaks from the associated wastewater lines, the East and West Pits, spills or other surface releases, or the nearby inactive tile drainage field discussed in Section 4.5.6. The following summarizes the potential environmental concerns associated with the GW-54 well cluster:

- An unknown subsurface soil source near or upgradient of the GW54 well cluster may be releasing TMPs and NNDPA to groundwater.

- Shallow groundwater at GW-54 may discharge to the surface water in the Off-Property West Ditch or may be transported at levels of concern off-property to the north and west, dependent on temporal changes in groundwater flow.

4.4.10 Deep Groundwater and Dense Layer

4.4.10.1 Deep Groundwater

The areas with elevated concentrations of site parameters in the deep groundwater (outside of the dense layer) are summarized below. The elevated concentrations of the below parameters in the deep groundwater near Plant B, Plant D, Lake Poly, and the Calcium-Sulfate Landfill coincide with elevated concentrations of these parameters detected in subsurface soil samples. The elevated concentrations at the GW-50D may be related to the flow of deeper groundwater (with higher concentrations) over the bedrock saddle which separates the Western Bedrock Valley from the Eastern Bedrock Valley. The elevated parameter concentrations in the lower Western Bedrock Valley are also attributed to the flow of deeper groundwater (with higher concentrations of ammonia, chloride, and sulfate) over a bedrock saddle which separates the bedrock low at GW-83D from the lower Western Bedrock Valley.

	Areas with Elevated Concentrations in the Deep Groundwater					
Parameters	Plant B Area	Plant D Area	Calcium Sulfate Landfill	Lake Poly Area	GW-50 Well Cluster	Lower Western Bedrock Valley GW-84D, 85D, 86D, and 87D
Ammonia	X		X	X	X	X
Chloride			X	X	X	X
Chromium				X		
Sulfate		X	X	X	X	X
Phthalates	X			X		
Phenol	X					
NNDPA	X	X	X	X	X	

The following summarizes the potential environmental concerns associated with deep groundwater:

- Contaminants from the deep groundwater at the Lower Western Bedrock Valley could potentially contribute solutes to the groundwater captured by the municipal supply wells.

4.4.10.2 Dense Layer

The dense layer is the result of the liquid waste material that was disposed into the unlined pits, lagoons, and Lake Poly. The extent of the dense layer is shown on Plate 2-10. The dense layer is an acidic aqueous liquid with high concentrations of dissolved solids consisting mainly of ammonia, chloride, chromium, sulfate, and sodium, accompanied by low concentrations of organics. These organics were components of the liquid wastes or were entrained in the dense layer as it flowed through areas with organics in the deep groundwater. The high specific gravity (1.03 to 1.11 g/cm^3) of the dense layer allowed it to sink through the overburden and groundwater until it reached a less permeable barrier, the top of the bedrock.

Investigations have shown that the greatest mass of inorganic contaminants resides in the dense layer. The distribution of the dense layer and the inorganics are controlled by the shape of the bedrock valleys and the gravity-driven flow of the dense layer to its current resting place in bedrock lows. Further characterization of the dense layer and the overlying shallow groundwater has shown that the contact is sharp and that the overlying shallow groundwater contains inorganic concentrations that are generally 100 to 1,000 times less than that found in the dense layer. Groundwater and geochemical modeling have concluded that the slow diffusion of inorganics from the dense layer is the main pathway for contaminant transport from the dense layer, and that the mobility of metals such as chromium are limited by precipitation and adsorption reactions.

The following summarizes the potential environmental concerns associated with the dense layer:

- Contaminants released from the dense layer by slow diffusion may eventually be transported into the groundwater system and subsequently discharged to surface water or potentially be captured by groundwater users such as the public supply wells.

- The concentrations of chromium in samples from the dense layer in the upper Western Bedrock Valley exceed the UCLs for groundwater.

- Significant changes in the existing groundwater flow system may affect the stable location of the dense layer, so that it has the potential to move in the future.

4.5 MISCELLANEOUS AREAS OF CONCERN

The remaining areas not discussed above in Section 4.4 are primarily the miscellaneous SWMUs identified in the CSA (CRA, 1993). These areas are impacted by low concentrations of site related chemicals and do not represent significant source areas. Generally, these areas are subject to response actions by Olin under the MCP or other regulatory programs.

Miscellaneous Areas of Environmental Concern

- Subsurface soils at Southwest corner of Warehouses
- Other miscellaneous localized areas (most denoted as SWMUs in the CSA (CRA, 1993))

4.5.1 Subsurface Soils at Southwest Corner of Warehouses

The results of subsurface soil samples collected from a depth of 8 to 10 feet at boring BH-15 show elevated concentrations of VOCs which are not typically detected at elevated concentrations in other subsurface soil samples collected across the site. This boring is located next to the southwest corner of the West Warehouse and is adjacent to an area of spills at the former Wytox handling area (SWMU 24) where soils were previously removed. The VOCs at elevated concentrations include ethylbenzene (2.3 mg/kg), styrene (3.3 mg/kg), TMPs (6.0 mg/kg), and toluene (4.8 mg/kg). Although this boring is located adjacent to Lake Poly, low concentrations of inorganics were detected in the sample. The detections of the above VOCs appear to be due to an isolated spill and do not appear to be related to releases at Lake Poly or the Wytox handling area (SWMU 24).

4.5.2 Hydrochloric Acid Tank (SWMU 3)

The hydrochloric acid tank (SWMU 3) consisted of a 10,000 gallon fiberglass above ground storage tank (AST) that contained a 33 percent solution of hydrochloric acid. This AST was located at Plant C on a concrete pad and within a concrete dike. Olin submitted closure plans for the acid tank to the MADEP and USEPA in April 1986. The closure of the acid tank was completed in 1986 in accordance with the closure plan which specified removal and disposal of remaining inventory and decontamination of the AST and the concrete pad.

4.5.3 Byproduct Ammonium Hydroxide Tanks (SWMUs 4 through 7)

These above ground storage tanks were formerly used as byproduct holding tanks in the Kempore® process to store ammonium hydroxide prior to neutralization and discharge to the publicly owned treatment works (POTW). All of these ASTs were constructed of carbon steel and were contained on a concrete pad and within a concrete dike. There are no known releases from these ASTs and all of the tanks were removed, with the exception of the ASTs at Plant B which is currently used as part of the ongoing groundwater treatment system in this area. The closure of the remaining ASTs was completed in 1986 in accordance with the closure plan submitted to the MADEP which specified removal and disposal of remaining inventory and decontamination of the AST and the concrete pad. The status and locations of the ASTs is listed below.

SWMU 4: Former Plant D Tank Farm - 8,000 gallon AST (removed)
SWMU 5: Former treatment plant - 10,000 gallon AST (removed)

SWMU 6: West of Plant C-3 - 15,100 gallon AST (removed)
SWMU 7: Former Plant B Tank Farm - 15,000 gallon AST (reused)

4.5.4 Wastewater Treatment Plant (SWMU 8 and SWMU 31)

The former wastewater treatment plant (SWMU 8) consisted of seven ASTs, one sump, and two settling ponds (Lined Lagoons). The acid wastewater streams were neutralized with lime and sent to the lined lagoons to allow settling of the calcium sulfate slurry. Supernatant was returned to a clarifier and discharged to the POTW. Other process streams were pre-treated for hydrochloric acid or ammonium hydroxide and then sent to the clarifier before discharge to the POTW. The closure of the remaining ASTs was completed in 1986 in accordance with the closure plan submitted to the MADEP which specified removal and disposal of remaining inventory and decontamination of the AST and the concrete pad. The Lined Lagoons were closed in 1986 which included removal of the calcium sulfate sludges, liners, and visibly contaminated soils for disposal at the Calcium-Sulfate Landfill. The wastewater treatment plant sump (SWMU 31) is scheduled to be cleaned and closed in 1997.

4.5.5 Buried Drums in Vicinity of Warehouses (SWMU 16 and SWMU 18)

There are two areas reportedly used prior to Olin's ownership of facility (1953 through 1980) for drum disposal in the vicinity of the East and West Warehouses

The magnetometer survey conducted in December 1991 through January 1992 as part of the CSA (CRA, 1993) and described in the "Geophysical Investigation Report" (CRA, 1993 - Appendix B) did not locate any anomalous areas around the warehouses. The soil vapor survey performed beneath the floor of the two warehouses in February 1992 as part of the CSA (CRA, 1993) and described in the "LGI Soil Gas Survey Report" (CRA, 1993 - Appendix C) located three areas which could potentially be indicative of buried wastes. The soil gas survey was followed by a series of nine test pit excavations to depths of up to 12 feet below grade in the anomalous areas. Based on visual inspection and field screening of soil samples for total volatile organics, no samples exhibited evidence of contamination. The complete details of the magnetometer survey, soil gas survey, and test pit program are contained within the CSA (CRA, 1993).

4.5.6 Inactive Septic Tanks and Tile Fields (SWMU 20 and SWMU 21)

These SWMUs were used and closed during the period in which Stephan Chemical Company and/or its predecessors National Polychemicals Inc. owned and operated the facility (1953 through 1980). One septic tank and tile field was located south and northwest, respectively, of Plant A and one septic tank and tile field was located east of Plant B (SWMU 21). The available files and records show that these septic tanks and tile fields handled sanitary wastes. Since these drawings and records were prepared prior to Olin's acquisition of the property, Olin cannot attest to the operation or design of these units.

4.5.7 PCB Capacitor (SWMU 22)

This SWMU was an electrical capacitor located at the northeast corner of the site. On August 24, 1985 the capacitor ruptured, spilling oils containing PCBs, and a minor fire erupted. The capacitor was subsequently removed to an appropriate offsite facility for disposal and corrective actions were taken to cleanup the spilled material and surrounding area. An emergency contractor (Clean Harbors, Inc.) performed the cleanup under oversight of the MADEP.

4.6 GROUNDWATER NEAR MAPLE MEADOW BROOK WELLFIELD

There are five wells comprising the Maple Meadow Brook wellfield: Butters Row 1, Butters Row 2, Chestnut Street 1, Chestnut Street 2/1A, and Town Park. The available historical analytical results include annual water supply analyses from 1986 through 1990 for inorganics including sulfate, chloride, and ammonia. Olin sampled all five wells as part of a toxicity assessment in September 1992. The toxicity assessment found that all organic compounds detected were below drinking water standards for public water supplies prior to treatment. Most of the inorganic compounds were also below Government standards with the exception of iron, manganese, and sodium which occur naturally at elevated concentrations. However, the concentration of these inorganics were three orders of magnitude below the dose established by toxicological studies and would have no adverse effect to health.

In addition, groundwater samples have been collected regularly since February 1992 and quarterly since 1994 for the site-specific indicator parameters: chloride, sulfate, chromium, and ammonium. Additional inorganic parameters, calcium, magnesium, sodium, and alkalinity have been analyzed since 1994 to provide quality assurance on the analytical results. The monitoring program was expanded in July 1996 to monthly sampling which also include the network of monitoring wells surrounding the production wells. The available analytical results are presented in Table 2-23, and graphs depicting the concentration of selected indicator parameters over time are provided in Figures 2-30 through 2-34. As discussed in Section 2.5.6, the data collected between July 1996 and April 1997 have shown that the concentrations of indicator parameters are within the range of concentrations historically detected in samples from the production wells.

Implementation of the June 1996 Groundwater Monitoring Program for the Western Bedrock included installation of additional wells in the Western Bedrock Valley. The data from these wells show that groundwater contamination is present beneath Maple Meadow Brook. However, it is significantly restricted to the deeper portions of the aquifer, and does not seem to be a source of high concentrations of contaminants to the shallower portions of the aquifer. In April 1997, a number of these wells have been added to the monthly sampling program to provide additional information on the water quality in the wellfield.

Other investigations to quantify the factors which may affect indicator parameter concentrations in the water supply wells are ongoing. As the program continues throughout the seasons, it is expected that the data can be evaluated for the presence of trends and that conclusions can be drawn. Final conclusions will be summarized in a separate report to be submitted to the MADEP at the end of one year of monitoring (September 1997).

Based on the data collected to date, the existing conditions within the wellfield do not appear to be significantly different than that observed over the last five years. In addition, unless major changes to the regional groundwater flow system take place, there is no reason to anticipate significant changes in water quality within this wellfield. A monitoring program is in place that will identify changes in water quality. A major data evaluation will be performed in September 1997 that will provide additional understanding of the relative importance of factors influencing water quality. This evaluation will be used to re-define the existing monitoring program, if necessary.

4.7 DELINEATION OF DISPOSAL SITE

The results of the Phase II Comprehensive Site Assessment (CRA, 1993) and the Supplemental Phase II Investigations were used to define the Disposal Site boundary shown on Figure 4-4. The extent of the Disposal Site is mainly controlled by the concentration of indicator parameters in the shallow groundwater, deep groundwater, and dense layer. The boundary of the Disposal Site also reflects the presence of other site contaminants in the surface water and sediment on- and off-property and the presence of site contaminants in the surface and subsurface soils on-property.

In general, the highest concentrations of inorganics were detected in samples from the dense layer and therefore, the Disposal Site includes the areas with bedrock lows where the dense layer is present. The limits of the area where the groundwater is impacted by site contaminants was determined by comparing the groundwater analytical results to the maximum background groundwater concentrations for the site indicator parameters (ammonia, chloride, chromium, and sulfate) which are presented in Section 4.1. Since elevated concentrations of chloride and ammonia could be related to other sources (e.g. septic systems, road salt, other disposal sites), the presence of two or more indicator parameters above maximum background groundwater concentrations were used to define groundwaters that lie within the Disposal Site.

The boundary of the Disposal Site shown on Figure 4-4 includes most of the Olin property with the exception of small areas in the northwestern, southeastern, and southwestern portions of the property. These areas are bedrock highs where groundwater is not impacted by appreciable concentrations of site contaminants.

The Disposal Site west of the property is defined by the presence of elevated inorganics in the dense layer and surrounding groundwaters. This area closely follows the Western Bedrock Valley and extends from the property west to the Maple Meadow Brook wetlands. The evaluation of analytical results concluded that the following off- property areas lie within the Disposal Site:

- The Disposal Site west of the property is defined by the presence of elevated inorganics in the dense layer and surrounding groundwaters. This area closely follows the Western Bedrock Valley and extends from the property west to Maple Meadow Brook wetlands.

- The East Ditch surface water and/or sediment extending from upstream sampling location SW-1 to beyond downstream sampling location SW-28.

- The small area immediately north of the property was mainly included in the Disposal Site due to the presence of low concentrations of organics (trimethylpentenes and phthalates) detected at East Ditch sampling location SW-1 and at monitoring wells (GW-31S/D and GW-32D) located at the north end of the property.

- The narrow area southwest of the Calcium-Sulfate Landfill was included in the Disposal Site due to the presence of inorganics (ammonia and sulfate) in the deep groundwater. The extent of this area was defined by a bedrock valley that begins at well GW-40S/D and extends past GW-75S/D. The Disposal Site boundary in this area is estimated (shown with a dashed line) since the elevated concentration of ammonia detected at GW-75D is not accompanied by elevated concentrations of other indicator parameters. Other nearby potential sources for ammonia may include the inactive municipal landfill for the City of Woburn.

5.0 CONTAMINANT FATE AND TRANSPORT CHARACTERISTICS

MCP 40.0835(4)(e)

CHAPTER SUMMARY

To properly evaluate the potential impacts of contaminants, it is necessary to understand both their fate and their potential to be transported. Fate is the way in which specific contaminants interact with their environment, such as the extent to which they are degraded or otherwise broken down, and the extent to which they are involved in certain kinds of reactions. Transport considers their ability to be moved via wind, flowing water, and other mechanisms. The Comprehensive Site Assessment (CSA) discussed both fate and transport aspects of the contaminants detected at the site (CRA, 1993, Appendix K). The Supplemental Phase II investigations provided additional information about the potential movement of the dense layer, and the extent to which geochemical effects influence the fate of inorganic contaminants.

This chapter brings together information relating to fate and transport of site contaminants. The key findings of this section are:

- Migration of contaminants via surface water and sediment transport is limited to the boundaries of the respective water body.

- Migration of shallow groundwater away from the Olin property is limited. Much of this groundwater is discharged locally to the East Ditch, South Ditch, and Off-Property West Ditch. A small amount of groundwater from the northern portion of the property by-passes these ditches, but is likely to be captured by the Altron supply wells.

- The bedrock formation acts as a barrier to dense layer and groundwater flow and does not appear to be acting as a significant pathway for contaminant migration.

- The mass transfer process (diffusion) whereby contaminants are moved vertically from the dense layer up into the overlying groundwater is a long, slow process that does not transfer significant concentrations of contaminants. However, this diffusion process appears to be the dominant mechanism by which contaminants are moved upwards from the dense layer. Therefore, although the dense layer will continue to provide a source of contamination to the groundwater, it is unlikely that the resultant concentrations will be significantly greater than currently observed.

- The geochemistry of the groundwater system aids in reducing mobility of certain metals, including chromium, aluminum, iron, and manganese. The solubility of these metals is controlled by precipitation of oxyhydroxide solids. For example, above a pH of 5.8, chromium is unlikely to be present in groundwater at concentrations above 0.1 milligrams per liter (mg/l), because precipitation of chromium hydroxide removes it from solution.

- The geochemical conditions in the Off-Property West Ditch and South Ditch tend to reduce the downstream migration of chromium. Upon discharge from the groundwater, the chromium is readily removed from the water column via precipitation to various oxyhydroxide and sulfate solids, thus creating the observed chromium-bearing flocculent. The existing controls, which include the weir and several sets of hay bales, serve to minimize off-site transport of the flocculent material.

- Groundwater flow modeling indicates that the municipal supply wells receive their water from rainfall, the wetlands, and portions of the aquifer which are not impacted by site contamination, with a smaller volume of water coming from contaminated portions of the aquifer. Between the Olin property and Main Street, much of the groundwater is captured by Altron's industrial supply wells.

- The results of the modeling suggest that there is little potential for future movement of the dense layer in the absence of significant changes to the existing groundwater flow system. The modeling results further suggest that the dense layer has not changed position appreciably since the mid-1980s.

- The results of the transport modeling indicate that transport of dissolved chromium from the dense layer to the municipal supply wells does not occur due to precipitation of chromium-iron hydroxides. Future concentrations of other inorganics (such as ammonia, chloride, sodium, and sulfate), which are know to be transported to the municipal supply wells, were predicted. The results of which give estimates of solute concentrations at or above historical observations. However, these predications are extremely conservative and ongoing monitoring of the surrounding sentinel wells will provide forewarning should any significant changes in groundwater quality and migration occur.

The documentation of these findings is presented in the sub-sections of this chapter as outlined below:

- Migration of contaminants, including migration pathways and fate processes, is discussed in section 5.1 Specific discussion about effective diffusion from the dense layer and geochemical considerations impacting fate and transport are provided.

- Groundwater flow and contaminant transport modeling are discussed in Section 5.2. The results of the groundwater flow modeling are provided in Section 5.2.1 and Appendix P; the results of the contaminant transport modeling are provided in Section 5.2.2 and Appendix Q.

5.1 MIGRATION OF CONTAMINANTS

As described in Section 4 above, contamination is present at the Olin site in wastes, surface and subsurface soils, groundwater, and surface water and sediment. A variety of types of releases resulted in the presence of these contaminants. In addition to spills and releases from chemical storage or handling areas, liquid wastes were disposed in unlined pits including the Lake Poly Liquid Waste Disposal Area (Lake Poly), and were allowed both to seep into the subsurface and to overflow onto the nearby ground surface or the ditches. In addition, the presence of buried drum debris indicates that certain wastes were disposed in the subsurface. The Calcium-Sulfate Landfill was constructed to contain wastes dredged from the lined lagoons.

Migration from the original source areas has resulted in an extended area affected by contamination. The classes of contaminants present in the environment consist of inorganics (including metals), volatile and semivolatile organics, and oily material. In general, it is possible for all of the contaminants in each of the media to be affected by their presence in the natural environment, and potentially to move via natural processes. The effects of these processes is discussed in more detail below.

5.1.1 Migration Pathways

The matrix in Figure 5-1 provides an overview of the transport mechanisms that enable contaminants to migrate from source areas into other media and other areas. Wastes at the site originally entered the environment by being spilled or released during storage and handling, or by being directly disposed in the environment, for example, drums buried in the subsurface, or liquid wastes discharged to unlined pits and ponds. In addition, calcium sulfate sludges were disposed in the Calcium-Sulfate Landfill. These activities had the potential to impact surface and subsurface soils and groundwater. Contaminants then moved from these media into other media as shown on Figure 5-1. For example, overflow of Lake Poly deposited wastes into the West Ditch, impacting surface water and sediment. Migration of groundwater carried certain contaminants to downstream receptors, such as the South and East Ditches, also affecting surface water and sediment. The potential migration pathways are shown on Figure 5-1. Infiltration is the process where contaminants located in unlined impoundments percolate downward into the subsurface. Leaching concerns the partitioning of contaminants between solids and liquids; as water (liquid phase) flows through contaminated soils or sediment (solid phase), contaminants will be leached from the solid to the liquid in accordance with partitioning relationships. The process of partitioning is described in more detail below. The ultimate individual potential receptors of site-related contaminants, both human and ecological, are described in detail in the risk assessment in Section 6.0.

The impact of contaminants on surface and subsurface soils is generally expected to be limited to the Olin property and the immediate vicinity. Impacted surface water and sediment are contained within the bounds of the affected stream. However, migration of groundwater and the dense layer is more extensive and complex.

Groundwater

The migration of contaminants in groundwater outside the dense layer is controlled by the hydraulic gradient. Directions of flow and gradients can be determined from the groundwater contour maps in Plates 2-3 through 2-6. In general, the movement of shallow groundwater away from the Olin property is limited. Much of this groundwater is discharged locally to the ditch system, either the East Ditch, the South Ditch, or the Off-Property West Ditch. A portion of the groundwater from the northwestern portion of the property by-passes the Off-Property West Ditch. This groundwater appears to be largely captured by the industrial pumping wells on the Altron property. As a result, the water quality of the shallow groundwater in the vicinity of Maple Meadow Brook is consistent with background, indicating that contaminants are not discharging to the wetlands in this area. Groundwater from the southwestern corner of the property, beneath a portion of the Calcium-Sulfate Landfill, appears to flow to the southwest.

Movement of contaminants into or through the bedrock formation is a possible alternative pathway for groundwater contamination. There are eight bedrock monitoring wells located throughout the study area (BR-1, GW-61BR, GW-62BR, GW-62BRD, GW-65BR, GW-68BR, GW-80BR and GW-81BR). Most of these wells are located within the Western Bedrock Valley. Concentrations of indicator parameters in GW-62BR indicate that the water quality in this well has been impacted by Olin-related contaminants. Because the dense layer is located at the top of the bedrock surface and its movement is controlled by density, it is expected that the dense layer could enter any fractures at the bedrock surface located within the limits of the dense layer. However, it is unlikely that the bedrock is acting as a significant pathway for migration of contamination, for the following reasons:

- The bedrock mapping, examination of cores, and downhole geophysics (Section 2.1) shows that many of the fracture zones are sealed or in-filled with clay weathering products. Permeability testing (Section 2.2 and the CSA) shows that the bedrock has an extremely low permeability. The bedrock formation lacks significant water-bearing zones. The low permeability and minimal storage capacity of the bedrock suggests that its ability to transmit significant amounts of contamination is limited. This is supported by the fact that concentrations of indicator parameters in the bedrock wells within the Western Bedrock Valley (excepting the GW-62 wells) are, at worst, only slightly higher than background and not at all typical of the dense layer.

- Three of the bedrock wells, GW-65BR, GW-80BR, and GW-81BR were located at the limits of the bedrock valleys to evaluate whether contamination may be leaving these valleys via migration through the bedrock. Concentrations of indicator parameters within these wells are typical of background water quality (see Table 2-22), again suggesting the limited ability of the bedrock to transmit significant contamination.

- A number of domestic water supply wells are located in the bedrock formation (see Section 3.4.6). These wells are typically deep (90 to 705 feet). Regular sampling of these wells has not show any evidence of impact from contamination related to the Olin property.

Dense Layer

As previously discussed, movement of the dense layer is controlled by the shape of the bedrock surface, moving downslope along the interface between the sand and gravel aquifer and the less permeable bedrock. Relative migration of individual parameters within the dense layer is affected by the geochemical reactions between the constituents and the aquifer matrix. The data obtained in the Supplemental Phase II investigations do not suggest that the dense layer is continuing to migrate at any significant rate. These data include:

- Borehole velocity monitoring in the dense layer (Section 2.2.4) shows low velocities; the data for wells in the dense layer are frequently inconsistent and non-reproducible, suggesting that flow rates are below the resolution of the instrument.

- Specific conductivity profiling (Section 2.6.1) between 1993 and 1995 has not shown any consistent trends or changes in the location of the dense layer interface around the periphery of the dense layer.

- Borehole inductance logging (Section 2.6.4) throughout the dense layer between 1992 and 1996 has not shown any consistent trends or changes in the location of the dense layer interface.

- Water quality within the dense layer has not shown significant changes since the CSA sampling in 1992.

It is possible that the dense layer is continuing to move, but very slowly, and that monitoring over 3 or 4 years is not adequate to observe such changes. To further evaluate the potential for future movement of the dense layer, groundwater modeling was performed. This is discussed in detail in Section 5.2 below.

5.1.2 Environmental Fate of Contaminants

The fate of contaminants includes all the processes acting on them that may affect their nature, distribution, concentration, and rate of movement. The typical fate processes include:

Adsorption onto soils/aquifer materials
Biodegradation
Volatilization
Dispersion and/or diffusion within the groundwater flow system
Geochemical reactions such as precipitation and/or dissolution

The fates of the contaminants in the surface and subsurface media are related both to the chemical and physical properties of the individual chemicals and that of the media. These factors interplay to determine the distribution and rate of migration of the chemicals in the environment.

The ability of some of the site contaminants to adsorb onto soils and aquifer materials are greater than other site contaminants. The organic carbon partitioning coefficient (K_{oc}) property of a chemical is a measure of the chemical's ability to be adsorbed onto organic matter in the subsurface. Chemicals with a relatively high K_{oc}, such as the n-nitrosodiphenylamine (NNDPA), are less mobile because of their affinity to adsorb onto available organic matter, compared to acetone, which is miscible in water and has a low K_{oc}.

Biodegradation occurs naturally in several environmental media by the action of microorganisms. This natural process has the ability to attenuate certain hazardous chemicals, altering them to other compounds which may not be a concern in the environment. For example, phthalates and toluene are readily biodegraded by a wide variety of microorganisms in the soil, which metabolize these compounds to simpler molecules. Other site chemicals, such as chromium, are not readily metabolized by microorganisms and so biodegradation is not a process which significantly affects the fate of these contaminants.

Volatilization involves the dispersion of a chemical in a vapor phase from one medium to another. Typically, this involves the migration of the chemical into the atmosphere from an exposed surface of water, soil, or sediment, or from the subsurface into buildings. It may also involve the movement of contaminants from groundwater to soil vapor and subsequently into indoor air. Volatilization is a significant fate mechanism for the volatile organic compounds (VOCs) such as toluene and trimethylpentenes; once in the atmosphere, the fate of both these VOCs are further controlled by photochemical reactions which break down these chemicals into non-hazardous constituents. Ammonia is also subject to volatilization at higher pH values (greater than 10). Volatilization is not a significant fate mechanism for other site organic compounds, such as phenol, which have low vapor pressure and Henry's Law Constants. Volatilization is also not a fate or transport mechanism for the metals of concern at this site, such as chromium.

A contaminant's aqueous solubility is another important factor in estimating its fate and transport in groundwater and surface water. Chemicals with higher solubility tend to enter the aqueous media, and migrate more readily through the media, than chemicals with lower solubilities. Acetone, ammonia, and phenol are examples of highly soluble chemicals at this site; toluene and most of the semivolatile organic compounds (SVOCs) have lower solubility.

A detailed discussion of the fate processes affecting the contaminants at the site is provided in Appendix K of the CSA (CRA, 1993), and so is not repeated here. However, the Supplemental Phase II investigations have provided additional insight into two processes that affect the fate of contaminants: dispersion and diffusion from the dense layer, and the geochemistry of the system. Each of these is discussed in more detail below.

5.1.2.1 Dispersion/Diffusion from the Dense Layer

The transport of contaminants from the dense layer into the overlying clean groundwater is affected by both physical and chemical factors. In the vertical direction, the dense layer is located at the base of the sand and gravel aquifer. It is separated from the overlying groundwater by a narrow transition layer that is at most a few feet thick, despite the fact that the dense layer has been in the subsurface for approximately 40 years. There are two processes that would act to mix the dense layer with the groundwater: mechanical dispersion and chemical diffusion. Mechanical dispersion is the process where contaminants are spread by moving through different pore spaces, around sand grains, and around permeability variations within the aquifer (heterogeneities). Chemical diffusion is the process where parameters move from areas of higher concentration to areas of lower concentration along a concentration gradient. Although vertical diffusion and dispersion should act to spread out the transition layer, the greater density and viscosity of the dense layer counteract this tendency, and suppress vertical mixing. The result is a thin transition layer, that is, a relatively sharp interface between the dense layer and the overlying groundwater.

The movement of contaminants from the dense layer can be examined quantitatively. Fick's Law describes diffusion: $J = -D\, dc/dx$

where J is the flux of chemical
D is a diffusion coefficient, dependent on the chemical and the medium, and
dc/dx is the concentration gradient.

Fick's Law shows that the amount of the chemical moving across an interface (the flux) is directly proportional to the concentration gradient. A high concentration gradient should result in a greater flux. Above the dense layer, there is a very high concentration gradient due to the sharp interface with the overlying groundwater. Therefore, there is the potential for contaminants to move upward via diffusion.

Diffusion coefficients for ions such as sodium, calcium, sulfate, and chloride in aqueous solutions (at about 10° Celsius [C]) are about 6×10^{-6} to 1×10^{-5} cm^2/sec (squared centimeters per second) (Freeze and Cherry, 1979). In porous media, the effective diffusion coefficients (i.e., mass

transfer coefficients) are significantly smaller, due to presence of the aquifer material. Laboratory measurements suggest values of 1 percent to 50 percent of the theoretical diffusion coefficients (Freeze and Cherry, 1979), or 6×10^{-8} to 5×10^{-6} cm^2/sec. As a demonstration of the diffusion process, Freeze and Cherry (1979) provide an example: with an effective diffusion coefficient of 5×10^{-6} cm^2/sec it would take contaminants 500 years to reach a concentration of 10 percent of the source concentration at a distance of 10 meters from the source. Clearly diffusion is a very slow process.

The data from the multilevel piezometers can be used to calculate coefficients of effective vertical diffusion from the dense layer up into the overlying groundwater using the following formula (Freeze and Cherry, 1979):

$$c_i = c_0 \operatorname{erfc}\left(\frac{x}{2\sqrt{D^* t}}\right)$$

where c_0 is the source concentration
c_i is the concentration at some distance x from the source
D^* is the effective diffusion coefficient
t is the time
erfc is the complimentary error function

Based on this formula, calculated effective diffusion coefficients for certain parameters are provided in Table 5-1. The calculated diffusion coefficients range from 1.8×10^{-7} to 2.4×10^{-6} cm^2/sec, which are typical of the range of values for these parameters in porous media. This similarity indicates that the vertical concentration profiles from the multilevel piezometers are consistent with diffusion being the dominant mechanism transferring contaminants from the dense layer to the overlying groundwater. In addition, evaluation of the major ion chemistry using modified Piper diagrams indicates that dense-layer constituents are transported disproportionately across the interface. In particular, the intermediate-TDS groundwaters are enriched in ammonia with respect to the other dense layer constituents. The disproportionate dense-layer constituent concentrations suggest that diffusion is an important transport mechanism at the interface between the dense layer and intermediate groundwater. If significant advective mixing (dispersion) were taking place, the disproportionate diffusion of different parameters would be obscured by the mixing process.

An additional insight into the diffusion process has been demonstrated in recent theoretical and experimental work (e.g., Hassanizadeh & Leijnse, 1995; Carey, et al., 1995). These researchers have been working with highly concentrated solutions, similar to the dense layer. They have shown that effective diffusion, or mass transfer, from such solutions is actually slower than predicted by Fick's Law, probably as a result of the higher density and viscosity which would tend to reduce mixing. Therefore, diffusion from the dense layer is likely to be a long, slow process that is not particularly effective at transferring contaminants from the dense layer into the overlying groundwater. However, this process appears to be the dominant mechanism by which

contaminants are moved from the dense layer. Therefore, although the dense layer will continue to provide a source of contamination to the groundwater, it is unlikely that the resultant concentrations will be significantly greater than currently observed.

The horizontal transport of contaminants in the dense layer is more complex than the vertical because of the density dependence of the fluid, longitudinal dispersion processes, and geochemical attenuation. As indicated by the geochemical modeling and the electro microprobe analysis (EMPA) studies, attenuation reactions primarily affect the mobility of selected metal ions that readily form oxyhydroxide solids (iron, chromium, aluminum, manganese) that have pH-dependent solubilities. The solubilities of other major ions, such as sulfate, chloride, ammonium, and sodium, are relatively unaffected by these reactions. This difference in reactivity has resulted in a chromatographic separation of the mobile ions from the immobile metals with increasing distance from the source areas. For example, sodium, ammonium, sulfate, and chloride concentrations within the central part of the dense layer (GW-36, GW-42D) are about 10^5 milligrams per liter (mg/l). These concentrations decrease in wells further from the source (GW-44D, GW-58D, GW-83D) by a factor of 10^3. This decrease is most likely due to longitudinal dispersion, that is, historical mixing with cleaner groundwater as the dense layer moved into new areas. In contrast, metals such as chromium and aluminum decrease over the same distance by a factor of 10^5 because of partitioning to the solid phase by attenuation reactions including precipitation and adsorption. The difference in behavior between the anions and cations is shown in Figure 5-2.

In addition, mass transfer vertically from the dense layer into the overlying groundwater introduces somewhat elevated concentrations of contaminants into the horizontal groundwater flow system. As these waters move away from the dense layer due to advective flow, they are subject to attenuation including hydrodynamic processes (e.g., dispersion) and geochemical processes as discussed below. These mass transfer processes most likely contribute to the concentrations of inorganic parameters above background that are observed in the intermediate wells beneath Maple Meadow Brook.

5.1.2.2 Geochemical Considerations

The factors controlling the fate and transport of inorganic compounds through groundwater systems include the chemical properties of the compound, the presence of other solutes that may enhance or reduce its solubility (i.e., complexing ligands), local geochemical environment (e.g., pH and oxidation-reduction potential), and the reactivity of the aquifer matrix. The solubility of compounds (i.e., the amount that can be dissolved in water) may be controlled by speciation and ligand complexation reactions, surface reactions such as adsorption, ion-exchange and coprecipitation, or through simple dispersion and dilution controls. Additional controls may involve biological reactions that change the form of the solute, such as the oxidation of ammonium (NH_4^+) to nitrate (NO_3^-), or sulfide (S^{-2}) to sulfate (SO_4^{-2}).

As described in Section 2.7 above, the fate and transport of metals such as chromium, manganese, and iron are strongly affected by oxidation-reduction reactions that cause these metals to precipitate as oxide or hydroxide solids in response to changes in pH or Eh (the oxidation potential of the system). Iron, for example, is soluble in groundwater in the reduced form as ferrous (Fe^{+2}) iron; however, when oxidized to the ferric (Fe^{+3}) form, it readily precipitates as ferric hydroxide [$Fe(OH)_3$]. Conversely, chromium is more soluble in the oxidized state as Cr^{+6} and relatively insoluble as Cr^{+3} due to the precipitation of chromium (III) hydroxide [$Cr(OH)_3$]. The solubility of chromium may also be reduced by coprecipitation with iron, whereby the precipitation of ferric hydroxide also incorporates chromium into the solid structure due to the similar ionic radius and charge of the two metals. Other metals such as sodium (Na^+) and potassium (K^+) are not affected by oxidation-reduction reactions because they exist in only one oxidation state, and in general, are relatively geochemically conservative.

Both geochemical modeling of the groundwaters and experimental results indicate that chromium(III) concentrations will be limited by the solubility of chromium hydroxide in the aquifer system. Above a pH of about 5.8, the solubility of chromium hydroxide is below the MCL for dissolved chromium (0.1 mg/l or $10^{-5.7}$ molar). The geochemical modeling indicates that there are no chromium complexes present in the groundwater that are likely to increase the solubility of chromium hydroxide.

Geochemical modeling results also indicate that aluminum concentrations are limited by the solubility of aluminum hydroxide. At pH less than 4.5, the concentrations of iron(III) and manganese(II) are also limited by the solubilities of their respective oxyhydroxide solids. At higher pH, iron(III) and manganese(II) do not appear to be solubility limited, possibly because of variabilities in thermodynamic constants or incomplete oxidation under the near anoxic conditions of the groundwater.

Analyses of aquifer solids by EMPA have identified the same minerals/amorphous precipitates that were predicted to be solubility controls by geochemical modeling. This finding provides strong evidence that the formation of metal hydroxides has partially attenuated the transport of metal solutes in the groundwater.

Adsorption of cations by the aquifer matrix generally increases as the pH of groundwater increases, because the surface charge on the matrix becomes progressively negative with increasing OH^- content. Similarly, as the pH of groundwater becomes more acidic (decreasing pH), the solubility of metals increase because of increasing positive charge on the matrix and because metal solid phases react with hydrogen ion (H^+) and dissolve at low pH. The presence of clay minerals in the aquifer also increases adsorption and facilitates cation exchange reactions. For example, chromium (III) readily substitutes for aluminum in clay minerals (e.g., illites) because of the similar size and charge of these metals.

The data from the partitioning study (Section 2.7.3.3) indicate that adsorption is not a dominant mechanism controlling migration rates for chromium away from the toe of the dense layer. Although chromium is often adsorbed strongly onto sandy aquifer materials (Thibault et al. 1990), adsorption tests using site-specific material indicate that even the low levels of chromium occurring along the leading edge of the dense layer plume (e.g., 0.06 mg/l at GW-83D) are not likely to be adsorbed by the aquifer materials. One possible reason for this lack of adsorption is the high concentrations of other cations that occur along the toe of the plume (e.g., iron concentration at GW-83D is 2,030 mg/l). These other cations out-compete the chromium for adsorption sites. The solubility of chromium (III) is instead controlled by precipitation of chromium oxyhydroxides.

The data from the partitioning study also indicate that sulfate is not adsorbed onto the site aquifer materials. At pH <4, sulfate concentrations are limited by the solubility of gypsum. However, at higher pH, the groundwaters are undersaturated with respect to gypsum, probably because of low calcium concentrations in the groundwaters mixing with the dense layer solutions. The groundwaters are oversaturated with various aluminum sulfate solids but theses appear to be ineffective solubility controls for both sulfate and aluminum. The solubility of aluminum is instead controlled by the precipitation of aluminum oxyhydroxides.

While the partitioning tests indicate that the site aquifer materials do have the capacity to adsorb ammonia, it is only weakly adsorbed ($K_{d\text{-}NH4+}$ = 0.4 l/kg). Therefore, adsorption is not likely to affect the rate of ammonia migration through the aquifer.

The adsorption tests also indicate that aluminum was adsorbed by aquifer materials $K_{d\text{-}Al}$ = 4.7 l/kg). Iron was strongly adsorbed by aquifer materials from GW-87D groundwaters ($K_{d\text{-}Fe}$ = 120 l/kg), but weakly adsorbed from GW-83D groundwaters ($K_{d\text{-}Fe}$ = 0.3 l/kg). Manganese demonstrated the opposite behavior, with aquifer materials capable of strongly adsorbing manganese from GW-83D groundwaters (K_{dMn} = 44 l/kg) and only weakly adsorbing manganese from GW-87D groundwaters (K_{dMn} = 1.4 l/kg).

Anions such as chloride and sulfate are generally more conservative in groundwater than cations because they are not readily adsorbed by the aquifer matrix (at the pH values of most natural systems) and because they do not participate strongly in precipitation reactions. Chloride (Cl^-) is the most conservative solute in groundwater and is often used in groundwater studies as a tracer because it is not appreciably retarded by geochemical reactions. Sulfate (SO_4^{-2}) is less conservative than chloride, and can form solid phases with various cations such as calcium (e.g., as the mineral gypsum, $CaSO_4{:}2H_2O$) and aluminum (e.g., as jurbanite, $AlOHSO_4{:}5H_2O$) when present at high concentrations. In general, however, the mobility of chloride and sulfate in groundwater is controlled primarily by advection and dispersion processes.

The solubility of metals can be increased by complexation with ligands such as chloride or sulfate which prevent the precipitation of solid phases until higher concentrations are encountered. For example, at neutral pH the solubility of iron is usually constrained by the solubility of amorphous ferric hydroxide [$Fe(OH)_{3(am)}$]. However, if sufficient sulfate or chloride is present they can form

soluble complexes with iron [e.g., $FeCl^{+}_{(aq)}$ and $FeSO_4^{0}{}_{(aq)}$] and impede the precipitation of $Fe(OH)_{3(am)}$. Complexation usually increases with increasing ligand concentration, and becomes less important in dilute waters.

A conceptualization of solute transport can be constructed from these data. In the dense layer, the concentrations of aluminum, chromium(III), iron(III), manganese(II), calcium, and sulfate are controlled by the solubilities of their respective oxyhydroxide and sulfate solids A portion of the solutes diffuse upwards towards the shallow aquifer where they mix and react with aquifer materials, resulting in a loss of metals from solution as oxyhydroxide precipitation. Consequently, the transport rate in the vertical direction is diffusion limited, resulting in only a narrow layer of elevated solute concentrations residing just above the dense layer. In the horizontal direction, the more mobile ions are transported downgradient with groundwater movement, whereas the reactive solutes, such as aluminum, chromium(III), iron(III) and manganese(II), are partitioned to the solid phase, hence are lost from solution.

Changes in geochemical conditions as groundwater discharges into the Off-Property West Ditch and South Ditch has resulted in the formation of chromium-bearing flocculent. Geochemical modeling demonstrates that the formation of flocculent in the Ditch streambed can be simulated by the mixing of shallow Ditch groundwater and surface water. The wells used in the modeling (GW-42S, GW-77S, and GW-78S) have sufficient dissolved aluminum, chromium, and/or iron to create a precipitate that is representative of the Ditch flocculent composition. Electron microprobe analysis of flocculent samples demonstrate that chromium is bound (by coprecipitation or adsorption) to Fe- and Al- solid phases either as amorphous hydroxides or precipitate as a mineral form (e.g., redingtonite). Amorphous solid phases of Al-, Cr-, and Fe-sulfate in aquifer samples and flocculent samples indicate that sulfate is actively being sequestered from solution during formation of these phases. In surface water, these phases restrict chromium solubility to a level below laboratory detection limits, effectively eliminating dissolved chromium from the South Ditch system.

The apparent continued accumulation of flocculent in the Off-Property West Ditch streambed following flocculent removal in 1994, suggests that shallow aquifer materials adjacent to the ditch system may contain chromium mineral phases that continue to be dissolved by groundwater as it flows towards the ditch system. This source is hypothesized to result from the slow dissolution/desorption/ion-exchange of "old" chromium from the geochemical reaction zone (Figure 2-108). Potential additional sources of chromium to groundwater may originate from the leaching of chromium-contaminated soil in the vicinity of the Central Pond and the former Acid Pits. However, near-neutral pH conditions in Ditch surface water (where chromium solubility is lowest) indicates that the geochemical environment in Ditch surface water strongly favors chromium immobilization. The downstream transport of flocculent and particulate chromium is restricted due to the installation of hay bales as sediment transport barriers.

Chromium migration from the Wilmington facility to the South Ditch system is being controlled by a combination of flocculation, sediment barriers (hay bales), ambient site geochemical conditions, and probable source depletion. Chromium is present solely in the trivalent form, as

particulate or solid phases of extremely low solubility. Migration of other analytes of concern to the Ditch system, such as Na, Cl, NH_4, and SO_4, appears to be limited to areas near GW-79S and the Central Pond. The concentration of these analytes is substantially lower in surface water locations relative to corresponding groundwater locations, suggesting that natural attenuation and dilution is controlling solute concentrations discharging to the Ditch system.

5.2 GROUNDWATER AND GEOCHEMICAL MODELING

In general, the fate and transport mechanisms at the site are straightforward and well understood. The environmental contamination has been in place for decades. Almost all of the original sources of contamination have been taken out of service or removed. Lake Poly is currently a grassy field; the East and West Ponds are located beneath warehouses. The Acid Pits were replaced with the Lined Lagoons which were dismantled in 1986. When manufacturing was discontinued in 1986, most of the manufacturing and storage facilities were dismantled. Because the facility is not currently manufacturing chemicals, there is little need for storing or handling significant amounts of chemicals, except for those used in the operation of the groundwater treatment system at Plant B. Therefore, there is little potential for future releases. Because of these changes in the use of the facility, concentrations of contaminants in the environment have generally decreased over time. This trend is generally expected to continue except in the Drum Areas, where future deterioration of drum integrity could increase impacts to the environment. In general, it can be assumed that existing conditions represent a worst-case for future impacts.

In contrast to conditions beneath most of the site, the potential future impacts associated with the dense layer are not as well understood. Its potential movement in the complex hydrogeologic and geochemical system is not readily inferred from existing information. If there is the potential for significant future movement of the dense layer, the receptor for the contamination is the municipal wellfield surrounding Maple Meadow Brook. The wellfield is clearly a sensitive receptor at which it is important to assess future risk associated with site contamination. Such an assessment was specifically requested by the Massachusetts Department of Environmental Protection (MADEP) in their letter to Olin dated March 22, 1995 (p. 2):

> "There is potential future risk due to the possibility of future contamination of the Town of Wilmington public water supply, therefore, this pathway should be addressed in the risk assessment."

In order to include this analysis in revisions to the risk assessment, it was necessary to predict maximum future concentrations of contaminants at the supply wells. Because of the complexity of the hydrogeology, the presence of the dense layer, and the influence of geochemistry in the subsurface, the only feasible approach to estimating future impacts to the supply wells was

through mathematical modeling. The use of modeling for this purpose was proposed in the "Scope of Work, Human Health Risk Assessment" (ABB Environmental Services [ABB-ES], 1996), which was approved by the MADEP in their letter dated May 16, 1996.

An approach to the modeling was developed that consisted of two phased subdivisions:

1. Groundwater Flow Modeling
2. Geochemical Fate and Transport Modeling

The results of the groundwater flow modeling were used to provide hydrogeologic input to the transport modeling. The results of the transport modeling provided estimated contaminant concentrations at the municipal supply wells for use in the risk assessment. The results of both of these modeling efforts are discussed below.

5.2.1 Groundwater Flow Modeling

The objectives of the groundwater modeling were as follows:

- To aid in more fully understanding the complex groundwater flow system in the area surrounding the site;
- To provide hydrogeologic inputs to the chemical transport modeling;
- To aid in understanding the potential movement of the dense layer; and,
- To determine the final location of the dense layer.

There are significant aspects of this site that make the modeling effort very complex. The hydrogeologic system consists of in-filled bedrock valleys with surface water interaction both on the site and in the extensive wetland surrounding Maple Meadow Brook; the municipal supply wells pump from this aquifer system. The existence of the dense layer adds significant complexity as its movement cannot be modeled using the more common methods that are based on constant-density waters. In addition, the top of the dense layer is characterized by a relatively sharp drop in concentrations to the overlying groundwater, suggesting very low vertical dispersivity. Finally, the concentrations of chromium within the dense layer and groundwater are strongly dependent on the geochemical conditions: at neutral pH, chromium typically precipitates out of solution and is not present in groundwater; at the acidic pHs within the dense layer, it is present and may be mobile.

Due to the complex nature of the system to be modeled, a phased approach was used for the modeling effort. These phases consisted of:

- Groundwater Flow Modeling at Basin Scale - The groundwater flow system within the entire drainage basin surrounding Maple Meadow Brook and including the site was modeled using a constant-density approach. The objective of this phase was to lay a strong foundation for later phases, particularly to provide a solid integrated understanding of the hydrogeologic system.

- Sharp-Interface Modeling at the Site Scale - The movement of the dense layer was modeled using a variable-density approach at a scale encompassing the source areas on site and extending beyond the center of the buried bedrock valley beneath Maple Meadow Brook. The modeling was based on a simplifying assumption that the interface between the dense layer and the overlying groundwater can be approximated as a sharp interface so that dispersion can be neglected. The objective of this phase was to aid in understanding potential movement of the dense layer and to determine where the dense layer will move in the future.

- Generation of Flowpaths - Based on the results of either or both of the first phases, groundwater flowpaths were generated to estimate future movement of contaminants. The flowpaths were initiated at the Butters Row No. 1 and Chestnut Street No. 1 wells and then backtracked upgradient to determine the area of groundwater contributing to water quality at the receptor. The percent contribution of each flowpath to the supply well was calculated, and the expected worst water quality along the flowpath was estimated. Flowpaths were generated for four scenarios:

  1. All wells pumping during the wet season
  2. Only Butters Row No. 1 pumping during the wet season at capacity (1 Mgd)
  3. All wells pumping during the dry season
  4. Only Butters Row No. 1 pumping during the dry season at capacity (1 Mgd)

  This information was provided to the transport modeling.

Using the flowpaths generated during the latter phase, fate and transport along individual flowpaths were modeled. This modeling estimated how contaminant concentrations changed along the flowpaths, including the effects of geochemistry, and provided estimated concentrations at the receptors. This transport modeling is described in Section 5.2.2 below.

The estimated exposure point concentrations resulting from the chemical transport modeling were used in evaluating potential human health risk at the municipal supply well exposure points. The results of the risk assessment are discussed in Section 6 below.

The groundwater flow modeling assumptions, procedures, and complete results are presented in detail in Appendix P. The results are summarized below:

- The groundwater flow system in the area surrounding the site is dominated by the hydrologic effects of the wetlands surrounding Maple Meadow Brook and the municipal supply wells. During wet periods of the year, groundwater flows towards both of these features. Groundwater is discharged to the wetlands in certain areas; surface water is also re-introduced from the wetlands back into the aquifer by the action of the supply wells. During drier periods, less groundwater discharges to the wetlands, but the supply wells continue to draw water from storage in the thick wetland sediments in this area.

- The results of the modeling show that there is a groundwater divide in the vicinity of Main Street. To the west of the divide, groundwater flow is towards the wetlands and/or the municipal supply wells; east of the divide, groundwater is captured by the supply wells at Altron. The location of this groundwater divide is based on pumping at the Altron wells; if these pumping rates change, the location of the divide will change.

- The results of the particle tracking simulations for the four scenarios are summarized on Table 5-2, which includes the beginning and ending point for each flowline, the percentage contribution to the supply well associated with each flowline, and the estimated characterization of the most degraded water quality along each flowline. A total of 34 flowlines were simulated from Butters Row No. 1 and Chestnut Street No. 1 each. These results show that most of the flowlines contributing to each well originate at shallow depths at the wetlands, or outside the area of contaminated groundwater. For example, for Scenario 1, 74 percent of the water contributing to Butters Row No. 1 is expected to be clean, and only 26 percent is expected to flow from areas impacted by contaminants.

- The particle flowlines with the entire wellfield in operation show that a number of the flowlines that reach the Butters Row No. 1 well pass through the contaminated groundwater beneath Maple Meadow Brook as shown on Figure 5-3. The capture zones for the other wells are then deflected to the north and south of the capture zone for Butters Row No. 1. Therefore, the location of Butters Row No. 1 relative to the intermediate-TDS groundwater is such that it actually captures relatively more of the water located beneath the center of Maple Meadow Brook. The capture zones for Butters Row No. 2 and both Chestnut Street wells extend further to the west as well as beneath the more northern and southern portions of the wetlands. Based on this result, it is expected that Butters Row No. 1 would be more likely to be impacted by site contaminants than the other supply wells. In fact, existing water quality data does show higher concentrations of indicator parameters at Butters Row No. 1 compared with the other supply wells.

- The potential worst-case situation with only Butters Row No. 1 in operation indicates that the capture zone would extend radially away from the well in all directions. Fewer of the particle flowlines pass through the intermediate-TDS waters beneath the wetlands. This suggests that the water quality at Butters Row No. 1 would actually be improved under such a scenario. This is discussed in more detail as part of the transport modeling (Section 5.2.2 below).

- The results of the modeling to simulate movement of the dense layer are shown on Figure 5-4. The simulated thickness of the dense layer matches the conditions observed in the field well. To achieve this match, it was necessary to simulate an existing subsurface barrier to flow in the vicinity of the bedrock saddle located just west of Main Street. The possible causes of such a barrier include: (1) the bedrock saddle may be higher than suggested by the limited number of data points, (2) glacial till overlying the bedrock may have a substantially lower permeability that impedes movement of the dense layer, or, (3) the leading edge of the dense layer itself may have created a zone of reduced porosity and permeability due to the precipitation of solids from solution at increasing pH. It should be noted that the existence of this barrier does not violate any field data. Without simulating this barrier, the model results place the current location of the dense layer throughout the bedrock valley beneath Maple Meadow Brook, with only a small amount remaining beneath Main Street, Jewel Drive and Eames Street - a situation that clearly does not exist. This situation is described in more detail in Appendix P.

- The dense layer simulation was then used to predict the potential for future movement. This result is presented in Figure 5-5 and shows little significant change in the future location of the dense layer. Based on this modeling, it does not appear that there is a potential for significant future movement of the dense layer.

- The results of the sharp-interface modeling also suggest that the dense layer essentially reached its present position in the mid-1980s. Therefore, the contribution of contaminants from the dense layer into the overlying flow system is also unlikely to have changed. As a result, in the absence of significant changes to the flow system, it is unlikely that the observed concentrations of contaminants will increase in the future. However, seasonal or short-term fluctuations may occur.

5.2.2 Transport Modeling

Transport modeling of selected inorganic compounds in groundwater to the Town of Wilmington pumping wells was conducted to provide exposure point concentrations to the Human Health Risk Assessment (Section 6.1 and Appendix R). Results of the groundwater flow modeling and particle tracking simulations formed the basis of the transport modeling, in addition to

geochemical data from Section 2.7 (Site Geochemical Characterization). The complete transport modeling approach and results are presented in Appendix Q. The objectives of the transport modeling were as follows:

- To evaluate the potential for transport of chromium in groundwater from the dense layer at monitoring well GW-58D to the Town of Wilmington pumping well Butters Row No. 1,

- To evaluate potential chromium transport in groundwater from the dense layer at monitoring well GW-83D to the Butters Row No. 1 and Chestnut Street No. 1 pumping wells,

- To estimate the maximum concentrations of groundwater indicator compounds (i.e., ammonia, chloride, sodium, and sulfate) and other inorganic solutes at the Butters Row No. 1 and Chestnut Street No. 1 pumping wells under differing scenarios of pumping and seasonality;

As outlined in Section 5.2.1 there are significant aspects of the site that make the modeling effort very complex. The CSA (CRA 1993) and Supplemental Phase II Investigation data indicate that the primary potential source of solutes at the site (i.e., the dense layer) appears to have reached stasis from a transport perspective (Sections 2.7 and 4). This hypothesis is supported by the results of Sharp-Interface variable density modeling of the dense layer (Section 5.2.1 and Appendix P) indicates that further downslope migration of the dense layer, both at the site and under Meadow Maple Brook wetland, has apparently ceased due to a combination of physical barriers (i.e., bedrock topography and/or reduced aquifer porosity) and dense layer solution properties (e.g., high specific gravity and viscosity).

Chemical data from multilevel piezometers (Section 2.7.4) and monitoring well clusters have established that steep chemical gradients exist across the interface between the dense layer and overlying groundwater, where solute concentrations decrease by as much as three orders of magnitude within a few feet, often reaching levels that approach that of site background (e.g., chloride and sodium) or are below the analytical detection limit (e.g., chromium and aluminum) (Section 2.7.4). The observed sharp decrease in solute concentrations in the vertical direction into the overlying groundwater is the result of a low mass transfer rate into the overlying groundwater. In addition, the physical properties of the dense layer suggest that this may be the result of non-Fickian diffusional processes which greatly reduces the flux of solutes from the dense layer to overlying groundwater even further (Section 5.1.2).

Chromium slowly diffuses from the dense layer into overlying groundwater where it is effectively prevented from being transported by the in situ formation of authigenic precipitates of chromium hydroxide and mixed chromium-iron hydroxide solids, which have very low solubilities at the higher pHs of the overlying groundwater (i.e., pH>5.5). Geochemical phenomena at the dense layer interface maintains chromium concentrations in the overlying groundwater below the analytical detection limit (0.015 mg/L) and well below the drinking water MCL of 0.1 mg/L for

chromium (Section 2.7). The general absence of chromium in groundwater that otherwise contains elevated levels of more geochemically conservative solutes (e.g., ammonia, chloride and sulfate) supports the hypothesis that chromium is not subject to transport outside of the region of low groundwater pH, but is instead controlled by geochemical processes at the dense layer interface.

Chemical data from shallow and intermediate depth groundwater indicates that surficial recharge to the sand and gravel aquifer has provided sufficient clean water to effectively dilute solute concentrations and buffer the groundwater pH at near-neutral conditions. Hence, hydrologic and geochemical conditions prevail favoring the precipitation of aluminum, chromium, iron and manganese hydroxides at or near the dense layer interface, and that reduce the concentrations of the geochemically conservative solutes such as ammonia, chloride, and sulfate.

Due to the complex nature of the system to be modeled, an integrated approach was used for the modeling effort consisting of:

- Equilibrium Geochemical Modeling - The geochemistry of chromium-bearing groundwater at monitoring wells GW-58D and GW-83D was characterized using the equilibrium geochemical code MINTEQ version 4.00 (Eary and Jenne, 1992). The objective of this modeling effort was to identify the saturation state of the groundwater with respect to various chromium-bearing mineral and amorphous solid phases that may be controlling the concentration and mobility of chromium at these locations. Electron microprobe analysis of available aquifer materials collected from monitoring well GW-83D was performed in support of geochemical modeling to determine if chromium-bearing solid phases estimated to exist by modeling were in fact present. MINTEQ modeling of all other Maple Meadow Brook wells was also conducted to determine the saturation state of these waters with respect to potential solid phases as well.

- Chemical Transport Modeling - Results from the equilibrium geochemical modeling were input to the geochemical transport code CTM (Erikson and Hostetler, 1992) to simulate the transport of chromium in groundwater from monitoring wells GW-83D and GW-58D to the pumping wells Butters Row No. 1 and Chestnut Street No. 1. This code was selected because it considers both hydrologic and geochemical effects on inorganic solute transport.

- Groundwater Mixing Modeling - Results of the groundwater flow and particle tracking modeling (Section 5.2.1 and Appendix P) and current water quality within the capture zones were used to estimate concentrations of inorganic solutes from source areas in groundwater under the Maple Meadow Brook wetland to the Butters Row No. 1 and Chestnut Street No. 1 pumping wells under two different pumping scenarios under both seasonally wet and dry conditions (a total of four scenarios). Particle tracking simulations were used to calculate the relative contribution of groundwater supplied to each pumping well by each of the particle

path flowlines (See Figure 5-3 in Section 5.2.1). Monitoring well data were then used to assign solute concentrations to each flowline which was then weighted according to the relative contribution of groundwater of each flowline. The weighted concentrations were summed for all flowlines to estimate potential solute concentrations at the pumping wells. The following assumptions were made in the mixing modeling:

1. The dense layer is not moving.

2. A steady state has been achieved between the rate at which dense layer constituents diffuse out of the dense layer into the overlying groundwater, and the rate at which they are then transported advectively away from the dense layer. Therefore, the average concentration of dense layer constituents in each of the wells in the Maple Meadow Brook area are representative of the current and future conditions and the different travel times associated with each flow path can be ignored when calculating the constituent concentration at the pumping wells.

The mixing model is very conservative because:

1. Particle flowpaths include solute concentrations that are the maximum concentrations observed along that flowpath.

2. These monitoring wells have solute concentrations higher than encountered by the particle, and

3. Although the particle flowline subsequently passes through cleaner water before it reaches a pumping well, the contribution of this cleaner groundwater is ignored.

4. Because the model assumes conservative transport, the resulting predictions are conservative.

5. The effects of diffusion and dispersion were not incorporated.

6. Some flow paths pass through the dense layer. Although SHARP modeling demonstrated that the dense layer is unlikely to be captured by the municipal pumping wells, the chemistry of the dense layer wells were used. Therefore, the mixing modeling made the conservative assumption that dense layer groundwater could be transported to the pumping wells.

The mixing modeling is also based on current observed hydrologic and geochemical conditions that define the range of seasonality conditions (i.e., wet season and dry season) at the site, as well as the current range of solute concentrations observed in groundwater.

Transport modeling was confined to simulations involving pumping wells Butters Row No. 1 and Chestnut Street No. 1 because groundwater chemical data has shown that the levels of inorganic solutes at these two wells are slightly elevated relative to the unaffected Town of Wilmington pumping wells (i.e., Chestnut Street No. 1A/2, Butters Row No. 2, and Town Park) (Section 4). Geochemical characterization of Maple Meadow Brook groundwater using Piper trilinear diagrams has established that the groundwater chemistry (ion ratios and total dissolved solids content) at the unaffected pumping wells and at Butters Row No. 1 and Chestnut Street No. 1 are substantially different than groundwater located in areas known to be impacted by site solutes (e.g., GW-83D) (Figure 5-6). Although the Butters Row No. 1 and Chestnut Street No. 1 groundwater chemistries indicate that contributions from the site are reaching these wells, the Piper diagrams indicate that the chemical composition of Butters Row No. 1 and Chestnut Street No. 1 are very similar to the chemical composition of the unaffected town wells. This indicates that substantial mixing of "clean" and impacted groundwater occurs during pumping, which results in solute concentrations that approach clean groundwater. Low mass transfer rates from the dense layer also contribute to this phenomenon.

Modeling Results

Complete model results, including the uncertainty analyses, are presented in Appendix Q. The results of the solute transport and geochemical modeling are summarized below:

- Using analytical data from the two sampling events at GW-83D (October 1996 and January 1997) as input to MINTEQ, modeling results indicate that groundwater at GW-83D is supersaturated with respect to several chromium-bearing iron solid phases represented by the generic formula: $Cr_xFe_{1-x}(OH)_3$. Additional solids phases indicating supersaturation include amorphous ferric hydroxide [$Fe(OH)_{3(am)}$], gibbsite [$Al(OH)_3$], $AlOHSO_4$, basaluminite [$Al_4(SO_4)(OH)_{10}{:}5H_2O$], goethite [FeOOH], and in some cases, gypsum [$CaSO_4{:}2H_20$]. These results indicate that solubility constraints on aluminum, chromium, and iron may be limiting the concentration of these elements in deep groundwater at GW-83D, and preventing their migration to shallower groundwater through precipitation reactions. The estimated supersaturation of several aluminum sulfate solids and gypsum suggest that some attenuation of sulfate and calcium may also be occurring at this location.

- Electron microprobe analysis of aquifer solids from GW-83D confirm the solubility predictions of MINTEQ4 with regard to chromium and iron solid phases (Figure 5-7). Electron microprobe photomicrographs demonstrate the presence of chromium at 0.32 and 0.34 weight percent contained within a matrix of amorphous ferric oxyhydroxide solids. A similar suite of Cr-bearing iron phases were found in aquifer samples collected from CPT-2 in the transition layer near multilevel piezometer MP-2, indicating that similar geochemical processes that control dense layer chromium and iron concentrations may be operating at both locations. In

addition, abundant iron oxide minerals (and some pyrite) were observed to exist in the aquifer material from GW-83D, indicating that sufficient iron was available to facilitate the precipitation of chromium from solution.

- Geochemical and transport modeling of chromium migration from GW-58D and GW-83D (the western extent of the dense layer) to Butters Row No. 1 indicates that the concentration of chromium decreases to below the analytical detection limit (<0.015 mg/L) within a few tens of feet after leaving GW-58D and GW-83D, and that detectable chromium is not estimated to reach Butters Row No. 1 (or Chestnut Street No. 1) under existing conditions. The decreases in chromium concentrations are due to the precipitation of chromium-iron hydroxides (represented by the formula $Cr_xFe_{1-x}[OH]_3$) and augmented by subsequent dispersion and dilution. Hence, natural attenuation reactions constrain further migration of chromium from its current location in the dense layer.

- The results of the four mixing modeling scenarios for the Butters Row No. 1 and Chestnut Street No. 1 pumping wells are presented in Table 5-3. These results indicate that under Scenario 1 (all five production wells pumping during the wet season), only ammonia and sodium exceed the historical range of concentrations observed at Butters Row No. 1 for these solutes. The majority of other analytes (with the exception of aluminum, iron, and manganese) are within the historical range of observed solute concentrations at this well. In contrast, the model predicts that the maximum concentrations of all four indicator analytes (ammonia, chloride, sodium, and sulfate) exceed the historical concentration ranges observed at Chestnut Street No. 1. The primary contributor of solutes to both Chestnut No. 1 and Butters Row No. 1 is GW-83D, which is predicted to contribute to these wells only during the wet season of Scenario 1.

- The results for Scenario 3 (all five production wells pumping during the dry season) indicate that none of the four indicator solutes exceed the historical range of solute concentrations at Butters Row No. 1, but that three indicator solutes (ammonia, chloride, and sulfate) and several other inorganics exceed the historical ranges at Chestnut No. 1.

- Under both scenario 2 (Butters Row No. 1 pumping during the wet season) and scenario 4 (Butters Row pumping during the dry season), the concentrations of all four indicator solutes is estimated to be within the historical range of concentrations observed at Butters Row No. 1. Exceptions under Scenario 4 consist solely of slightly elevated concentrations of iron and potassium.

- The mixing model also indicates that relatively elevated concentrations of aluminum, manganese and iron are expected at Butters Row No. 1 and Chestnut Street No. 1 under all four simulated scenarios. This is not surprising since naturally-occurring iron and manganese are already present in relatively high

concentrations in regional groundwater. Because of this, the Butters Row Treatment Plant was designed to remove excess iron and manganese from groundwater. The biggest contribution of these metals to the pumping wells is from GW-83D, where the dense layer is present. It should be noted that for many years, the Town of Wilmington has been pumping water selectively from all five production wells on an "as-needed" basis. The influent to the Butters Row Treatment Plant is a composite of water from the production wells being pumped at any time. Consequently, the future composite water to be treated will have concentrations of solutes significantly lower that predicted by the mixing model. In addition, through selective pumping from the five wells, a treatable composite influent should be attainable under the worse case scenario.

- As mentioned above, particle tracking simulations show that groundwater from the GW-83 well cluster (consisting of GW-83S, GW-83M, and GW-83D), can be captured by both Butters Row No. 1 and Chestnut Street No. 1 pumping wells under Scenario 1 (all production wells pumping during the wet season). Monitoring well GW-83D, screened in the deepest portion of a deep bedrock topographic depression at this location, intersects the dense layer which exhibits high concentrations of solutes and total dissolved solids (up to 35,500 mg/L TDS). The mixing routine for these particle paths is very conservative in that the particle flowline passing through GW-83D is assigned solute concentrations from that well even though the flowline subsequently passes through cleaner groundwater before it reaches either pumping well. Because of this, the mixing model indicates that groundwater from GW-83D is primarily responsible for the estimated elevated concentrations of solutes at these pumping wells. Because the groundwater flow and particle tracking modeling used in the mixing model do not consider the effects of groundwater density or viscosity, it is not thought probable that advecting groundwater (represented in the model by the particle flowline) will actually enter the dense layer along its flow path. In addition, because Sharp-Interface modeling predicts no future movement of the dense layer beyond the bedrock depression at GW-83D, the actual transportable solute concentrations from GW-83D to Butters Row No. 1 and Chestnut Street No. 1 will probably be less than the levels estimated by the model, given the physical and chemical condition of the dense layer at this location. Therefore, the estimated solute concentrations that could reach the pumping wells probably represent a very conservative worst case scenario.

- It should also be noted that if migration of high levels of solutes from GW-83D to the pumping wells actually occurs, it is estimated from the particle tracking simulations that it would take approximately 10 years for the groundwater from GW-83D to reach Butters Row No. 1, and approximately 14 years to reach Chestnut Street No. 1 (Appendix P). Olin Corporation has installed a sentinel monitoring well network between GW-83D and the pumping wells to provide

forewarning of any significant changes in groundwater chemistry that could indicate movement of the dense layer or migration of elevated levels of solutes in overlying groundwater.

- Particle tracking simulations also indicate that groundwater from other deep monitoring wells (e.g., GW-87D, GW-86D, and GW-85D) that are located closer to both Butters Row No. 1 and Chestnut Street No. 1 than GW-83D will also be captured by these wells. However, the mixing model results indicate that the estimated solute concentrations contributed by these wells is significantly less than that contributed by GW-83D (see Table 5-3 "w/out GW-83D" headings), such that the estimated solute concentrations at Butters Row No. 1 and Chestnut Street No. 1 that are contributed from these deep flowpaths are generally within the historical range of solute concentrations observed at the pumping wells. This is due to the significantly lower solute concentrations present at these wells, which are input to the model.

- Because dispersion and dilution were not factored into the solute concentrations assigned to each particle flowpath in the mixing model, the estimated solute concentrations at Butters Row No. 1 and Chestnut Street No. 1 are conservative. Many of the calculated particle flowline solute concentrations could be lower given dispersion and dilution effects which are especially significant in the vicinity of the pumping wells where groundwater velocities increase substantially.

- MINTEQ simulations confirm that solutes present in intermediate and shallow groundwater are not appreciably affected by geochemical processes (i.e., precipitation and adsorption) that are known to control the concentrations and migration of chromium, aluminum and iron in the dense and transition layers. With the possible exception of iron, the concentrations of ammonia, chloride, sodium, and sulfate in intermediate and shallow groundwater are substantially less than required to exceed the saturation indices of the conjugate solid phases of these solutes, thus precluding precipitation reactions as a viable natural attenuation mechanism (Attachment A in Appendix Q). Experimental adsorption data (Section 2.7) using aquifer solids and groundwater from Maple Meadow Brook wells indicate that adsorption of ammonia, sodium and sulfate is weak and is not a significant factor controlling the mobility or concentrations of these species in groundwater, supporting the premise that dispersion and dilution are the primary processes controlling solute concentrations in the aquifer.

- Solute transport and geochemical modeling results (this section and Section 2.7) in conjunction with SHARP-interface modeling (Section 5.2.1) and contaminant nature and extent data (Section 4), provide evidence that the groundwater system downgradient of the Olin site may have reached steady-state conditions with respect to inorganic compound transport and solute concentrations at the Butters Row No. 1 and Chestnut Street No. 1 pumping wells. If existing hydrologic

conditions remain stable, and the dense layer has reached stasis, then future variations in solute concentrations at the pumping wells are likely to be due only to short-term influences such as seasonal or climatic changes and slow diffusion of solutes from the dense layer to the overlying groundwater. However, it is important to note that the estimated solute concentrations predicted to arrive at the pumping wells by the mixing model is based on the current understanding of the hydrogeologic and geochemical conditions at the site. Because most of the data for the Maple Meadow Brook monitoring wells is very recent (most were installed in the Fall 1996), there is less than 1 years worth of data from which to use in the model. This provides little data from which to compare the model results to and incorporates a degree of uncertainty in the results.

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6.0 RISK CHARACTERIZATION SUMMARY

MCP 40.0835(4)(h)

CHAPTER SUMMARY

A public health, safety, and welfare risk characterization and an ecological risk characterization have been conducted to assess the risks to public health and the environment posed by oil and hazardous material (OHM) detected at or having migrated from the Wilmington facility Section 6.1 presents the human health, safety and public welfare risk assessments. Section 6.2 presents the ecological risk characterization. This section presents a summary of the risk characterization technical approach and results. The key findings of the human health and ecological risk characterizations are:

Human Health Risk Summary

- **Current Land Use and Site Conditions** - Cancer and noncancer risks for all identified receptors, including site workers, on-property utility workers, off-property industrial workers and neighborhood residents are below the MCP Cumulative Receptor Cancer Risk limit and the Cumulative Receptor Noncancer Risk Limit.

- **Future Land Use and Site Conditions** - Cancer and noncancer risks for all elevated receptors, including site maintenance workers, on-property utility workers, full-time, long-term on-property industrial workers, off-property industrial workers, and neighborhood residents are equal to or below the MCP Cumulative Receptor Cancer Risk Limit and the Cumulative Receptor Noncancer Risk Limit.

- These conclusions are dependent, in part, on Notices of Limitation with Respect to Groundwater, Downgradient Property Status, Groundwater Management Plans, and an Activity and Use Limitations. The limitations for the facility are based on an Activity and Use Limitation; limits on groundwater use at residences along Main Street are based on abandoned wells, public water supply availability, Town of Wilmington regulations, and Notices of Limitation with Respect to groundwater; and limits on groundwater use at downgradient industrial properties are provided by availability of public water, Town of Wilmington regulations, and provisions of groundwater management plans that are associated with Downgradient Property Status.

- Concentrations of 1,2-dichloroethane, 1,2-dichloroethene, benzene, chloroform, bis(2-ethylhexyl)phthalate, methylene chloride, trichloroethene, vinyl chloride, chromium, nitrate, arsenic, and lead in the Zone II of the Town Supply Wells exceed corresponding health-based MCLs, which are considered applicable public health standards. Because of these exceedances, a condition of non significant risk of harm to health does not exist. Available data indicate that 1,2-dichloroethane, 1,2-dichloroethene, methylene chloride, trichloroethene, vinyl chloride, arsenic, and lead may not be associated with the site.

Safety Risk Summary

- Future excavation of materials in the Drum areas may pose a threat of fire or explosion due to the possible presence of Opex® and Kempore®. Therefore, a condition of no significant risk of harm to safety does not exist.

Public Welfare Risk

- The groundwater Upper Concentration Limits (UCLs) for chromium, nickel, and cadmium are exceeded at the dense layer hot spot in the Zone II area.

- The groundwater UCLs for indeno(1,2,3,-cd)pyrene and bis(2-ethylhexyl)phthalate are exceeded in the Plant B Hot Spot.

- The groundwater UCLs for chromium and zinc are exceeded in the Zone II area.

- Non-aqueous phase liquid (NAPL) with thickness greater than one-half inch was detected in the Plant B area, a condition that by definition exceeds a UCL.

- Secondary drinking water standards, which are based on aesthetic considerations, for aluminum, chloride, iron, manganese, sulfate, and zinc are exceeded in the Zone II area.

- Manipulations and/or modifications of Butters Row Treatment Plant operations may be required to maintain adequate chlorine residuals to insure disinfection of treated drinking water.

- Public groundwater resources have been impacted.

Therefore, a condition of no significant risk of harm to public welfare does not exist.

Ecological Risk Summary

The conclusions concerning the ecological risk are as follows:

- The results of a food chain model, which are based on site-specific biological tissue data, support a conclusion of no significant risk of harm to representative terrestrial and semi-aquatic receptors (i.e., the American woodcock, red fox, and green heron) from food chain exposures.

- The results of earthworm toxicity tests and an evaluation of surface soil data support a conclusion of no significant risk to terrestrial receptors (i.e., the woodcock) from decreased prey abundance.

- The FETAX toxicity test results combined with an amphibian population model suggest that we cannot conclude no significant risk of harm to aquatic receptor (i.e., the green frog) from exposure to sediment in the On-Property West Ditch, based on a predicted subpopulation decrease of greater than 25 percent for amphibians in that area.

- The FETAX toxicity test results and amphibian population model support a conclusion of no significant risk of harm to semi-aquatic receptors (i.e., the green heron) from decreased prey abundance.

- A comparison of surface water concentrations with published surface water toxicity data for amphibian suggest that we cannot conclude no significant risk of harm to aquatic receptors from exposure to surface water in the Off-Property West Ditch, South Ditch, and Ephemeral Drainage Ditch.

6.1 HUMAN HEALTH, SAFETY, AND PUBLIC WELFARE RISK CHARACTERIZATION

The human health, safety, and public welfare risk characterizations are summarized below. The risk assessment technical approach is presented in Section 6.1.1, the risk assessment findings are presented in Section 6.1.2, and the conclusions of the human health, safety, and public welfare risk assessments are presented in section 6.1.3.

6.1.1 Risk Assessment Approach

The Method 3 human health risk characterization was prepared consistent with the Massachusetts Contingency Plan (MCP) promulgated under Chapter 21E on October 3, 1988 (310 CMR 40.000) as amended through February 28, 1997, and in substantial compliance with the National Contingency Plan (NCP), March 8, 1990, as amended (40 CFR 300); "Guidance for Disposal Site Risk Characterization" (MADEP, 1995); and, the "Risk Assessment Shortform Residential

Scenario, Version 1.6" (MADEP, 1992). Supplemental guidance was provided by "Risk Assessment Guidance for Superfund, Volume I, Human Health Evaluation Manual (Part A), Interim Final" (USEPA, 1989a) and the "Human Health Evaluation Manual, Supplemental Guidance: Standard Default Exposure Factors" (USEPA, 1991a). The Method 3 human health risk characterization was conducted in a manner consistent with 310 CMR 40.0993. The risk to safety was evaluated per 310 CMR 40.0960 and the risk of harm to public welfare was conducted per 310 CMR 40.0994. The risk characterization is consistent with the Scope of Work, Human Health Risk Assessment, Olin Corporation Wilmington Facility, DEP RTN: 3-0471, April 1996 that was submitted to the Massachusetts Department of Environmental Protection (MADEP). The MADEP provided Conditional Approval of that scope of work in a letter of May 16, 1996 (MADEP, 1996a).

The risk assessment process consists of four components: hazard identification, dose-response assessment, exposure assessment and risk characterization. The hazard identification determines what substances are present at a site, whether a substance causes adverse effects, and identifies those effects. The dose response assessment describes the relationship between the level of exposure and the likelihood and/or severity of an adverse effect. The exposure assessment identifies potential routes of exposure; characterizes the populations exposed; and determines the frequency, duration, and extent of exposure. The risk characterization, combines the information from the previous three steps to describe the type (e.g., carcinogenic and non-carcinogenic) and magnitude of risks to the exposed populations. It also identifies the uncertainty in the characterization of risks. Each of these steps is described in later sections of this document.

6.1.1.1 Hazard Identification

Hazard identification consists of the presentation of analytical data, selection of data for risk characterization and the selection of the OHM that will be included in the risk characterization. Media evaluated at the site included surface soil, subsurface soil, surface water, sediments, groundwater, and air. The analytical data and selection of OHM for the risk assessment for each of the media are presented in section 2.0 of Appendix R. The chemical constituents that are the focus of the risk assessment because of their high frequency of detection and elevated concentrations include chromium, ammonia, sulfate, chloride, sodium, n-nitrosodiphenylamine (NNDPA), trimethylpentenes, phthalate compounds, acetone, and to a lesser extent, polycyclic aromatic hydrocarbons (PAHs) and Opex® and Kempore®. Most of the detected OHM in each of the media were retained in the risk assessment. Only analytes determined to be consistent with background conditions or those OHM detected two or fewer times and at very low concentrations in a given medium were not retained in the risk assessment.

6.1.1.2 Dose Response Assessment

The purpose of the dose-response assessment is to characterize the relationship between the dose of contaminant administered or received and the incidence of adverse health effects associated with exposure. From this quantitative dose-response relationship, toxicity values (e.g. slope factors, reference dose values or reference concentrations) are derived that can be used to estimate the

likelihood of adverse effects as a function of human exposure to an agent. These toxicity values are used in the risk characterization process to estimate the potential for adverse effects occurring in humans at different exposure levels.

The dose-response relationship(s) for each OHM which has been selected as a Chemical of Potential Concern (CPC) is presented in this section. The dose-response information may be divided into two major categories:

> Toxicity information associated with threshold (non-carcinogenic) health effects.
>
> Toxicity information concerning carcinogenicity, either from human epidemiological data, or from laboratory studies.

All the chemicals selected as CPCs are evaluated for potential non-carcinogenic health effects. In addition, any substance considered to be a known, probable, or possible human carcinogen is also evaluated for its potential carcinogenic effect. The classification of a chemical as a carcinogen does not preclude an evaluation of that same chemical for potential non-carcinogenic health risks, as all potentially carcinogenic chemicals may also exert non-carcinogenic health effects.

The dose response values utilized in this assessment include Cancer Slope Factors (CSFs) and Cancer Unit Risk (CUR) values, and for non-carcinogenic effects, Reference Doses (RfDs) and Reference Concentrations (RfCs). The values for each of the OHM considered in the risk assessment have been assembled and applied in the risk characterization. The sources of these dose response values, consistent with MADEP guidance (MADEP, 1995) are identified below.

The main source of dose-response values is the United States Environmental Protection Agency (USEPA) Integrated Risk Information System (IRIS) which is a data base established by USEPA containing all validated data on many toxic substances found at hazardous waste sites. This data base was used to identify the CSFs and RfDs applied in this risk assessment. Where no information was found in IRIS, USEPA Health Effects Assessment Summary Tables (HEAST) were used. When toxicity values from IRIS or HEAST were not available, alternative toxicity values available from USEPA or MADEP sources were used. The hierarchy of sources used to obtain dose response data for carcinogenic and non-carcinogenic effects, per MADEP guidance (MADEP, 1995), is as follows:

Slope factors and unit risk values:

1. Integrated Risk Information System; current as of January, 1997 (USEPA, 1997)

2. Health Effects Assessment Summary Tables; current as of fiscal year 1995 (including July updates) (USEPA, 1995)

3. California Environmental Protection Agency (CAEPA, 1994)

Reference doses and reference concentrations:

1. Integrated Risk Information System (IRIS); current as of January, 1997 (USEPA, 1997)

2. Health Effects Assessment Summary Tables (HEAST); current as of fiscal year 1995 (including July updates) (USEPA, 1995)

3. Toxicity values developed by MADEP, ORS (MADEP, 1992; 1994)

4. Agency for Toxic Substances and Disease Registry (ATSDR) Toxicity Profile Documents

5. Allowable threshold concentrations (ATCs) developed by MADEP (MADEP, 1995)

6. Dose-response values used to develop drinking water standards and guidelines (USEPA, 1996)

RfDs and RfCs are provided in IRIS, HEAST, and by MADEP ORS. In addition, RfDs used to develop drinking water standards and guidelines for some chemicals are provided in USEPA's 'Drinking Water Regulations and Health Advisories". For some chemicals, ATSDR publishes minimal risk levels (MRLs) for threshold effects via the oral or inhalation exposure route. MRLs are derived using the modified risk assessment methodology the USEPA uses to derive RfDs and RfCs. MRLs were used as surrogate RfDs and RfCs when these values were not available in IRIS, HEAST or MADEP sources. ATCs are derived from threshold effects exposure limits (TELs), which represent 20 percent of the allowable dose. Therefore, ATCs were developed by multiplying the TEL by five, as described in MADEP (1995). ATCs were used as surrogate RfCs when RfCs were not available in other sources.

An RfD developed by ABB Environmental Services, Inc. (ABB-ES) for the two isomers of 2,4,4-trimethylpentene (diisobutylene) was also used. Documentation for the 2,4,4-trimethylpentene value is presented in Attachment 4 of Appendix R. In addition, ABB-ES developed RfDs for Opex® and Kempore®. The documentation for these RfDs is contained in Attachment 4. Finally, an RfD for cobalt developed by USEPA's National Center for Environmental Assessment (NCEA) is used in this assessment. The documentation for this RfD is provided in Attachment 4 of Appendix R.

6.1.1.3 Exposure Assessment

The exposure assessment includes the identification of receptors, the current and foreseeable land use of the site and surrounding area (and any assumed limitations on activities and uses at the site and surrounding area), identification of exposure points (including a hot spot evaluation), exposure routes, development of exposure profiles , calculation of exposure point concentrations, and the calculation of appropriate average doses or concentrations for use in risk calculations. The calculation of doses also

includes a Relative Absorption Factor (RAF) that is used to insure that the medium and route of exposure and the dose-response value are compatible. Each of the components of the exposure assessment has been conducted in a manner consistent with the MCP and the MADEP guidance (MADEP, 1995).

Given current use of land and groundwater, the following bullets identify potential human receptors and associated exposure pathways:

> Neighborhood resident to the west of the site who consumes Town drinking water and who may play in surface water and sediments outside of the fenced perimeter of the facility
>
> Neighborhood resident to the east of the site who consumes Town drinking water and who may play in surface water and sediments outside of the fenced perimeter of the facility
>
> On-property Worker who consumes Town drinking water and is occasionally exposed to on-property surface soil, and surface water and sediment in the on-property ditches
>
> Worker at an Adjacent Industrial Property who consumes Town drinking water, is exposed to vapors that may migrate from shallow groundwater through the building slab to indoor air, and is exposed to groundwater that is used as process water at the property
>
> On-property Utility Worker who is exposed to surface soil and subsurface soil at the facility

There are several limitations on groundwater use that have been incorporated into the exposure assessment for current land use. All downgradient residential landowners have been contacted concerning abandonment of wells and implementation of a Notice of Limitation With Respect to Groundwater. As an interim risk reduction measure, all private drinking water wells downgradient of the facility have been abandoned and all of the associated properties are connected to the Public Water Supply. A Notice of Limitation With Respect to Groundwater is in place or planned for several properties along Main Street (including those where wells have been abandoned) west of the site. Two downgradient residential properties have unused non-potable private wells that have not yet been abandoned and negotiations with the landowner are on-going. All downgradient industrial/commercial properties within the disposal site boundary have been granted or offered Downgradient Property Status (DPS), which includes a Groundwater Management Plan. Also it is assumed, per the Activity and Use Limitation for the facility property, that the groundwater interceptor system that is currently operating to prevent discharge of non-aqueous phase liquid (NAPL) and OHM from the subsurface to the East Ditch will continue to operate and be maintained.

Residential properties - Main Street. As a temporary risk reduction measure at the downgradient residential properties listed in Table 6-1 and shown in Figure 10 of Appendix R, existing private wells (if any existed) have been abandoned, residences have been connected to the public water supply, and Notice of Limitations With Respect to Groundwater are in place or planned to insure no additional wells are installed at these residential properties. The wells at two properties (M-25/L-12 and M-25/L-9) are not drinking water wells, are not used currently, and have not yet been abandoned. Negotiations for abandonment are on-going. The use of or installation of private water supply wells for drinking water or any other purpose (with some exceptions related to testing of the water) at several residential properties along Main Street west of the site is restricted by the Notice of Limitation With Respect to Groundwater. The Notice was issued in conjunction with a Licensed Site Professional's (LSP) Groundwater Limitation Opinion. In addition, installation of new residential wells would require prior approval by the Town of Wilmington Board of Health under the Town's ordinances. Since the Board is aware of the groundwater issues, it is extremely unlikely the board would approve of new wells. In a recent draft guidance document (MADEP, 1997), the MADEP indicates that in such circumstances, the installation of new private wells is otherwise adequately regulated, and separate mechanisms to prevent installation of new private wells would not be necessary with respect to the MCP. Although this draft guidance is not official policy, it does indicate the Agency's current thinking on this matter. Therefore, for current land use, there are no downgradient properties with potable wells and measures have been taken to insure that no private residential wells will be installed at downgradient properties.

Industrial Properties - West of the Facility. Limitations with respect to groundwater use by owners/operators of industrial properties have been assumed in the risk assessment. It has been assumed that, with the exception of the Altron facility that currently has two operating wells that extract groundwater for non-potable industrial use, none of the downgradient industrial properties within the boundaries of the disposal site would install and use wells that would result in human exposure to site-related constituents. This assumption is supported by the following information: there is only one downgradient industrial property that uses groundwater for non-potable purposes (Altron); Olin has informed all of the downgradient property owners of the contamination; DPS, with an associated groundwater management plan (so that groundwater use may be managed in such a way as to prevent exposure to the contaminants), will be achieved at most, and perhaps all of the properties identified on Figure 11 of Appendix R; the Town of Wilmington's Board of Health Code of Regulations, Section 6 prohibits the installation of any well without permission from the Board of Health; the Board of Health may deny a permit when public health might be at risk; the Board of Health has been informed of the contamination and is unlikely to allow installation of a well that would access the contamination; and the properties are currently connected to the Public Water Supply. All of these factors, in combination, suggest that installation of wells on these industrial properties with associated human exposure is not a reasonably foreseeable use of the groundwater in this area. It is also assumed groundwater will not be put to potable use, since the groundwater in this area is categorized as GW-2 and GW-3.

Notice of DPS and an associated LSP's DPS Opinion have been offered by Olin to owners/operators of industrial properties, identified by map number and lot number per Table 6-2 and Figure 11 of Appendix R, as M-24/L-207, M-24/L-208, M-24/L-208B, and M-24/209A (related to both shallow and deep groundwater) and M-24/L-205, M-24/L-206, M-24/L-31A, M-24/L-31C, M-24/L-31D, M-24/L-33A, M-38/L-1, M-38/L-2, and M-26/L-2 (related to deep groundwater). These properties are the only industrial properties located in the area where impacted groundwater has migrated to the west of the site. With the Notice in place, current or foreseeable direct exposure to groundwater at these properties is controlled. Indirect exposures, such as vapor migration, are evaluated as part of the risk assessment.

The Notice of DPS includes a DPS Groundwater Management Plan. The purpose of the Management Plan is to prevent the exposure of human and environmental receptors to hazardous material at the downgradient property that is subject to the DPS, prevent an act by the owner/operator of the DPS property to cause the release to become worse than it otherwise would be, and avoid an activity that could prevent or impede the implementation of reasonably likely response actions in the future. Activities and uses inconsistent with the LSP's DPS Opinion include the use of groundwater for potable purposes and use of groundwater that are inconsistent with the above-stated purpose of the DPS Groundwater Management Plan. The groundwater management plans also require that installation of new wells be undertaken only after a thorough evaluation of risks associated with a proposed well. In addition, installation of a well would require prior approval from the Town of Wilmington. The owner/operators have an obligation to maintain compliance with the conditions of the Notice of DPS and the associated Groundwater

Management Plan in order to maintain the DPS. Examples of the Notice of DPS and the LSP DPS Opinion are provided in Attachment 3 of Appendix R.

Therefore, the two non-potable wells at the Altron facility are the only wells expected at downgradient industrial/commercial properties for both current and future land use.

Groundwater Interceptor System. An ongoing groundwater interceptor system in place in the Plant B area contains the groundwater, preventing discharge of OHM from the groundwater to the East Ditch. It is assumed here that this system will remain in place until such time that there is no longer a threat of release associated with the area and therefore there is no foreseeable future exposure to OHM in the groundwater in the Plant B area. The Activity and Use Limitations (AUL) for the property will contain an obligation to maintain and operate the groundwater interceptor system until there is no longer a threat of release, until there is no longer NAPL on the groundwater surface or until it is demonstrated that continued operation of the system would result in no measurable benefit.

There are two direct groundwater exposure pathways that are evaluated for current land use: worker exposure to process water at the Altron facility and ingestion of drinking water that is distributed from the Butters Row Treatment Plant.

Altron Wells. At the present time, there is only one commercial/industrial property with water wells in use in the area downgradient of the facility (between the facility and Main street). Two industrial wells at the Altron facility produce on average 180,000 gallons per day (gpd) for non-potable, industrial use. The process water is used primarily to wash printed circuit boards. Direct skin contact with the process water by workers is minimal. However, since the water is used in open tanks, any OHM that might volatilize from the water represents a potential inhalation exposure for workers. Inhalation exposures have not been documented at the facility, although site-related OHM, particularly ammonia, have been detected in the Altron wells.

Town of Wilmington Public Water Supply. At the present time, although elevated ammonia and sulfate have been detected in two of the Town of Wilmington groundwater supply wells, people are not actually exposed to this untreated water. The groundwater is combined with water from three other supply wells in the pipeline prior to entering the Butters Row Treatment Plant, where it is treated for removal of trichloroethyene from some other unknown source (which seems to no longer impact the Town water), for removal of naturally occurring iron and manganese, and for disinfection via chlorination. The disinfection process effectively eliminates ammonia from the treated drinking water. Therefore, all water distributed to Town residents from the Butters Row Treatment Plant meets current Federal and Massachusetts drinking water standards. Consumption of Town drinking water by Town residents and facility and Altron employees has been evaluated for current and foreseeable land use.

Given future use of land and groundwater, the following bullets identify potential human receptors and associated exposure pathways:

- Neighborhood resident who consumes Town drinking water, who may play in surface water and sediments (both on and off the facility property) and who may contact surface soils within the perimeter of the facility.

- On-property Worker who consumes Town drinking water and is occasionally exposed to on-property surface soil, and surface water and sediment in the on-property ditches

- Full time, long-term on-property worker who consumes Town drinking water and is frequently exposed to on-property surface soil, and surface water and sediment in the on-property ditches

- Worker at an Adjacent Industrial Property who consumes Town drinking water, is exposed to vapors that may migrate from shallow groundwater through the building slab to indoor air, and is exposed to groundwater that is used as process water at the property

- On-property Utility Worker who is exposed to surface soil and subsurface soil

- Construction Worker who is exposed to surface soil and subsurface soil at the facility property

There are assumed limitations on activities and uses that have been incorporated into the evaluation of risks for future land and groundwater use. The limitations with respect to groundwater that have been previously discussed are among those limitations. In addition, there are limitations that apply to the property itself.

6.1.1.4 Facility Property - Activity and Use Limitations

A notice of AUL is in preparation that will apply to the property at 51 Eames Street, preventing activities and uses that would be inconsistent with the findings of this risk assessment. The Notice will be issued in conjunction with an LSP's AUL Opinion. The current use of the former industrial property is largely unused office and warehouse space. Property maintenance activity is associated with the unused portions of the site. In the foreseeable future, use of the facility and the surrounding land is anticipated to remain industrial/commercial. The AUL identifies the following activities and uses as being inconsistent with the AUL Opinion: use of the property for residential purposes, for schools, for daycare facilities, for active outdoor recreational purposes without prior risk evaluation by an LSP, for retail or wholesale commercial use that involves regular presence of children, excavation of surface or subsurface soil or other materials in the Calcium-Sulfate Landfill; excavation of subsurface soils (deeper than 3-feet bgs) and placement of same at the soil surface, excavation activities in areas identified as surface soil or subsurface soil hot spots, installation of a water supply well within the property, and withdrawal of groundwater for any purpose other than sampling, testing, or containment/treatment. The above-identified uses and activities are considered inconsistent with the AUL Opinion because the risk assessment conducted here did not specifically evaluate the risks associated with them. The AUL contains a provision that any of these activities or uses could be permitted provided an LSP determines that associated risks are consistent with MCP requirements.

The AUL for the property will require operation of the groundwater interceptor system until there is no longer a threat of release to the East Ditch, until there is no longer NAPL on the groundwater surface, or until it is demonstrated that continued operation of the system would result in no measurable benefit.

Future use of groundwater at and downgradient of the facility is expected to remain similar to current use. The temporary risk reduction measures described previously for downgradient residential properties will continue to be effective in preventing potable use and non-potable direct exposure to groundwater. In addition, for the reasonably foreseeable future, the Altron wells are expected to be the only operational wells at the industrial/commercial properties downgradient of the site (between the facility and Main Street) for the reasons discussed above.

Based on existing data, the dense layer appears to be moving very slowly, if at all. Disposal of material into unlined pits/lagoons and Lake Poly began in about 1953 and continued until a treatment plant was installed and connected to the Metropolitan District Commission sewer in 1972. Although discharge of materials began more than 40 years ago and ceased 20 years ago, the migration has been limited in extent. There is not now an imminent threat to the Town of

Wilmington public water supply, whose wells are located about 4,000 feet to the west of the facility. The slope of the bedrock surface is a factor in the migration of this material. It is likely that migration of the material slowed with time due to flattening of the bedrock slope away from the disposal area. The cessation of disposal would also have resulted in a decrease in driving head for migration of the denser material. In addition, the geochemical behavior of the inorganic material potentially results in the precipitation of some constituents. Additional data necessary to provide an understanding of the behavior and potential for migration of the inorganic plume has been collected. These data have been used to model the migration of groundwater, dissolved constituents and constituents of the dense layer. The modeling data have been used to estimate future concentrations in water brought into the Wilmington public water supply system from each of the five water supply wells that contribute water to the Butters Row Treatment Plant. Consumers of water from the Town water system are evaluated as potential future receptors. It is assumed that the Butters Row Treatment Plant will continue to operate indefinitely and that disinfection of the water via chlorination will effectively remove ammonia from the water.

6.1.1.5 Risk Characterization

Potential risks to human health associated with exposure to contaminants at the Wilmington site have been characterized. Risk is a function both of exposure and toxicity. The extent of risk depends on the nature, duration, and frequency of exposure to contaminants and characteristics of the exposed population. Information presented in the Exposure Assessment section, combined with the dose-response toxicity data presented in the Dose-Response Assessment section, is the basis for this risk characterization. Per the requirements of the MCP at 310 CMR 40.0993(6), risk of harm to human health is characterized by:

1. Comparing Cumulative Receptor Cancer Risk to the Cumulative Receptor Cancer Risk Limit (10^{-5}) for current and future land use; and

2. Comparing Cumulative Receptor Non Cancer Risk to the Cumulative Receptor Non Cancer Risk Limit (Hazard Index of 1) for current and future land use; and

3. Comparing exposure point concentrations to applicable or suitably analogous public health standards.

The details of the risk calculations are presented in Section 5.0 of Appendix R. These calculations were made consistent with the MCP and MADEP guidance (MADEP, 1995).

Potential risk to public welfare was evaluated per 310 CMR 40.0994, using two major components:

1. A consideration of such factors as the existence of nuisance conditions, loss of property value, the unilateral restriction of the use of another person's property, and any monetary or non-pecuniary costs not otherwise considered in the

characterization of risk of harm to health, safety, and the environment but which may accrue due to the degradation of public or private resources directly attributable to the release of the oil and/or hazardous material, and

2. Comparison of the concentrations of oil and/or hazardous material to the Upper Concentration Limits (UCLs) in soil and groundwater as described in 310 CMR 40.0996.

The evaluation of risk to public welfare is presented in its entirety in Section 6.0 of Appendix R.

Potential risk to safety was evaluated per 310 CMR 40.0960 by determining if any release-related conditions do, or might in the future, pose a threat of harm or bodily injury to people. The complete evaluation of the risk to safety is presented in Section 6.0 of Appendix R.

6.1.2 Human Health, Safety, and Public Welfare Risk Assessment Findings

6.1.2.1 Human Health Risks Associated with Current Land Use

Potential cancer risks and noncancer hazard indices for current land use are presented in Table 6-3. Total receptor cancer risks were less than the MCP Cumulative Receptor Cancer Risk Limit (1×10^{-5}) and within the NCP target carcinogenic risk range of between 1×10^{-6} and 1×10^{-4} for all receptors, including site workers, utility workers, neighborhood residents, and off-property industrial workers. Total receptor non-cancer risks were below the MCP Cumulative Non-Cancer Risk Limit of a Hazard Index of 1 for all receptors.

6.1.2.2 Human Health Risks Associated with Foreseeable Future Land Use

Table 6-4 presents the potential carcinogenic risks and noncancer hazard indices associated with foreseeable future land use. Total receptor cancer risks were less than or equal to the MCP Cumulative Receptor Cancer Risk Limit (1×10^{-5}) and within the NCP target carcinogenic risk range of between 1×10^{-6} and 1×10^{-4} for all receptors, including future site workers, on-property utility workers, construction workers, full-time, long-term industrial workers, neighborhood residents, and off-property industrial workers.

6.1.2.3 Comparison to Applicable or Suitably Analogous Public Health Standards

Groundwater exposure point concentrations were compared to applicable or suitably analogous public health standards. Massachusetts Maximum Contaminant Levels (MMCLs) promulgated under 310 CMR 22.00 are applicable public health standards for groundwater classified as category GW-1. Groundwater at and downgradient of the facility classified as GW-1 is identified on Figure 14 in Appendix R. For GW-1 groundwater within the Zone II area, the following groundwater constituents have maximum concentrations (exposure point concentrations) that exceed drinking water standards: 1,2-dichloroethane, 1,2-dichloroethene, benzene, chloroform, bis(2-ethylhexyl)phthalate, methylene chloride, trichloroethene, vinyl chloride, chromium, nitrate,

arsenic, and lead. It does not appear that 1,2-dichloroethane, 1,2-dichloroethene, methylene chloride, trichloroethene, vinyl chloride, arsenic, or lead are related to the Wilmington facility. Within the GW-1 area located in the northern portion of the facility adjacent to Eames Street, one groundwater constituent has a maximum concentration (exposure point concentration) that exceeds a drinking water standard: bis(2-ethylhexyl)phthalate. While there is an exceedance of the drinking water standard in this area, it is unlikely that any human exposure to this groundwater will occur in the foreseeable future. In addition, an ongoing groundwater interceptor system will remain in place until drinking water standards are achieved. Because there are Exposure Point Concentrations (EPCs) within the Zone II of the Public Water Supply (PWS) that exceed the applicable public health standards, it cannot be concluded that a condition of no significant risk of harm to health exists.

For an off-site industrial worker, there is no indication that workplace Air Standards might be exceeded as a result of industrial groundwater use.

The Massachusetts Surface Water Quality Standards are not considered applicable public health standards for this site. The human health components of these standards are based on either use of water as drinking water or on ingestion of fish or other organisms. Neither of these situations is applicable to this site.

No applicable public health standards published by MADEP exist for any of the chemicals in the other media where exposures could potentially occur (soil and sediment).

6.1.2.4 Characterization of Risk of Harm to Safety and Public Welfare

As part of the Method 3 risk characterization, an evaluation of risk of harm to safety and public welfare is conducted by evaluating such factors as the existence of nuisance conditions, loss of property value, and the unilateral restriction of the use of another person's property, and by comparing the exposure point concentrations of oil and/or hazardous material to the UCLs in soil and groundwater as described in 310 CMR 40.0996.

Safety. As documented in the Phase II report, there is currently no threat of explosion or direct contact with releases from this site which is likely to endanger public safety or welfare. A previous imminent hazard evaluation was conducted as part of an Immediate Response Action (IRA) focused on Drum Area A and Drum Area B (Olin, 1995). These areas are currently fenced and no access to these areas is currently allowed. Although some materials identified in the subsurface were identified as possible fire/explosion hazards, these materials have been in place in the subsurface for more than 20 years without any such hazardous conditions being observed. Under current site conditions, no nuisance conditions are known to exist. Access to the site is restricted and no noxious fumes or noises are associated with the site. A condition of no significant risk of harm to safety exists at the site for current land use conditions. Under future land use conditions, potential fire/explosion hazards may exist if the Drum Area A and Drum Area B areas are excavated. Therefore, it cannot be concluded that a condition of no significant risk of harm to public safety exists.

Public Welfare. The results of the evaluation of risks to public welfare are summarized below. Although Limitations With Respect To Groundwater have been implemented on another person's property (properties along Main Street), these limitations have not been applied unilaterally. The owners of these properties have been compensated for the limitations on groundwater use. In a like manner, the Groundwater Management Plans that are part of the DPS for the neighboring industrial properties are not applied unilaterally and they benefit those property owners because their responsibility to investigate or remediate the groundwater is reduced by the DPS status.

The existence of OHM within the Zone II of the Town of Wilmington PWS may represent a loss of available public resources because it limits future available groundwater resources for the Town. A recent report indicates that the Town of Wilmington has identified the need to develop an additional 1.8 million gallons per day drinking water capacity over the next several years. The presence of the dense layer may prevent the development of additional water supply wells in the Maple Meadow Brook area. Further study will be required to determine if that area would have been a realistic drinking water resource in the absence of the dense layer. If it is confirmed that the presence of the dense layer has resulted in reduction in public groundwater/drinking water resources, a condition of no significant risk of harm to public welfare would not exist for future land use.

There is evidence that due to potential future loadings of ammonia to the Butters Row treatment plant, manipulations and/or modifications of Butters Row Treatment Plant operations may be required to maintain adequate chlorine residuals to insure disinfection of treated water.

Massachusetts Secondary Maximum Contaminant Levels (SMCLs) are designed to protect aesthetic characteristics of water resources (such as taste and odor issues) and therefore could be considered applicable or suitably analogous public welfare standards for the groundwater beneath this site. SMCLs are not legally enforceable standards. A Zone II is the area of an aquifer that contributes to a public water supply well under the most severe recharge and pumping conditions that can be realistically anticipated.

Concentrations of aluminum, chloride, iron, manganese, sulfate, and zinc within the Zone II of the water supply exceed their respective Secondary Drinking Water Standards. For groundwater within areas of GW-1 groundwater, the maximum detected concentration is the EPC, since each monitoring well or supply well is considered a separate exposure point. Rather than compare concentrations in each individual well to UCLs, the maximum concentration from all wells has been is compared to the UCL as a streamlining measure.

There were no EPCs in surface soil or subsurface soil that exceed corresponding UCLs. The groundwater UCLs for chromium, nickel and cadmium are exceeded at the dense layer hot spot in the non-Zone II area. There are also exceedances of indeno(1,2,3-cd)pyrene and bis(2-ethylhexyl)phthalate UCLs in Plant B hot spot, and chromium and zinc concentrations exceed UCLs in Zone II groundwater. The presence of NAPL with thickness greater than one-half inch in the area of Plant B is by definition, an exceedance of a UCL.

Because there are groundwater EPCs that exceed corresponding UCLs, there is NAPL present that exceeds the corresponding UCL, groundwater EPCs exceed Secondary Drinking Water Standards, and public groundwater resources have been impacted, there is not a condition of no significant risk of harm to public welfare.

6.1.3 Conclusions

The assessment of risk of harm to human health indicates that for current land use and current site conditions, cancer and noncancer risks are below the corresponding MCP Cumulative Receptor Cancer Risk Limit (1×10^{-5}) and Cumulative Receptor Non-Cancer Risk Limit (hazard index = 1) for all receptors.

For future land use and future site conditions, cancer and noncancer are below the corresponding MCP Cumulative Receptor Cancer and Non-Cancer Risk Limit for all receptors.

The water currently distributed from the Butters Row Treatment Plant currently meets all Massachusetts Maximum Contaminant Levels (MMCLs). The sodium concentrations in the water distributed from the Butters Row Treatment Plant do exceed the Massachusetts Drinking Water Guideline for sodium. However, the background groundwater concentrations of sodium also exceed this conservative drinking water guideline. The concentrations of sodium in water distributed from Butters Row Treatment Plant do not appear to be related to the Wilmington facility.

Groundwater modeling has been conducted to estimate future concentrations of the most mobile site-related constituents in groundwater. The results of this modeling indicate that for the reasonably foreseeable future the water distributed from the Butters Row Treatment Plant would continue to meet the Massachusetts Drinking Water Standards (MMCLs) and Guidelines (with the exception of sodium as noted above. The modeling indicates that chromium would not migrate to the water supply wells.

For groundwater within the Zone II of the Town of Wilmington PWS (classified as GW-1 groundwater per the MCP), concentrations of 1,2-dichloroethane, 1,2-dichloroethene, benzene, chloroform, bis(2-ethylhexyl)phthalate, methylene chloride, trichloroethene, vinyl chloride, chromium, nitrate, arsenic, and lead exceed the corresponding Massachusetts MCLs (MMCLs), which are considered applicable public health standards. It does not appear that 1,2-dichloroethane, 1,2-dichloroethene, methylene chloride, trichloroethene, vinyl chloride, arsenic, or lead are associated with the Wilmington facility.

For current land use, it can be concluded that a condition of no significant risk of harm to human health exists. For future land use, although noncancer risks are less than the MCP Cumulative Receptor Non-Cancer Risk Limit and the cancer risks are less than the MCP Cumulative Receptor Cancer Risk Limit, because groundwater exposure point concentrations within the Zone II of the PWS exceed MMCLs which are applicable public health standards, it cannot be concluded that, for future land use, a condition of no significant risk of harm to health exists.

Because of potential danger of fire or explosion associated with future excavation in the drum areas, it cannot be concluded that a condition of no significant risk of harm to safety exists.

The risk of harm to welfare was evaluated by comparing site concentrations to UCLs as well as impacts to property value and drinking water sources. Because the exposure point concentration for chromium and other groundwater constituents in groundwater are above corresponding UCLs, because NAPL with thickness greater than one-half inch is present in the Plant B area, because Secondary Drinking Water Standards are exceeded in the Zone II of the Public Water Supply, and because a public groundwater drinking water source has been impacted, it cannot be concluded that a condition of no significant risk of harm to public welfare exists.

6.2 ENVIRONMENTAL RISK CHARACTERIZATION

An ERC was conducted to assess the risks to ecological receptors posed by OHM detected in the study area including Olin Corporation's Wilmington, Massachusetts facility. The primary goal of the risk characterization was to determine whether there is an indication of the potential for ecological harm and /or evidence of ecological harm associated with OHM at the facility.

6.2.1 Environmental Risk Characterization Approach

This ERC was conducted in a manner consistent with the MADEP's Guidance for Disposal Site Risk Characterization, Interim Final Policy (WSC/ORS-95-141) (MADEP, 1995) and the 'Method 3-Environmental Risk Characterization" published in April 1996, which comprised Chapter 9 of the Interim Final Policy. This policy provides additional guidance to that contained within the regulations (310 CMR 40.0995 (4)) regarding the conduct of environmental risk characterizations. This ERC is consistent with the Scope of Work, Stage II Environmental Risk Characterization, Olin Corporation Wilmington Facility, DEP RTN: 3-0471, January, 1997 (Olin, 1997) which was submitted to the MADEP. The MADEP provided conditional approval of that Scope of Work in a letter dated February 7, 1997 (MADEP, 1997a).

The ERC consists of the following three steps: Problem Formulation, Analysis, and Risk Characterization. The problem formulation defines the purpose and scope of the ERC and discusses the nature and distribution of OHM in media at the site. OHM of potential concern (OHMPC) are also identified for quantitative evaluation in the ERC. Likely ecological receptors and exposure pathways are also identified, and a site conceptual model is developed. Finally, the problem formulation presents the specific assessment and measurement endpoints for the ERC. Assessment endpoints represent the particular ecological effect being evaluated in the assessment (e.g., a significant reduction in amphibian populations at the site) whereas the measurement endpoints approximate or provide a measure of the assessment endpoint (e.g., results of frog toxicity tests).

In this ERC, media evaluated at the site included surface soil, surface water, and sediment. Surface soil was evaluated for all areas at the facility containing suitable terrestrial habitat. Surface water and sediment were evaluated separately for five areas at the facility (On-Property West Ditch, Off-Property West Ditch, South Ditch, Ephemeral Drainage, and Central Pond). The East Ditch was not evaluated because it provides minimal cover and contains few prey items to attract foraging wildlife.

Representative ecological receptors evaluated included the green frog, green heron, American woodcock, and red fox. Exposure pathways for the frog include direct contact with surface water and sediment, while those for the remaining wildlife receptors included food-chain exposure and incidental ingestion of surface soil or surface water and sediment.

The assessment endpoint for each of the selected receptor species was a reduction in population size. The measurement endpoints used to evaluate this endpoint for the frog included the results of a toxicity test, as well as the results of a population model, field observations, and a comparison of published amphibian toxicity data to surface water data. A 25 percent decrease in population size was assumed to represent a significant effect to amphibian species. The measurement endpoint used to evaluate the assessment endpoint for the green heron was a comparison of predicted dietary exposures (using a food chain model) with published toxicity data for bird species. An additional assessment endpoint evaluated for the heron was a reduction in population size as a result of reduced prey abundance. The results of the frog population model were used as a measurement endpoint to evaluate this endpoint for the heron. A 50 percent decrease in frog population size was assumed to represent a significant effect to herons that could rely on frogs and other amphibians for part of their diet. The higher percentage used for this measurement endpoint is due to the fact that, because frogs are not the only source of food for a predator such as the heron, it would take a larger decrease in frog population size to adversely affect the heron population. The results of a food chain model were also compared with published toxicity data for bird or mammal species to evaluate risks to the terrestrial receptors (woodcock and fox).

The analysis step contains the exposure and effects assessments. In order to obtain site-specific information regarding exposure and toxicity, both biological tissue sampling and toxicity tests were performed. Biological tissue samples were chemically analyzed, and the analytical data were incorporated into food chain models used to help characterize risks to semi-aquatic and terrestrial wildlife receptors. Earthworm toxicity tests were conducted in which laboratory-reared earthworms were exposed to surface soil samples from the site. A type of frog toxicity test, referred to as the Frog Embryo Teratogenesis Assay-Xenopus (FETAX) test, was conducted in which frog embryos were exposed to sediment elutriate samples from the site. The results of the earthworm tests were used to help characterize risks to terrestrial wildlife receptors that may rely on soil invertebrates as prey items. Earthworm tissue concentrations were also obtained from these tests which were incorporated into the food chain model. The results of the FETAX tests were used to help characterize risks to amphibians as representative aquatic receptors, and to semi-aquatic receptors that may rely on amphibians as prey items.

A population model was used to help relate the results of the toxicity tests, which looked at embryo mortality and malformation, to potential population-level impacts.

6.2.2 Environmental Risk Characterization Findings

The risk characterization combines information regarding ecological exposures and effects to characterize the potential risk of harm to ecological receptors at the site. A weight-of-evidence approach is used to consider the various lines of evidence (e.g., toxicity test results, population model results, etc.) to determine whether or not a condition of no significant risk of harm to the environment exists at the site. It also contains a comparison to Applicable or Suitably Analogous Standards (ASASs), which for this ERC consist of Massachusetts Water Quality Standards.

6.2.2.1 Aquatic Receptors

Risks to aquatic receptors (i.e., the green frog) were evaluated based on results of FETAX toxicity tests, results of a population model, field observations, and concentrations of OHMPCs in surface water and sediment elutriate relative to published RTVs. Table 6-5 contains a summary of the risk evaluation for the green frog. The results of the toxicity tests indicate significant toxicity at two locations in the On-Property West Ditch. The population model, which incorporated the results of the toxicity tests, indicated a greater than 25 percent reduction in frog subpopulations in the On-Property West Ditch. These results are given greater consideration in the overall weight of evidence evaluation because they are based on site-specific information and a model which directly relates the results of the toxicity tests to a population level effect, which is the selected assessment endpoint. Sediment elutriate concentrations were compared with amphibian RTVs in an attempt to identify chemicals responsible for the toxicity observed in the tests; no trends were noted, and a regression analysis indicated that there is no correlation between any of the OHMPCs and the observed toxicity.

A comparison of surface water concentrations with amphibian RTVs resulted in Hazard Indeces (HIs) greater than 1, particularly in the Off-Property West Ditch, South Ditch, and Ephemeral Drainage areas. Chromium, ammonia, and di-n-octyl phthalate are risk contributors for historical data. Concentrations and associated HIs for recent data are considerably lower than historical data in both the Off-Property West Ditch and the Ephemeral Drainage. The primary site-related risk contributor from the more recent data is ammonia. Aluminum and iron are also identified as potential risk contributors.

6.2.2.2 Semi-Aquatic Wildlife Receptors

Risks to semi-aquatic wildlife receptors (i.e., the green heron) associated with exposures to OHMPC were evaluated based on results of a food chain model, which evaluated food chain exposures based on site-specific tissue concentrations for likely prey items (e.g., frogs and crayfish), as well as surface water and sediment ingestion exposures. Table 6-6 contains a summary of the risk evaluation for the green heron. Results of the model indicated that HIs for each of the ditch areas evaluated are less than one, indicating that there is no significant risk of harm to semi-aquatic receptors from exposure to OHMPCs at the site. Indirect impacts to semi-aquatic wildlife receptors from reduced prey abundance

were also evaluated, based on the FETAX toxicity test results which were incorporated into the frog population model. A 50 percent reduction in abundance is unlikely at all locations except possibly the On-Property West Ditch. This ditch comprises only a portion of potential habitat for the heron at the site, and since a significant reduction in prey items at other areas of the site is not predicted, an overall 50 percent reduction in abundance is unlikely.

6.2.2.3 Terrestrial Wildlife Receptors

Risks to terrestrial wildlife receptors (i.e., the woodcock and red fox) associated with exposures to OHMPC were evaluated based on results of a food chain model, which incorporates site-specific tissue concentrations for likely prey items (e.g., earthworms and small mammals) as well as incidental ingestion of surface soil. Table 6-7 contains a summary of the risk evaluation for terrestrial wildlife. Results of the model indicated that the HI for the fox is below 1, while that for the woodcock is 1.9. All OHMPC-specific HQs for the woodcock were below 1; the analyte contributing the most to this HI is aluminum, with an HQ of 0.83. These results support a conclusion of no significant risk of harm to terrestrial wildlife receptors at the site.

Indirect impacts to terrestrial wildlife receptors from reduced prey abundance were also evaluated, based on the earthworm toxicity test results. No significant toxicity was observed in any of the soil samples tested. However, in the chronic earthworm toxicity test, potential reproductive effects were indicated by low cocoon production relative to the laboratory control. Low cocoon production was also noted in the off-site reference location. This low cocoon production does not appear to be chemical related, as it was similar at all locations tested, regardless of chemical concentrations present in the samples used for the tests. Low cocoon production is attributed to a reflection of differences in the physical characteristics of the local soils (grain size, percent clay, amount of organic material) relative to those of the formulated soil used in the laboratory control. The overall results of this evaluation indicate that there is no significant risk of harm to terrestrial wildlife receptors from reduced prey abundance resulting from exposure to OHMPCs at the site.

6.2.2.4 Comparison to ASASs

Surface water concentrations of several metals, including aluminum, chromium, copper, iron, and lead, and ammonia at one or more surface water locations at the site exceed Massachusetts Surface Water Quality Standards, which are considered ASASs. Because these ASASs are exceeded, the MCP states that a condition of no significant risk of harm to the environment has not been achieved. These ASASs consist of criteria which are not truly appropriate for the types of aquatic receptors that would occur in surface water bodies at this site, because they are protective of sensitive cold water fish species such as trout which would not be expected to occur at this site, and they should therefore be given a low overall weight of evidence relative to the other findings of this ERC.

6.2.3 Conclusions

The results of the ERC support a finding of no significant risk of harm to terrestrial receptors exist at the Olin Wilmington facility. However, for aquatic receptors, a condition of no significant risk of harm to the environment does not exist. Future studies or remedial actions should focus on addressing sediment-related risk in the On-Property West Ditch (i.e., a Tier 1 Toxics Identification Evaluation [TIE]), and potential surface water-related risks in the Off-Property West Ditch, South Ditch, and Ephemeral Drainage Areas.

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7.0 CONCLUSIONS

The purposes of the Supplemental Phase II Field Investigation program were met by obtaining additional site-specific information needed to adequately evaluate remedial alternatives. This investigation supplements and enhances the findings presented in the Comprehensive Site Assessment (CSA) (CRA 1993). The specific original objectives of this investigation which have been achieved include the following (BCM, 1993):

- To define the morphology and depth of the bedrock surface in areas further west, east and southwest of the facility where bedrock valleys have been identified;
- To define the limits of the dense, inorganic layer to the west, east and southwest of the property;
- To evaluate the geochemistry of the dense layer and its effect on migration and potential remedial alternatives;
- To evaluate aquifer characteristics and the potential rate of dense layer migration;
- To determine the relationship between the shallow groundwater and surface water in the ditch system, and evaluate the geochemistry of the observed flocculent; and
- To obtain additional information about the soil quality at the former Lake Poly Liquid Waste Disposal Area (Lake Poly).

Based in part on the preliminary results of the Supplemental Phase II Field Investigation, additional work plans were prepared, submitted and approved by the Massachusetts Department of Environmental Protection (MADEP). The objectives for these work plans were also addressed during this study, and included the following:

- To investigate the nature of the terrain conductivity anomalies detected in the deepest parts of the bedrock valley underlying this area;
- To perform additional seismic reflection surveys and installation of additional borings to improve understanding of the bedrock morphology in certain areas;
- To provide an understanding of the relationship between water quality in the supply wells and nearby monitoring wells in conjunction with other influences, such as pumping rate;
- To supplement existing information in specific areas, including hydrology of the South Ditch System, vertical behavior of the dense layer, and background chemical concentrations; and

- To obtain data in response to MADEP comments on the original Risk Assessment (ABB-ES, 1993a), and more updated analytical data for use in the current Risk Assessments.

The following conclusions have been reached which are supported by the original CSA (CRA 1993) and the detailed studies and findings of the Supplemental Phase II Investigation. Conclusions concerning the bedrock and chemicals of concern are presented in Section 7.1 and 7.2, respectively. The areas of environmental concern, associated with the major source areas and impacted media, are discussed in Section 7.3. The findings of the geochemical conditions of the aquifer material, the dense layer, surface water and flocculent, in addition to geochemical modeling conclusions, are presented in Section 7.4.

Conclusions concerning the fate and transport of site contaminants is presented in Section 7.5. Section 7.6 presents the conclusions of the human health and ecological risk assessment. The conclusions concerning areas of potential environmental concern are presented in Section 7.7, followed in Section 7.8 by the Supplemental Phase II Investigation Completion Statement.

7.1 BEDROCK

The Supplemental Phase II Investigations have shown that the bedrock at the site acts as a barrier to the vast majority of contaminant migration. The water quality results show that bedrock wells in close proximity to the dense layer are impacted by dense layer constituents, however, the results wells away from the dense layer (Cook Avenue Wells, GW-65BR, GW-80BR, and GW-81BR) show generally background levels of dense layer constituents. The bedrock has a low permeability, minimal storage capacity, and lacks significant water-bearing zones. Yields are less than 1 gallon per minute (gpm), with calculated transmissivities of 0.035 to 2.3 square feet per day (ft^2/day).

Defining the bedrock surface is important because the extent of the dense layer, the media with the greatest mass of inorganic contaminants, is controlled by the shape of the bedrock surface. Specifically, the buried bedrock valleys at the site have controlled the direction of the past movement of the dense layer from source areas to its current position in the closed bedrock depressions. The primary features of the bedrock surface at the site are:

- A Western Bedrock Valley extends from the facility towards the west and reaches depths of over 120 feet beneath the wetland surrounding Maple Meadow Brook. The Western Bedrock Valley has been determined to terminate slightly north of where the former Middlesex Canal crosses the wetland,

- An Eastern Bedrock Valley, identified in the CSA, does not extend significantly beyond the eastern facility boundary, and

- A possible Southwestern Bedrock Valley located along the southwestern property line and extending towards Breed Avenue has been determined not to exist.

7.2 CHEMICALS OF CONCERN

Under the MADEP Risk Assessment Guidelines, over 140 organic and inorganic compounds could be considered chemicals of concern based on frequency of detections and background concentrations. The Phase II Human Health Risk Assessment presented in Appendix R considered all chemicals detected through laboratory analyses. In order to provide a more meaningful context of identifying areas of concern, the results of the CSA (CRA, 1993) and the Supplemental Phase II Investigations were reviewed to identify the frequently detected parameters in each media and the parameters that are consistently detected in most media.

The results of the Supplemental Phase II Investigations confirmed the CSA finding that inorganic compounds are the most prevalent class of contaminants associated with the site. Ammonia, chloride, sulfate, and chromium are the main inorganic compounds that were detected at high concentrations in all media on-property and in the sediment and the dense layer off-property. The results of the Supplemental Phase II Investigations also confirmed the CSA finding that organic compounds are less prevalent off-property and are present at the highest concentrations in sediment, subsurface soil, and subsurface waste samples on-property These parameters provide a representative picture of the distribution of the different types of contaminants in all media and are expected to be useful in future monitoring of conditions and later evaluations of the effectiveness of response actions, if needed. The parameters selected to define the major areas of concern are listed below:

Inorganics:	ammonia, chloride, chromium, sulfate
Volatiles:	acetone and total trimethylpentenes
Semivolatiles:	n-nitrosodiphenylamine, total phenols, total phthalates
Pesticides:	total pesticides

7.3 AREAS OF ENVIRONMENTAL CONCERN

The results of the Supplemental Phase II Investigations and the CSA were used to identify the following main impacted media associated with the major source areas. These areas also represent the portions of the site with the highest concentrations of contaminants, but do not necessarily represent the areas with the greatest potential for exposure. The risk associated with each of the impacted media and the need for response actions is summarized in Section 7.7. Olin has already acted to protect the public and on-property workers from potential risks in several of these areas including Plant B, the Off-Property West Ditch and South Ditch, and the Drum Areas. The major source areas and the impacted media are listed below:

On-Property

- Groundwater and soils in the former Plant B Production and Tank Farm Areas
- Soils and sediment in the area of former Lake Poly
- Soils and waste in Drum Area A and Drum Area B
- Subsurface soils associated with the East and West Pits
- Subsurface soils associated with the Acid Pits and Lined Lagoons
- Shallow and Deep Groundwater at GW-54 well cluster

Off-Property

- Off-Property Deep Groundwater and Dense Layer
- Groundwater, surface water, and sediment near the South and West Ditches

Groundwater and Soils - Plant B Production and Tank Farm Areas

Separate-phase oil and groundwater are contained by the three interceptor wells that prevent the discharge of oil and hazardous material into the East Ditch. Soils are impacted by low levels of VOCs, moderate levels of inorganics, and high concentrations of SVOCs including bis(2-ethylhexyl)phthalate, di-n-octylphthalate, and NNDPA. The oil in the area around Plant B contains organics including high concentrations of TMPs and bis(2-ethylhexyl)phthalate, and lesser amounts of NNDPA and di-n-octylphthalate. The groundwater results show much lower concentrations of these organics and moderate concentrations of inorganics including ammonia, chloride, chromium, manganese, and sulfate which do not appear to be related to the floating oil.

Potential environmental concerns at Plant B are subsurface soils, groundwater, and the floating oil.

Soils and Sediment - Lake Poly

The subsurface and surface soils at Lake Poly contain high concentrations of SVOCs, VOCs, and inorganics. The most prevalent compounds at the highest concentrations in the subsurface and surface soils are bis(2-ethylhexyl)phthalate, NNDPA, TMPs, ammonia, and chromium. In the groundwater, phthalates, TMPs, acetone, ammonia, and chromium are also detected in the shallow groundwater downgradient of Lake Poly.

Potential environmental concerns at Lake Poly are subsurface soils, sediments, and the groundwater.

Soils and Waste - Drum Area A and Drum Area B

An Imminent Hazard Evaluation for Drum Area A and Drum Area B was submitted to MADEP in November, 1994. The evaluation concluded that buried drum areas could pose an imminent hazard because the presence of Opex® and Kempore® may represent a threat of explosion or fire if

subjected to heat, flame, or friction. An Immediate Response Action (IRA) Completion Report was submitted to MADEP on June 28, 1996. This report concluded that Olin had completed all components of the IRA, that a condition of Substantial Release Migration does not exist for Opex® and Kempore®, and that No Imminent Hazard currently exists due to the buried drum areas on the Olin facility. As fencing is maintained around the two buried drum areas and the facility to prevent access to these areas, the need to continue the IRAs is eliminated.

The subsurface waste and/or soils in Drum Area A contained elevated concentrations of TMPs, bis(2-ethylhexyl)phthalate, NNDPA, and inorganics including ammonia, chloride, chromium, sodium, and sulfate. Surface soil exhibited elevated concentrations of phthalates, NNDPA, several inorganics including aluminum, chromium, and potassium. In addition, well GW-35D, adjacent to Drum Area A, shows Kempore® at a concentration of 3.8 mg/l (milligrams per liter), suggesting that Kempore® is also present in the soils or waste material in this area.

The subsurface waste and/or soils in Drum Area B contained elevated concentrations of acetone, toluene, phenol, and inorganics including chromium and sulfate. Surface soil exhibited elevated concentrations of phthalates, phenols, NNDPA, several inorganics including aluminum, ammonia, and chromium.

Potential environmental concerns at both Drum Areas A and B are surface soils, subsurface soils and subsurface waste. Shallow groundwater may also be a potential environmental concern; however, results suggest that the Drum Areas may be contributing only trace to low concentrations of contaminants to the groundwater in the immediate vicinity of the Drum Areas.

Groundwater, Surface Water and Sediment - South and West Ditches

Off-Property West Ditch and South Ditch

An IRA has been implemented to prevent chromium-bearing precipitate from migrating off site. A temporary weir was installed in July 1994 across the South Ditch to increase the water level in the Off-property West Ditch, thereby allowing the flocculent to settle out of the surface water. The increased surface water head also minimized the discharge of shallow groundwater to the ditches, and reduced the creation of additional flocculent. Hay bale filter dams located downstream across the South Ditch prevents downstream migration of flocculent. An associated monitoring program begun in April 1994 has continued.

Surface water and shallow groundwater data collected since the CSA (CRA, 1993) have also shown that the inorganic concentrations in the Off-property West Ditch and South Ditch have greatly decreased since the installation of the weir. The geochemical environment in the Ditch System surface water strongly favors the immobilization of dissolved chromium through precipitation and adsorption reactions. The increased heads created by the weir have generally improved water quality, but may have moved some inorganics and VOCs in the shallow groundwater further west and may have changed the location of the dominant groundwater discharge points along the South Ditch

The CSA identified elevated concentrations of ammonia, chromium, and sulfate in the surface water. Sediment results in both branches of the Ditch System and surface soils (SWMU 30) show elevated concentrations of chromium, sulfate, phthalates, and NNDPA, while 4-bromophenyl-phenylether and 4-chlorophenyl-phenylether were only detected in the South Ditch sediments.

Potential environmental concerns at Off-Property West Ditch and South Ditch are shallow groundwater, sediments, and surface soils.

On-Property West Ditch

The sediments and adjacent surface soils have elevated concentrations of TMPs, phthalates, NNDPA, PAHs, and inorganics including ammonia, chloride, chromium, and sulfate. Pesticides were detected in sediments and surface soils in these areas. In the surface water, ammonia, phthalates, and VOCs were detected at low concentrations and sulfate was detected at moderate concentrations. The On-property West Ditch is not a groundwater discharge area, and consequently NNDPA, phthalates, phenols, and pesticides are not detected in the surface water. Overall, the concentrations of inorganics and organics present in surface water are lower than the concentrations found in the South Ditch or Off-property West Ditch surface water.

Potential environmental concerns at the On-property West Ditch are surface soils and sediments.

Central Pond, Central Wetland, and Surrounding Area

The Central Pond sediment contained oils and elevated concentrations of TMPs, NNDPA, bis(2-ethylhexyl)phthalate, 4-bromophenyl-phenylether, pesticides, and inorganics including aluminum, chromium, sodium and sulfate. The Central Pond surface water has elevated levels of chromium and sulfate. The surface soils in the surrounding area (grid Area 8) have elevated concentrations of bis(2-ethylhexyl)phthalate, PAHs, pesticides, and inorganics including chromium and sulfate. The highest concentrations of these parameters were observed in pockets of residual wastes that were typically found in the drainage areas that lead to the Central Pond. Surface soils in the Central Wetland within grid Area 9 have elevated concentrations of 1,1,1-trichloroethane, bis(2-ethylhexyl)phthalate, and ammonia.

Potential environmental concerns at the Central Pond, Central Wetland, and Surrounding Area are sediments (Central Pond and Central Wetland) and surface soils (Central Pond, Central Wetlands, and Surrounding Area).

Ephemeral Drainage

The Ephemeral Drainage sediment and surface water has elevated concentrations of sulfate. The surface soils in the area near the GW-55 well nest (SWMU 33) have elevated concentrations of bis(2-ethylhexyl)phthalate, pesticides, chromium, and sulfate. The shallow groundwater beneath the eastern-most and western-most portions of the Ephemeral Drainage show elevated

concentrations of ammonia, chloride, and sulfate. The deep groundwater beneath these portions of the Ephemeral Drainage show elevated concentrations of VOCs, phthalates, ammonia, chloride, and sulfate, while the area near the GW-55 well nest also show elevated concentrations of chromium and pesticides. The parameters detected at elevated concentrations in these areas are believed to be unrelated to the sediments. Shallow groundwater may contribute contaminants to the surface water.

Potential environmental concerns at the Ephemeral Drainage are surface soils, sediments, and shallow and deep groundwater.

East Ditch

The East Ditch has elevated concentrations of TMPs, NNDPA, phthalates, PAHs, and inorganics including ammonia, chloride, chromium, sodium, and sulfate. The East Ditch surface water has elevated concentrations of TMPs, chromium, and sulfate. The CSA (CRA, 1993) concluded that the presence of toluene in the surface water and trichloroethene, 1,2-dichloroethene, methylene chloride, and chlororethene in sediments in the East Ditch are not associated with the facility and are from an off-site source.

Potential environmental concerns at the East Ditch are sediments.

Subsurface Soils - East and West Pits

Subsurface soil samples immediately south of the East and West Pits, on-property, contained elevated chromium concentrations. No surface soils samples were collected due to the coverage of asphalt and buildings. The groundwater in this area is impacted mainly by inorganics; however, it is unknown whether the pits are a source of these chemicals due to the proximity to Lake Poly and Drum Area A.

Potential environmental concerns at the East and West Pits are subsurface soils and possibly groundwater.

Subsurface Soils - Acid Pits and Lined Lagoons

Elevated concentrations of TMPs, acetone, phthalates, NNDPA, ammonia, and chromium were detected in the subsurface soils in the vicinity of the Acid Pits. Specific parameters are difficult to associate with groundwater results for surrounding wells due to the proximity of other potential source areas and the presence of a groundwater divide in this area.

Soils in the area of the lined lagoons are impacted by high concentrations of calcium and sulfate. The shallow groundwater results show elevated sulfate downgradient of the Lined Lagoons. Elevated concentrations of sulfate in surface soils in the drainage areas northwest of the Central Pond may be associated with releases from the Lined Lagoons.

Potential environmental concerns at the Acid Pits and Lined Lagoons are subsurface soils, which may be a source of contaminants to shallow groundwater.

Calcium-Sulfate Landfill

Surface soils in the area have low concentrations of phthalates and inorganics, and subsurface soils/wastes have high concentrations of ammonia, calcium, and sulfate. The surface water and sediment of the nearby Ephemeral Drainage have low to moderate concentrations of sulfate and phthalates. The groundwater analytical results surrounding the area show that only inorganic compounds are consistently detected in the groundwater. Elevated concentrations of sulfate have been detected in the well (SL-03) monitoring the southern side of the landfill and elevated concentrations of ammonia, chloride, sulfate have been detected in the wells (SL-05 and SL-06) monitoring the eastern side of the landfill.

Potential environmental concerns at the Calcium-Sulfate Landfill are subsurface soils and calcium sulfate wastes. The groundwater in the immediate vicinity of the Calcium-Sulfate Landfill may also be a potential environmental concern; however, it is unclear whether the groundwater in areas downgradient are impacted by site contaminants or are impacted by other off-site sources such as the adjacent City of Woburn Landfill.

Plant D and Surrounding Area

Subsurface soils are contaminated with SVOCs in the vicinity of the former Plant D and the associated solid waste management units (SWMUs): the Plant D Drum Storage Area (SWMU 1), the Plant D Byproduct Tank of ammonia hydroxide (SWMU 4), and the low-lying black area where soils were removed near Plant D (SWMU 26). Surface soil results for these areas show elevated concentrations of phthalates, PAHs, gamma-BHC (lindane), and sulfate. The results of subsurface soil investigations detected elevated concentrations of TMPs and sulfate at SWMU 26 and elevated concentrations of ammonia at the former Plant D storage tank area. Plant D soils are impacted by low levels of chromium, moderate levels of ammonia, and high levels of VOCs and phthalates.

There appears to be no significant environmental concerns at Plant D and the Surrounding Area. Potential environmental concerns in this area may include surface and subsurface soils; however, these potential sources do not appear to be contributing appreciable concentrations of contaminants to the groundwater or other receptors.

Shallow and Deep Groundwater - GW-54 well cluster

The shallow and deep groundwater results show elevated concentrations of TMPs, NNDPA, chromium, and sulfate: being generally higher in the deep groundwater. Subsurface soil samples from BH-26 adjacent to Plant C-1 also show elevated concentrations of TMPs, NNDPA,

chromium, and sulfate. Other potential sources include Lake Poly and leaks from the associated wastewater lines, the East and West Pits, spills or other surface releases, or the nearby inactive tile drainage field.

Potential environmental concerns at the GW-54 well cluster may be an unidentified subsurface soil source and the shallow groundwater.

Deep Groundwater and Dense Layer

Elevated concentrations of ammonia, chloride, sulfate, phthalates, phenol, and NNDPA have been detected in the deep groundwater (outside of the dense layer). These parameters have been elevated near Plant B (ammonia, phthalates, phenol, and NNDPA), Plant D (sulfate and NNDPA), Calcium-Sulfate Landfill (ammonia, chloride, sulfate, and NNDPA), Lake Poly area (ammonia, chloride, sulfate, phthalates, and NNDPA), GW-50 cluster (ammonia, chloride, sulfate, and NNDPA), and the lower Western Bedrock Valley (ammonia, chloride, and sulfate). The elevated concentrations near Plant B, Plant D, Lake Poly, and the Calcium-Sulfate Landfill coincide with elevated concentrations detected in subsurface soils. The elevated concentrations at GW-50D may be related to the flow of deeper groundwater (with higher concentrations) over the bedrock saddle which separates the Western Bedrock Valley from the Eastern Bedrock Valley. The elevated parameter concentrations in the lower Western Bedrock Valley are also attributed to the flow of deeper groundwater (with higher concentrations of ammonia, chloride, and sulfate) over a bedrock saddle which separates the bedrock low at GW-83D from the lower Western Bedrock Valley.

The dense layer, an acidic liquid with high concentrations of dissolved solids, consists mainly of ammonia, chloride, chromium, sulfate, and sodium and is accompanied by low concentrations of organics. These organics were components of the liquid wastes or were entrained in the dense layer as it flowed through soils containing organics in the deep groundwater. The high specific gravity (1.03 to 1.10 g/cm^3) allowed it to sink through the overburden and groundwater until it reached a less permeable barrier, the top of the bedrock. The results of modeling and direct observation suggest that the dense layer is not moving appreciably and that there is little potential for future movement of the dense layer in the absence of significant changes to the groundwater flow system.

The contact between the dense layer and the overlying shallow groundwater is sharp, with the overlying shallow groundwater containing inorganic concentrations that are generally 1/100 to 1/1,000 of those found in the dense layer. The mass transfer process whereby contaminants are moved from the dense layer up into the overlying groundwater is a long and slow process that does not transfer significant concentrations of contaminants. Therefore, although the dense layer will continue to provide a source of contamination to the groundwater, it is unlikely that the resultant concentrations will be significantly greater than currently observed.

The potential environmental concern for the deep groundwater and the dense layer is the continued migration of contaminants in the aquifer. Although the results of modeling suggest that the continued migration of these contaminants is unlikely to greatly change the currently observed conditions, significant changes to the groundwater flow system could alter these conditions.

Miscellaneous Areas of Environmental Concern

The remaining areas of concern at the facility are the miscellaneous SWMUs identified in the CSA (CRA, 1993). These areas were impacted by low concentrations of site related chemicals and do not represent significant source areas. They are presented in this report for the purpose of completeness only. These miscellaneous areas of concern include subsurface soils at the southwest corner of the warehouses, the hydrochloric acid tank (SWMU 3), byproducts ammonium hydroxide tanks (SWMUs 4 through 7), the wastewater treatment plant (SWMUs 8 and 31), buried drums in vicinity of warehouses (SWMUs 16 and 18), the inactive septic tanks and tile fields (SWMUs 20 and 21), and the PCB capacitor (SWMU 22).

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7.4 GEOCHEMICAL CONDITIONS

General Geochemical Conditions

- Over the range of Eh and pH values observed in the site groundwater, Chromium (VI) [hexavalent chromium] is thermodynamically unstable, and is not expected to occur. Matrix interferences during the analysis for Chromium (VI) using EPA Method 6010 has likely resulted in false detections. Sampling results support the conclusion that dissolved chromium is mainly present as Chromium (III), and not Chromium (VI). In addition, the presence of ferrous iron throughout the groundwater system precludes (thermodynamically) the existence of Chromium (IV).

- The dense layer's leading edge reacts with aquifer materials and is partially neutralized, resulting in an increase in pH (increasing from 3.5 to 5) which effectively reduces the solubility and mobility of chromium. Consequently, the waters at the leading edge of the dense layer plume contain high total dissolved solids (TDS) concentrations (e.g., 35,000 mg/l) but very little chromium.

- Given that the major ions (e.g., sodium, ammonia, sulfate, chloride) are present at different proportions in the dense layer than in the intermediate-TDS groundwaters found immediately above the dense layer, the mechanism by which the dense layer constituents are transferred to the overlying groundwater is diffusion rather than advection. Once these constituents have diffused into the intermediate-TDS groundwater, advection may then be the dominant method of continued constituent migration.

Characterization of Aquifer Material

XRF and Electron Microprobe (EMPA)

- Analysis of aquifer soils by EMPA have identified the same minerals/amorphous precipitates that were predicted to be solubility controls by geochemical modeling. This finding provides strong evidence that the formation of metal hydroxides has partially attenuated transport of certain metal solutes in the groundwater. Specifically, the increases in the chromium and iron concentrations at depth observed in the aquifer solids demonstrate that geochemical precipitation or adsorption reactions have removed chromium and iron from solution and incorporated it into solid phases.

- The hydroxides that have been identified as coatings on layer silicates indicate that hydrolysis reactions have caused local neutralization of the dense layer to the extent that the precipitation or adsorption of small amounts of secondary solids, including metal hydroxides and sulfates, has occurred. The hydroxides may serve as solubility controls for the major cationic metals [i.e., chromium (III), iron (III), aluminum] present as solutes in high concentration in the dense layer. The sulfate-containing solids may result from precipitation or adsorption reactions of the dense layer with the aquifer materials.

Chromium Solubility and Partitioning

- Chromium (III) concentrations in dense layer solution undergoing neutralization reactions with aquifer solids is predicted by the solubility of chromium (III) hydroxide. This is an important consideration for modeling the fate and transport of chromium (III) in the groundwater system contacting the dense layer.

- The results of experiments on samples from the dense layer with the highest solute concentrations and aquifer material samples from near the property show a sharp increase in the removal of chromium (III) from solution with increasing solid to solution ratio, again indicating that the aquifer materials react with the dense layer, causing an increase in pH and precipitation of chromium (III) hydroxide.

- The adsorption tests using the aquifer matrix and dense layer samples from Maple Meadow Brook wetlands indicate that low levels of chromium occurring along the leading edge of the dense layer (e.g., 0.06 mg/l at GW-83D) are sparingly adsorbed by the aquifer matrix. One possible reason for this lack of adsorption is the high concentrations of other cations that occur along periphery of the dense layer (e.g., the iron concentration at GW-83D is 2,030 mg/l). The other cations may initially out-compete the chromium for adsorption sites. However, continued transport and precipitation of amorphous hydroxides would tend to create more adsorption sites and continue to scavenge chromium from the groundwater through precipitation and/or adsorption reactions.

- For the three major dense layer constituents - ammonia, sulfate, and chromium - the data indicate that adsorption is not a dominant mechanism controlling their migration rates away from the toe of the dense layer plume. In addition, while the tests indicate that the site aquifer materials do have the capacity to adsorb ammonia, it is only weakly adsorbed. Therefore, adsorption is not likely to affect the rate of ammonia migration through the aquifer.

- The adsorption tests also indicate that aluminum was adsorbed by aquifer materials as well as iron (strongly adsorbed by aquifer materials from GW-87D groundwaters but weakly adsorbed from GW-83D). Manganese was strongly adsorbed by aquifer materials from GW-83D and only weakly adsorbed from GW-87D.

Vertical Characterization of the Dense Layer

- The dramatic compositional changes that occur between the fourth port of each piezometer and the next overlying port (2.5–3.5 foot interval) indicate a relatively sharp boundary exists between the dense layer and the groundwaters that overlie it. The compositional changes that occur across this boundary indicate that dense-layer constituents are transported disproportionately across it. The disproportionate dense-layer constituent concentrations suggest that, although little interaction and movement occurs across the interface, diffusion is a slow, but important transport mechanism at the interface between the dense layer and overlying groundwater.

- Calcium sulfate sludge and other residual soil contamination found in some onsite soils has not been completely flushed from the aquifer and may continue to affect the quality of the shallowest groundwaters in the vicinity of MP-1, located near the former Lined Lagoons and Acid Pits.

- The presence of chromium throughout the groundwater column in the vicinity of MP-2 indicates that all groundwaters at this location have been affected by the wastes released at the site, either during the initial infiltration event, or as a result of constituent transport away from the dense layer at the base of the aquifer.

- The major ion composition, TDS, and specific conductance profile in the groundwater from MP-2 indicate that infiltration of surface waters from the ditch system may be enhancing the flushing of dense-layer constituents from the intermediate-depth groundwater system.

Geochemical Modeling

Geochemical modeling has been used to identify the factors controlling the fate and transport of inorganic compounds through groundwater systems including the chemical properties of the compound, the presence of other solutes that may enhance or reduce its solubility (i.e., complexing ligands), local geochemical environment (e.g., pH and oxidation-reduction potential), and the reactivity of the aquifer matrix. The main conclusions of geochemical modeling are:

- Both geochemical modeling of the groundwaters and experimental results indicate that chromium(III) concentrations will be limited by the solubility of chromium hydroxide in the aquifer system. Above a pH of about 5.8, the solubility of chromium hydroxide is below the MCL for dissolved chromium (0.1 mg/l).

- Analyses of aquifer solids by EMPA showed the presence of the metal hydroxide solids indicated to be solubility controls by the geochemical modeling results. This finding provides strong evidence that the formation of metal hydroxides such as $Cr(OH)_3$ has partially attenuated the transport of metal solutes in the groundwater.

- The results of partitioning experiments indicate that chromium(III) will be strongly attenuated by adsorption and/or precipitation to the aquifer material under conditions of undersaturation with chromium hydroxide solubility, particularly for pH > 4.4.

- Manganese, iron, and silica are present in high concentrations in the dense layer. Given that these solutes were not known to have been released in the original waste stream, they were probably derived from the dissolution of silicate and oxyhydroxide minerals present in the aquifer. Dissolution reactions generally consume acid, and hence have probably moderated the pH of the dense layer solutions to some extent, although they have also contributed mineral acidity to the dense layer.

- The slow rates at which silicate mineral dissolution reactions consume acidity and the persistence of the acidic dense layer solutions in the aquifer indicate that natural attenuation reactions are unlikely to significantly reduce the dense layer acidity.

- Dissolved chromium in the groundwater is present only as chromium(III). Previous analytical results indicating the presence of chromium(VI) were in error because of matrix interferences. This result is consistent with knowledge of the thermodynamics and kinetics of chromium(VI) and reduction by iron(II), which is present throughout the groundwater.

- The geochemical modeling indicates that there are no chromium complexes present in the groundwater that are likely to increase the solubility of chromium hydroxide.

- As with chromium(III), the geochemical modeling results indicate that aluminum concentrations are limited by the solubility of aluminum hydroxide.

- At pH <4.5, the concentrations of iron(II) and manganese(II) are also limited by the solubilities of their respective oxyhydroxide solids. At higher pH, iron(II) and manganese(II) do not appear to be solubility limited, possibly because of variabilities in thermodynamic constants or incomplete oxidation under the near anoxic conditions of the groundwater.

- At pH <4, sulfate concentrations are limited by the solubility of gypsum. However, at higher pH, the groundwaters are gypsum undersaturated, probably because of low calcium concentrations in the groundwaters mixing with the dense layer solutions. The groundwaters are oversaturated with various aluminum sulfate solids but these appear to be ineffective solubility controls for both sulfate and aluminum.

- Differences in the reactivity of different solutes with the aquifer material has resulted in a separation of the mobile anions from the less mobile cationic metals in areas downgradient from the dense layer. The concentrations of metal solutes [i.e., chromium(III), aluminum, iron(II), and manganese(II)] are limited by the pH-dependent solubilities of metal hydroxide solids, hence are largely confined to the low pH regions of the dense layer. The concentrations of the anions (i.e., chloride and sulfate) are not appreciably limited by solubility controls, hence their concentrations are affected primarily by dilution.

Surface Water and Flocculent Geochemistry

- Several indicator analytes (sulfate, chloride, ammonia and nitrate) occur at elevated concentrations in the Off-property West Ditch and South Ditch surface water before attenuation downstream. These analytes may result from localized aquifer flushing of constituents from the site to these portions of the Ditch System. In contrast to indicator analytes, consistent detections of chromium in groundwater near the Ditch System is limited and is virtually absent in Ditch surface water samples. Therefore, there is no current impact from chromium on Ditch water quality and the source of chromium to surface water may be undergoing depletion.

- Chromium solubility is restricted by the formation of amorphous solid phases of Al-, Cr-, and Fe- sulfate to a level below laboratory detection limits, effectively eliminating dissolved chromium from the South Ditch system. In the flocculent, the chromium is along with Fe- and Al- solid phases as amorphous oxyhydroxides or precipitate as a mineral. The apparent continued accumulation of flocculent in the Off-Property West Ditch streambed following flocculent removal in 1994 suggests that a source of chromium in the shallow aquifer matrix immediately adjacent to the South Ditch may still be present, possibly resulting from the slow dissolution/desorption/ion-exchange of "old" chromium from a geochemical reaction zone or residual soil contamination in source areas.

- In summary, chromium migration from the facility to the South Ditch system is being controlled by a combination of flocculation, sediment barriers (hay bales), ambient site geochemical conditions, and probable source depletion. Chromium is present solely in the trivalent form (chromium (III), as particulate or solid phases of extremely low solubility. Migration of other analytes of concern to the Ditch system, such as Na, Cl, NH_4, and SO_4, appears to be limited to areas near GW-79S and the Central Pond. The concentration of these analytes is substantially lower in surface water locations relative to corresponding groundwater locations, suggesting that natural attenuation and dilution is controlling solute concentrations discharging to the Ditch system.

7.5 FATE AND TRANSPORT

The key findings regarding the fate and transport of site contaminants are based on the results of an extensive sampling program, groundwater modeling, and geochemical testing and modeling. The main conclusions are:

- Migration of contaminants via surface water and sediment transport is limited to the boundaries of the respective water body.

- Migration of shallow groundwater away from the Olin property is limited. Much of this groundwater is discharged locally to the East Ditch, South Ditch, and Off-Property West Ditch. A small amount of groundwater from the northern portion of the property by-passes these ditches, but is likely to be captured by the Altron supply wells.

- The bedrock formation acts as a barrier to dense layer and groundwater flow and does not appear to be acting as a significant pathway for contaminant migration.

- The mass transfer process (diffusion), whereby contaminants are moved vertically from the dense layer up into the overlying groundwater, is a long, slow process that does not transfer significant concentrations of contaminants. However, this diffusion process appears to be the dominant mechanism by which contaminants move upwards from the dense layer. Therefore, although the dense layer will continue to provide a source of contamination to the groundwater, it is unlikely that the resultant concentrations will be significantly greater than currently observed.

- The geochemistry of the groundwater system aids in reducing mobility of certain metals, including chromium, aluminum, iron, and manganese. The solubility of these metals is controlled by precipitation of oxyhydroxide solids. For example, above a pH of 5.8, chromium is unlikely to be present in groundwater at concentrations above 0.1 milligrams per liter (mg/l), because precipitation of chromium hydroxide removes it from solution.

- The geochemical conditions in the Off-Property West Ditch and South Ditch tend to reduce the downstream migration of chromium. Upon discharge from the groundwater, the chromium is readily removed from the water column via precipitation to various oxyhydroxide and sulfate solids, thus creating the observed chromium-bearing flocculent. The existing controls, which include the weir and several sets of hay bales, serve to minimize off-site transport of the flocculent material.

- Groundwater flow modeling indicates that the municipal supply wells receive their water from rainfall, the wetlands, and portions of the aquifer which are not impacted by site contamination, with a smaller volume of water coming from contaminated portions of the aquifer. Between the Olin property and Main Street, much of the groundwater is captured by Altron's industrial supply wells.

- The results of the modeling suggest that there is little potential for future movement of the dense layer in the absence of significant changes to the existing groundwater flow system. The modeling results further suggest that the dense layer has not changed position appreciably since the mid-1980s.

- The results of the transport modeling indicate that transport of dissolved chromium from the dense layer to the municipal supply wells does not occur due to precipitation of chromium-iron hydroxides. Future concentrations of other inorganics (such as ammonia, chloride, sodium, and sulfate), which are know to be transported to the municipal supply wells, were estimated; the results of transport modeling yield estimates of solute concentrations at or above historical observations. However, these predications are extremely conservative. Ongoing monitoring of the surrounding sentinel wells will provide forewarning should any significant changes in groundwater quality and migration occur.

7.6 RISK ASSESSMENT

Human Health Risk

Current Land Use and Site Conditions

- Cancer and noncancer risks for all identified receptors, including site workers, on-property utility workers, off-property industrial workers and neighborhood residents are below the MCP Cumulative Receptor Cancer Risk limit and the Cumulative Receptor Noncancer Risk Limit.

Future Land Use and Conditions

- Cancer and noncancer risks for all evaluated receptors, including site maintenance workers, on-property utility workers, full-time, long-term on-property industrial workers, off-property industrial workers, and neighborhood residents are equal to or below the MCP Cumulative Receptor Cancer Risk limit and the Cumulative Receptor Noncancer Risk Limit

These conclusions are based on assumed limitations on future activities and uses of the Facility and surrounding environment. The limitations for the facility are based on an Activity and Use Limitation, limits on groundwater use at residences along Main Street are based on abandoned wells, public water supply availability, Town of Wilmington regulations, and Notices of Limitation with Respect to groundwater, the availability of public water at all downgradient industrial properties, Town of Wilmington regulations, and provisions of groundwater management plans that are associated with Downgradient Property Status.

- Concentrations of 1,2-dichloroethane, 1,2-dichloroethene, benzene, chloroform, bis(2-ethylhexyl)phthalate, methylene chloride, trichloroethene, vinyl chloride, nitrate, arsenic, chromium, and lead, in the Zone II of the Town Supply Wells exceed corresponding health-based MMCLs, which are considered applicable public health standards. Because of these exceedances, a condition of no significant risk of harm to health does not exist. Available data from Zone II wells indicate that 1,2-dichloroethane, 1,2-dichloroethene, methylene chloride, trichloroethene, vinyl chloride, arsenic, and lead in the Zone II groundwater may not be associated with the site.

Safety Risk

- Future excavation of materials in the Drum areas may pose a threat of fire or explosion due to the possible presence of Opex® and Kempore®. Therefore, a condition of no significant risk of harm to safety does not exist.

Public Welfare Risk

- The groundwater Upper concentration limits (UCLs) for chromium, nickel, and cadmium are exceeded at the dense layer hot spot in the non-Zone II area.

- The groundwater UCLs for indeno(1,2,3-cd)pyrene and bis(2-ethylhexyl)phthalate are exceeded in the Plant B Hot Spot.

- The groundwater UCLs for chromium and zinc are exceeded in the Zone II area.

- Non-aqueous phase liquid (NAPL) with thickness greater than one-half inch was detected in the Plant B area, a condition that by definition, exceeds a UCL.

- Secondary drinking water standards, which are based on aesthetic considerations, for iron, manganese, and sulfate are exceeded in the Zone II area.

- Manipulations and/or modifications of Butters Row Treatment Plant operations may be required to maintain adequate chlorine residuals to insure disinfection of treated drinking water.

- Public groundwater resources have been impacted.

Therefore, a condition of no significant risk of harm to public welfare does not exist.

Ecological Risk

The risks to ecological receptors for surface water are associated with only a few contaminants, namely aluminum, ammonia, chromium, iron, and di-n-octylphthalate. The risks to ecological receptors for sediments could not be attributed to any one contaminant. The conclusions concerning the ecological risk assessment are as follows:

- The results of a food chain model, which use site-specific biological tissue data, support a conclusion of no significant risk of harm to representative terrestrial and semi-aquatic receptors (i.e., the American woodcock, red fox, and green heron) from food chain exposures.

- The results of earthworm toxicity tests and an evaluation of surface soil data support a conclusion of no significant risk to terrestrial receptors (e.g., the woodcock) from decreased prey abundance.

- The FETAX toxicity test results combined with an amphibian population model suggest that we cannot conclude no significant risk of harm to aquatic receptors (e.g., the green frog) from exposure to sediment in the On-Property West Ditch, based on a predicted subpopulation decrease of greater than 25 percent for amphibians in that area.

- The FETAX toxicity test results and amphibian population model support a conclusion of no significant risk of harm to semi-aquatic receptors (e.g., the green heron) from decreased prey abundance.

- A comparison of surface water concentrations with published surface water toxicity data for amphibians suggests that we cannot conclude no significant risk of harm to aquatic receptors from exposure to surface water in the Off-Property West Ditch, South Ditch, and Ephemeral Drainage areas.

Response actions, if needed, should focus on potential sediment-related risks in the On-Property West Ditch.

Response actions, if needed, should address potential surface water-related risks in the Off-Property West Ditch, South Ditch, and Ephemeral Drainage areas.

7.7 AREAS OF CONCERN

With respect to the areas of potential environmental concern, the following conclusions can be made:

NAPL ≠ SOIL

Groundwater and ~~Soils~~ in the Former Plant B Production and Tank Farm areas are associated with UCL exceedances (NAPL and groundwater organics) and represent a continuing threat of release that necessitates the continued operation of the groundwater interceptor wells.

Soils and sediment in the area of former Lake Poly are associated with risks to aquatic receptors(sediment only). Subsurface soils may also be a source to groundwater.

Soils and waste in Drum Area A and Drum Area B are associated with a potential fire or explosion hazard associated with future excavation activities. Subsurface soils and wastes may also be a source to groundwater.

Groundwater, surface water and sediment near the South and West Ditches are associated with risks to aquatic receptors (sediments in particular) and the shallow groundwater represents a potential source of contamination to the environment.

Subsurface soils associated with the East and West Pits are not associated with significant risks, although subsurface soils may also be a source to the groundwater.

Subsurface soils associated with the Acid Pits and Lined Lagoons are not associated with significant risks although subsurface soils may also be a source to the groundwater.

Shallow and Deep Groundwater at GW-54 well cluster is not associated with significant risks but may be a source of contamination to the environment.

Deep Groundwater and Dense Layer are associated with chromium UCL exceedances, MMCL exceedances in the Zone II of the Town Wells, Secondary Drinking Water exceedances in the Zone II of the Town Wells and represent a continuing source of contamination to the environment.

7.8 COMPLETION STATEMENT

Based on the findings of this Supplemental Phase II Investigations and Risk Assessments, in conjunction with the findings of the CSA, additional comprehensive response actions are necessary at this site. In the opinion of the License Site Professional, all Phase II requirements have been met in accordance with all applicable provisions of orders, permits (Tier I Permit No. 83004), and approvals identified in this submittal.

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