ADDENDUM TO DRAFT REPORT

REMEDIAL INVESTIGATION

RESPONSES TO COMMENTS

Stratford Army Engine Plant
Stratford, Connecticut

Contract Number DACW41-96D-8014

Prepared for

United States Army Corps of Engineers
New England District
696 Virginia Road
Concord, MA 01742-2751

October 2001

Prepared by

URS

201 Willowbrook Boulevard
Wayne, New Jersey 07470

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RESPONSES TO COMMENTS

FROM

ENVIRONMENTAL PROTECTION AGENCY

ATTACHMENT I

GENERAL COMMENTS

1. *The volumes comprising this report contain an enormous amount of information, resulting from years of detailed work by numerous investigators at this site. URSGWC has done a good job of compiling high-quality data from their own work as well as from previous studies. The organization of this report and clarity of presentation is generally very good The discussion of the site geology, both surficial deposits and bedrock characterization, is well-done and thorough. The geologic cross-sections give a reasonable three-dimensional overview of the complex stratigraphy underlying the site.*

   In general, interpretations are consistent with the data presented herein, with some exceptions (noted in the following comments).

Response:

Duly noted.

2. *Contaminated groundwater discharging to surface water at Frash Pond could impact the pond ecosystem and may also pose potential human health risks. Existing information regarding the pond should be reviewed and evaluated in conjunction with the new hydrogeologic information to determine whether there needs to be further evaluation.*

Response:

The majority of contamination at the site is on the opposite side of a groundwater divide from Frash Pond. Flow in this contaminated area is toward the river. There is a cis-1,2-DCE plume near Frash Pond. Since the pond would dilute concentrations, which are already low, impact on the pond would be minimal. Although, further investigation may be necessary and will be dealt with in the Feasibility Study.

3. *Data in the RI suggest that there is a potential for DNAPL to collect in a depression on the bedrock surface underlying the northern end of Building B-2 This feature indicates the need for additional deep overburden and bedrock monitoring wells to verify the extent of contamination under this portion of the site. To accomplish this additional investigation, alternative drilling technologies, such as directional drilling techniques, should be explored.*

Response:

It is questionable that a bedrock aquifer investigation is warranted since there are no drinking water supply wells in the area, which would be threatened by the presence of DNAPL. The cost for this type of investigation would need to be justified based on the concerns presented to the environment and human health. Any necessary further investigation and remedial decisions will be addressed during the Feasibility Study.

4. *The arsenic fate and transport scenario is incomplete. The two possible sources for arsenic exceedances should be clearly identified, and the subsurface conditions that favor arsenic mobilization should be defined. All available information regarding historical arsenic use, quantities and disposal practices at the site should be discussed. It is noted that a background data set for surface water and surface sediments exists against which samples from the river and tidal flats are compared. There is no attempt to establish site-wide background for soils or groundwater. Possible links between a bedrock source for arsenic, arsenic in overburden soil and arsenic found in overburden groundwater should be developed, including a review of redox*

conditions in the overburden and associations with other dissolved metals, such as iron and manganese.

Response:

Two possible sources for arsenic exceedances are (1) natural geologic sources including naturally occurring arsenic in the sediment and/or arsenic-containing minerals present in rocks, and (2) agricultural and industrial activities involving arsenic use on the site. No data were available about the background soil concentration on the site or in the State of Connecticut, but according to the Technical and Administrative Guidance Memorandum #4046 (New York State Department of Environmental Conservation, 1994), the background soil arsenic concentration in the Eastern U.S. is about 3 –12 ppm. Arsenic was detected at concentrations exceeding 10 ppm across the main portion of the SAEP facility. Thus, naturally occurring arsenic may account for at least some of the arsenic found. Although the bedrock beneath SAEP has been identified as black schist with greenstone, few whole-rock geochemical data exist for the rock types in the lithogeochemical groups where elevated arsenic concentrations were present in soil and groundwater. Without such data, the effect of the specific bedrock types on arsenic concentrations in soil and groundwater can not be determined. In addition, the unconsolidated sediments, which were deposited under glacial conditions, are not derived from the underlying bedrock. Based on the geologic formation of the site, it is unlikely that the bedrock would be a significant contributor to the detection of arsenic.

As stated in Section 2.1, land in the SAEP vicinity was historically used for agricultural and residential purposes prior to 1929 when it was developed for manufacture and assembly of aircraft or engines. However, no data were available regarding the potential release of arsenic from the previous land use. Additionally, investigation of the redox status of groundwater reveals that most of the arsenic exceedances occurred when groundwater was in reductive condition, indicating the reductive condition of ground water favors the dissolution of arsenic bearing minerals and/or soils.

Overall, the widespread of arsenic in soil indicates naturally occurring arsenic could be a source for at least some of the arsenic found on the site; previous agricultural and industrial activities are also possible sources of arsenic in some instances. The relative importance and the interrelation of the two sources are not clear. More data are needed before a complete fate and transport of arsenic can be established.

5. *Intertidal mudflats sediment sampling and outfall drainage sampling results raise the question of whether or not the tidal-flat PCBs can be attributed to site-related sources. The types and concentrations of PCBs to be expected along the shore of the estuary away from the SAEP site should be reviewed in order to assess the potential impacts from the site itself.*

Response:

The final scope and approach of the ecological risk assessment for the SAEP site was developed after numerous meetings with, and comments from, Region I EPA and CDEP from December 1996 through October 1999. While complete summary and characterization of sediment PCB occurrences upstream of the site might provide a better understanding of the site itself, it was beyond the scope of the assessment. Reference stations were chosen in conjunction with EPA and CDEP. EPA also requested that data collected at the Raymark site be used as reference for soil and sediment metal concentrations. The scope did include assessment of constituent concentrations in environmental media, results of toxicity tests and tissue analyses and benthic community data to characterize potential risk from constituents present at the site areas. Based on these data sets it was concluded that

- There is a threat to benthic communities and individual receptors feeding on benthos in certain localized areas of the intertidal mudflats, due to the presence of several patches of elevated sediment PCBs.
- Soil invertebrates (already limited by poor physical habitat conditions) on the Causeway may also be threatened by soil PCB concentrations.

- There is also a potential threat to individual terrestrial receptors in the Marine Basin area and the Causeway due to PCBs in prey items.

6. *The RI report fails to compile and make full use of data bearing on the reduction-oxidation (redox) state of the site groundwater. Field data for oxidation-reduction potential (ORP) and dissolved oxygen (DO) collected during the low-flow purging and sampling of drivepoints, piezometers, and monitoring wells should be assembled Maps and cross-sections should be compiled to display the distribution of redox conditions in relation to high concentrations of VOCs and inorganics. This will allow an assessment of the dominant processes affecting the contaminants of concern. There is emphasis in the RI on the formation of 1,1-dichloroethene (1,1-DCE) by microbial degradation of trichloroethane (TCA), noting that this process occurs under methanogenic conditions. In areas of the site where both of these compounds are present, are measurements of oxidation-reduction potential consistent with methanogenic conditions?*

Response:

Oxidation-reduction potential (ORP) and dissolved oxygen (DO) data have been compiled (Table 1). Following guidelines presented in the U.S. EPA <u>Technical Protocol for Evaluating Natural Attenuation of Chlorinated Solvents in Ground Water</u>, 29 of the 60 wells sampled exhibit conditions in which reductive pathways are likely, 13 of the 60 wells exhibit conditions in which reductive pathways are possible. Figure 1 shows areas of reductive and oxidative conditions in combination with VOC contamination. Reductive conditions exist in many of the locations where VOC contamination is present. In addition, the available methane concentration data (Table 1) shows sufficient amounts of methane present to indicate that methanogenic conditions are likely to exist at the site.

The distribution of reductive conditions and chlorinated solvent contamination demonstrates two large areas where methanogenic pathways are likely to exist. One area covers almost the entire length of the site along the Housatonic River. The area of reductive conditions nearly encompasses all of the chlorinated solvent plumes. The cis-1,2-DCE plume contour is consistent with the expected location in relation to the PCE and TCE plumes, providing further evidence that chlorinated solvents are degrading through methanogenic pathways in the area along the river. The other area of reductive conditions includes the wells around the west parking lot. Although cis-1,2-DCE is not as prevalent, it is possible that the degradation of PCE through methanogenic pathways can occur in the area of the west parking lot.

Eighteen of the 60 wells exhibit oxidative conditions. Oxidative conditions do exist at the site, in small pockets. These conditions exist mostly where there is little contamination from chlorinated solvents, such as the locations around wells WC5-1D and PZ-17D. Only three of those 18 wells had TCA and 1,1-DCE at levels above 5 ppb. Only one well, WC-12S, showed high levels of both TCA and 1,1-DCE. TCA will readily degrade to 1,1-DCE in an oxidative environment through chemical processes. Evidently, chemical degradation of TCA is occurring in the area surrounding WC-12S. The largest pocket of oxidative conditions lies east of Frash Pond. Although, TCE (below SWPC) and cis-1,2-DCE (between 68 and 970 ppb) are both present in wells PZ-17D, PZ-9D and WC-20D1. Based on this data alone, a conclusion can not be drawn concerning the degradation pathways occurring in the area east of Frash Pond.

7. *There are many groundwater samples where cis-1,2-DCE is present in high concentrations. The distribution of cis-1,2-DCE is critical to understanding the predominant transport mechanisms for chlorinated VOCs at the site. The relationships between PCE, TCE, and cis-1,2-DCE should be explored and incorporated into a revised conceptual model for chlorinated VOCs at the site.*

Response:

Table 1 presents a summary of VOC groundwater data and reduction-oxidation data. Figure 1 presents contours for PCE, TCE, 1,1-DCE, *cis*-1,2-DCE and reduction-oxidation conditions. Where TCE and *cis*-1,2-DCE concentrations are high, reductive conditions also exist. It is likely that TCE source material is being reduced through methanogenic pathways to its most common daughter product *cis*-1,2-DCE in these areas.

8. *In many areas of the site, the presence of 1,1-DCE is not necessarily diagnostic of degradation of 1,1,1-TCA. It should be noted that TCE is also known to produce at least some 1,1-DCE as a decay product. It is also possible to form 1,1-DCE by abiotic degradation of 1,1,1-TCA The possible derivation of 1, 1-DCE by abiotic oxidation of TCA should be addressed and incorporated into the revised conceptual model for the chlorinated compounds. All VOCs should be shown, in order to present a complete picture of the interrelationships between compounds detected, even if some do not exceed the regulatory criteria.*

Response:

In the area surrounding monitoring well WC-12S, oxidative conditions exist. It is likely that in this area, the abiotic degradation of 1,1,1-TCA forming 1,1-DCE is occurring (see response to General Comment number 6.) TCA can be chemically transformed to 1,1-DCE under almost any conditions, therefore, it is likely that abiotic degradation of 1,1,1-TCA to 1,1-DCE is occurring throughout the site. Where reductive conditions exist, some 1,1-DCE can be produced as a daughter product of TCE through methanogenic pathways. But this pathway would be responsible for only a small portion of the total 1,1-DCE present. The amount of 1,1-DCE formed from TCE may be small enough to be considered negligible in the Conceptual Model. See Table 1 and Figure 1 for a summary of VOC groundwater data and reduction-oxidation data.

9. *The Human Health Risk Assessment evaluates the risk to construction workers in the hydraulic/plating area. The possibility of exposure to concrete and alteration products of concrete is apparently not considered. Cores taken through the concrete floor beneath the plating area in Building 2 showed bright yellow staining. It is possible that this represents reactions between chromium plating solutions and the concrete, resulting in new Cr(VI)-bearing phases. Thus, a construction worker who disturbs concrete in the floor may be exposed to dust containing high concentrations of Cr(VI). The human health risk assessment seems to have overlooked this scenario by focusing only on the natural media. Consider whether this potential exposure scenario is/could be significant.*

Response:

Since current and future use in this area is industrial, this pathway is not addressed as a public health issue or release to the environment as part of the risk assessment. OSHA regulations would cover exposure to current and future site-specific construction worker activities such as the use of personal protective equipment and/or engineering controls to minimize exposure.

10. *The discussion of chromium fate and transport is somewhat incomplete – the role of solid phases containing hexavalent chromium, formed as a result of plating solution releases is not addressed, yet such phases may be critical to assessing chromium mobility and the persistence (or attenuation) of the hexavalent species. The wells with the highest groundwater hexavalent chromium exceedances were not sampled for VOCs, but it is inferred that these wells are within the contours delineating exceedances of TCE and 1,1-DCE. Since the occurrence of 1,1-DCE is attributed to the degradation of TCA, reportedly under methanogenic conditions, how is the presence of a strong oxidant (CrO_4^{2-}) reconciled with the strongly reducing environment required for methanogenesis? Aside from its apparent coexistence with the chlorinated compounds, how does CrO_4^{2-} remain mobile (i.e., unreduced) at near-neutral pH conditions in soils with an organic fraction of 0.005? Is it possible to establish that the Cr(VI) "plume" is at steady-state, i.e. Cr(VI) may be emanating from a source mass in the soil beneath a portion of Building B-2 and spreading due to lateral dispersion and advection? The extent of the "plume" may become limited by* *reduction of Cr(VI) and precipitation as Cr(III)-containing solid phases. Site-specific information, including pH, ORP and associations with other dissolved metals, should be incorporated into the discussion of chromium transformation.*

Response:

This portion of the project was conducted and the chromium fate and transport was developed by Harding Lawson and Associates (HLA). The chromium fate and transport will be further detailed in the Feasibility Study.

11. *It should be noted that in areas closer to the shore, within the zone of influence of tidal fluctuations, dispersion can be driven by these periodic components to the groundwater velocity. Given the very low mean groundwater velocities estimated for portions of the site, this tidally driven dispersion may predominate over the dispersion associated with the mean flow, as represented in the conventional model discussed. The conceptual model for contaminant transport at the site should acknowledge this possibility. The information from the tidal study should be summarized and presented graphically in more detail. The amplitude of the fluctuations at each well monitored in the tidal study should be extracted from the data, and plotted against distance to the river. This will provide a quick, visual, qualitative tool for evaluating the importance of water-level fluctuations at any point at the site. In addition, the variation of amplitude with distance can be exploited to provide an independent estimate of key hydraulic properties. A fit to these data can yield an estimate of the hydraulic diffusivity, or, with an independent estimate of conductivity, can give an estimate of the storativity.*

Response:

The amplitude fluctuations at each well was plotted against the distance to the river (Figure 2). The area significantly effected by tide fluctuations is limited to a narrow stripe along the river, approximately 100 feet wide. The data did not fit the exponential curve sufficiently to extract a reasonable estimate of the hydraulic diffusivity. An attempt was made to remove anomalous data points in order to achieve a better fit. An acceptable fit was not achieved until only six data points remained (Figure 3). This is a poor statistical sample that may not be representative of conditions at the site. Any hydraulic calculations based on this limited sample would not be representative of actual conditions. See Table 2 for a summary of tide amplitudes and distances.

12. *The RI does not outline the principles and methods that guided the location of the soil samples. A brief discussion should be included. Did the investigation include a thorough search of as-built drawings for engineered, subsurface structures such as floor drains, waste pipes, sumps, dry wells, oil-water separators, underground storage tanks, etc.? The RI should document the efforts made to sample soils in locations most likely to have accumulated contaminants.*

Response:

Sample locations were biased towards areas most likely to have accumulated contaminants. These areas were identified through a thorough search of as-built drawings for engineered, subsurface structures such as floor drains, waste pipes, sumps, dry wells, oil-water separators, underground storage tanks, etc.

SPECIFIC COMMENTS

1. *__Executive Summary, Hydraulic/Plating Area, Page ES-9.__ The Executive Summary notes an important conclusion from the human-health risk assessment to the effect that there are no unacceptable health effects for construction workers engaged in activities in the Hydraulic/Plating Area. However, it is noted that the risk assessment did not address exposure to concrete from the floor, which has been found to be stained a bright yellow, apparently by reaction with chromium plating solutions. The potential risks due to exposure to dust, should construction penetrate or break up the floor, should be considered Please see related to General Comment*

Response:

Since current and future use in this area is industrial, this pathway is not addressed as a public health issue or release to the environment as part of the risk assessment. OSHA regulations would cover exposure to current and future site-specific construction worker activities such as the use of personal protective equipment and/or engineering controls to minimize exposure.

2. *__Section 1.2.5, Former Lagoons RCRA Investigation, Page 1-6, Bullet 3.__ The text indicates that PCE and TCE in groundwater from LW-12 are decreasing gradually and at a steady rate. However, it is stated elsewhere (same page, second bullet discussing the March 200 sampling trends) that groundwater from LW-12 contains PCE and TCE at levels that are "slightly higher than 1998 concentrations (4.9 ppb vs. 3.8 ppb)." It is unclear whether this difference is significant. This apparent contradiction should be resolved.*

Response:

The intention was to express that the trend in groundwater from LW-12 showed declining concentrations of PCE and TCE. The 1.1 ppb increase from the 1998 data is not considered significant and does not negate the general trend of decreasing concentrations.

3 *__Section 1.2.5. , Preliminary Assessment Screening, Page 1-8, Paragraph 1.__ The drainage ditch that originates at SAEP and flows into the Marine Basin carries storm water runoff as well as the discharge from the Chemical Waste Treatment Plant (CWTP) The text indicates that the CWTP effluent is currently permitted by NPDES, but that the facility has been discharging to this drainage ditch since 1958. Is the last sentence of this paragraph meant to imply that VOCs; SVOCs. PCBs, metals, and cyanide were included in the CWTP waste stream and discharged to the drainage ditch prior to NPDES permitting? The nature if this discharge and consequences for deposition of contaminants in this area should be clarified*

Response:

SAEP maintains two NPDES permits for the Chemical Waste Treatment Plant (CWTP) and Oil Abatement Separator (OATP) until final disposal of the facility. The nature of the discharge from CWTP is not clear at this stage and further information may be obtained from the Army personnel.

4 *__Section 1.2.5 Phase II Investigation. Page 1-10 , Paragraph 9__ (see also related General Comment above) Conclusion from the Phase II report are stated in a manner that reaffirms the likelihood of off-site sources, particularly for chlorinated compounds. As discussed elsewhere in additional comments in this review, the distribution of monitoring wells near Frash Pond in sparse, and groundwater flow direction in this area is poorly-constrained, especially in the deep aquifer. The degradation pathways called upon to explain the distribution of daughter products focus primarily on the formation of 1,1-DCE from TCA. Neglecting the more common route by which PCE and TCE degrade to cis-1,2-DCE, In summary, any conclusions from the Phase II report should be evaluated substantially, and support for these conditions should be explicitly provided before inclusion in the RI*

Response:

Based on compiled reduction-oxidation and VOC groundwater data (Table 1, Figure 1), most wells near Frash Pond, specifically those in the area of the west parking lot, that have detected levels of VOCs exhibit reductive conditions. It is likely that TCE and PCE are degrading through methanogenic pathways to cis-1,2-DCE. In addition, TCA will degrade to 1,1-DCE under almost all conditions. It could be expected that both processes are occurring in this area. (See response to General Comments 6, 7 and 8.)

Wells to the East and Northeast exhibit conflicting evidence for degradation pathways. For example, Wells PZ-9D and WC-20D1 are with an area of oxidative conditions and yet they have high concentrations of cis-1,2-DCE, 970 ppb and 780 ppb respectively. Any conclusions supporting methanogenic or chemical pathways can not be drawn from these data. It is agreed that further investigation may be necessary. (See response to General Comment 2.)

5 ***Section 2.4.2 Bedrock Coring Page 2-10***. *Five-foot bedrock cores were collected from three monitoring wells, which provide confirmation that refusals were indeed bedrock encounters, confirmation of bedrock lithology, and limited look at the nature of the bedrock (e.g. weathering of the top of bedrock, presence of fractures and evidence of fluids in bedrock fractures, etc,) It is noted that, if a decision is made to pursue open questions regarding potential DNAPL, this bedrock characterization may be necessary to characterize fractures (e.g. fracture orientation and frequency orientation and frequency from borehole imaging) and fracture flow of groundwater (e.g. borehole geophysical logs heat-pulse flow metering, etc.)*

Response:

The U.S. Army believes that a bedrock DNAPL investigation is not warranted because the aquifer does not serve as a potable water source and there are no potential human health risks associated with groundwater contamination at the depth. Any necessary further investigation will be addressed during the Feasibility Study. (See response to General Comment number 3.)

6 ***Section 2.4.3.2 Operable Unit 2 NCRA Investigation, Soil, Vapor Survey, Page 2-11***. *The soil vapor survey provides a direct measure of impacts of VOCs residuals in the vadose zone or dissolved in underlying groundwater on the soil gas and overlying indoor air. The text states that the soil vapor survey results generally correlated well with the VOC concentrations in nearby groundwater, but notes on exception in the southeastern end of B-2, where TCE was found in soil vapor at a concentrations in excess of the CTTDEP ICVC. The statement that this location in "an exception" to the correlation with groundwater contamination is not well supported, which raises a concern with respect to groundwater characterization in this area. From Figure 2-1, it appears that there is good coverage of the southeastern end of B-2 with respect to soil borings, and that these indeed revealed no significant VOCs in soils. However, there appears to be no coverage at all of groundwater in this area: no direct-push sampling was done, nor were monitoring wells installed. With no data on groundwater in this area, on what basis is the statement made that the soil gas results are uncorrelated? Given the success elsewhere in matching soil vapor results with underlying groundwater concentrations for chlorinated VOCs, this raises the possibility of uncharacterized VOC contamination in groundwater beneath this end of the building. This potential data gap takes on additional significance when viewed in the larger context of the conceptual model advanced in the RI for chlorinated VOCs in groundwater site-wide. In particular, Figure 3-1 shows two NE-SW trending plumes defined by PCE exceedances, one corresponding roughly to the west parking lot, and one offset to the southeast and trending northeastward through building B-3 and B-4. The area under the southeastern portion of B-2 is critical to delineating these PCE plumes as separate bodies, with implied separate sources.*

Response:

The reason for insufficient groundwater sampling points in this area was that it is occupied primarily by the large building B-2 and groundwater monitoring wells may not be conveniently and economically established on the site. In Figure 3-1, the PCE exceedance plume trending northeastward from eastern of B-2 through buildings B-3 and B-4 shows an significant increase of PCE concentration, i.e., the PCE concentration was the lowest in the eastern end of B-2, indicating a possibility of uncharacterized VOC contamination source in the eastern side of B-2 is low. However, no information on groundwater flow or quantity in this area is available. More information may be obtained from HLA who conducted the investigation.

7 <u>*Section 3.3.2, Monitoring Well Installation, and Section 3.3.3, Monitoring Well Sampling, Page 3-5*</u>*. The test details the stabilization criteria used for both well development and for low-flow sampling. The development procedure allows for termination of pumping based on volume rather than stabilization, and the low-flow sampling protocol similarly allows for sampling based on volume or time of purging rather than stabilization in the event that stability of one or more parameters is difficult to achieve. A few words in these sections regarding the experience in the field with stabilization would improve the presentation. With what frequency was stability not achieved, and the fall-back on purge volume invoked? A brief summary would help the reader to gauge the potential impact of the drilling or direct-push disturbance on the groundwater samples.*

Response:

See revised pages for Section 3.3.2, Monitoring Well Installation, and Section 3.3.3, Monitoring Well Sampling, Page 3-5. Also see Well Development Logs and Groundwater Purge & Sample Data Sheets included in the RI.

8 <u>*Section 3.3.8.1, Chromium-Focused Investigations, Page 3-10, Paragraph 3*</u>*. The text refers to "[t]he following Hach test kits...," but the list appears to have been dropped. This should be checked and completed.*

Response:

The appropriate changes have been made, see revised page.

9 <u>*Section 3.4.3, Tidal Study, Page 3-25.*</u> *The tidal study provides important data to characterize the influence of tidal fluctuations on groundwater levels across the site. The filtering scheme seems to yield meaningful mean water levels, and has helped constrain the mean groundwater potential field It is suggested that the statistics on the fluctuations themselves (e.g., mean amplitude of the tidal signal at each monitoring location) should be exploited, too, in order to extract additional information This includes the amplitude of the tidal signal as a function of distance from the river (providing an opportunity to estimate the hydraulic diffusivity of the shallow overburden), as well as the possible influence of the tidal fluctuations on dispersion. See related General Comment.*

Response:

See response to General Comment number 11.

10 <u>*Section 4.4, Results, VOCs Page 4-2.*</u> *The reader is referred to Figure 4-1 for the location of surface water sampling, while it appears that the intended reference is to Figure 4-2.*

Response:

The appropriate change in text has been made, see revised page.

11 ***Section 5.4.1.1, Intertidal Mudflats, PCBs, Page 5-3.*** *The sediment sampling from the intertidal mudflats revealed detections of three Aroclors, 1248, 1254 and 1260. It is interesting to note that sampling in the Outfall 008 Drainage (p. 5-5, sec. 5.4.1.3) found only Aroclor 1260. This raises the question of whether or not the tidal-flat PCBs can be attributed to site-related sources Were any "reference" or "background" sediment samples taken from, say, Nell's Island (as is the case for surface water, p. 4-1, sec. 4.3.1)? It may be desirable to determine and/or report on the types and concentrations of PCBs to be expected along the shore of the estuary away from the SAEP site in order to assess the potential impacts from the site itself.*

Response:

Background PCB sediment concentrations were analyzed from samples collected from three locations and are provided in Table 6 of Appendix 9A. Three stations US-001, US-002 and US-003 were located upstream of the jetty in an area considered to be outside of the immediate influence of the site. All sediment PCB concentrations were below detection at detection limits ranging from 11 to 14 µg/kg.

In addition, a control location was also selected in the Marine Basin portion of the site. Similar sediment PCB concentrations were measured at this station. All sediment PCB concentrations were below detection at detection limits ranging from 11 to 14 µg/kg.

Please see response to General Comment 5.

12 ***Section 7.4.3, Hydrodynamics, Horizontal Groundwater Gradients, Page 7-11.*** *The RI acknowledges that there is a broad area characterized by flat horizontal gradients from Building 2 toward the west parking lot. The shallow groundwater potential maps (Figures 3-7 and 3-8) show a "saddle" in the area of the northwest end of Building 2, with flow converging from the southeast and northwest, and diverging toward the river (NE) and Frash Pond (SW). This is apparently a complex area, and resolving the potential surface is difficult with the control available. However, it is a critical area, as it bears strongly on the transport of contaminants from and to areas of known concentrations. The picture that emerges from the water levels may be consistent with the observed distribution of contaminants originating in the area of the plating shop, which appear not to have moved far from their release point. However, the shallow groundwater picture is difficult to reconcile with the conceptual model of an off-site source for the chlorinated solvents detected in wells in the west parking lot The highest detection of chlorinated VOCs in the west parking lot is at WC-19S; the November sampling round found PCE at 740 µg/L Figure 3-5 shows a "finger" of PCE exceedances coming from the south to encompass WC-19S and the other wells in the parking lot The implied transport direction is perpendicular to the flow direction implied by the shallow potential map (Figures 3-7 and 3-8). This is a weakness of the conceptual model that has apparently guided the mapping of this plume This apparent contradiction should be addressed. The deeper groundwater (Figures 3-9 and 3-10), in contrast, is mapped with a gradient from southwest to northeast. The deeper screen at WC-19D1 showed PCE at 480 µg/L, and is again apparently ascribed to the "finger" of contamination originating to the south. This is again difficult to reconcile with the mapped deep hydraulic gradient. In general, more effort is needed to delineate the chlorinated solvent plume that crosses the west parking lot area, to establish continuity with its source area, and to reconcile the contaminant distribution with what is known about the predominant transport processes.*

Response:

Groundwater contours and the PCE exceedances "finger" in the area of the west parking lot were based on limited data. The source of PCE is not known. There is evidence for an off site source of chlorinated solvents to the south and west of the west parking lot. The mapping of groundwater contours and the chlorinated solvent plume are the best representation of the available data. It is possible that the apparent contradiction could be attributed to the tendency of DNAPL to follow preferential pathways rather than the general groundwater flow direction. Further data points are needed in order to reconcile the contradiction in flow.

13. *__Section 7.4.3, Hydrodynamics, Horizontal Groundwater Gradients, Page 7-11, Paragraph 2.__* *The text notes that water levels in deep wells are generally lower than levels in shallow wells in the central portion of the site. This is consistent with the equipotentials and flow directions sketched in Figure 7-6, which shows downward flow at intermediate depths. Overlaid on Figure 8-13, which shows the VOC exceedances along the same section, this implies downward, advective transport of the VOCs in the vicinity of the plating shop. This adds to a general concern for downward transport of the chlorinated solvents and possible bedrock involvement. Downward advective transport of dissolved VOCs could potentially compound the downward transport of free-phase DNAPL, and bring more mass to depth, possibly entering fractured bedrock. The ultimate fate of the chlorinated solvents is not well established, and needs further work.*

Response:

Any further investigation will be dealt with during the Feasibility Study.

14. *__Section 7.4.3, Hydrodynamics, Storativity and Specific Yield, Page 7-14.__* *The estimate of specific yield based on the rate of water level decline during a period of no recharge is a worthwhile calculation, and makes good use of the site-wide water-level data to infer aquifer properties. While the result of the calculation is a reasonable value for specific yield, one qualification might be noted in the development. The approach as described makes the implicit assumption that all groundwater discharging from the site to the river on the north is derived from the site itself. That is, the calculation is essentially a mass balance calculation that states that the net of outflow rates minus inflow rates is equal to the rate of change of stored water. The model posed, however, does not allow for any groundwater inflow from off-site, say, from the area regionally upgradient to the southwest. Thus, it requires that all of the discharge to the river is derived from storage within the site aquifer volume in a period of no recharge. In other words, the quantity Q used in the calculation is the total discharge from the river side of the volume, rather than the net of the outflow to the river minus the inflow on the upgradient side. The result may be a significant overestimate of Q, and, therefore, an overestimate of specific yield.*

Response:

The calculated specific yield fell within the expected range of values of .01 to .30. This range is typical for unconfined aquifers, according to Driscoll. It should be noted that the text indicates that the calculation of the specific yield was considered an estimate. Since the calculated value was within the range of specific yields for this type of aquifer, URS felt it would be useful to present. However, URS never presented it as an absolute value since we considered that we would not be able to accurately compensate for groundwater flow into the site.

15. *__Section 7.4.4, Hydrogeologic Conceptual Model, Page 7-17.__* *The RI states, "Frash Pond appears to be located downgradient of the northwest portion of the SAEP facility," and this is certainly borne out by the map of the shallow groundwater potential. This raises a concern for groundwater discharging to the pond, potentially carrying contaminants from the site upgradient. The RI program did not include sampling of Frash Pond surface water or sediment, in part because there was existing information that had been collected for the Raymark project. Based on information presented in the RI, a discussion should be included regarding the potential of the site to impact Frash Pond.*

Response:

Impact on Frash Pond from the site is expected to be limited since the majority of contamination lies on the other side of a groundwater divide. (See response to General Comment number 2.) Further investigation may be necessary and will be addressed during the Feasibility Study.

16. ***<u>Section 7.4.4, Hydrogeologic Conceptual Model, Page 7-18, Paragraph 2</u>.*** *The text should be corrected: "Groundwater fluctuations of one to two feet were recorded... "*

Response:

The appropriate change has been made. See revised page.

17. ***<u>Section 8.2.2, Inorganic Characterization, Arsenic, Page 8-2</u>.*** *Figure 8-8, referred to in this section, gives concentration ranges in the map key in units of ug/L. However, these are soil analyses, presumably reported in mg/kg. Clarify.*

Response:

The appropriate correction has been made to reflect the correct units, mg/kg. See revised figure.

18. ***<u>Section 8.3.1, Organic Characterization, 1,1-DCE, Page 8-3</u>.*** *The text suggests that the widespread 1,1-DCE may be present due to degradation of 1,1,1-TCA. While this transformation is recognized, and may indeed be the source of some or even all of the 1,1-DCE detected, the support for this conceptual model is not well developed. In particular, what are the spatial correlations of 1,1,1-TCA and its degradation products, as well as PCE, TCE, and their degradation products? The latter are important in understanding the former, as 1,1-DCE is also a known degradation product of TCE, albeit not the predominant one. If these transformations are redox sensitive, what are the correlations with redox conditions at this site? Please see related General Comment.*

Response:

Table 1 and Figure 1 summarize reduction-oxidation conditions in correlation with areas of contamination. TCA will readily degrade to 1,1-DCE under almost all conditions through chemical processes. See responses to General Comments number 6, 7, 8 and 11.

19. ***<u>Section 8.3.1, Organic Characterization, 1,1-DCE, Page 8-3</u>.*** *The text states, "An additional off-site source of 1,1-DCE extending from the southwest of the West Parking Lot in a northeastern direction across Main Street towards Building B-2... " Please revise this sentence for clarity. On Figure 3-5, the 1,1-DCE "plume" is drawn on the basis of analyses from two monitoring wells, PZ-11D (where 1,1-DCE was found at 340 ug/L in November 1999) and WC3-11 (2900 ug/L). These are insufficient data to define a "plume" or even the direction toward a "source." It is noted that the hydrogeologic section D-D' (Fig. 7-6) shows groundwater flow at the depths of these screens from PZ-11D to WC3-11. Thus, the proposed plume represents transport upgradient, against the groundwater flow as mapped. This inconsistency should be addressed. In addition, the apparent gradient in 1,1-DCE concentration from WC3-11 to PZ-11D is highly uncertain. There is no guarantee that the screen placements at these locations happen to coincide with the maximum 1,1-TCE concentrations. The direct-push sampling in this area (Figure 3-1) offered considerably denser coverage, but still fails to delineate the plume. The highest hit of 1,1-TCE found was at DP3-2 at 7800 ug/L. This concentration was found at the deepest sampling interval (35 ft), so the vertical distribution is unbounded at depth. Concentration are lower both to the north and to the southwest. Therefore there is no indication that the 1,1-TCE has been traced "upgradient" toward its source. Further work is needed to establish the extent of chlorinated VOC contamination in the area of the west parking lot, as well as its source and the transport pathways. The purported off-site source to the southwest cannot be established until the plume is defined both on and off the SAEP site.*

Response:

It is agreed that there is insufficient data to accurately determine groundwater quality and flow in the west parking lot area. The apparent upgradient transport of 1,1-DCE will need to be addressed by further investigation. Plume delineation and additional groundwater elevation data in the area of the west parking lot may need to be included as part of the Feasibility Study. Also, there is evidence of off-site sources to the south and west of the west parking lot. Off-site sample results indicate higher concentrations of 1,1-DCE in this area. Documentation of this data can be provided by the Army.

20. *__Section 8.3.1, Organic Characterization, PCE, Page 8-4__. The text notes that the highest PCE detection in groundwater was 4300 µg/L at DP3-2 on the southeast side of the West Parking Lot. It should be noted that this PCE detection was at only 8 ft below ground surface, and that this area has been mapped as being just off the "shoulder" of a divide in the shallow groundwater. One might expect a component of downward flow in this area, as reflected in the hydrogeologic cross-sections. This seems to indicate that the PCE release may have been quite local to the detections at DP3-2 or immediately to the east. The potential contours shown on Figures 3-7 and 3-8 suggest flow to the west in this area. Please reconcile the flow direction with the inferred plume "...extending from southwest of the West Parking Lot in a northeastern direction...": Again, more evaluation is needed to delineate the VOC distribution in this area and to develop a conceptual model consistent with available knowledge of site (and off-site) use, known or suspected disposal practices, contamination distribution, and hydrology.*

Response:

Further investigation in the area of the west parking lot may be necessary and will be addressed in the Feasibility Study. See response to Specific Comment number 19.

21. *__Section 8.3.1. Organic Characterization, TCE, Page 8-4__. The text states, "A second TCE plume originating from an adjacent off-site property has migrated to monitoring wells (MW-191 and MW-19S) located in the west parking lot The off-site source was confirmed with the installation of direct push borings (DP3-7 through DP3-13)..." It is agreed that the source appears to be just outside the parking lot perimeter. It might be noted in the text that the monitoring wells cited are immediately upgradient of the maximum TCE hit of 16,000 µg/L in DP3-8 at 27-30 ft bgs. WC-19D1 is screened at a similar depth only a short distance upgradient, but showed only 36 µg/L TCE. Also, decreases in concentration to three downgradient direct-push holes (DPE System-310 at 2500 µg/L, DP3-11 at 5300 µg/L and DP3-9 at 7000 µg/L) seem to suggest that the release must be very close to DP3-8 Is this consistent with any "working model" for the nature of historic disposal practices from SAEP or the adjacent site?*

Response:

The Army will provide additional information regarding the disposal practices in this area.

22. *__Section 8.3.1, Organic Characterization, TCE Hotspots, Page 8-6, Paragraph 4__. The RI recognizes the strong indication of TCE DNAPL as suggested by the very high concentrations of dissolved TCE persisting to bedrock (e.g. at WC2-3D, 3100 µg/L). This issue remains open A thorough investigation targeted at potential DNAPL is called for in order to delineate the extent of TCE contamination at depth, particularly in view of the bedrock through identified with the seismic work, extending to the northwest from the area of highest TCE concentrations associated with the plating shop area The lack of deep exploration in the area of the bedrock trough represents a data gap*

Response:

Any necessary further investigation in this area will be dealt with in the Feasibility Study. See response to Specific Comment number 5.

23. ***Section 8.3.1, Organic Characterization, 1.1.1-TCA Hotspots, Page 8-7, Paragraph 1.*** *The RI again acknowledges the potential for 1,1,1-TCA DNAPL originating from the area of high dissolved concentrations in the central portion of Building B-2. This again highlights the lack of data to characterize deep overburden and bedrock in the vicinity of the mapped bedrock trough extending northwest from the area of deep, high-concentration chlorinated VOC detections.*

Response:

Any necessary further investigation in this area will be dealt with in the Feasibility Study. See Specific Comment number 5.

24. ***Section 8.4, Contaminant Transport, Hydrodynamic Dispersion, Page 8-10.*** *The units on the dispersions noted for TCE and PCE should be in* ft^2*/day.*

Response:

The appropriate corrections have been made. See revised page.

25 ***Section 8.4, Contaminant Transport, Contaminant Advection Calculations, Page 8-10.*** *The citation of Ogata (1970) is not included in Section 12, References.*

Response:

The appropriate correction has been made. See revised page.

26 ***Section 8.4, Contaminant Transport, Page 8-11, Paragraph 1*** *The reader is referred to Table 8-1, which summarizes results of the calculations using the Ogata solution. The table includes columns labeled* C_{log} *and* C_{obs} *without explanation. Are these concentrations calculated using velocities computed from the logarithmic and arithmetic mean conductivities cited previously? Please clarify.*

Response:

C_{log} used the logarithmic mean hydraulic conductivity, C_{obs} use the arithmetic mean conductivity as cited in Section 7.4.3.

27. ***Section 8.4, Contaminant Transport, Contaminant Retardation Calculations, Page 8-11.*** *The text refers to the application of the Ogata solution as a "diffusion" calculation. It would be more precise to call this an "advection/dispersion calculation." Diffusion, while included in a full evaluation of the dispersion coefficient, is typically negligible. In addition, the solution accounts not only for dispersion, but also for advection. The term "diffusion" is repeated several times in succeeding pages (see, e.g., p. 8-13. Para. 5) in reference to the Ogata solution. Please replace with "advection/dispersion."*

Response:

The appropriate changes have been made. See revised page.

28 ***Section 8.4, Contaminant Transport, Contaminant Retardation Calculations, Page 8-11.*** *The retardation estimates invoke an organic carbon fraction of 0.005. Is this estimate supported by site-specific data? If so, please refer here to the appropriate section of the report. It is also noted that the highest concentrations of VOCs are found in the alluvial sands It might be expected that the alluvial and outwash sands are lower in organic carbon than the silts. The RI states in sec. 2.3.1 (p. 2-3) that soil samples were analyzed for TOC. Are direct measurements of organic carbon available for each of the hydrostratigraphic units, and are the appropriate values used here to model retardation of the VOCs?*

Response:

The soil samples analyzed for TOC were from the unsaturated zone only. This data would not be appropriate for use in an organic carbon fraction value that would be used to represent the saturated zone. The organic carbon fraction of 0.005 represents the upper range presented for fine to medium sand deposited in a glaciofluvial or glaciolacustrine environment in EPA document Technical Protocol for Evaluating Natural Attenuation of Chlorinated Solvents in Ground Water, page C3-27.

29. ***Section 8.4, Contaminant Transport, Contaminant Retardation Calculations, Page 8-11.*** *The calculations summarized in this section are for advective travel times to the river, allowing for retardation. This is a useful reference calculation. However, it might be noted that it is not "conservative" as an estimate of travel time due to neglect of dispersion. Dispersion will cause contaminants to arrive ahead of the advective front, albeit at lower concentration. This may prove to be particularly significant given the relatively large dispersion coefficients estimated for the RI. It is noted that it is not necessary to separate the advection/diffusion calculation embodied in the Ogata solution from the assessment of the effects of retardation, as is done here. The Ogata solution still holds for a system with retardation, one only need replace v_s with v_s/R and D_L/R (retardation affects both the advection and the dispersion). Setting $R = 1$ will recover the calculations presented in the section "Contaminant Advection Calculations," and allowing $R > 1$ will provide an evaluation of the influence of retardation on "breakthrough" at the river, accounting for dispersion, as well.*

Response:

Contaminant advection calculations were performed using the retardation factor as described in Specific Comment number 29. The results of these calculations are summarized in Table 3. The resulting concentration in the Ogata solution was set to equal the Surface Water Protection Criteria (SWPC). Using the "Goal Seek" feature in Microsoft Excel, the Ogata solution was solved to find the time at which the SWPC concentration would reach the river. Using the arithmetic mean hydraulic conductivity yielded the most conservative estimates, having SWPC concentrations of 1,1-DCE, TCE and PCE reach the river in 21, 32 and 66 years, respectively.

30. ***Section 11.1.2, Summary of Chemical Characterization, Inorganics, Page 11-7.*** *The summary remarks concerning chromium in groundwater state, "The extent of the horizontal distribution is limited to the area under and surrounding the Chromium Plating Facility due to the relatively flat horizontal gradient in this area " While the slow groundwater flow velocity in this area would certainly result in slow advective transport, the chemistry of the plating solutions, the concrete floor slab, the groundwater, and the solid matrix of the aquifer may also play important roles in limiting the horizontal spread of hexavalent chromium The extremely acidic and oxidizing plating solutions (assumed a composition containing Cr_2O_3 and sulfuric acid) almost certainly*

would have reacted instantly with any materials contacted upon release (e.g., the concrete slab, mineral grains comprising the underlying soil, organic carbon in the aquifer matrix, etc.) Such reactions would have consumed protons and may also have reduced Cr(VI) to Cr(III), leading to a rise in pH and precipitation of chromium-bearing solid phases. However, depending on the kinetics of the reactions involved, some of these phases – in particular, those resulting from the reaction between plating solutions and the concrete slab – may still contain chromium as Cr(VI) phases are relatively water-soluble, these reaction products would continue to serve as a potential source of Cr(VI) as long as the solid mass persists. The possibility that this scenario prevails should be explored further, perhaps with additional sampling of subsurface materials beneath Building B-2.

Response:

The fate and transport of the chromium will be addressed in the Feasibility Study.

31. ***Section 11.1.2, Summary of Chemical Characterization, Inorganics, Page 11-7.*** *The summary remarks concerning arsenic in groundwater note that exceedances of the SWPC were found along the berm area, as well as other locations, and that "...no direct source can be attributed." However, all available data from the RI, as well as previous investigations, should be marshaled to provide an assessment of the roles of anthropogenic sources and indirect, anthropogenic impacts on natural sources. In particular, the presence of elevated arsenic in groundwater should be weighed against the discovery of arsenic in soil at a maximum concentration of 3550 mg/kg at 5-7 ft bgs in the north end of Building B-10 (soil borings SB13G1-1C). This strongly suggests that arsenic was used in some form in some process in the facility (bronzing? as a hardener? as a metal adhesive?). Possible site uses should be researched and discussed where appropriate. In addition, it is noted elsewhere in the RI that there is widespread evidence of degradation of chlorinated VOCs, particularly in areas peripheral to the main plume bodies. The reducing environment typically associated with degradation of chlorinated VOCs is also likely to reduce natural iron oxyhydroxides on the aquifer solids and release sorbed arsenic that was present under natural conditions. In such a scenario, both arsenic and iron are naturally-occurring and are probably derived from minerals such as pyrite or other sulfides that are commonly found in litholgies such as the schists, phyllites, and gneisses that comprise the underlying bedrock. The frequent reference to 'micaceous sand' in many of the boring logs (Appendix B) suggests that the glacial overburden is indeed locally-derived from the adjacent bedrock However, the elevated concentrations of dissolved arsenic in groundwater may be a consequence of site activities, such as the release of hydrocarbons and/or solvents The subsequent microbial degradation of these compounds results in reducing conditions that then mobilize arsenic otherwise bound in solid phases*

Response:

This comment was addressed under General Comment number 4.

32 ***Appendices H and I, Monitoring Well Purge Data Sheets.*** *It is noted that groundwater in monitoring well WC-12S is apparently oxidizing during the 7/8/99 sampling round (ORP is reported at ~ 220 mV), and reducing (of the order of –147 mV on 11/17/99). This difference should be explained*

Response:

The difference in ORP at monitoring well WC-12S from July to November can likely be attributed to equipment malfunctions.

BASELINE RISK ASSESSMENT
(VOLUME 4 OF 9, PARTS 1 AND 2)

GENERAL COMMENTS

1. *One major concern upon review of the human health risk assessment portions of the RI involves the inconsistencies in selection of screening values for the areas of concern. The screening values listed in Tables 10.2-17 through 10.2-42 are not consistent or correct for many of the chemicals. In general, the human health screening values listed in Tables for the Main Site are correct. In most cases, the corrected screening value does not affect the outcome of the COPC selection. However, in some instances, the correct screening value adds chemicals to the list of COPCs for the area. Where this is the case, the chemicals and area are specified in a specific comment below. It is recommended that a thorough review of the screening values be conducted.*

Response:

There were some errors and inconsistencies in the COPC selection tables. Where this is the case, the corrected screening values did not change the COPCs for the area except for chloroethane in surface soil at the Causeway/Dike. The addition of chloroethane as a COPC will not significantly affect the risk results given the low frequency of detection and slight exceedance of the screening value.

2. *A QC of the risk assessment tables in Part 2 should be performed because it appears that there may be some linkage problems with the spreadsheets. Errors with Exposure Point Concentrations have been pointed out in the specific comments below*

Response:

A linkage error was found in Table 10.11-6 where the oral toxicity values were incorporated rather than the inhalation toxicity values. The corrected slope factors and reference doses will result in a lower than reported risk and Hazard Index.

3. *Evaluation of the noncarcinogenic effects of exposure to all PCBs has not been performed in this risk assessment EPA Region 1 accepts the use of a surrogate reference dose for evaluation of these noncarcinogenic effects. The noncarcinogenic effects of exposure to PCBs should be evaluated using surrogate information.*

Response:

Only those PCBs with verified toxicity values were evaluated in the risks assessment.

4. *Tables 10.2-18 through 10.2-25 summarize data for chemicals that were eliminated from selection as COPCs based upon frequency of detection. Review of these tables indicates that almost all of these chemicals could have been eliminated from consideration as COPCs based upon comparison of their highest detections with the human health screening criteria, which is a more supportable rationale for elimination from further consideration The few chemicals that exceeded screening levels and were infrequently detected may represent "hot spots" of contamination. These potential hot spots should be evaluated spatially and compared to the known site history*

before being eliminated as COPCs. Additional discussion should be presented in the text regarding the possibility of hot spots of contamination.

Response:

As stated in Section 10.2.3.3, chemicals detected at low frequency were also compared to health-based screening values before being eliminated from the risk assessment. This was done to assure that no potential risk driver was eliminated and to minimize the potential that a hot spot would be missed. A chemical was retained for further evaluation if it exceeded the health screening value. The uncertainty related to elimination of chemicals based upon frequency of detection will be discussed in future reports.

5. *There is inconsistency in the methods employed for eliminating chemicals as COPCs based on low frequency in Tables 10.2-18 through 10.2-25. For example, Table 10.2-21 is titled COPCs Eliminated Based on Low Frequency/Causeway/Dike. This table lists vinyl chloride as a chemical of concern that was eliminated for evaluation of risk associated with exposure to surface soil. This elimination is reportedly based on the low frequency of detection. Table 10.2-37 lists concentrations of Chemicals of Potential Concern in Surface Soil for the Causeway/Dike. Vinyl chloride is included on this list even though it was listed as an "eliminated" chemical of potential concern in the previous table. Perform a thorough review of the frequency of detection tables to ensure that chemicals with a maximum detected concentration that exceeded the screening concentration were not eliminated from consideration in the risk assessment. In addition, provide an explanation for the differences in "Frequency of Detection" between Tables 10.2-17 through 10.2-25 and Tables 10.2-26 through 10.2-42. The frequency of detection listed for vinyl chloride in surface soil in the Causeway/Dike is not consistent among the tables.*

Response:

Vinyl chloride was the only chemical which exceeded the screening value from Table 10.2-17 through Table 10.2-25 and was included in the risk assessment. The correct frequency of detection for vinyl chloride in surface soil at the Causeway/Dike is listed in Table 10.2-37. See response to Comment #7.

6. *The tables presenting the Risk Assessment calculations (Tables 7 and 8) should not present an Intake Factor in the column representing the chemical's Intake. USEPA RAGS Part D tables specify (chemical concentration specific) "Intake" values be listed in the risk assessment tables.*

Response:

The current tables present a soil concentration and an Intake Factor which are multiplied to obtain the chemical-specific Intake used to assess risk. Intakes listed in the USEPA RAGS D Tables should be multiplied by their respective chemical concentration to meet the definition of Intake for Tables 7 and 8.

7. *Throughout the risk assessment, screening values employed for chromium concentration comparison have been for the less stringent trivalent chromium. The screening value for hexavalent chromium is employed for the selection of chemicals of potential concern in finfish at the Marine Basin (Table 10.2-32). EPA guidance requires that unless there is speciated data, the more conservative hexavalent chromium screening values and toxicity data must be used. The inconsistencies in the selection of screening values should be addressed.*

Response:

The use of the hexavalent chromium toxicity factor in finfish was an error. The trivalent factor was the correct toxicity factor. Hexavalent chromium is not as environmentally stable as trivalent chromium. Given that the site is located next to an area that is subject to tidal influences and brackish water, it is unlikely the hexavalent would remain for extended periods. Additionally, some environmental media (sediments, surface water) were analyzed for hexavalent chromium and all samples were non-detect. Therefore, the use of trivalent chromium was considered appropriate.

8. *There were no toxicological profiles submitted with Baseline Risk Assessment Toxicological profiles for the contaminants of concern should be provided in either Part 1 or Part 2 of the Baseline Risk Assessment.*

Response:

Toxicological profiles will be provided in future documents.

9. *The intake parameters and the results of the IEUBK and ALEM lead models should be provided in tabular format with appropriate area of concern in both Parts of the risk assessment.*

Response:

The intake parameters for the ALEM lead model is provided in Table 4-66. The results of the IEUBK and ALEM models are provided in Section 10.4.4.2 in tabular format. The results of the predicted blood lead levels for relevant receptors and media were also calculated and presented in Section 10.4.4.2 since the methodology differs from typical risk characterization.

SPECIFIC COMMENTS-PART 1 of 2

1 ***<u>Section 10.3.2.2, Identification of Exposure Points and Exposure Routes, Page 10-16.</u>*** *This section provides an identification of exposure points and exposure routes evaluated in this HHRA. There is a conflict with one of the receptors and pathways as required by the Work Plan. The construction worker's exposure to surface water and sediment in the Intertidal Flats was not evaluated. A sentence is provided to correlate the exposure to the recreational adult and the short-term construction worker's exposure in the Intertidal Flats area. Provide a description of the similarities and/or differences between the exposure parameters for the construction worker and recreational adult to support this statement.*

Response:

The construction worker is only considered to be exposed to surface water/sediment 20-40 days/year for a duration of 1 year, while the recreator is assumed to be present at the site 100-150 days/year for 9-30 years. These assumptions result in significantly less exposure to the construction worker over a lifetime.

2 ***<u>Section 10.4.1.6, Soil/sediment Ingestion Rate, Page 10-21</u>*** *The ingestion rate selected for soil for the construction worker is 100 mg/day This value is below the default exposure parameter for high end exposure of 480 mg/day published in Risk Update #2, August 1994 EPA Region 1. The appropriate ingestion rate for a short-term construction worker is not 100 mg/day Use of the 480 mg/day ingestion rate will increase the intake factor and affect the resulting calculated risks. The*

construction worker scenario should be re-evaluated using the appropriate ingestion rate as accepted by EPA Region 1.

Response:

At the present time there are no reliable studies to accurately estimate soil ingestion rates among adult construction worker populations. For the construction worker scenario, it is assumed that work performed at the site will be more "contact intensive" with soil/sediment than for the site worker. Hawley (1985) proposed a high-end soil ingestion rate of 480 mg/day for contact intensive activities such as yard work, assuming dermal adherence of soil to hands of 3.5 mg/cm^2 and ingestion of all soil adhered to a certain portion of the hand. New data from USEPA-sponsored studies reported by Kissel et al. (1996) show that even for contact intensive activities, adherence of soil to hands is substantially less than 1 mg/cm^2. USEPA's Exposure Factors Handbook (USEPA, 1997) recommends using the Kissel data to estimate soil adherence to skin. Following this recommendation, a soil ingestion value of 60 mg/day can be estimated by adjusting the Hawley data using the experimentally determined dermal adherence factor for a farmer's hands of 0.44 mg/cm^2 (480 mg/day x 0.44/3.5 = 60 mg/day). This value is lower than, but of similar magnitude to the 100 mg/day suggested by USEPA for high-end adult residential 24 hour exposure (USEPA, 1991b). Therefore, a high-end soil ingestion rate of 100 mg/day (RME) is used to estimate RME ingestion for the construction worker with high contact with soil or sediment.

3. ***<u>Section 10.7.2.1, Commercial Fisherman Page 10-46.</u>*** *A reference or supporting documentation should be provided for the statement that "any shellfish harvested by commercial fishermen within the Intertidal Flats area would be transported out into Long Island Sound, where they would spend the next two years 'cleaning out'."*

Response:

The statement regarding shellfish harvesting is referenced on page 10-7. This reference is included in the reference list. However, we do concur that the reference should have been cited on page 10-46, as well.

4 ***<u>Section 10.7.2.1, Commercial Fisherman Page 10-47.</u>*** *This section presents results of exposure assessment for the commercial fisherman's receptor in the Intertidal Flats. The final paragraph in this section summarizes the RME cancer risk and indicates that the RME cancer risk exceeds the USEPA target risk. This statement is not correct. The RME cancer risk for the commercial fisherman is $8 X 10^{-5}$, which is within the range of $1 X 10^{-4}$ to $1 X 10^{-6}$. This statement should be corrected*

Response:

Concur.

5 ***<u>Section 10.9.2.1, Construction Worker, Page 10-53.</u>*** *This section presents results of the exposure assessment for the Site Perimeter-Construction Worker. The first sentence of this section incorrectly states that the construction worker was assumed to be exposed to contaminated sediments and surface water. For the Site Perimeter, the construction worker was evaluated for exposure to COPCs in total soil and groundwater, not sediments and surface water. This statement should be corrected.*

Response:

Concur.

6. ***Section 10.9.2.1, Construction Worker, Page 10-54.*** *The target risk range shown in the last paragraph of the section (describing the Construction Worker's risk assessment results) is incomplete. The error should be corrected.*

Response:

Concur. The target risk range should read 10^{-4} to 10^{-6}.

7. ***Section 10.11.2.1, Construction Worker, Page 10-58.*** *This section presents the results of the exposure assessment for the Main Site-Construction Worker The first sentence of this section indicates that the construction worker was assumed to be exposed via ingestion and dermal contact. Inhalation should be included as pathway for evaluation for the construction worker. The construction worker was also evaluated for adverse health effects associated with inhalation of contaminants in indoor air. This information should be provided in the introductory paragraph of this section. An explanation should be provided for separately evaluating the risks to the construction worker for exposure to contaminants in indoor air.*

Response:

Concur.

8. ***Section 10.13, Uncertainties and Limitations, Page 10-61.*** *A description of the uncertainties associated with the combination of data sets (described in Section 10.2) should included in this section.*

Response:

Concur.

9. ***Table 10.2-19, Chemicals Eliminated Based on Low Frequency of Detection in Surface Soil at the Site Perimeter.*** *The units specified in the table column heading for the Screening Criteria (ug/kg) are incorrect and should be corrected to mg/kg Also, the footnote pertaining to di-n-octyl phthalate should be corrected to diethyl phthalate*

Response:

Concur. The units should be mg/kg in Table 10.2-19 and the footnote should refer to diethyl phthalate.

10 ***Table 10.2-21, Chemicals Eliminated Based on Low Frequency of Detection in Surface Soil at the Causeway/Dike.*** *The correct PRG chloroethane in surface soil is 3.0 mg/kg. The PRG listed in Table 10.2-21 is 1600 mg/kg. Based on this correction, chloroethane should be added to the list of COPCs for surface soil at the Causeway/Dike.*

Response:

The correct PRG is 3 mg/kg which results in chloroethane being retained as a chemical of potential concern. The addition of chloroethane is unlikely to significantly affect the calculated risk given the low frequency of detection and low concentration.

11 ***Table 10.2-25, Chemicals Eliminated Based on Low Frequency of Detection in Soil at the Hydraulic/Plating Area.*** *The PRG listed for 2,4-dinitrobenzene could not be verified. This table should be reviewed to ensure that the appropriate chemical and screening value are listed*

Response:

2,4-Dinitrobenzene does not have an RSR or PRG. 1,3-Dinitrobenzene should have been used as a surrogate. The number shown likely did not get changed from the previous compound. However, the number shown is more conservative than either the 1,3-dinitrobenzene RSR or PRG. Therefore, the erroneous screening value did not impact the conclusions of the risk.

12. ***Table 10.2-26, Selection of Potential Chemicals of Concern in Sediment at the Intertidal Flats.*** *This table summarizes the selection of COPCs in sediment at the Intertidal Flats and indicates that 2-hexanone has been selected as a COPC, but it does not indicate the rationale. The rationale for selecting this contaminant as a COPC should be presented in this table for consistency with other COPCs in this table and for consistency with other COPC selection tables.*

Response:

2-Hexanone was retained as a COPC because no screening value was available. Footnote "D" should have been listed as the reason for keeping 2-hexanone. 2-Hexanone was evaluated in the uncertainties section. No impact to the conclusions.

13. ***Table 10.2-27, Selection of Potential Chemicals of Concern in Surface Water at the Intertidal Flats.*** *The human health screening value listed for bromodichloromethane is not the lowest screening value applicable to this chemical. The lowest screening value is the PRG which is 0.18 ug/L. In addition, the entire risk assessment should be reviewed in terms of the appropriate screening value for this chemical. In some instances, the screening value listed does not apply. The screening value listed does not apply. The screening value listed for antimony in surface water should be verified.*

Response:

The Water Quality Criteria (WQC) should preferentially be used for screening surface water. The screening value for antimony was reported in units of μg/L rather than mg/L therefore antimony should be included as a COPC. The addition of antimony to the risk evaluation does not change the conclusions since the calculated Hazard Quotient is still significantly less than one.

14 ***Table 10.2-36, Selection of Potential Chemicals of Concern in Groundwater at the Site Perimeter.*** *The reference for the screening value for bromodichloromethane should be corrected. The screening value listed is correct, but the reference should be PRG, not RSR*

Response:

Concur.

15 ***Table 10.2-38, Selection of Potential Chemicals of Concern in Total Soil at the Causeway/Dike.*** *The screening value for 1,2,3-trimethylbenzene should be verified. Also, correct the human health screening value references for carbazole and chrysene The reference should be PRG.*

Response:

1,2,3-Trimethylbenzene should be 1,3,5-trimethylbenzene. The screening value for 1,2,4-trimethylbenzene should have been used as a surrogate for 1,3,5-trimethylbenzene. However, the erroneous screening value caused no impact on the conclusions of the risk assessment. We concur that the screening value references for carbazole and chrysene are incorrect, but there is no impact to the conclusions of the risk assessment.

16. ***<u>Table 10.2-39, Selection of Potential Chemicals of Concern in Groundwater at the Causeway/Dike.</u>*** *Verify that the appropriate screening value for 1,1, 1-trichloroethane in groundwater is listed. The current RSR for 1,1, 1-trichloroethane in groundwater is 200 ug/L, not 1,000 ug/L. If the screening value used was incorrect, revise all groundwater screening tables in the risk assessment. This correction may add 1,1, 1-trichloroethane as a COPC.*

Response:

An incorrect RSR was used for 1,1,1-Trichloroethane in groundwater at the Causeway/Dike and Site Perimeter which resulted in 1,1,1-Trichloroethane being selected as a COPC at the Site perimeter. Chemicals present at similar concentrations and with similar toxicity factors did not result in a significant risk at this site; therefore, the addition of 1,1,1-TCA would not significantly impact the conclusions of the risk assessment.

17. ***<u>Table 10.4-2, Intake Assumptions for Incidental Ingestion of Soil (Construction Worker).</u>*** *This table indicates that the reasonable maximum ingestion rate of soil for the construction worker is 100 mg/day, and the central tendency ingestion rate is 50 mg/day. These rates are based upon a study by Calabrese (1997) involving only six adults who were not engaged in earth moving activities. Because the construction workers are assumed to be engaged in direct and intense contact with the soil, a RME value of 480 mg/day is recommended (based upon Hawley 1985 as cited in EPA 1997). A central tendency value of 100 mg/day is recommended. Also refer to Specific Comment #2 listed above. The necessary corrections to the risk calculations should be made.*

Response:

At the present time there are no reliable studies to accurately estimate soil ingestion rates among adult construction worker populations. For the construction worker scenario, it is assumed that work performed at the site will be more "contact intensive" with soil/sediment than for the site worker. Hawley (1985) proposed a high-end soil ingestion rate of 480 mg/day for contact intensive activities such as yard work, assuming dermal adherence of soil to hands of 3.5 mg/cm^2 and ingestion of all soil adhered to a certain portion of the hand. New data from USEPA-sponsored studies reported by Kissel et al. (1996) show that even for contact intensive activities, adherence of soil to hands is substantially less than 1 mg/ cm^2. USEPA's Exposure Factors Handbook (USEPA, 1997) recommends using the Kissel data to estimate soil adherence to skin. Following this recommendation, a soil ingestion value of 60 mg/day can be estimated by adjusting the Hawley data using the experimentally determined dermal adherence factor for a farmer's hands of 0.44 mg/cm^2 (480 mg/day x 0.44/3.5 = 60 mg/day). This value is lower than, but of similar magnitude to the 100 mg/day suggested by USEPA for high-end adult residential 24 hour exposure (USEPA, 1991b). Therefore, a high-end soil ingestion rate of 100 mg/day (RME) is used to estimate RME ingestion for the construction worker with high contact with soil or sediment.

18. ***<u>Table 10.4-40, Concentrations of Potential Chemicals of Concern in Groundwater Adjusted for Dermal Absorption Site Perimeter.</u>*** *Clarify the USEPA 1996 reference found in the footnote of* *this table There is no USEPA 1996 reference in the list provided with the text In addition, the permeability coefficient listed for chromium assumes that the species present is trivalent chromium USEPA RAGS Supplemental Guidance for Dermal Risk Assessment lists the permeability coefficient for hexavalent chromium as 2.0E-03. Confirm that the appropriate permeability coefficient was selected according to the known species of chromium.*

Response:

The USEPA, 1996 reference should be USEPA, 1996c. Trivalent chromium was considered the appropriate valence for this site. See response to comment #7.

19. ***Table 10.5-2, Reference Doses for Noncarcinogenic Chemicals of Potential Concern.*** *This table presents the Reference Doses for noncarcinogenic chemicals of potential concern for the SAEP. The Inhalation Reference Dose for 1,1, 1-trichloroethane could not be verified. Verify and provide supporting information for the chronic reference dose listed for inhalation of 1,1, 1-trichloroethane. In addition, there is an asterisk footnote on this table referencing the slope factor table. The purpose in referencing the cancer slope factor table is unclear and in some instances, unnecessary. For example, thallium is footnoted and there is no listing for thallium on the cancer slope factor table.*

Response:

The inhalation reference dose for 1,1,1-TCA should be 0.29 mg/kg-day. The RfD is an EPA-NCEA provisional value. The RfD is currently 0.63, so there may have been some confusion between the old and the new RfD, depending on when the change occurred. Inhalation is generally not a significant contributor to risk; therefore, this change would not change the conclusions of the risk assessment. The footnote to the slope factor table is erroneously referenced for some chemicals.

20 ***Table 10.5-3, Slope Factors for Carcinogenic Chemicals of Potential Concern.*** *This table presents the cancer slope factors for carcinogenic chemicals of potential concern for the SAEP. Verify and provide supporting information for the oral cancer slope factor listed for benzene. Verify and provide supporting information for the inhalation slope factor and oral slope factor for vinyl chloride. Also, provide a reference for the inhalation slope factor for dibenz(a,h)anthracene.*

Response:

The inhalation and slope factors and oral slope factors for benzene, vinyl chloride, and dibenz(a,h)anthracene was verified by IRIS, HEAST 1997, and NCEA, respectively. For dibenz(a,h)anthracene, the source should be referenced as "3", based on benzo(a)pyrene. We feel that the use of provisional values by Region IX is adequate background for their use in this risk assessment. All the provisional values can be found in the Region IX PRG document provided in the Technical Memorandum (URSG-WC, November 1999).

SPECIFIC COMMENTS-PART 2 of 2

1. ***Table 5.1, Non-Cancer Toxicity Data–Oral/Dermal.*** *Table 5.1 lists oral reference doses for chromium III and 1,1,2-trichloroethane which do not correspond to the oral reference doses listed in Table 10.5-2 in Part 1. Verify that the appropriate reference doses for subchronic and chronic exposure assessment were utilized.*

Response:

Table 10.11-6 was linked to the subchronic oral reference dose rather than the inhalation reference dose. Editing the table results in a lower calculated risk. The correct toxicity values for chromium III (1.0, subchronic and 1.5, chronic) were used in the risk calculations but were listed incorrectly in Table 10.5-2 and Table 5.1.

2. ***Table 5.2, Non-Cancer Toxicity Date–Inhalation.*** *Table 5.2 lists an inhalation reference dose for mercury. The mercury inhalation reference dose is not provided in Table 10.5-2. Please verify that the correct inhalation reference dose was utilized for mercury*

Response:

The inhalation reference dose for mercury was not included on Table 10.5-2 since the inhalation pathway did not apply to this chemical. The correct inhalation reference dose is 8.6 E^{-5} for elemental mercury.

3. ***Tables 9.*.* and 10.*.*.*** *In accordance with USEPA RAGS guidance, these tables serve to provide a summary of receptor risks and hazards for COPCs as well as a summary of the risk assessment. The tables in this Part are confusing because some are blank and the purpose of the table is unclear. For example, Table 10.4 CT for the Marine Basin, Outfall 8 does not have any numbers for shellfish cancer risk or hazard quotient. Perform a thorough review of these tables and add necessary information to improve the clarity of these tables.*

Response:

The risk drivers were presented in the risk assessment summary tables. If the total risk or Hazard Quotients did not approach the target range, only the total risk or Hazard Quotient across all media and all exposure routes was shown.

4. ***Table 4.1, Intertidal Flats, Values Used for Daily Intake for Adult Fisherman.*** *The skin surface areas presented in this section do not agree with the surface areas presented in Table 10.4-17 of Part 1 Confirm the correct surface areas were utilized in the calculation of risk This surface area intake parameter is listed for other areas of the site in which the Adult Fisherman's risk is evaluated*

Response:

Table 10.4-7 is correct and these values were used in the risk calculations.

5. ***Table 4.3, Intertidal Flats, Values Used for Daily Intake for Child Recreator.*** *The surface area of skin exposed for dermal contact for the child recreator is equal to that of the adult The uncertainty associated with estimating the child's available surface area is explained in the footnote to this table and that of Table 10.4-17. This uncertainty should be discussed in Part 1, Section 10.13, Uncertainties and Limitations.*

Response:

Concur.

6 ***Table 4.4, Intertidal Flats, Values Used for the Adult Fisherman.*** *Footnote (1) addresses accidental sediment ingestion. This should be corrected to "accidental surface water ingestion."*

Response:

Concur.

7. ***Table 7.3 RME, Intertidal Flats, Calculation of Non-Cancer Hazards.*** *Verify that the total non-cancer hazard index is correct.*

Response:

The total non-cancer hazard index in Table 7.3 RME should be 2.0.

8. ***Table 9.6 CT, Intertidal Flats, Summary of Receptor Risks and Hazards for COPCs.*** *The Total [NOEL] H] listed in this table should be corrected to Total [liver] H].*

Response:

Concur.

9. ***Marine Base/Outfall 8 Risk Tables.*** *For consistency, provide the Values Used for Daily Intake Calculation for the Construction Worker with this Section of risk calculation tables.*

Response:

Concur.

10 ***Tables 7.2 CT and 7.2 RME, Marine Basin/Outfall 8, Calculation of Non-cancer Hazards for the Adult Fisherman.*** *For the dermal pathway evaluation, incorrect exposure point concentrations are provided for bromodichloromethane, chloroform, tetrachloroethane and trichloroethene. Verify that the EPC's are consistent with those listed in Table 10.8-4*

Response:

The exposure point concentrations for the dermal pathway are incorrect in several tables due to transposing errors; however, the risk calculations are the same as Part I.

11 ***Tables 7.7 CT and 7.7 RME, Marine Basin/Outfall 8, Calculation of Non-cancer Hazards for the Adult Recreator.*** *For the dermal pathway evaluation, incorrect exposure point concentrations are provided for chloroform, tetrachloroethane and trichloroethene. Verify that the EPC's are consistent with those listed in Table 10.8-17.*

Response:

The exposure point concentrations for the dermal pathway are incorrect in several tables due to transposing errors; however, the risk calculations are the same as Part I.

12. ***Table 7.9 CT and 7.9 RME, Marine Basin/Outfall 8, Calculation of Non-cancer Hazards for the Adult Recreator.*** *For the dermal pathway evaluation, incorrect exposure point concentrations are provided for bromodichloromethane, chloroform, tetrachloroethene and trichloroethene. Verify that the EPC's are consistent with those listed in Table 10.8-17.*

Response:

The exposure point concentrations for the dermal pathway are incorrect in several tables due to transposing errors; however, the risk calculations are the same as Part I.

13. ***Table 7.13 CT and 7.13 RME, Marine Basin/Outfall 8, Calculation of Non-cancer Hazards for the Recreational Child.*** *For the dermal pathway evaluation, incorrect exposure point concentrations are provided for chloroform, tetrachloroethene and trichloroethene. Verify that the EPC's are consistent with those listed in Table 10.8-24.*

Response:

The exposure point concentrations for the dermal pathway are incorrect in several tables due to transposing errors; however, the risk calculations are the same as Part I.

14. ***Table 8.2 CT and 8.2 RME, Marine Basin/Outfall 8, Calculation of Cancer Risks for the Commercial Fisherman.*** *For the dermal pathway evaluations, incorrect exposure point concentrations are provided for bromodichloromethane, chloroform, tetrachloroethene, trichloroethene and vinyl chloride. Verify that the EPC's are consistent with those listed in Table 10.8-4.*

Response:

The exposure point concentrations for the dermal pathway are incorrect in several tables due to transposing errors; however, the risk calculations are the same as Part I.

15. ***Table 8.6 CT, Marine Basin/Outfall 8, Calculation of Cancer Risks for the Construction Worker.*** *In accordance with EPA guidance, EPA recommends the use of the converted concentrations for the PAH's to assess inhalation exposure. The EPC's listed in this and other inhalation exposure tables are not the TEF equivalent concentrations.*

Response:

The EPC's listed in the inhalation exposure tables were not adjusted by the TEF equivalent, only soil concentrations were adjusted by this factor for PAHs. Although the EPCs used in the inhalation exposure tables were not adjusted by TEFs, the concentrations used are more conservative than if they had been adjusted. Since inhalation was not a major contributor to the risk, there is no significant impact on the conclusions of the risk assessment.

16 ***Tables 8.9 CT, Marine Basin/Outfall 8, Calculation of Cancer Risks for the Adult Recreator.*** *For the dermal pathway, incorrect exposure point concentrations are provided for bromodichloromethane, chloroform, tetrachloroethene, trichloroethene and vinyl chloride. Verify that the EPC's are consistent with those listed in Table 10.8-17.*

Response:

The exposure point concentrations for the dermal pathway are incorrect in several tables due to transposing errors; however, the risk calculations are the same as Part I.

17 ***Table 8.3 CT, Main Site, Calculation of Cancer Risks for the Construction Worker.*** *The intake factors for the dermal and ingestion pathways are reversed on this table. Verify that the appropriate exposure parameters were utilized in calculating the cancer risk.*

Response:

The intake factors for the dermal and ingestion pathway are reversed in Table 8.3 CT due to transposing errors; however, the risk calculations are the same as Part I.

ATTACHMENT II

The following are EPA's comments on the draft Ecological Risk Assessment (ERA) portion of "Remedial Investigation, Stratford Army Engine Plant, Stratford, Connecticut". The document is dated September, 2000. The document was received by EPA on October 25,2000.

EPA's review of ERA issues focused on volumes 1 and 3 of the RI, including identification of chemicals of concern (COCs) through Tier I and II screening, the ecological conceptual site model, selection of assessment endpoints and receptor species, exposure and toxicity assessments, risk characterization, weight-of-evaluation, and conclusions.

GENERAL COMMENTS

Several issues were identified in the approach used to select COCs. Of particular concern was:

- *The selection of COCs does not comply with EPA's guidance (USEPA, 1997) because background and blank samples data were used in the process of screening for COCs. EPA has previously commented on the proposed approach to selecting COCs (USEPA, 1999). The presence of contamination in blank samples is a data validation consideration the precedes, and is separate from, the ecological risk assessment. Data accepted through validation (true hits or "J" qualifier data) should be carried through the screening step according to USEPA (1997).*

Response:

Surficial sediments and soils were the only environmental media screened for ECOC in the Ecological Risk Assessment. The initial screening of these media is provided in Section 9.3 of the document. This screening approach was very conservative and consistent with guidance. While the description of the approach to sediment screening (Section 9.3.2.1) listed both background and blank data, these data never resulted in the exclusion of any potential ECOC. For soils, the only constituent eliminated based on a background comparison was aluminum. An additional conservative tier was added to the approach later in the assessment so that the most likely constituents resulting in potential risk were comprehensively evaluated in the exposure analysis step.

Excluding mention of these two criteria in the document will not change the outcome of the assessment. The assessment resulted in the identification of the true potential ecological risk drivers at the site.

- *Redundant lines-of-evidence were presented as though they were completely independent of one another. The ERA characterized risks using equilibrium-partitioning (EqP) approach in a hypothetical exposure of fish to sediment pore water, and benthic macroinvertebrates in the same manner. While the characterizations seem to concern separate receptors, they are in essence the same information and not independent.*

Response:

Section 9.5.5.3 specifically states "Similar to sediment TRVs for benthic invertebrates, TRVs for fish, were developed using the EqP approach for organic chemicals. TRVs for metal ECOCs [for fish] were developed using the values and sources derived for effects on benthic invertebrates." Exposing fish to pore water is substantially more conservative than exposure to ambient surface water would be. While the same conservative approach was used for both receptor groups, both groups are important components of the site food web and are included as part of complete pathways in the CSM for the site. Hence, this "redundancy" is warranted and insures an appropriate level of protection for these receptors and predators that feed on them.

- *As discussed in the specific comments section, if a line-of-evidence points to the presence of risk from an ECOC the finding should not be discounted in the conclusions unless other evidence was presented to offset or refute a finding of ecological risk.*

Response:

It is not readily apparent which finding is being referenced by EPA.

- *The ERA used an EqP approach to estimate risks to fish from bulk sediment chemistry. The narrative lacked rationale for assessing fish pore water exposure by EqP, and was essentially the same as the assessment for benthic macroinvertebrates. The ERA should be revised so the chemical body burden data can be compared directly to tissue residue levels of effect (U.S. Army ERED, 2000; Jarvinen and Ankley, 1999) which constitute a separate line-of-evidence.*

Response:

Section 9.6.4 addresses fish receptors. EqP was one of several approaches that could be used to assess potential threats to fish receptors. As discussed in that section, forage fish inhabiting and feeding in the sediment water interface can be exposed to pore water. This pathway represents a more conservative approach, since constituent concentrations in pore water are greater than ambient surface water concentrations. In addition, this approach incorporates site-specific data and conditions and was therefore, selected over a literature-based approach using chemical body burden data.

Lines of evidence comparing constituent levels in fish tissue to literature values provides an extremely weak inference into site-specific risks to fish. This type of comparison includes a high level of uncertainty, given the number of complicating variables such as differences in fish species, life stages, mobility, tolerances, feeding behaviors, species-specific predatory avoidance abilities as compared to site-specific constituent concentrations, seasonality factors, site-specific constituent suites, and even localized food abundance. Nevertheless, a cursory comparison was made using data presented in Eisler (1986) for chromium, a known metal of concern at the site.

Without a comprehensive histological investigation including organ dissection and tissue analysis (and the appropriate tissue for comparison) it is unlikely that meaningful conclusions can be reached concerning individual or multiple constituent effects as they compare to literature data (given the risk drivers at the site are generally metals and PCBs, and PCBs issues are estuarine wide). In fact, other qualitative site-specific factors such as the mobility of fish populations combined with the tidal regime of the intertidal mudflats, the qualitative observations of juvenile and adult species, the fish diversity and richness observed during sample collection and throughout the estuary, inter-station comparison of tissue concentrations, and the lack of noticeable external anomalies, could all be equally weighted to draw conclusions regarding risks to this community.

Note also, that the primary DQO for collection of fish tissue data was to characterize prey concentrations as input to dose calculations to higher order receptors (i.e., birds and mammals), not to assess risk to prey.

- *The weight-of-evidence approach used in the ERA failed to make the most use of the available lines-of-evidence to evaluate risk (e.g., see previous paragraph). Measures of effect (e g., chemistry, residues, toxicity, benthic community) were not weighted a priori and were contrary to* *the ERA method referenced (Menzie et al., 1996). Make the results more clear and succinct through revisions to the narrative weight-of-evidence. For example, sue plus (+) or minus (-) symbols to report the results qualitatively. These symbols may be combined so one plus or minus I assigned as a results of the benchmark screening, two pluses or minuses assigned to indicate toxicity observed in laboratory tests, and three of either symbol assigned to indicate toxicity observed in laboratory tests, and three of either symbol assigned to indicate benthic community impairment. The combination of symbols (---, --, -, or +, ++, +++) could be displayed in a single summary table to aid the reader in interpreting the overall weight-of-evidence by location.*

Response:

Reference to Menzie et al., 1996 in Section 9.6.3.6 of the ERA was included based on comments provided by EPA. The referenced document does provide a discussion of qualitative weight-of-evidence comparisons and the ERA evaluation was an adaptive approach based on the provided reference. Conclusions based on the approach taken would be interpreted the same, regardless of the presentation.

- *The method of Menzie et al. (1996) used in the ERA had limited objectivity because relative "inference weights" were not assigned a priori for various measures of exposure and effect in the ERA. This is a basic requirement of the method.*

Response:

Agreed. Please see response to General Comment Bullet 5.

- *Finally, risks were identified for birds using less conservative than usual exposure input parameters for: area use factors, toxicity reference values (TRVs), and EPCs using mean concentrations. It was therefore significant that, even so, certain COCs were found to pose an ecological risk. The likelihood of underestimating risks was significant because less conservative than usual input parameters were used, and also because the COCs were found to pose an ecological risk. The likelihood of underestimating risks was significant because less conservative than usual input parameters were used, and also because the COCs were selected using a process other than that considered acceptable to EPA New England (USEPA, 1997)*

Response:

Based on the approach taken, it is significant that certain ECOC were found to pose ecological risk. This was one of the objectives of the conservative tiered assessment.

It is not clear what "less conservative than usual exposure input" refers to. A three tiered approach was used to evaluate exposure ranging from the most conservative (i.e., receptors forage at the station of maximum concentration, 100% of the year), to least conservative. In the third tier, where available, standard <u>default</u> values were modified using site specific information to make the assessment more realistic, but still very conservative. As stated in the text, in the third tier, a consideration of foraging area and seasonal use was included in the calculation of area use factor. The <u>maximum</u> prey concentration and <u>mean</u> sediment or soil concentration was used to calculate the EPC in the third tier.

Since there is no single correct TRV, values were selected based on professional judgement of appropriateness considering such factors as receptor group, data source and test type. The model could easily be rerun for any TRV that may be deemed appropriate.

Based on this approach there is no realistic, greater likelihood of underestimating risks. Realistic site-specific assumptions were used in place of worst case assumptions. The ECOC was sufficiently rigorous and conservative, so that all constituents posing a threat were identified (See Response to General Comment Bullet 1).

SPECIFIC COMMENTS

1. ***Page 15, Section 9.3.2.1, Approach***

The method of using QA chemical data from blanks to screen for COCs is not considered appropriate or acceptable EPA. The approach presented does not follow EPA New England guidance (USEPA, 1997). Blanks data are relevant to the data validation process and are used before the ERA to determine data quality. Blanks data should not be used to screen for COCs.

EPA recommends deleting the method of using reference or background chemical data to screen for COCs. Again, the approach presented does not follow EPA New England guidance (USEPA, 1997). While it is acceptable in the risk screening to identify and discuss chemical background concentrations that exceed those on site, if the maximum concentration of a chemical detected on site exceeds its screening-level benchmark, it must be identified as a COC and carried into the ERA (exceptions are chemicals that function as nutrients or that are toxic at only very high concentrations).

A COC that has background concentrations in excess of the site can be excluded from the data analysis phase (toxicity and exposure assessments), but must be carried as a COC and the situation discussed in the risk characterization.

Response:

See Response to General Comment Bullet 1.

2 ***Page 76, Section 9.5.3.3, TRVs for Fish***

The evaluation of tissue chemical data should include a comparison of measured levels to residue effect levels reported in EPA (Jarvinen and Ankley, 1999) or U.S. Army Corps of Engineers (USACE ERED, 2000) databases. Comparisons may be done by chemical and species.

Response:

See Response to General Comment Bullet 4.

3 ***Page 100, Section 9.6.6.1.2, Marine Basin***

The section cites the presence of successful breeding populations of many species of birds in the estuary as evidence that ecological risks by other lines-of-evidence are overestimated in the Basin. However, the argument is not supported by an analysis such as comparison of current and historic breeding or population data The statement that risks are overestimated in the Basin should be deleted unless revisions to the ERA include a comparative or other analysis in support of the argument.

Response:

Agreed. The statement that risks are overestimated in the Basin can be deleted from the final document.

4 ***Table 9-11 Surface Soil Data Summary and Selection of ECOCs***

The approach of using average background concentrations as the primary tier for selecting COCs should be revised The approach conflicts with EPA New England guidance (USEPA, 1997) The first level of COC selection should be entirely risk-based using maximum detected concentrations. See an earlier specific comment on the subject for additional detail.

Response:

As requested by the EPA, Raymark data were considered background in the evaluation of ECOC. Note that this step was only done for the assessment of metals. Note also, that of the 25 metals analyzed, all but five were retained as ECOC. The five metals that were eliminated from further assessment were aluminum, calcium, magnesium, potassium and sodium. Of these, all but aluminum are essential nutrients.

5. ***Table 9-28, Benthic Invertebrate Summary for Weight of Evidence***

In addition to general comments above on the weight-of-evidence analysis, the following specific comments apply to the summary table:

- *There were inconsistencies in reporting % survival data. For example, in this table Station TD-7 Polychaeta survival was 96%, but in an earlier tables it was reported as 60% For Station TC-6, % survival was 0%, but listed as 5% elsewhere in the report. Review the % survival data and make any necessary corrections to the revised report.*

Response:

In Table 9-28, Station TD-7 Polychaeta survival was 96%; similarly, in Table 9-19, survival data for the polychaete *Neanthes arenaceodentata* was also listed as 96%. In both Tables 9-20 and 9-28, % survival for the amphipod *Leptocheirus plumulosus* was listed as 0%. These tables appear to be consistent. No other inconsistencies in toxicity testing survival data are noted in any of the tables or text.

- *Two species of Amphipoda were tested, but the table reports amphipod % survival in only one row of data. Review amphipod % survival data and make any necessary corrections to this table and overall report*

Response:

Table 9-28, Footnote (1) states that the [only]amphipod test organism for the 1994 investigation was *Ampelisca abdita*, and the [only] amphipod test organism for the 1999 investigation was *Leptocheirus plumulosus*. Including data for both amphipods on the same line in the table may have lead to some confusion. The data are correct; no corrections to the table or overall report are warranted.

- *Table 9-28 should be revised to help focus the readers attention on important aspects of the summary data discussed in the text. Fonts, shading or some other visual can help highlight those measures which signified a potential risk, such as reduced survival relative to control or low species richness*

Response:

Test results of toxicity testing provided in Tables 9-19 and 9-20 are footnoted where results are statistically significantly different from control results. No shading or other 'flagging' was included on Table 9-28 since no statistical evaluations were performed on the other data sets.

CONCLUSIONS

The ERA used a process of selecting COCs that is not generally acceptable to EPA and did not follow EPA New England guidance (USEPA, 1997) The issue was also expressed in earlier comments on a proposed approach for ERA at the site (USEPA, 1999).

The ERA evaluated risks to wildlife using less conservative than usual exposure input parameters · area use factors, mean concentrations, and toxicity reference values. Because of this, risks were probably underestimated, and when a hazard quotient (HQ) exceeded 1 (unity), it is likely than an ecological risk was indicated, but this was not done.

Several revisions to the ERA were recommended so that existing data can be constructed in a more comprehensive array of line-of-evidence. Risks to fish should be evaluated through comparisons of existing fish tissue data to residue effect levels reported in readily available databases (Jarvinen and Ankley, 1999; USACE ERED, 2000).

Finally, if a line-of-evidence points to the presence of risk from COCs, the finding should not be discounted unless other evidence of comparable "weight" can be presented to offset or refute the finding of risk.

Response:

All of the statements made in the conclusion portion of Attachment II were made in both the General and Specific Comments. The reader is referred to responses to the General and Specific Comments above for this information.

REFERENCES:

New York State Department of Environmental Conservation. 1994. Technical and Administrative Guidance Memorandum #4046, Appendix A, Table 4.

USEPA. 1998. Technical Protocol for Evaluating Natural Attenuation of Chlorinated Solvents in Ground Water, EPA/600/R-98/128, September 1998.

TABLE 1
Reduction-Oxidation and Chlorinated Solvent Groundwater Data
Stratford Army Engine Plant

	MW-1		MW-2		MW-4		MWCD-99-01A		MWCD-99-01B	
	07/12/1999	Nov 1999	07/12/1999	Nov 1999	07/12/1999	Nov 1999	July 1999	11/12/1999	July 1999	11/10/1999
Redox Potential (mV)	174	NS	-113.6	NS	98.8	NS	NS	-200.6	NS	-155.6
Dissolved Oxygen (mg/L)	2.89	NS	0.73	NS	1.68	NS	NS	0.44	NS	0.29
pH	6.53	NS	6.61	NS	7.03	NS	NS	8.18	NS	6.8
1,1,1-TCA (ug/L)	1.2 J	NS	<5	NS	<1	NS	NS	1.5	NS	<1
1,1,2-TCA (ug/L)	<5	NS	<5	NS	<1	NS	NS	<1	NS	<1
1,1-DCA (ug/L)	<10	NS	<10	NS	1.5 J	NS	NS	12	NS	<2
1,1-DCE (ug/L)	1.6 J	NS	<5	NS	<1	NS	NS	<1	NS	<1
1,2-DCA (ug/L)	<5	NS	<5	NS	<1	NS	NS	<1	NS	<1
cis-1,2-DCE (ug/L)	9.1	NS	2.1 J	NS	0.78 J	NS	NS	6.8	NS	0.71 J
PCE (ug/L)	69	NS	<5	NS	<1	NS	NS	<1	NS	<1
trans-1,2-DCE (ug/L)	<5	NS	<5	NS	0.32 J	NS	NS	<1	NS	<1
TCE (ug/L)	83	NS	<5	NS	<1	NS	NS	0.5	NS	<1
Vinyl Chloride (ug/L)	<5	NS	2.2 J	NS	<1	NS	NS	<1	NS	<1
Nitrate-Nitrite (mg/L)	NA	NS	NA	NS	NA	NS	NS	NA	NS	NA
Sulfate (mg/L)	NA	NS	NA	NS	NA	NS	NS	NA	NS	NA
TOC (mg/L)	NA	NS	NA	NS	NA	NS	NS	NA	NS	NA
ethane (mg/L)	NA	NS	NA	NS	NA	NS	NS	NA	NS	NA
ethene (mg/L)	NA	NS	NA	NS	NA	NS	NS	NA	NS	NA
methane (mg/L)	NA	NS	NA	NS	NA	NS	NS	NA	NS	NA
Arsenic (mg/L)	<0.01	NS	0.017	NS	<0.01	NS	NS	<0.01	NS	0.0154
Chromium (mg/L)	<0.01	NS	<0.01	NS	<0.01	NS	NS	<0.01	NS	<0.01

	MWCD-99-02A		MWCD-99-02B		PZ-1D		PZ-4D		PZ-5D	
	July 1999	11/11/1999	July 1999	11/11/1999	07/06/1999	11/15/1999	07/06/1999	11/10/1999	07/06/1999	11/11/1999
Redox Potential (mV)	NS	65.2	NS	-197.1	-54.7	-368	-10.7	-194.2	-106.4	-133
Dissolved Oxygen (mg/L)	NS	1.88	NS	0.3	8.87	0.07	4.93	0.33	7.06	0.08
pH	NS	8.71	NS	7.32	6.72	7.07	6.66	6.64	6.39	6.54
1,1,1-TCA (ug/L)	NS	<1	NS	<1	<25	2.8	<1	1.3	<1	<1
1,1,2-TCA (ug/L)	NS	<1	NS	<1	<25	<1	<1	<1	<1	<1
1,1-DCA (ug/L)	NS	0.75 J	NS	<2	180	130	80	110	1.6 J	0.73 J
1,1-DCE (ug/L)	NS	<1	NS	<1	140	70	2.1	2.4	<1	<1
1,2-DCA (ug/L)	NS	<1	NS	<1	<25	1.4	<1	0.6 J	<1	0.61 J
cis-1,2-DCE (ug/L)	NS	3	NS	<1	390	350	110	130	0.66 J	0.62 J
PCE (ug/L)	NS	3.4	NS	<1	850	660	<1	<1	<1	<1
trans-1,2-DCE (ug/L)	NS	<1	NS	<1	<25	<1	7.2	9.4	2.30	<1
TCE (ug/L)	NS	0.23	NS	<1	2400	2500	<2.2	4.2	0.7 J	<1
Vinyl Chloride (ug/L)	NS	<1	NS	<1	25	22	230	2800	<1	0.44 J
Nitrate-Nitrite (mg/L)	NS	NA	NS	NA	NA	NA	NA	NA	NA	NA
Sulfate (mg/L)	NS	NA	NS	NA	NA	NA	NA	NA	NA	NA
TOC (mg/L)	NS	NA	NS	NA	NA	NA	NA	NA	NA	NA
ethane (mg/L)	NS	NA	NS	NA	NA	NA	NA	NA	NA	NA
ethene (mg/L)	NS	NA	NS	NA	NA	NA	NA	NA	NA	NA
methane (mg/L)	NS	NA	NS	NA	NA	NA	NA	NA	NA	NA
Arsenic (mg/L)	NS	<0.01	NS	<0.01	0.0047 J	<0.01	0.018	<0.01	0.038	0.0495
Chromium (mg/L)	NS	0.008	NS	<0.01	<0.01	<0.01	0.019	0.0068	<0.01	0.0092

NA = Not analyzed
NS = Not sampled
* = Equipment malfunction, could not get a reliable reading

TABLE 1
Reduction-Oxidation and Chlorinated Solvent Groundwater Data
Stratford Army Engine Plant

	PZ-7D		PZ-8D		PZ-9D		PZ-11D		PZ-13D	
	07/02/1999	Nov 1999	07/07/1999	11/16/1999	07/01/1999	11/08/1999	07/01/1999	11/12/1999	07/07/1999	11/16/1999
Redox Potential (mV)	13.8	NS	-40.5	-353.4	78.7	69.2	236.3	-308.2	137.4	325.6
Dissolved Oxygen (mg/L)	*	NS	0.38	0.26	0.45	0.54	0.48	0.22	2.66	6.93
pH	6.22	NS	7.13	*	7.01	7.3	7.26	7.64	6.27	6.49
1,1,1-TCA (ug/L)	<1	NS	86	3300	<50	<1	20 J	29	<25	13
1,1,2-TCA (ug/L)	<1	NS	<1	6.4	<50	2.6	<25	<1	<25	2.1
1,1-DCA (ug/L)	<2	NS	100	220 J	<100	16	28 J	30	<50	6.1
1,1-DCE (ug/L)	<1	NS	560	3100	24	32 J	230	340	15 J	26
1,2-DCA (ug/L)	<1	NS	19	25	<50	0.86 J	26	32	<25	<1
cis-1,2-DCE (ug/L)	0.82 J	NS	270 J	270	500	970	<25	5	71	69 J
PCE (ug/L)	<1	NS	63	75	<50	2.5	320	370	600	630
trans-1,2-DCE (ug/L)	<1	NS	<1	<1	<50	2.3	<25	<1	<25	<1
TCE (ug/L)	<1	NS	6800	5900	1300	1500	190	250	790	970
Vinyl Chloride (ug/L)	<1	NS	7.1	18	<50	3.2	<25	<1	<25	<1
Nitrate-Nitrite (mg/L)	NA	NS	NA	NA	NA	0.118	NA	NA	NA	1.15
Sulfate (mg/L)	NA	NS	NA	NA	NA	86	NA	NA	NA	52.6
TOC (mg/L)	NA	NS	NA	NA	NA	<5	NA	NA	NA	5.5
ethane (mg/L)	NA	NS	NA	NA	NA	<1.5	NA	NA	NA	<1.5
ethene (mg/L)	NA	NS	NA	NA	NA	<1.5	NA	NA	NA	<1.5
methane (mg/L)	NA	NS	NA	NA	NA	7	NA	NA	NA	4.7
Arsenic (mg/L)	<0.01	NS	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
Chromium (mg/L)	<0.01	NS	0.0063	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01

	PZ-16D		PZ-17D		WC-1S		WC-2D		WC-3S	
	07/01/1999	11/11/1999	07/02/1999	11/09/1999	07/08/1999	11/15/1999	07/07/1999	11/09/1999	07/12/1999	11/11/1999
Redox Potential (mV)	-27.3	-109.3	134.3	126.7	-74.6	-340.7	-96.6	-149.2	-334.6	-355.2
Dissolved Oxygen (mg/L)	0.36	0.42	0.5	0.83	0.64	0.36	1.03	2.18	0.76	0.25
pH	6.43	6.73	6.76	6.91	6.71	7	6.38	6.36	7.73	8.01
1,1,1-TCA (ug/L)	<1	<1	<1	<1	<10	1.7	<25	<50	<1	<1
1,1,2-TCA (ug/L)	<1	<1	1	<1	<10	<1	<25	<50	<1	<1
1,1-DCA (ug/L)	55	34	37	25	110	25	82	74	5.2	4.3
1,1-DCE (ug/L)	73	38	60	31	52	10	<25	<50	<1	<1
1,2-DCA (ug/L)	2.70	1.6	<	0.71 J	<10	<1	<25	<50	<1	<
cis-1,2-DCE (ug/L)	310	300	110	68	34	11	320	470	1.1	0.65 J
PCE (ug/L)	3.20	1.6	5.7	2.3	11	2.1	<25	<50	<1	<1
trans-1,2-DCE (ug/L)	3.20	1.4	1.2	0.49 J	<10	<1	6.8 J	<50	0.37 J	<1
TCE (ug/L)	6.9	3.9 J	640	490	12	2.1	18	15 J	<1	<1
Vinyl Chloride (ug/L)	100	70	14	7.9	37	7.7	1900	1900	<1	<1
Nitrate-Nitrite (mg/L)	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Sulfate (mg/L)	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
TOC (mg/L)	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
ethane (mg/L)	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
ethene (mg/L)	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
methane (mg/L)	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Arsenic (mg/L)	0.0051 J	0.0061 J	<0.01	<0.01	0.0061 J	0.00736 J	<0.01	<0.01	0.005 J	<0.01
Chromium (mg/L)	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.05	<0.01

NA = Not analyzed
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TABLE 1
Reduction-Oxidation and Chlorinated Solvent Groundwater Data
Stratford Army Engine Plant

	WC-4S		WC-5S		WC-9D2		WC-9S		WC-10S	
	07/08/1999	11/16/1999	07/08/1999	11/16/1999	07/01/1999	11/08/1999	06/30/1999	11/08/1999	07/07/1999	11/16/1999
Redox Potential (mV)	-61 5	-361 9	-107 5	-351	-124 3	-137	148 5	140 6	125 1	-314 3
Dissolved Oxygen (mg/L)	0 53	0 27	0 67	0 25	0 38	1 03	0 67	3 11	3 08	0 39
pH	6 57	6 79	6 46	6 55	6 89	7 15	6 23	6 67	6 92	6 98
1,1,1-TCA (ug/L)	770	64 00	<100	<10	<1	<1	<1	<1	<25	15
1,1,2-TCA (ug/L)	<100	<1	<100	<10	<1	<1	<1	<1	<25	<1
1,1-DCA (ug/L)	240	94 00	<100	<20	<2	<2	1 J	<2	29 J	11
1,1-DCE (ug/L)	730	94 00	<100	<10	<1	<1	3 3	<1	280	49
1,2-DCA (ug/L)	<100	1 10	<100	<10	<1	<1	<1	<1	<25	1 4
cis-1,2-DCE (ug/L)	59 J	18	8900	13	<1	<1	1 1 J	<1	190	52
PCE (ug/L)	<100	2 80	<100	<10	<1	<1	0 72 J	<1	5 2 J	5 5
trans-1,2-DCE (ug/L)	<100	<1	37 J	9 5 J	<1	<1	<1	<1	<25	<1
TCE (ug/L)	67 J	19	<100	6 5 J	<1	<1	2 5	0 29 J	210	71
Vinyl Chloride (ug/L)	<100	64	3000	<10	<1	<1	<1	<1	<25	0 99 J
Nitrate-Nitrite (mg/L)	NA	NA	NA	NA	NA	NA	NA	NA	NA	1 75
Sulfate (mg/L)	NA	NA	NA	NA	NA	NA	NA	NA	NA	40 5
TOC (mg/L)	NA	NA	NA	NA	NA	NA	NA	NA	NA	7 7
ethane (mg/L)	NA	NA	NA	NA	NA	NA	NA	NA	NA	<1 5
ethene (mg/L)	NA	NA	NA	NA	NA	NA	NA	NA	NA	<1 5
methane (mg/L)	NA	NA	NA	NA	NA	NA	NA	NA	NA	20
Arsenic (mg/L)	0 0055 J	0 0066 J	<0 01	0 0057 J	0 05	0 065	<0 01	<0 01	<0 01	<0 01
Chromium (mg/L)	<0 01	<0 01	<0 01	0 0059 J	<0 01	<0 01	<0 01	<0 01	0 006	0 012

	WC-12S		WC-13S		WC-14S		MW-19D1(WC-19D1)		WC-19S	
	07/08/1999	11/17/1999	July 1999	11/16/1999	07/13/1999	Nov 1999	06/30/1999	11/17/1999	06/29/1999	11/17/1999
Redox Potential (mV)	219 8	-147 5	NS	340 1	4 8	NS	64 6	-338 1	168 9	-221 4
Dissolved Oxygen (mg/L)	3 44	2 81	NS	7 60	2 74	NS	0 75	0 33	0 62	0 34
pH	6 61	7	NS	6 50	6 70	NS	6 03	*	5 76	*
1,1,1-TCA (ug/L)	1800	1700	NS	24	0 51 J	NS	<25	3 3	9	25
1,1,2-TCA (ug/L)	<100	<1	NS	<1	<1	NS	<25	<1	<5	<1
1,1-DCA (ug/L)	270	110	NS	1 4 J	3 60	NS	<50	3 10	3 6 J	12
1,1-DCE (ug/L)	960	660	NS	<1	<1	NS	20 J	19	4 9 J	11
1,2-DCA (ug/L)	<100	5 4	NS	<1	<1	NS	<25	1 9	<5	<1
cis-1,2-DCE (ug/L)	36 J	100	NS	0 54 J	2 6	NS	44	64	50	180
PCE (ug/L)	<100	4	NS	5 3	<1	NS	400	480	290	740
trans-1,2-DCE (ug/L)	<100	1	NS	<1	<	NS	<25	<1	<25	<1
TCE (ug/L)	140	270	NS	2 7	21	NS	24 J	36	8 8	22
Vinyl Chloride (ug/L)	<100	11	NS	<1	<1	NS	<25	<1	<25	20
Nitrate-Nitrite (mg/L)	NA	4 26	NS	NA	NA	NS	NA	NA	NA	NA
Sulfate (mg/L)	NA	67 6	NS	NA	NA	NS	NA	NA	NA	NA
TOC (mg/L)	NA	5 6	NS	NA	NA	NS	NA	NA	NA	NA
ethane (mg/L)	NA	<1 5	NS	NA	NA	NS	NA	NA	NA	NA
ethene (mg/L)	NA	<1 5	NS	NA	NA	NS	NA	NA	NA	NA
methane (mg/L)	NA	<1	NS	NA	NA	NS	NA	NA	NA	NA
Arsenic (mg/L)	<0 01	<0 01	NS	<0 01	0 0074 J	NS	<0 01	<0 01	<0 01	<0 01
Chromium (mg/L)	0 35	0 522	NS	<0 01	<0 01	NS	<0 01	<0 01	<0 01	<0 01

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NS = Not sampled
* = Equipment malfunction, could not get a reliable reading

TABLE 1
Reduction-Oxidation and Chlorinated Solvent
Groundwater Data
Stratford Army Engine Plant

	WC-20D1		WC-20D2		WC-20S		WC-21D1		WC-21D2	
	July 1999	11/09/1999	July 1999	11/09/1999	July 1999	11/09/1999	July 1999	11/10/1999	July 1999	11/10/1999
Redox Potential (mV)	NS	84 4	NS	133 7	NS	184 7	NS	-112 3	NS	-116 3
Dissolved Oxygen (mg/L)	NS	0 35	NS	0 24	NS	0 26	NS	0 28	NS	0 25
pH	NS	7 11	NS	6 86	NS	6 24	NS	7 11	NS	6 61
1,1,1-TCA (ug/L)	NS	<1	NS	<1	NS	<1	NS	<1	NS	<1
1,1,2-TCA (ug/L)	NS	2	NS	<1	NS	<1	NS	1 8	NS	<1
1,1-DCA (ug/L)	NS	<2	NS	<2	NS	<2	NS	2 3	NS	0 79 J
1,1-DCE (ug/L)	NS	3	NS	<1	NS	<1	NS	17	NS	<1
1,2-DCA (ug/L)	NS	0 47 J	NS	<1	NS	<1	NS	8	NS	<1
cis-1,2-DCE (ug/L)	NS	780	NS	0 53 J	NS	<1	NS	610	NS	4 2 J
PCE (ug/L)	NS	1 3	NS	<1	NS	<1	NS	18	NS	<1
trans-1,2-DCE (ug/L)	NS	2 7	NS	<1	NS	<1	NS	1 1	NS	<1
TCE (ug/L)	NS	180	NS	0 59 J	NS	<1	NS	60	NS	45
Vinyl Chloride (ug/L)	NS	0 59 J	NS	<1	NS	<1	NS	3 1	NS	<1
Nitrate-Nitrite (mg/L)	NS	NA	NS	NA	NS	NA	NS	NA	NS	NA
Sulfate (mg/L)	NS	NA	NS	NA	NS	NA	NS	NA	NS	NA
TOC (mg/L)	NS	NA	NS	NA	NS	NA	NS	NA	NS	NA
ethane (mg/L)	NS	NA	NS	NA	NS	NA	NS	NA	NS	NA
ethene (mg/L)	NS	NA	NS	NA	NS	NA	NS	NA	NS	NA
methane (mg/L)	NS	NA	NS	NA	NS	NA	NS	NA	NS	NA
Arsenic (mg/L)	NS	0 0119	NS	0 054	NS	<0 01	NS	<0 01	NS	<0 01
Chromium (mg/L)	NS	<0 01	NS	<0 01	NS	<0 01	NS	<0 01	NS	<0 01

	WC-21S		WC1-1S		WC2-1D		WC2-1S		WC2-1I	
	July 1999	11/10/1999	07/02/1999	Nov 1999	07/06/1999	Nov 1999	07/07/1999	Nov 1999	07/02/1999	Nov 1999
Redox Potential (mV)	NS	-135 6	76 8	NS	-63 8	NS	-63 7	NS	-45 5	NS
Dissolved Oxygen (mg/L)	NS	0 32	1 02	NS	2 50	NS	1 95	NS	*	NS
pH	NS	6 47	5 96	NS	6 76	NS	6 16	NS	6 37	NS
1,1,1-TCA (ug/L)	NS	<1	<1	NS	<1	NS	<1	NS	<1	NS
1,1,2-TCA (ug/L)	NS	<1	<1	NS	<1	NS	<1	NS	<1	NS
1,1-DCA (ug/L)	NS	<2	<2	NS	<2	NS	0 78 J	NS	0 77 J	NS
1,1-DCE (ug/L)	NS	<1	<1	NS	<1	NS	<1	NS	<1	NS
1,2-DCA (ug/L)	NS	<1	<1	NS	<1	NS	0 38 J	NS	<1	NS
cis-1,2-DCE (ug/L)	NS	1 2	0 69 J	NS	<1	NS	1 4	NS	<1	NS
PCE (ug/L)	NS	<1	0 88 J	NS	<1	NS	<1	NS	<1	NS
trans-1,2-DCE (ug/L)	NS	<1	<1	NS	<1	NS	0 9 J	NS	<1	NS
TCE (ug/L)	NS	<1	<1	NS	<1	NS	3 9	NS	<1	NS
Vinyl Chloride (ug/L)	NS	<1	<1	NS	<1	NS	<1	NS	<1	NS
Nitrate-Nitrite (mg/L)	NS	NA	NA	NS	NA	NS	NA	NS	NA	NS
Sulfate (mg/L)	NS	NA	NA	NS	NA	NS	NA	NS	NA	NS
TOC (mg/L)	NS	NA	NA	NS	NA	NS	NA	NS	NA	NS
ethane (mg/L)	NS	NA	NA	NS	NA	NS	NA	NS	NA	NS
ethene (mg/L)	NS	NA	NA	NS	NA	NS	NA	NS	NA	NS
methane (mg/L)	NS	NA	NA	NS	NA	NS	NA	NS	NA	NS
Arsenic (mg/L)	NS	<0 01	<0 01	NS	0 024	NS	<0 01	NS	0 0086 J	NS
Chromium (mg/L)	NS	<0 01	<0 01	NS	<0 01	NS	<0 01	NS	<0 01	NS

NA = Not analyzed
NS = Not sampled
* = Equipment malfunction, could not get a reliable reading

TABLE 1
Reduction-Oxidation and Chlorinated Solvent Groundwater Data
Stratford Army Engine Plant

	WC2-2D		WC2-2I		WC2-3D		WC2-3I		WC2-3S	
	07/08/1999	11/10/1999	07/08/1999	11/12/1999	07/07/1999	11/12/1999	07/13/1999	11/12/1999	07/13/1999	11/12/1999
Redox Potential (mV)	-112 9	-106 1	-32 2	-317	46 1	46 4	39 9	20 9	-234 3	-309 8
Dissolved Oxygen (mg/L)	3 85	2 76	6 68	0 4	4 91	0 56	1 54	0 51	8 11	0 1
pH	6 84	6 84	6 68	6 99	6 51	6 56	6 36	6 5	6 74	6 78
1,1,1-TCA (ug/L)	<1	<1	<25	<1	<1	<1	<50	<1	5 1	2 9
1,1,2-TCA (ug/L)	<1	<1	<25	<1	<1	<1	<50	<1	<1	<1
1,1-DCA (ug/L)	3 9 J	2 80	<50	1 3 J	0 55 J	<2	<100	4 4	2 2	8 5
1,1-DCE (ug/L)	1 1 J	0 59 J	<25	1 8	2 9	2	<50	9 9	<1	<1
1,2-DCA (ug/L)	<1	<1	<25	<1	<1	<1	<50	0 68 J	<1	<1
cis-1,2-DCE (ug/L)	15	12	66	230	45 J	64	530	660	1 6	2
PCE (ug/L)	<1	<1	19 J	1 5	99	33	17 J	22	<1	<1
trans-1,2-DCE (ug/L)	0 38 J	<1	<25	2 1	<1	<1	27 J	19	<1	<1
TCE (ug/L)	<6 6	4 2	320	24	3100	620	320	400	<1	0 57
Vinyl Chloride (ug/L)	<1	<1	28	36	<1	1 4	<50	62	2 1	<1
Nitrate-Nitrite (mg/L)	NA	<0 5	NA	<0 1	NA	<0 5	NA	NA	NA	NA
Sulfate (mg/L)	NA	1610	NA	211	NA	387	NA	NA	NA	NA
TOC (mg/L)	NA	11 2	NA	6 8	NA	<5	NA	NA	NA	NA
ethane (mg/L)	NA	<1 5	NA	0 49 J	NA	3 9	NA	NA	NA	NA
ethene (mg/L)	NA	<1 5	NA	1 5	NA	14	NA	NA	NA	NA
methane (mg/L)	NA	6 8 J	NA	89	NA	45	NA	NA	NA	NA
Arsenic (mg/L)	0 02	0 0274	0 0074 J	0 0134	<0 01	<0 01	0 0059 J	0 00832 J	<0 01	<0 01
Chromium (mg/L)	<0 01	0 0761 J	<0 01	<0 01	<0 01	<0 01	<0 05	<0 01	<0 05	<0 01

	WC2-4I		WC2-4S		WC2-5I		WC2-5S		WC2-6I	
	July 1999	11/15/1999	07/13/1999	11/15/1999	July 1999	11/15/1999	07/13/1999	11/10/1999	July 1999	11/15/1999
Redox Potential (mV)	NS	-96	58	247 1	NS	-377 4	176	78 8	NS	1 2
Dissolved Oxygen (mg/L)	NS	0 31	2 98	2 39	NS	0 14	3 94	1 02	NS	0 64
pH	NS	7 04	6 94	7 01	NS	6 95	6 3	6 58	NS	6 5
1,1,1-TCA (ug/L)	NS	6 5	1 7	<1	NS	25	0 37 J	3 5	NS	<1
1,1,2-TCA (ug/L)	NS	<1	<1	<1	NS	<1	<1	<1	NS	<1
1,1-DCA (ug/L)	NS	120	1 9 J	<2	NS	510	9 2	130	NS	11
1,1-DCE (ug/L)	NS	34	0 59 J	<1	NS	100	0 84 J	17	NS	<1
1,2-DCA (ug/L)	NS	<1	<1	<1	NS	<1	<1	0 36 J	NS	<1
cis-1,2-DCE (ug/L)	NS	150	2 4	<1	NS	230	3 6	48	NS	330
PCE (ug/L)	NS	22	11	0 54 J	NS	11	1 7	12	NS	610
trans-1,2-DCE (ug/L)	NS	0 73 J	<1	<1	NS	1 3	<1	0 37 J	NS	1 6
TCE (ug/L)	NS	14	15	0 63	NS	53	18	23	NS	3700
Vinyl Chloride (ug/L)	NS	41	<1	<1	NS	86	<1	13	NS	21
Nitrate-Nitrite (mg/L)	NS	NA	NA	NA	NS	<0 5	NA	NA	NS	NA
Sulfate (mg/L)	NS	NA	NA	NA	NS	387	NA	NA	NS	NA
TOC (mg/L)	NS	NA	NA	NA	NS	<5	NA	NA	NS	NA
ethane (mg/L)	NS	NA	NA	NA	NS	3 9	NA	NA	NS	NA
ethene (mg/L)	NS	NA	NA	NA	NS	14	NA	NA	NS	NA
methane (mg/L)	NS	NA	NA	NA	NS	45	NA	NA	NS	NA
Arsenic (mg/L)	NS	0 0674	<0 01	<0 01	NS	0 00827	<0 01	<0 01	NS	<0 01
Chromium (mg/L)	NS	<0 01	<0 01	<0 01	NS	<0 01	<0 01	<0 01	NS	<0 01

NA = Not analyzed
NS = Not sampled
* = Equipment malfunction, could not get a reliable reading

TABLE 1
Reduction-Oxidation and Chlorinated Solvent
Groundwater Data
Stratford Army Engine Plant

	WC3-1D		WC3-1I		WC3-2D		WC3-2I		WC5-1D	
	July 1999	11/18/1999	July 1999	11/17/1999	July 1999	11/17/1999	July 1999	11/17/1999	07/13/1999	11/08/1999
Redox Potential (mV)	NS	-290	NS	-588 5	NS	-588 3	NS	-562 3	-40 4	127 6
Dissolved Oxygen (mg/L)	NS	2 21	NS	0 17	NS	0 17	NS	0 13	9 79	0 44
pH	NS	6 3	NS	6 76	NS	6 73	NS	6 83	6 63	6 85
1,1,1-TCA (ug/L)	NS	<1	NS	250	NS	<1	NS	15	<1	<1
1,1,2-TCA (ug/L)	NS	<1	NS	<1	NS	<1	NS	<1	<1	<1
1,1-DCA (ug/L)	NS	<2	NS	47 J	NS	<2	NS	18	<2	<2
1,1-DCE (ug/L)	NS	0 8 J	NS	2900	NS	<1	NS	24	<1	<1
1,2-DCA (ug/L)	NS	<1	NS	160	NS	<1	NS	1 4	<1	<1
cis-1,2-DCE (ug/L)	NS	14	NS	17	NS	0 3 J	NS	17	<1	<1
PCE (ug/L)	NS	350	NS	590	NS	3	NS	130	<1	<1
trans-1,2-DCE (ug/L)	NS	<1	NS	110	NS	<1	NS	<1	<1	<1
TCE (ug/L)	NS	680	NS	<1	NS	3 9	NS	110	<1	<1
Vinyl Chloride (ug/L)	NS	1 3	NS	3 3	NS	<1	NS	<1	<1	<1
Nitrate-Nitrite (mg/L)	NS	<0 5	NS	NA	NS	NA	NS	NA	NA	NA
Sulfate (mg/L)	NS	387	NS	NA	NS	NA	NS	NA	NA	NA
TOC (mg/L)	NS	<5	NS	NA	NS	NA	NS	NA	NA	NA
ethane (mg/L)	NS	3 9	NS	NA	NS	NA	NS	NA	NA	NA
ethene (mg/L)	NS	14	NS	NA	NS	NA	NS	NA	NA	NA
methane (mg/L)	NS	45	NS	NA	NS	NA	NS	NA	NA	NA
Arsenic (mg/L)	NS	<0 01	NS	0 0138	NS	<0 01	NS	<0 01	<0 01	0 0069
Chromium (mg/L)	NS	<0 01	NS	0 021	NS	<0 01	NS	<0 01	0 052 J	<0 01

	WC5-1S		WC5-2I		WC5-2S		WC5-3S		WC6-1I	
	07/06/1999	11/09/1999	06/30/1999	11/08/1999	07/01/1999	Nov 1999	07/01/1999	11/08/1999	06/30/1999	Nov 1999
Redox Potential (mV)	108 2	87 5	190 2	140 4	138 2	NS	251 2	230 9	173 5	NS
Dissolved Oxygen (mg/L)	0 34	0 26	0 8	0 45	0 69	NS	2 66	0 64	0 34	NS
pH	6 59	7 46	6 15	6 47	6 14	NS	5 81	5 86	5 87	NS
1,1,1-TCA (ug/L)	<1	<1	<1	1 3	<1	NS	<1	<1	<1	NS
1,1,2-TCA (ug/L)	<1	<1	<1	<1	<1	NS	<1	<1	<1	NS
1,1-DCA (ug/L)	7 50	<2	4 8	1 9 J	<2	NS	<2	<2	<2	NS
1,1-DCE (ug/L)	57 00	0 41	1 2	<1	<1	NS	<1	<1	<1	NS
1,2-DCA (ug/L)	0 97 J	<1	<1	<1	<1	NS	<1	<1	<1	NS
cis-1,2-DCE (ug/L)	27	0 32	2 9	1 9	<1	NS	<1	<1	0 4 J	NS
PCE (ug/L)	1 40	<1	7 8	4	<1	NS	<1	<1	0 28 J	NS
trans-1,2-DCE (ug/L)	0 26 J	<1	<1	<1	<1	NS	<1	<1	<1	NS
TCE (ug/L)	160	2 9	4 2	1 7	<1	NS	<1	<1	7 4	NS
Vinyl Chloride (ug/L)	<1	<1	<1	<1	<1	NS	<1	<1	<1	NS
Nitrate-Nitrite (mg/L)	NA	1 98	NA	NA	NA	NS	NA	0 73 J	NA	NS
Sulfate (mg/L)	NA	18	NA	NA	NA	NS	NA	35 6	NA	NS
TOC (mg/L)	NA	<5	NA	NA	NA	NS	NA	<5	NA	NS
ethane (mg/L)	NA	<1 5	NA	NA	NA	NS	NA	<1 5	NA	NS
ethene (mg/L)	NA	<1 5	NA	NA	NA	NS	NA	<1 5	NA	NS
methane (mg/L)	NA	86	NA	NA	NA	NS	NA	0 85 J	NA	NS
Arsenic (mg/L)	<0 01	<0 01	<0 01	<0 01	<0 01	NS	<0 01	<0 01	<0 01	NS
Chromium (mg/L)	<0 01	<0 01	<0 01	<0 01	<0 01	NS	<0 01	<0 01	<0 01	NS

NA = Not analyzed
NS = Not sampled
* = Equipment malfunction, could not get a reliable reading

TABLE 2
Tide Amplitude Versus Distance from River
Stratford Army Engine Plant

Well	Distance from River (feet)	Amplitude of Tide (feet)
LW-5D*	1269.53	0.03425
LW-5SI	1253.91	0.16783
LW-5S	1261.72	0.12075
WC-9D2*	617.19	0.1002
LW-10I	726.56	0.0176
PZ-9D*	609.38	0.1127
WC-9S	613.28	0.0059
WC2-1S	125.00	0.1678
WC2-1I	105.47	0.8624
WC2-1D	93.75	0.5647
WC-5S*	74.22	0.4608
PZ-5D	78.13	0.5843
PZ-4D	85.94	0.2495
WC-4S	82.03	0.6502
WC2-31	82.03	0.3188
WC-3S	78.13	1.371
WC2-2D*	70.31	0.4537
PZ-1D*	70.31	0.518
WC-1S*	70.31	0.359

* indicates well data used in Figure 3

TABLE 3
Result of Advection/Dispersion Calculations Including Retardation
Stratford Army Engine Plant

Compound	Hydraulic Conductivity	Criteria (ppb)	Time Compound Reaches River at Criteria Concentration (years)
1,1-DCE	logarithmic mean	96	28.19
TCE	logarithmic mean	2340	37.75
PCE	logarithmic mean	88	85.05
1,1-DCE	arithmetic mean	96	21.69
TCE	arithmetic mean	2340	32.06
PCE	arithmetic mean	88	66.72

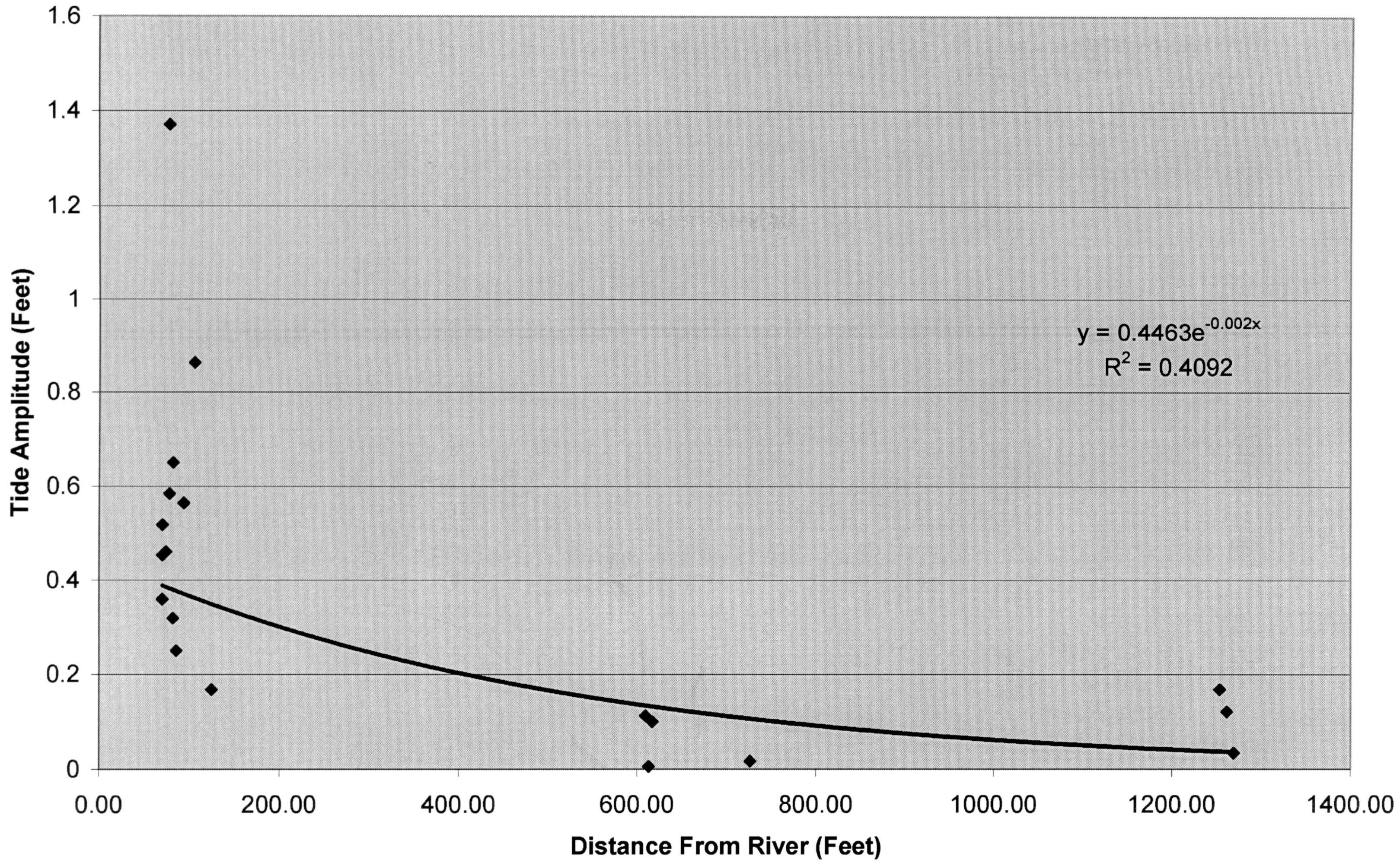
Retardation factors used:

$R_{DCE} = 7.69$

$R_{TCE} = 5.69$

$R_{PCE} = 10.34$

[redacted]



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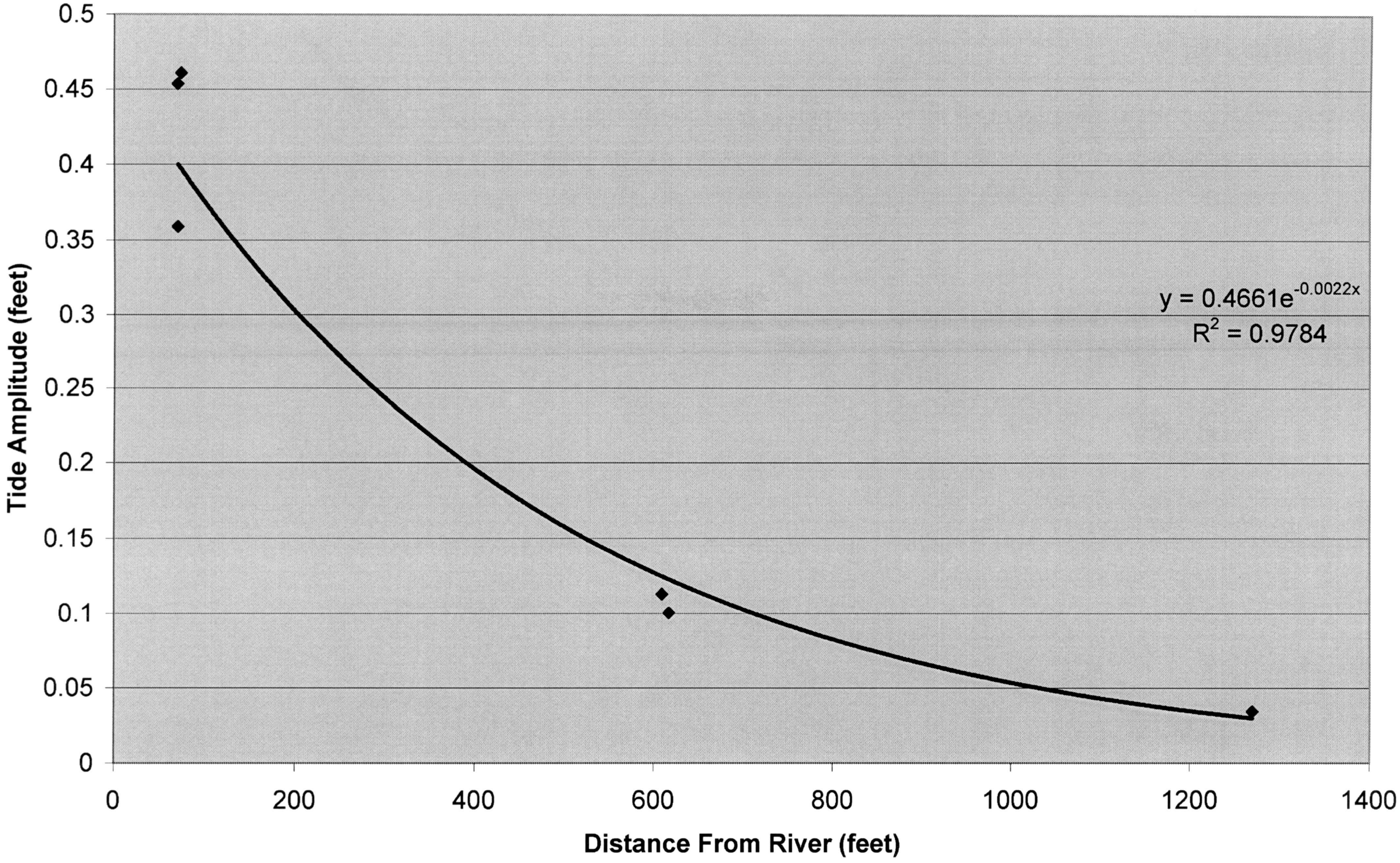
TIDE AMPLITUDE VERSUS DISTANCE FROM RIVER
STRATFORD ARMY ENGINE PLANT
STRATFORD, CONNECTICUT

URS
WAYNE, NEW JERSEY

DR. BY	SCALE AS SHOWN	DWG. NO. 98104059	PROJ. NO. R98104
CK'D. BY DK	DATE OCT 11, 2001	FIG. NO. 2	



K:\Cadd\R98104\98104058.dwg, 10/25/01 01:34:55 PM

TIDE AMPLITUDE VERSUS DISTANCE FROM RIVER
STRATFORD ARMY ENGINE PLANT
STRATFORD, CONNECTICUT

URS
WAYNE, NEW JERSEY

DR. BY	SCALE AS SHOWN	DWG. NO. 98104058	PROJ. NO. R98104
CK'D. BY DK	DATE OCT 11, 2001	FIG. NO. 3	

3.1 OVERVIEW

This section describes the remedial hydrogeological investigation of the facility. The remedial hydrogeological investigation involved reviewing the literature on aquifers in the Stratford area, Geoprobe and microwell groundwater sampling, installing monitoring wells, two rounds of groundwater sampling, long-term tidal study, aquifer tests, and a seepage meter pilot study to evaluate the aquifers underlying the Site. In addition, hydrogeologic investigations conducted by Harding Lawson Associates (HLA) at SAEP are summarized in this section and incorporated into the hydrogeologic conceptual models presented in Sections 7.0 and 8.0. Data collected from C.A Rich and Sound Environmental Solutions in conjunction with the wastewater treatment plant and former lagoons RCRA investigation are also incorporated into the hydrogeologic conceptual models presented in Sections 7.0 and 8.0. A summary of the wastewater treatment plant and former lagoons RCRA investigation is presented in Section 1.2. The HLA hydrogeologic investigation tasks included chromium and VOC focused Geoprobe groundwater sampling events, cone penetrometer survey, and aquifer testing. The combined geologic investigations permitted comprehensive assessment of the physical and chemical geologic system at the Site.

Section 3.2 describes the objectives of the hydrogeological investigation, Section 3.3 reviews the methods and analyses used, and Section 3.4 describes the results obtained. The results of the remedial hydrogeological investigation are discussed in Sections 7.0 and 8.0 and incorporated in the hydrogeologic physical and chemical conceptual models.

The Site has been divided into seven on and off-site areas critical to the determination of exposure pathways in which collection of additional groundwater information will be necessary for a complete understanding of groundwater flow and contaminant transport. This division into seven areas is based on the conceptualization of the overall hydrologic regime at the Site and an evaluation of the relative amount of groundwater information available from an area:

- Area 1 – Northern portion of the Site with potential groundwater flow from the Site to both Frash Pond and intertidal mudflats.
- Area 2 – Eastern portion of the Site with groundwater flow from the Site to the intertidal mudflats.
- Area 3 – West of the Site with groundwater flow onto the Site from the airport
- Area 4 – West of the Site with potential groundwater flow both onto the Site from off-site and, into Frash Pond from off-site, and to Frash Pond from on-site.
- Area 5 – Area in central portion of the Site (primarily Buildings B-2 and B-3) where no information on groundwater flow or quantity is available.
- Area 6 – Southern portion of the Site where groundwater flow is both to the intertidal mudflats and the drainage channel
- Area 7 – Area south of the Site

These areas are outlined in Figure 1-2.

3.2 OBJECTIVES

The objectives of the remedial hydrogeologic investigation were to 1) better define and quantify the groundwater flow, horizontal and vertical gradients and flow quantities in areas where data gaps exist, and 2) define the nature and extent of groundwater contamination. Understanding the groundwater flow regime will allow for an identification of receptors through an assessment of the potential for contamination originating at the Site to be conveyed to off-site locations and to surface water bodies by way of the groundwater. Understanding the groundwater quality will allow for an evaluation of risk to identified potential receptors.

Specific goals were to determine:

1. The extent that contamination originating at the Site has impacted on-site groundwater
2. Whether contamination has migrated or may migrate to off-site locations via groundwater
3. The extent of groundwater impacts
4. What potential remedies may be effective in preventing or eliminating further contamination

In addition, an understanding of the groundwater interactions with the soil, sediment, and the surface water will identify or eliminate other potential exposure pathways.

3.3 METHODS AND ANALYSES

The hydrogeological investigation involved six main activities by URSGWC:

1. Direct push groundwater sampling
2. Monitoring well installations
3. Two rounds of groundwater sampling
4. Tidal study
5. Slug testing
6. Seepage meter pilot study

In addition, a summary of HLA's groundwater investigation is included in this section and intergrated into the Site's physical and chemical conceptual models presented later in the report.

3.3.1 Direct Push Groundwater Sampling

A total of 39 direct push borings were advanced by URSGWC during the remedial investigation for initial characterization of groundwater. The direct push study was used for VOC groundwater screening and placement of permanent monitoring wells (DP5-5 through DP5-9 and DP6-1 through DP6-4), assessment of contaminant impacts to the tidal flat areas (DP2-1 through DP2-4), groundwater screening of a potential chromium hotspot at Building 3 (DP5-1 through DP5-11) and groundwater screening of on-site and off-site VOC contamination impacting Area 3

(DP3-1 through DP3-13). The 39 direct push boring were installed in four areas (Areas 2, 3, 5, and 6) on the facility. Figure 3-1 depicts the locations of all direct push borings.

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

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

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

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

During both direct push groundwater sampling events, a mobile laboratory was used on-site to analyze the VOC samples from Areas 2, 3, 5, and 6. A gas chromatograph was utilized to analyze the groundwater samples. In addition, samples collected from the Building 3 area (DP5-1 through DP-10) were analyzed in the field using field screening kits from the Hach Company for total chromium, hexavalent chromium, and ferrous iron. For QA/QC purposes, split samples for the USACE were collected from 10% of the groundwater samples and submitted to Quanterra Environmental Services of West Sacramento, California for analysis.

The results for the direct push groundwater samples are presented in Section 3.4.1.

3.3.2 Monitoring Well Installation

A total of 24 monitoring wells were installed by URSGWC during the remedial investigation in order to better define groundwater conditions at SAEP in areas where wells were not present. The 24 monitoring wells (8 shallow wells, 10 intermediate wells, and 6 deep wells) were installed at 15 locations in five areas (Areas 1, 2, 3, 5, and 6) on the facility. The 24 single-cased monitoring wells were installed in the overburden soils to determine groundwater elevations and to obtain groundwater samples for field and laboratory analyses. Figure 3-2 depicts the locations of all newly installed monitoring wells, as well as, existing URSGWC monitoring wells.

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

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

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

Boreholes for monitoring wells WC2-2D, WC2-3D, and WC5-1D were cored five feet into the bedrock. The cores were collected for analysis and classification. The methodologies of the rock coring are detailed further in Section 2.4.2. The interval from which the core was collected was grouted to the top of the bedrock prior to installation of the monitoring well.

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

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

A summary of well constructions details for all monitoring wells included in the RI study are presented on Table 3-1. Well construction logs detailing the newly installed wells are included in Appendix G of this report. Well development logs for the newly installed monitoring wells are also included in Appendix G. The monitoring well installation SOPs are outlined in the RIWP dated October 1998.

3.3.3 Monitoring Well Sampling

Two rounds of monitoring well groundwater sampling were conducted in June/July 1999 and November 1999 during the remedial investigation using low flow rate purging and sampling techniques. Groundwater samples were collected in order to determine the current constituents, concentration, and the extent and migration of the contaminants in the groundwater at the facility. The groundwater sampling program included newly and previously installed monitoring wells throughout the facility in Areas 1, 2, 3, 4, 5, and 6 of the Site. The monitoring wells are depicted on Figure 3-2.

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

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

3.3.3.1 Round One

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

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

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

The results for the first round of monitoring well groundwater samples are presented in Section 3.4.2.

3.3.3.2 Round Two

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

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

The results for the second round of monitoring well groundwater samples are presented in Section 3.4.2.

3.3.4 Tidal Study

A tidal study consisting of long-term groundwater level monitoring of 34 monitoring wells and three surface water stilling wells was conducted from July 20 through August 18, 1999. The purpose of the tidal study was to compare tidal variations in surface waters versus groundwater, determine the tidal fluctuations in groundwater across the Site, and determine mean groundwater elevations and hydraulic gradients at SAEP. A total of seventeen monitoring wells were monitored for a one-week period. The remaining seventeen monitoring wells and three stilling wells were monitored for a four-week period in order to gauge the full range of changes during a tidal cycle. The locations of the monitoring wells included in the tidal study are presented on Figure 3-2.

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

The results of the tidal study are presented in Section 3.4.3.

3.3.5 Slug Tests

Slug tests were conducted from November 15 through November 19, 1999 on a total of 34 monitoring wells. The monitoring wells included each of the newly installed monitoring wells including HLA's monitoring wells installed on the Causeway (MWCD-99-01A, MWCD-99-01B, MWCD-99-02A, and MWCD-02B) as well as six previously existing monitoring wells (LW-3S, LW-5S, LW-2S, PZ-5D, PZ-9D and WC9-D2) as requested by the USACE. Slug tests are an in-situ measurement of hydraulic conductivity in the area surrounding the well screen and filter pack. Hydraulic conductivity is a measurement of a porous medium's ability to transmit water with units of distance divided by time.

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

The results of the slug tests are presented in Section 3.4.4.

3.3.6 Seepage Meter Pilot Study

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

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

The results of the seepage meter pilot study are presented in Section 3.4.5.

3.3.7 Location and Elevation Survey

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

3.3.8 Harding Lawson Associates Investigations

This section briefly describes the methods and analyses used during the three investigations conducted by HLA at SAEP from January 1998 through February 2000. The investigations include chromium-focused and chlorinated VOC groundwater sampling through a contract with AlliedSignal from January 1998 through June 1999, pre-design activities conducted by FW/HLA from August to December 1999 under the OU2 NCRA, and pre-design activities conducted by FW/HLA on the Causeway and Dike from September to October 1999 through a TERC with the USACE. The HLA investigations are presented in more detail in their Draft PDIR for the OU2 NCRA and EE/CA for the Causeway and Dike both dated March 2000.

3.3.8.1 Chromium-Focused Investigations

The following subsections present summaries of the investigations conducted by HLA at SAEP from January 1999 through June 1999 through a contract with AlliedSignal. These investigations included chromium-focused soil and groundwater sampling with a GeoProbe, VOC groundwater sampling with a GeoProbe, and VOC groundwater sampling with a cone penetrometer.

Geoprobe Investigation

In January and February of 1999, HLA collected 182 groundwater samples from 32 sampling locations (WP-99-01 through WP-99-32) in and around the Chromium Plating Facility using a GeoProbe. Figure 2-2 depicts the HLA's Geoprobe sampling locations

The "drag" technique was employed for collection of multiple groundwater samples within each boring. This technique involved driving the GeoProbe rods to total depth at the exploration, and pushing off the disposable stainless steel drive point. The 4-foot, 0.010-inch slotted stainless steel screen was then exposed by retracting the outer rods. Approximately 3 to 5 gallons of groundwater was then purged from the sample interval with a peristaltic pump, followed by sample collection. The screen and rods were then retracted to the next sample interval, followed by purging and sample collection. This process was repeated until sampling from the required intervals was completed.

Groundwater samples were analyzed on-site by HLA for hexavalent chromium and ferrous iron using Hach™ test kits. The following Hach™ test kits were employed for analysis of groundwater samples during the chromium-focused investigations:

Hach™ Catalog #2228-00:	low range (0-1.5 mg/L) Total CR, Hex CR, Tri CR
Hach™ Catalog #2227-02:	med-high range (0-1000 mg/L) Hex CR
Hach™ Catalog #22672-00:	Ferrous Iron (0-10 mg/L)

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

The results of the chromium Geoprobe investigation are presented in Section 3.4.6.

Monitoring Well/Piezometer Installation and Sampling

HLA installed 13 piezometers for water level measurement and analytical sampling in the vicinity of the Chromium Plating Facility Investigation during the chromium-focused investigation in January and February 1999. Locations of these wells/piezometers are presented in Figure 2-2.

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

Groundwater samples were collected from select wells and tested on-site for hexavalent chromium and ferrous iron using Hach™ test kits. Due to the high concentrations of hexavalent chromium present in groundwater, many of the samples required dilution with deionized water prior to analysis by the Hach™ test kits. Note that groundwater samples from PZ-99-01A through PZ-99-05A were collected from the GeoProbe rods during advancement of the borings, not from the completed piezometer.

The results of the monitoring well/piezometer sampling are presented in Section 3.4.6.

3.3.8.2 Chlorinated VOC Investigations - Groundwater

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

GeoProbe Investigations

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

The "drag" technique was employed for collection of multiple groundwater samples within each boring. This technique involved driving the GeoProbe rods to total depth at the exploration, and pushing off the disposable stainless steel drive point. The 4-foot, 0.010-inch slotted stainless steel screen was then exposed by retracting the outer rods. Approximately 3 to 5 gallons of groundwater was then purged from the sample interval with a peristaltic pump, followed by sample collection. The screen and rods were then retracted to the next sample interval, followed by purging and sample collection. This process was repeated until sampling from the required intervals was completed.

HLA collected 126 groundwater samples from the 40 boring locations. A mobile laboratory was used on-site to analyze the groundwater samples for select VOCs (PCE, TCE, cis-1,2-DCE and trans-1,2-DCE, vinyl chloride, 1,1-DCE, and 1,1,1-TCA). Groundwater samples were analyzed by modified USEPA Method 8021B, using a HP 5890 GC with an electrolytic conductivity detector (ELCD). In addition, 10 groundwater samples were submitted for confirmation analysis at an off-site laboratory for VOCs.

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

Off-site confirmatory water samples were analyzed for VOCs by USEPA Method 8260B at Averill Environmental Laboratories to provide definitive data for confirmation of the on-site laboratory screening data set (USEPA, 1993). Samples were split in the field at the time of sample collection and submitted to both the on-site and off-site laboratories.

The results of the VOC Geoprobe investigation are presented in Section 3.4.6.

Monitoring Well Sampling.

Nine monitoring wells were sampled by HLA around the chrome plating facility to assess the potential for groundwater contamination in the area resulting from historic chrome plating activities. The samples collected from all but one piezometer were analyzed for selected volatile organic compounds including 1,1,1-TCA, 1,1-DCE, PCE, TCE, and vinyl chloride. Samples collected from piezometer PZ-Pilot-03 were analyzed for TAL metals.

The results of the VOC monitoring well sampling are presented in Section 3.4.6.

Cone Penetrometer Investigations.

In April and May 1999, HLA conducted additional VOC groundwater investigations using a cone penetrometer to access depths not attainable by the GeoProbe equipment. Nineteen total cone penetrometer (CP-99-01 through CP 99-18) explorations were completed for in-situ stratigraphic and hydrogeologic data acquisition and collection of groundwater samples. The cone penetrometer locations are depicted on Figure 2-2.

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

The PCPT technology employs an electronically tipped, stainless steel cone (probe), which, using hydraulic rams, is pushed smoothly and vertically into the overburden. Data was recorded continuously using a data logger and computer as the cone was advanced into the ground. The recorded data included tip (cone) resistance, sleeve friction, instantaneous pore pressure, and soil electrical conductivity. The combination of tip resistance, sleeve friction, and pore pressure data provided continuous stratigraphic logs for geologic and hydrogeologic interpretation and assessment. Tip resistance was highest in the gravelly sand strata at SAEP, lower and more variable in the fine to medium sand zone, and consistently very low in the silt zone. Sleeve friction data were generally similar in the gravelly sand and fine to medium sand zones, with a marked change in the silt.

Pore pressure was monitored continuously during the PCPT soundings using a transducer and porous element located on the cone tip. Pore pressure is first registered at the water table when the pen plotter departs from zero. Pore pressure increases steadily with depth below the water table, as a function of hydrostatic pressure. As the cone is pushed into the soil, excess pressures are generated which, in low permeability materials, cannot rapidly dissipate. At SAEP, excess pore pressure peaks correlated with decreases in cone tip resistance, which indicated that silty or clay strata were being penetrated.

In addition to the basic data (i.e., tip resistance, sleeve friction, and pore pressure), these data are combined into a line plot showing the ratio of the friction sleeve reading to tip resistance. Increases in this ratio result from decreases in tip resistance, often coupled with increases in sleeve friction, indicative of silty or clay-rich transition zone strata.

Soil electrical conductivity (EC) measurements were collected in each exploration. Soil EC is controlled by the conductance of both the soil particle and the soil pore fluids. Low EC values in saturated soils can be indicative of anomalous geochemistry. In particular, depressed EC zones immediately at the water table may be associated with floating (Light Non-Aqueous Phase Liquids [LNAPLs]) compounds. Very low EC zones at interfaces between aquifers and aquitards may be associated with either floating (LNAPL) or sinking (Dense Non-Aqueous Phase Liquids [DNAPLs]) compounds. Gravel interference must be considered when evaluating depressed EC zones in saturated soils. Elevated EC values in saturated soils can be due to increased soil clay content or to increased dissolved salts in the ground water. Increased clay contents are evaluated based on the CPTU-EC piezometric data and soil type information.

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

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

The results of the cone penetrometer sampling are presented in Section 3.4.6.

3.3.8.3 Operable Unit 2 NCRA Investigations

The following subsections present discussions of FW/HLA pre-design activities conducted under the OU2 NCRA through the TERC contract with the USACE. Activities included piezometer and extraction well installation and aquifer testing.

Piezometers and Extraction Well Installations

In August 1999, 7 piezometers and one extraction well were installed in the vicinity of the Chromium Plating Facility at SAEP. The objectives of these explorations were to: 1) obtain groundwater chemical data, 2) collect groundwater samples for bench-scale testing of hexavalent chromium and VOC treatment technologies, and 3) provide a pumping well and water level observation points for aquifer testing of the subsurface hydraulic conditions. These activities are discussed in the following subsection.

Piezometers

A total of seven piezometers were installed from August 8, 1999 through August 20, 1999 at three separate locations (soil borings PZ-99-01 through PZ-99-03) in the vicinity of the Chromium Plating Facility. Two sets of three nested piezometers (PZ-99-01 A, B, C and PZ-99-02 A, B, C), consisting of a shallow, mid-depth and deeper piezometer, were set in two borings, with a single shallow piezometer being placed at PZ-99-03. The borings were drilled using traditional HSA techniques, and using split-spoons for sample collection. The locations of HLA OU2 NCRA piezometers are depicted on Figure 2-2.

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

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

Extraction Well

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

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

Monitoring Wells

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

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

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

The four Causeway monitoring wells were sampled during URSGWC second round of groundwater sampling. The results of the Causeway monitoring wells are presented in Section 3.4.6.

Aquifer Testing

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

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

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

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

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

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

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

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

The results of HLA's aquifer testing are presented in Section 3.4.6.

3.4 RESULTS

3.4.1 Direct Push Investigation

The direct push study was performed for VOC groundwater screening and placement of permanent monitoring wells (DP2-5 through DP2-9, DP5-12, and DP6-1 through DP6-4), assessment of contaminant impacts to the tidal flat areas (DP2-1 through DP2-4), groundwater screening of a potential chromium hotspot at Building 3 (DP5-1 through DP5-11) and groundwater screening of on-site and off-site VOC contamination impacting Area 3 (DP3-1 through DP3-13). All samples results are compared to the Connecticut Remediation Standard Regulations. Samples collected from depths of 0 to 15 ft bgs are compared to CTDEP ICVC for Groundwater and all results are compared to CTDEP SWPC for Substances in Groundwater. The CTDEP SWPC do not include a criterion for total chromium, therefore the criterion for trivalent chromium (1,200 ppb) was used to compare the analytical results.

3.4.1.1 Along the Berm – Area 2

Four direct push groundwater samples (DP2-1 through DP2-4) were advanced along the berm to assess the potential of contaminant impacts to the tidal flat areas. These samples were analyzed for TCL VOCs, SVOCs, PCBs, TAL Metals, and CN. The following is a summary of analytical results from direct push groundwater samples collected along the berm.

DP2-1: Five groundwater samples were collected at location DP2-1 from 7 to 60 ft bgs. Benzene was detected above the SWPC of 710 parts per billion (ppb) at 15 ft bgs with a concentration of 760 ppb. Phenanthrene was detected at 7 ft exceeding the SWPC of 0.077 ppb with a concentration of 15.3 ppb. Arsenic was detected above the SWPC of 4 ppb at concentrations ranging from 5.56 ppb to 21.3 ppb at the depths of 7 to 15 ft and then again at 50 to 60 ft. Lead was detected above the SWPC of 13 ppb at 50 ft with a concentration of 37.5 ppb. Zinc was detected above the SWPC of 123 ppb at 7 ft with a concentration of 236 ppb.

DP2-2: Groundwater samples were collected at location DP2-2 at ten-foot intervals from 8 to 60 ft bgs. Samples collected at 8 ft exceeded the ICVC for 1,1-DCE (criterion: 6 ppb) and vinyl chloride (criterion: 2 ppb) with concentrations of 220 ppb and 20 ppb, respectively. The concentration of 1,1-DCE also exceeds the SWPC of 96 ppb. Arsenic was detected above the SWPC of 4 ppb at 30 ft, 40 ft and 60 ft with concentrations of 6.46 ppb, 6.1 ppb and 13.6 ppb. Copper was detected above the SWPC of 48 ppb at 40 ft with a concentration of 53 ppb. Zinc was detected above the SWPC of 123 ppb at 40 ft and 50 ft at concentrations of 183 ppb and 180 ppb, respectively.

DP2-3: Five groundwater samples were collected at location DP2-3 from 8 ft to 60 ft bgs. Arsenic was detected above the SWPC of 4 ppb at 15 ft and 60 ft at concentrations of 13.7 ppb and 19 ppb, respectively. Total chromium was detected above the SWPC for trivalent chromium (1200 ppb) at 60 ft with a concentration of 1,370 ppb. Copper was detected above the SWPC of 48 ppb at 8 ft with a concentration of 54.5 ppb. Lead was detected above the SWPC of 13 ppb at 8 ft and 60 ft with concentrations of 21.1 ppb and 43.6 ppb, respectively. Zinc was detected above the SWPC of 123 ppb at 60 ft with a concentration of 326 ppb.

DP2-4: Groundwater samples were collected at location DP2-4 at ten-foot intervals from 8 ft to 55 ft bgs. Samples collected from the intervals between 25 ft and 55 ft exceeded the SWPC for both PCE (criterion: 88 ppb) and arsenic (criterion: 4 ppb) with concentrations ranging from 95 ppb to 410 ppb and 6.35 ppb to 51 ppb, respectively. Samples collected from the intervals between 35 ft and 55 ft bgs exceed the SWPC for copper (criterion: 48 ppb) with concentrations ranging from and 51.3 ppb to 723 ppb. Lead was detected above the SWPC of 13 ppb at 35 ft and 45 ft bgs at concentrations of 98.6 ppb and 56.7 ppb, respectively.

Sample results from direct push borings along the berm are presented in Table 3-2 and on Figure 3-1.

3.4.1.2 Southern End of the Property – Monitoring Well Screening

Eleven direct push locations (DP2-5 through DP2-9, DP5-12, and DP6-1 through DP6-4), were advanced and sampled at the southern end of the property for VOC groundwater screening and placement of permanent monitoring wells monitoring well screening basis. All samples were analyzed for VOCs utilizing an on-site laboratory. The following is a summary of analytical results from direct push groundwater samples collected from the southern end of the property.

DP2-5: Groundwater samples were collected at location DP2-5 at ten-foot intervals from 8 ft to 50 ft bgs. 1,1-DCE was detected above the ICVC of 6 ppb at 8 ft with a concentration of 65 ppb. 1,1-DCE was detected above the SWPC of 96 ppb, at 20 ft and 40 ft bgs with concentrations of 335 ppb and 265 ppb, respectively. Vinyl chloride was detected above the ICVC of 2 ppb at 8 ft with a concentration of 130 ppb.

DP2-6: Groundwater samples were collected at location DP2-6 at ten-foot intervals from 8 ft to 40 ft bgs. The 8 ft sample results indicated concentrations of 1,1-DCE (45 ppb) and vinyl chloride (10 ppb) above the ICVC of 6 ppb and 2 ppb respectively. 1,1-DCE was detected above the SWPC of 96 ppb at intervals between 20 ft and 40 ft bgs at concentrations ranging from 210 ppb to 320 ppb.

DP2-7: Groundwater samples were collected at location DP2-7 at ten-foot intervals from 8 ft to 60 ft bgs. The 8 ft sample results indicated concentrations of 1,1-DCE (9 ppb) and vinyl chloride (62 ppb) above the ICVC of 6 ppb and 2 ppb, respectively. 1,1-DCE was detected above the SWPC of 96 ppb at 30 ft and 40 ft with concentrations of 255 ppb and 210 ppb, respectively. PCE was detected above the SWPC of 88 ppb at the intervals between 20 ft and 50 ft bgs with concentrations ranging from 275 ppb to 1,600 ppb. TCE was detected above the SWPC of 2,340 ppb at the intervals between 30 ft and 50 ft with concentrations ranging from 3,100 ppb to 4,100 ppb.

DP2-8: Groundwater samples were collected at location DP2-8 at ten-foot intervals from 8 ft to 60 ft bgs. 1,1-DCE was detected above the criterion of 96 ppb at 20 ft and 30 ft with concentrations of 175 ppb and 155 ppb, respectively. PCE was detected above the SWPC of 88 ppb between the intervals of 20 ft and 60 ft with concentrations ranging from 240 ppb to 750 ppb. TCE was detected above the SWPC of 2,340 ppb at 30 ft with a concentration of 2,400 ppb. Vinyl chloride was detected above the ICVC of 2 ppb at 8 ft with a concentration of 21 ppb.

DP2-9: Groundwater samples were collected at location DP2-9 at 8 ft and 23 ft bgs. The 8 ft sample results indicated concentrations of 1,1-DCE (70 ppb) and vinyl chloride (30 ppb) above the ICVC of 6 ppb and 2 ppb respectively. The 23 ft sample results indicated concentrations of 1,1-DCE (410 ppb) and PCE (165 ppb) above the SWPC of 96 ppb and 88 ppb, respectively.

DP5-12: Groundwater samples were collected at location DP5-12 at ten-foot intervals from 17 ft to 57 ft bgs. PCE was detected above the SWPC of 88 ppb in the sample collected between 47 ft to 50 ft bgs with a concentration of 120 ppb.

DP6-1: Groundwater samples were collected at location DP6-1 at ten-foot intervals from 8 ft to 60 ft bgs. No analytes were detected above the ICVC or the SWPC.

DP6-2: Groundwater samples were collected at location DP6-2 at ten-foot intervals from 8 ft to 60 ft bgs. PCE was detected above the SWPC of 88 ppb at 40 ft with a concentration of 325 ppb.

DP6-3: Groundwater samples were collected at location DP6-3 at ten-foot intervals from 8 ft to 60 ft bgs. 1,1-DCE was detected above the ICVC of 6 ppb and the SWPC of 96 ppb at 8 ft with a concentration of 3,900 ppb. 1,1-DCE was also detected above the SWPC of 96 ppb at 50 ft bgs with a concentrations of 110 ppb. Vinyl chloride was detected above the ICVC of 2 ppb at 8 ft with a concentration of 970 ppb.

DP6-4: Groundwater samples were collected at location DP6-4 at ten-foot intervals from 8 ft to 60 ft bgs. 1,1-DCE was detected above the ICVC of 6 ppb at 8 ft with a concentration of 12 ppb and above the SWPC of 96 ppb at 30 ft with a concentration of 124 ppb.

Sample results from direct push borings in the southern end of the property are presented in Table 3-2 and on Figure 3-1.

3.4.1.3 West Parking Lot- Area 3

Fourteen direct push locations (DP3-1 through DP-3-14) were installed and sampled in the west parking lot. The west parking lot investigation was performed as an initial groundwater screening of on-site and off-site VOC contamination impacting Area 3. All samples were analyzed for VOCs utilizing an on-site laboratory. The following is a summary of analytical results from direct push boring located in the west parking lot.

DP3-1: Groundwater samples were collected at location DP3-1 from 8 ft and 23 ft bgs. No analyses were detected above the ICVC or the SWPC.

DP3-2: Three groundwater samples were collected at location DP3-2 between 8 ft to 35 ft bgs. 1,1-DCE was detected above the SWPC of 96 ppb at 35 ft with a concentration of 7,800 ppb. The 8 ft sample results indicate concentrations of PCE (4300 ppb) and vinyl chloride (240 ppb) above the ICVC of 3820 ppb and 2 ppb, respectively. The PCE concentration detected from the 8 ft sample also exceeds the SWPC of 88 ppb. PCE was detected above the SWPC of 88 ppb at 21 and 35 ft bgs with concentrations of 560 ppb and 2400 ppb, respectively.

DP3-3: Groundwater samples were collected at location DP3-3 at 8 ft and 24 ft bgs. PCE was detected above the SWPC of 88 ppb at 24 ft with a concentration of 275 ppb.

DP3-4: Groundwater samples were collected at location DP3-4 at ten-foot intervals from 21 ft to 43 ft bgs. 1,1-DCE was detected above the SWPC of 96 ppb at 33 ft with a concentration of 240 ppb. PCE was detected above the SWPC of 88 ppb at 21 ft and 33 ft with concentrations of 710 ppb and 1,200 ppb, respectively.

DP3-5: Groundwater samples were collected at location DP3-5 at 8 ft and 20 ft bgs. PCE was detected above the SWPC of 88 ppb at 20 ft with a concentration of 110 ppb.

DP3-6: Groundwater samples were collected at location DP3-6 at ten-foot intervals from 7 ft to 46 ft bgs. Samples collected from the depth of 36 ft exceeded the SWPC for 1,1-DCE (criterion: 96 ppb) and PCE (criterion: 88 ppb) with concentrations of 750 ppb and 428 ppb, respectively.

DP3-7: Groundwater samples were collected at location DP3-7 at two intervals from 17 ft to 20 ft and 27 ft to 30 ft bgs. PCE was detected above the criterion of 88 ppb at the interval of 27 ft to 30 ft with a concentration of 600 ppb.

DP3-8: Groundwater samples were collected at location DP3-8 at ten-foot intervals from 17 ft to 60 ft bgs. PCE was detected above the SWPC of 88 ppb at the two intervals between 17 ft and 30 ft with concentrations of 90 ppb and 170 ppb, respectively. TCE was detected above the SWPC of 2,340 ppb at the interval between 27 ft and 30 ft with a concentration of 16,000 ppb.

DP3-9: Groundwater samples were collected at location DP3-9 at ten-foot intervals from 17 ft to 60 ft bgs. Samples collected from the two intervals between 17 ft and 30 ft exceed the SWPC for both PCE (criterion: 88 ppb) and TCE (criterion: 2,340 ppb) with concentrations of 150 ppb and 260 ppb for PCE and 4,700 ppb and 7,000 ppb for TCE.

DP3-10: Groundwater samples were collected at location DP3-10 at ten-foot intervals from 17 ft to 60 ft bgs. TCE was detected above the SWPC of 2,340 ppb at the interval between 37 ft and 40 ft with a concentration of 2,500 ppb.

DP3-11: Groundwater samples were collected at location DP3-11 at ten-foot intervals from 17 ft to 60 ft bgs. TCE was detected above the SWPC of 2,340 ppb at the two intervals between 27 ft and 40 ft with concentrations of 2,600 ppb and 5,300 ppb.

DP3-12: Groundwater samples were collected at location DP3-12 at ten-foot intervals from 17 ft to 57 ft bgs. No analyses were detected above the ICVC or the SWPC at either location.

DP3-13: Groundwater samples were collected at location DP3-13 at ten-foot intervals from 17 ft to 60 ft bgs. No analyses were detected above the ICVC or the SWPC at either location.

DP3-14: Groundwater samples were collected at location DP3-14 at two intervals between 37 ft to 47 ft bgs. 1,1-DCE was detected above the SWPC of 96 ppb at the interval from 37 ft to 40 ft with a concentration of 290 ppb.

Sample results from direct push borings in the west parking lot are presented in Table 3-2 and on Figure 3-1.

3.4.1.4 Building 3 – Area 5

Eleven direct push locations (DP5-1 through DP5-11) were installed and sampled around Building 3. The Building 3 investigation was conducted to perform groundwater screening of a potential chromium hotspot at Building 3. All samples were analyzed for VOCs, total chromium, hexavalent chromium and ferrous iron. All of the inorganic results were below the SWPC detection limits and therefore will not be reported in this section. The following is a summary of analytical results from direct push groundwater samples collected in the vicinity of Building 3.

DP5-1: Groundwater samples were collected at location DP5-1 at ten-foot intervals from 8 ft to 60 ft bgs. PCE was detected above the SWPC of 88 ppb at 8 ft, 31 ft and 60 ft with concentrations of 244 ppb, 102 ppb and 96 ppb, respectively.

DP5-2: Groundwater samples were collected at location DP5-2 at ten-foot intervals from 8 ft to 30 ft bgs. PCE was detected above the SWPC of 88 ppb at 20 ft and 30 ft bgs with concentrations of 140 ppb and 450 ppb, respectively.

DP5-3: Groundwater samples were collected at location DP-5-3 at ten-foot intervals from 10 ft to 50 ft bgs. PCE was detected above the SWPC of 88 ppb at 30 ft and 40 ft with concentrations of 160 ppb and 120 ppb, respectively. Vinyl chloride was detected above the ICVC of 2 ppb at 10 ft with a concentration of 24 ppb.

DP5-4: Groundwater samples were collected at location DP5-4 at ten-foot intervals from 30 ft to 50 ft bgs. 1,1-DCE was detected above the SWPC of 96 ppb at 40 ft with a concentration of 165 ppb. PCE was detected above the SWPC of 88 ppb at 30 ft with a concentration of 410 ppb.

DP5-5: Groundwater samples were collected at location DP5-5 at ten-foot intervals from 10 ft to 60 ft bgs. 1,1-DCE was detected above the SWPC of 96 ppb at 40 ft with a concentration of 175 ppb. PCE was detected above the SWPC of 88 ppb at 40 ft and 50 ft with concentrations of 150 ppb and 100 ppb, respectively.

DP5-6: Groundwater samples were collected at location DP5-6 at ten-foot intervals from 8 ft to 60 ft bgs. PCE was detected above the SWPC of 88 ppb at 60 ft with a concentration of 350 ppb.

DP5-7: Groundwater samples were collected at location DP5-7 at ten-foot intervals from 8 ft to 60 ft bgs. PCE was detected above the SWPC of 88 ppb at 30 ft and 40 ft with concentrations of 125 ppb and 340 ppb, respectively.

DP5-8: Groundwater samples were collected at location DP5-8 at ten-foot intervals from 8 ft to 60 ft bgs. PCE was detected above the SWPC of 88 ppb at 40 ft with a concentration of 180 ppb. Vinyl chloride was detected above the ICVC of 2 ppb at 8 ft with a concentration of 40 ppb.

DP5-9: Groundwater samples were collected at location DP5-9 at ten-foot intervals from 8 ft to 60 ft bgs. PCE was detected above the SWPC of 88 ppb at 40 ft with a concentration of 215 ppb.

DP5-10: Groundwater samples were collected at location DP5-10 at ten-foot intervals from 8 ft to 60 ft bgs. PCE was detected above the SWPC of 88 ppb at 40 ft with a concentration of 165 ppb.

DP5-11: Groundwater samples were collected at location DP5-11 at ten-foot intervals from 8 ft to 60 ft bgs. PCE was detected above the SWPC of 88 ppb at 40 ft bgs with a concentration of 88 ppb.

Sample results from direct push borings in the vicinity of Building 3 are presented in Table 3-2 and on Figure 3-1. A table containing the analytical results for all direct push groundwater samples is included in Appendix K.

3.4.2 Monitoring Well Sampling

3.4.2.1 Round One

In June through July 1999, 42 monitoring wells were sampled and analyzed for VOCs, SVOCs, TAL metals, PCBs, chloride and CN. Ten of the 42 monitoring wells were additionally analyzed for Remedial Design Parameters In addition, samples collected from 10 monitoring wells (WC5-3S, PZ-9D, WC-10S, WC-12S, WC2-3D, WC2-1D, WC2-2I, WC5-1s, WC2-2D and PZ-13D) were analyzed for design-related parameters. The design parameters consisted of pH, conductivity, salinity, total and dissolved manganese and iron, TOC, chemical oxygen demand, biological oxygen demand, Eh, sulfate, sulfide, nitrate, nitrite, ammonia, alkalinity as bicarbonate, dissolved oxygen, nitrogen, chlorides, phosphates. All sample results were compared to the Connecticut Remediation Standard Regulations. Samples collected from depths of 0 to 15 ft bgs were compared to CTDEP ICVC for Groundwater and all results were compared to CTDEP SWPC for Substances in Groundwater. All of the SVOC results were below the SWPC detection limits and are therefore not reported in this section. The following is a summary of the analytical results.

Volatile Organic Results

1,1-Dichloroethylene (1,1-DCE) was detected in 22 of the 42 monitoring well samples ranging from 0.59 ppb to 3,700 ppb. Of these 22 samples, six samples exceed the SWPC of 96 ppb with concentrations ranging from 140 ppb in PZ-1D to 3,700 ppb in PZ-8D. The ICVC of 6 ppb was exceeded in two samples (WC-4S and WC-12S) with results of 730 ppb and 960 ppb, respectively. The purge water from monitoring well PZ-1D was observed to exhibit an oily sheen. Three additional samples exceed the ICVC of 6 ppb with concentrations of 6.8 ppb, 52 ppb and 57 ppb in wells WC-19S, WC-1S and WC5-1S, respectively.

Tetrachloroethylene (PCE) was detected in 21 of the 42 monitoring well samples ranging from 0.28 ppb to 850 ppb. Of these 21 samples, seven samples exceed the SWPC of 88 ppb with concentrations ranging from 99 ppb in WC2-3D to 850 ppb in PZ-1D.

Trichloroethylene (TCE) was detected in 27 of the 42 monitoring well samples ranging from 0.7 ppb to 6,800 ppb. Of these 27 samples, three samples exceed the SWPC of 2,340 ppb with concentrations of 2,400 ppb, 3,100 ppb and 6,800 ppb in wells PZ-1D, WC2-3D and PZ-8D, respectively.

Vinyl chloride was detected in twelve of the 42 monitoring well samples ranging from 2.1 ppb to 3,000 ppb. None of the samples exceed the SWPC of 15,750 ppb, however five samples collected from shallow monitoring wells exceed the ICVC of 2 ppb. These samples range from 2.1 ppb in WC2-3S to 3,000 ppb in WC-5S. An oily sheen was observed on the purge water from monitoring well WC-5S.

A summary of organic results is presented in Table 3-3 and on Figure 3-3. A table containing the analytical results for all monitoring well samples during the first round of groundwater sampling is included in Appendix L.

Inorganic Results

Arsenic was detected in fifteen of the 42 monitoring well samples. All fifteen of the samples exceed the SWPC for arsenic of 4 ppb with concentrations ranging from 4.7 ppb in PZ-1D to 50 ppb in WC-9D2.

Total chromium was analyzed for in all monitoring well samples. The CTDEP does not have a criterion for total chromium, therefore the CTDEP SWPC for hexavalent chromium of 110 ppb was used for comparison. Chromium was detected in five samples with concentrations ranging from 6 ppb to 350 ppb. Of these sample results, one concentration, 350 ppb found in WC-12S was above the hexavalent chromium CTDEP SWPC.

Lead was detected in three of the 42 monitoring well samples with concentrations of 1.6 ppb, 8.6 ppb and 19 ppb in wells PZ-1D, PZ-8D and PZ-4D, respectively. Only the sample result from PZ-4D was above the SWPC criterion of 13 ppb.

Silver was detected in four of the 42 monitoring well samples with concentrations ranging from 5 ppb to 50 ppb. Only the sample result from WC5-1D was in exceedance of the SWPC criterion of 12 ppb.

Cyanide was detected in three of the 42 monitoring well samples with concentrations of 12 ppb, 13 ppb and 78 ppb in wells WC2-3S, WC-5S and WC-12S, respectively. Only the sample result from WC-12S was in exceedance of the SCPC criterion of 52 ppb.

A summary of the inorganic results is presented in Table 3-4 and on Figure 3-4. A table containing the analytical results for all monitoring well samples during the first round of groundwater sampling is included in Appendix L.

3.4.2.2 Round Two

In November 1999, 51 monitoring wells were sampled during the second round of groundwater sampling. Groundwater samples collected from monitoring wells which were sampled during the first round of groundwater sampling were submitted for analysis of TCL VOCs, SVOCs, and TAL Metals. Samples from monitoring wells, which were not sampled during the first sampling round, and newly installed monitoring wells, were submitted for analysis of TAL VOCs, SVOCs, PCBs, TAL Metals, and CN. All sample results were compared to the Connecticut Remediation Standard Regulations. Samples collected from depths of 0 to 15 ft bgs were compared to CTDEP ICVC for Groundwater and all results were compared to CTDEP SWPC for Substances in Groundwater. All of the SVOC results were below the SWPC detection limits and are therefore not reported in this section. The following is a summary of the analytical results.

Volatile Organic Results

1,1-DCE was detected in 28 of the 51 monitoring well samples ranging from 0.41 ppb to 3,100 ppb. Of these 28 samples, seven samples exceed the CTDEP SWPC of 96 ppb with concentrations ranging from 96 ppb in PZ-1D to 3,100 ppb in PZ-8D. Of these seven sample results, two also exceed the ICVC of 6 ppb. These samples were collected from WC-4S and WC-12S and indicated concentrations of 99 ppb and 700 ppb, respectively. An oily sheen was observed on the purge water from monitoring well PZ-1D. Three samples exceed the ICVC of 6 ppb with concentrations of 10 ppb, 11 ppb and 49 ppb in wells WC-1S, WC-19S and WC-10S, respectively.

PCE was detected in 30 of the 51 monitoring well samples ranging from 0.54 ppb to 740 ppb. Of these 30 samples, nine samples exceed the SWPC of 88 ppb with concentrations ranging from 130 ppb in WC3-2I to 740 ppb in WC-19S.

TCE was detected in 41 of the 51 monitoring well samples ranging from 0.23 ppb to 5,900 ppb. Of these 41 samples, three samples exceed the SWPC of 2,340 ppb with concentrations of 2,500 ppb, 3,700 ppb and 5,900 ppb in wells PZ-1D, WC2-6I, and PZ-8D, respectively.

Vinyl Chloride was detected in 25 of the 51 monitoring well samples ranging from 0.44 ppb to 3,400 ppb. None of the samples exceed the SWPC of 15,750 ppb, however five of the samples collected from shallow monitoring wells exceed the ICVC of 2 ppb. These five sample results range from 7.7 ppb in WC-1S to 64 ppb in WC-4S.

A summary of organic results is presented in Table 3-5 and on Figure 3-5. A table containing the analytical results for all monitoring well samples during the second round of groundwater sampling is included in Appendix M.

Inorganic Results

Arsenic was detected in 16 of the 51 monitoring well samples. All 16 of the samples exceed the SWPC for arsenic of 4 ppb with concentrations ranging from 6 ppb in PZ-16D and WC-5S to 67 ppb in WC2-4I.

Chromium was detected in eleven of the 51 monitoring well samples with concentrations ranging from 6 ppb to 522 ppb. The CTDEP does not have a criterion for total chromium, therefore the CTDEP SWPC for hexavalent chromium of 110 ppb was used for comparison. Two concentrations 160 ppb and 522 ppb were detected in WC2-2D and WC-12S, respectively, and exceed this criterion.

Lead was detected in six of the 51 monitoring well samples with concentrations ranging from 1 ppb to 25 ppb. Of these six samples, two samples were detected above the criterion of 13 ppb with concentrations of 24 ppb and 25 ppb in wells PZ-4D and WC3-1I, respectively.

Zinc was detected in 22 of the 51 monitoring well samples ranging from 7 ppb to 133 ppb. Of these 22 samples, two samples were detected above the criterion of 123 ppb with concentrations of 130 ppb and 133 ppb in wells WC2-2D and WC3-1I.

Cyanide was detected in two samples ranging from 13 ppb to 221 ppb in wells WC-5S and WC-12S, respectively. The sample result from WC-12S (221 ppb) is above the SWPC criterion of 52 ppb.

A summary of the inorganic results is presented in Table 3-6 and on Figure 3-6. A table containing the analytical results for all monitoring well samples during the second round of groundwater sampling is included in Appendix M.

3.4.3 Tidal Study

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

Tidal fluctuations in surface water bodies produce progressive pressure waves in adjacent aquifers. As these pressure waves propagate inland, groundwater levels and hydraulic gradients continuously fluctuate, creating a situation where a single set of water level measurements cannot be used to accurately characterize groundwater flow. At any point where groundwater tidally fluctuates, the magnitude and direction of the hydraulic gradient fluctuates about the mean or regional hydraulic gradient. The net effect of these fluctuations on groundwater flow can be determined using the mean hydraulic gradient which is calculated by filtering the groundwater level measurements to obtain a mean groundwater elevations. Filtering methods using 71 consecutive hourly water level observations to accurately determine the mean level were utilized in the tidal study (Serfes, 1991).

The filtering method detailed in Serfes, 1991, is used to effectively remove all diurnal and semidiurnal lunar and solar harmonics from 71 consecutive hourly observations. Using moving averages it yields a filtered mean level for the median time of the 71 hours. First, a sequence of mean is computed for 24 observations, starting with observation one for the first mean and observation 48 for the last, yielding a total of 48 means. Second, a similar series of means is computed for 24 of those means yielding 25 means. Last, the mean of those 25 means is computed yielding the mean level at hour 36.

The filtering method can be expressed mathematically as:

Let the consecutive hourly water level values be O(1), O(2), O(3), …, O(71):

The first sequence of means (X_i) is

$$X_i = \sum_{K=0}^{23} \frac{O(K+i)}{24} \quad \text{where I} = 1, 2, 3, \ldots, 49;$$

the second sequence of means (Y_j) is

$$Y_j = \sum_{i=0}^{23} \frac{X_i + j}{24} \quad \text{where j} = 1, 2, 3, \ldots, 25;$$

then the mean level (M) at hour 36 is

$$M = \sum_{j=1}^{25} \frac{Y_j}{25}$$

The groundwater level monitoring raw data recorded by the data logger was converted into elevations and five 71-hour time periods were chosen for statistical filtering. The five time periods were July 20 through July 23, July 24 through July 26, July 27 through July 29, August 3 through August 5, and August 10 through August 12, 1999. The first two time periods included all 34 monitoring wells and 3 stilling wells. The remaining three time periods included only the wells monitored for four weeks and the three stilling wells. A table summarizing the mean groundwater elevations for the five periods is attached as Table 3-7.

The mean groundwater elevations from the first two 71-hour periods (7/20 through and 7/23 and 7/24 through 7/26) were used to construct groundwater elevation contour maps. Groundwater elevations for each 71-hour period were divided into two categories, shallow (<50 ft bgs) and deep (>50 ft bgs) and groundwater elevation contour was developed for each. The shallow and deep aquifer groundwater elevation contour maps are included as Figure 3-7 through Figure

3-10. Appendix N contains graphs which depict the data filtering process for each well included in the tidal study.

3.4.4 Slug Tests

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

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

where:

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

r_w = horizontal distance from well center to undisturbed aquifer

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

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

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

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

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

The hydraulic conductivity values obtained from the slug tests ranged from 0.000238 ft/minute (ft/min) to 0.187 ft/min. Hydraulic conductivity values for shallow screened wells ranged from a low of 0.000238 ft/min at LW-3S to a high of 0.118 ft/min at WC5-3S. The wells installed within the intermediate zone ranged from a low of 0.000293 at PZ-9D to a high of 0.187 ft/min at WC6-1I. The hydraulic conductivity values for the deep monitoring wells ranged from a low of 0.0025 ft/min at WC2-1D to a high of 0.0336 ft/min at WC9-D2. A summary table of slug test results is presented in Table 3-8. Slug test raw data is attached in Appendix O. The slug test plots are presented in Appendix P.

3.4.5 Seepage Meter Pilot Study

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

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

3.4.6 Harding Lawson Associates Investigations

This section briefly describes the results of the three investigations conducted by HLA at SAEP from January 1998 through February 2000. The investigations include chromium-focused and chlorinated VOC groundwater sampling through a contract with AlliedSignal from January 1998 through June 1999, pre-design activities conducted by FW/HLA from August to December 1999 under the OU2 NCRA, and pre-design activities conducted by FW/HLA on the Causeway and dike from September to October 1999 through a TERC with the USACE. The HLA investigations are presented in more detail in their Draft PDIR for the OU2 NCRA and EE/CA for the Causeway and Dike both dated March 2000. The groundwater analytical results summary tables from PDIR are attached as Appendix D and the groundwater analytical results summary tables from EE/CA are attached as Appendix E.

3.4.6.1 Chromium-Focused Investigations

The following subsections present summaries of the investigations conducted by HLA at SAEP from January 1999 through June 1999 through a contract with AlliedSignal. These investigations included chromium-focused soil and groundwater sampling with a GeoProbe, and monitoring well and piezometer sampling.

Geoprobe Investigation

Thirty-two borings were advanced with the direct push method by HLA around the chrome plating facility to assess the potential for groundwater contamination in the area resulting from historic chrome plating activities. The samples collected were analyzed for heavy metals including cadmium, chromium, copper, CN, reactive CN, ferrous iron, hexavalent chromium, iron, and nickel. An examination of the analytical data for the samples revealed constituent concentrations exceeding the SWPC in 19 of the 32 borings, which are overviewed in this section. The CTDEP does not have a criterion for total chromium, therefore the CTDEP SWPC for trivalent chromium of 1,200 ppb was used for comparison.

WP-99-01: Groundwater samples were collected at location WP-99-01 at intervals from 10 ft to 25 ft bgs. The SWPC for copper (48 microgram per liter [µg/L]) was exceeded at 10 ft with a concentration of 61 µg/L.

WP-99-02: Groundwater samples were collected at location WP-99-02 at intervals from 20 ft to 71 ft bgs. Chromium and hexavalent chromium were detected above the respective SWPC at depths of 20 ft, 28 ft, and 34 ft with concentrations of 1,500 µg/L, 350,000 µg/L and 75,000 µg/L, respectively.

WP-99-03: Groundwater samples were collected at location WP-99-03 at intervals from 9 ft to 49 ft bgs. Chromium was detected above the SWPC of 1,200 µg/L at 9 ft, 15 ft, 25ft, and 30 ft bgs, with concentrations of 7,000 µg/L, 9,000 µg/L, 7,170 µg/L, and 8,000 µg/L respectively. Hexavalent chromium was detected above the SWPC of 110 µg/L at the same intervals with concentrations of 7,000 µg/L, 9,000 µg/L, 16,780 µg/L, and 8,000 µg/L, respectively.

WP-99-04: Groundwater samples were collected at location WP-99-04 at intervals from 10 ft to 66 ft bgs. Chromium was detected above the SWPC of 1,200 µg/L at 10 ft (4,500 µg/L), at 20 ft (95,000 µg/L), and at 26 ft (700,000 µg/L). Hexavalent chromium was detected at the same levels over the same intervals, exceeding its SWPC of 110 µg/L.

WP-99-05: Groundwater samples were collected at location WP-99-05 at intervals from 10 ft to 60 ft bgs. Hexavalent chromium was detected above the SWPC of 110 µg/L at 50 ft bgs, with a concentration of 150 µg/L.

WP-99-07: Groundwater samples were collected at location WP-99-07 at 9 ft bgs. Chromium was detected above the SWPC of 1,200 µg/L with a concentration of 13,800 µg/L. Hexavalent chromium was detected above the SWPC of 110 µg/L with a concentration of 26,400 µg/L.

WP-99-08: Groundwater samples were collected at location WP-99-08 at intervals from 12 ft to 70 ft bgs. Chromium and hexavalent chromium were detected above the SWPC of 1,200 µg/L and 110 µg/L, respectively at all intervals between 12 and 45 ft bgs with concentrations ranging from 5,000 µg/L at 19 ft to 500,000 µg/L at 25 ft bgs.

WP-99-09: Groundwater samples were collected at location WP-99-09 at intervals from 11ft to 60 ft bgs. Copper was detected above the SWPC of 48 µg/L at 11 ft with a concentration of

85 µg/L. Hexavalent chromium was detected above the SWPC of 110 µg/L at the same depth, with a concentration of 1,338 µg/L.

WP-99-10: Groundwater samples were collected at location WP-99-10 at intervals from 12 ft to 50 ft bgs. Chromium was detected above the SWPC of 1,200 µg/L at 24 ft (4,000 µg/L), and at 29 ft (4,500 µg/L). Hexavalent chromium was detected above the SWPC of 110 µg/L at depths of 12 ft, 19 ft, 24 ft, and 29 ft with concentrations of 600 µg/L, 900 µg/L, 4,000 µg/L, and 4,500 µg/L, respectively.

WP-99-11: Groundwater samples were collected at location WP-99-11 at intervals from 9 ft to 51ft bgs. Cadmium was detected above the SWPC of6 µg/L at 51ft bgs with a concentration of 6.6 µg/L. Chromium was detected above the SWPC of 1,200 µg/L at depths of 28 ft, and 34ft bgs with concentrations of 55,200 µg/L, and 60,000 µg/L, respectively. CN was detected above the SWPC of 52 µg/L at 34ft bgs with a concentration of 91µg/L. Hexavalent chromium was detected above the SWPC of 110 µg/L at 9 ft, 38 ft, and 34 ft bgs with concentrations of 500 µg/L, 54,200 µg/L, and 135,000 µg/L, respectively.

WP-99-12: Groundwater samples were collected at location WP-99-12 at intervals from 9 ft to 50 ft bgs. Copper was detected above the SWPC of 48 µg/L at 9 ft bgs with a concentration of 423 µg/L. CN was detected above the SWPC of 52 µg/L at 24 ft with a concentration of 182 µg/L.

WP-99-15: Groundwater samples were collected at location WP-99-15 at intervals from 9 ft to 60 ft bgs. Chromium and hexavalent chromium were detected above the SWPC of 1,200 µg/L and 110 µg/L, respectively at all intervals from 9 to 50 ft bgs with identical concentrations ranging from 3,000 µg/L to 950,000 µg/L. CN was detected above the SWPC of 52 µg/L at 30 ft with a concentration of 535 µg/L.

WP-99-18: Groundwater samples were collected at location WP-99-18 at intervals from 12 ft to 50 ft bgs. Chromium was detected above the SWPC of 1,200 µg/L at 12 ft, and 19 ft with concentrations of 2,250 µg/L, and 2,760 µg/L, respectively. Hexavalent chromium was detected above the SWPC of 110 µg/L at 12 ft, with a concentration of 2,250 µg/L.

WP-99-20: Groundwater samples were collected at location WP-99-20 at intervals from 12 to 60 ft bgs. Chromium and hexavalent chromium were detected above the SWPC of 1,200 µg/L, and 110 µg/L, respectively at all intervals between 19 ft and 40 ft with identical concentrations ranging from 20,940 µg/L to 450,000 µg/L, respectively.

WP-99-23: Groundwater samples were collected at location WP-99-23 at intervals from 12 ft to 25 ft bgs. Chromium and hexavalent chromium were detected above the SWPC of 1,200 µg/L, and 110 µg/L, respectively only at 25 ft with an identical concentration of 40,000 µg/L.

WP-99-26: Groundwater samples were collected at location WP-99-26 at intervals from 12 ft to 26 ft bgs. Chromium was detected above the SWPC of 1,200 µg/L only at 26 ft with a concentration of 12,000 µg/L. Hexavalent chromium was detected above the SWPC of 110 µg/L at all intervals with concentrations ranging from 300 µg/L to 12,000 µg/L.

WP-99-27: Groundwater samples were collected at location WP-99-27 at intervals from 12 ft to 50 ft bgs. Chromium and hexavalent chromium were detected above the SWPC of 1,200 µg/L ,and 110 µg/L, respectively, at 12 ft, and 16 ft with identical concentrations of 3,200 µg/L, and 4,500 µg/L, respectively.

WP-99-31: Groundwater samples were collected at location WP-99-31 at intervals from 12 ft to 24 ft bgs. Chromium was detected above the SWPC of 1,200 µg/L at 12 ft, and 19 ft with a concentration of 2,500 µg/L, and 4,500 µg/L, respectively. Hexavalent chromium was detected above the SWPC of 110 µg/L at 12 ft, 19 ft, and 24 ft with concentrations of 2,500 µg/L, 4,500 µg/L, and 150 µg/L, respectively.

Monitoring Well/Piezometer Sampling

Nine monitoring wells were sampled by HLA around the chrome plating facility to assess the potential for groundwater contamination in the area resulting from historic chrome plating activities. The samples collected from all but one piezometer were analyzed for selected VOCs including 1,1,1-TCA, 1,1-DCE, PCE, TCE, and vinyl chloride. Samples collected from piezometer PZ-Pilot-03 were analyzed for TAL metals. An examination of the analytical data for the samples revealed exceedances of the SWPC in seven of the nine wells, which are overviewed in this section. The CTDEP does not have a criterion for total chromium, therefore the CTDEP SWPC for trivalent chromium of 1,200 ppb was used for comparison. In a few cases concerning 1,1-DCE and PCE, the reporting limit was higher than their respective criteria due to sample dilution at the laboratory. Those instances are not listed in the paragraphs below.

EW-99-01: A groundwater sample was collected at extraction well EW-99-01 from 20 ft to 40 ft bgs. TCE was detected above the SWPC of 2,340 µg/L with a concentration of 400,000 µg/L.

PZ-Pilot-03: A groundwater sample was collected at piezometer PZ-Pilot-03 from 6 ft to 8 ft bgs. Chromium and hexavalent chromium were detected above their respective SWPC at an identical concentration of 22,500 µg/L.

PZ-99-02B: A groundwater sample was collected at piezometer PZ-99-02B from 25 ft to 30 ft bgs. TCE was the only compound detected above its SWPC of 2,340 µg/L, with a concentration of 82,000 µg/L.

PZ-99-03: A groundwater sample was collected at piezometer PZ-99-03 from 4 ft to 9 ft bgs. 1,1-DCE was detected above the SWPC of 96 µg/L, with a concentration of 9,400 µg/L.

PZ-99-04I: A groundwater sample was collected at piezometer PZ-99-04I from 30 ft to 35 ft bgs. 1,1-DCE was detected above the SWPC of 96 µg/L, with a concentration of 320 µg/L. PCE was detected slightly above the SWPC of 88 µg/L, at a concentration of 93 µg/L. TCE was detected above its SWPC of 2,340 µg/L, with a concentration of 45,000 µg/L.

PZ-99-12I: A groundwater sample was collected at piezometer PZ-99-12I from 16 ft to 21 ft bgs. 1,1-DCE was the only compound detected above its SWPC of 96 µg/L, with a concentration of 1,200 µg/L.

3.4.6.2 Chlorinated VOC Investigations - Groundwater

The following subsections present summaries of the investigations conducted by HLA at SAEP from January 1999 through June 1999 through a contract with AlliedSignal. These investigations included VOC groundwater sampling with a GeoProbe, and VOC groundwater sampling with a cone penetrometer.

GeoProbe Investigation

Thirty-eight direct push borings were advanced by Harding Lawson Associates (HLA) around the plant's premises to assess the potential for groundwater contamination resulting from historic activities. The samples collected were analyzed for selected VOCs including 1,1,1-TCA, 1,1-DCE, PCE, TCE, and vinyl chloride. An examination of the analytical data for the samples revealed exceedances of the SWPC in 27 of the 38 cone direct push borings, and these exceedances are overviewed in this section. In many cases concerning 1,1-DCE and PCE, the reporting limit was higher than their respective criteria due to sample dilution at the laboratory. Those instances are not listed in the paragraphs below. (HLA, 2000)

WP-99-33: Groundwater samples were collected at location WP-99-33 at intervals from 30 ft to 80 ft bgs. TCE was detected above the SWPC of 2,340 µg/L, at depths of 30 ft, 40 ft, and 50 ft, with a concentration of 830,000 µg/L, 12,000 µg/L, and 2,900 µg/L, respectively. PCE was detected above the SWPC of 88 µg/L at 80 ft, with a concentration of 340 µg/L.

WP-99-34: Groundwater samples were collected at location WP-99-34 at intervals from 20 ft to 40 ft bgs. TCE was detected above the SWPC of 2,340 µg/L at 30 ft and 40 ft, with a concentration of 390,000 µg/L, and 31,000 µg/L, respectively.

WP-99-35: A groundwater sample was collected at location WP-99-35 from 22 ft to 26 ft bgs. 1,1-DCE was detected above the SWPC of 96 µg/L, at a concentration of 2,500 µg/L. TCE was detected above the SWPC of 2,340 µg/L, at a concentration of 12,000 µg/L.

WP-99-36: Groundwater samples were collected at location WP-99-36 at intervals from 22 ft to 48 ft bgs. TCE was detected above the SWPC of 2,340 µg/L at depths of 34 ft, and 48 ft with a concentration of 23,000 µg/L, and 10,000 µg/L, respectively. PCE was detected above the SWPC of 88 µg/L at 48 ft, with a concentration of 260 µg/L.

WP-99-37: Groundwater samples were collected at location WP-99-37 at intervals from 20 ft to 44 ft bgs. TCE was detected above the SWPC of 2,340 µg/L at 30 ft, and 44 ft, with concentrations of 18,000 µg/L, and 17,000 µg/L, respectively.

WP-99-38: A groundwater sample was collected at location WP-99-38 from 22 to 26 ft bgs. TCE was detected above the SWPC of 2,340 µg/L, at a concentration of 5,000 µg/L.

WP-99-40: Groundwater samples were collected at location WP-99-40 at intervals from 20 ft to 31 ft bgs. TCE was detected above the SWPC of 2,340 µg/L only at 31 ft with a concentration of 29,000 µg/L.

WP-99-41: Groundwater samples were collected at location WP-99-41 at intervals from 10 ft to 81 ft bgs. 1,1- DCE was detected above the SWPC of 96 µg/L at 22 ft, with a concentration of 3,700 µg/L. PCE was detected above the SWPC of 88 µg/L at 40 ft, at a concentration of

300 µg/L. TCE was detected above the SWPC of 2,340 µg/L at 30 ft, and 40 ft with concentrations of 71,000 µg/L, and 7,600 µg/L, respectively.

WP-99-42: A groundwater sample was collected at location WP-99-42 from 19 to 23 ft bgs. 1,1-DCE was detected above the SWPC of 96 µg/L, at a concentration of 160 µg/L.

WP-99-43: Groundwater samples were collected at location WP-99-43 at intervals from 10 ft to 82 ft bgs. 1,1-DCE was detected above the SWPC of 96 µg/L at 10 ft, 20 ft, and 30 ft with a concentration of 1,500 µg/L, 2,400 µg/L, and 440 µg/L, respectively. PCE was detected above the SWPC of 88 µg/L at 20 ft, 30 ft, and 40 ft, with concentrations of 120 µg/L, 480 µg/L, and 180 µg/L, respectively.

WP-99-44: Groundwater samples were collected at location WP-99-44 at intervals from 29 ft to 69 ft bgs. 1,1-DCE was detected above the SWPC of 96 µg/L only at 29 ft at a concentration of 200 µg/L. PCE was detected above the SWPC of 88 µg/L at 29 ft with a concentration of 580 µg/L. TCE was detected above the SWPC of 2,340 µg/L at 29 ft, with a concentration of 5,900 µg/L.

WP-99-45: Groundwater samples were collected at location WP-99-45 at intervals from 15 ft to 60 ft bgs. 1,1-DCE and PCE were detected above their respective SWPC, with concentrations of 4,500 µg/L, and 1,900 µg/L, respectively. TCE was detected above the SWPC of 2,340 µg/L at 15 ft, 30 ft, and 40 ft, with concentrations of 5,900µg/L, 4,000 µg/L, and 6,000 µg/L, respectively.

WP-99-47: Groundwater samples were collected at location WP-99-47 at two intervals from 6 to 10 ft bgs, and from 17 ft to 21 ft bgs. 1,1-DCE was detected above the SWPC of 96 µg/L at both intervals with concentrations of 4,200 µg/L and 8,700 µg/L, respectively.

WP-99-48: Groundwater samples were collected at location WP-99-48 at two intervals from 5 to 9 ft bgs, and from 15 ft to 19 ft bgs. 1,1-DCE was detected above the SWPC of 96 µg/L at both intervals, with concentrations of 8,500 µg/L, and 4,000 µg/L, respectively.

WP-99-49: Groundwater samples were collected at location WP-99-49 at intervals from 8 ft to 43 ft bgs. 1,1-DCE was detected above the SWPC of 96 µg/L only at 21 ft, with a concentration of 250 µg/L. TCE was detected above the SWPC of 2,340 µg/L only at 43 ft, at a concentration of 7,100 µg/L.

WP-99-50: Groundwater samples were collected at location WP-99-50 at intervals from 13 ft to 55 ft bgs. PCE was detected above the SWPC of 88 µg/L at 55 ft, at a concentration of 160 µg/L. TCE was detected above the SWPC of 2,340 µg/L at 13 ft, and 54 ft, with a concentration of 8,400 µg/L, and 44,000 µg/L, respectively.

WP-99-53: Groundwater samples were collected at location WP-99-53 at intervals from 10 ft to 32 ft bgs. 1,1-DCE was detected above the SWPC of 96 µg/L at all intervals, with concentrations ranging from 44,000 µg/L to 110 µg/L. PCE was detected above the SWPC of 88 µg/L at 32 ft, at a concentration of 670 µg/L. TCE was detected above the SWPC of 2,340 µg/L at 32 ft, with a concentration of 7,900 µg/L.

WP-99-54: Groundwater samples were collected at location WP-99-54 at intervals from 20 ft to 62 ft bgs. PCE was detected above the SWPC of 88 µg/L at 20 ft, at a concentration of

210 µg/L. TCE was detected above the SWPC of 2,340 µg/L at 50 ft with a concentration of 3,000 µg/L.

WP-99-56: Groundwater samples were collected at location WP-99-56 at intervals from 10 ft to 52 ft bgs. PCE was detected above the SWPC of 88 µg/L only at 52 ft, at a concentration of 140 µg/L.

WP-99-61: Groundwater samples were collected at location WP-99-61 at intervals from 10 ft to 28 ft bgs. 1,1-DCE was detected above the SWPC of 96 µg/L at 20 ft, and 28 ft with concentrations of 160 µg/L, and 530 µg/L, respectively. PCE was detected slightly above the SWPC of 88 µg/L only at 28 ft, at a concentration of 100 µg/L.

WP-99-62: Groundwater samples were collected at location WP-99-62 at intervals from 10 ft to 63 ft bgs. 1,1-DCE was detected slightly above the SWPC of 96 µg/L at 50 ft, at a concentration of 120 µg/L. PCE was detected above the SWPC of 88 µg/L at 20 ft, 30 ft, and 50 ft, at concentrations of 130 µg/L, and 180 µg/L, and 1,100 µg/L, respectively. TCE was detected above the SWPC of 2,340 µg/L at 50 ft, and 63 ft, with concentrations of 11,000 µg/L, and 570 µg/L, respectively.

WP-99-63: Groundwater samples were collected at location WP-99-63 at intervals from 10 ft to 40 ft bgs. TCE was detected above the SWPC of 2,340 µg/L only at 30 ft, at a concentrations of 4,400 µg/L.

WP-99-68: Groundwater samples were collected at location WP-99-68 at intervals from 10 ft to 80 ft bgs. 1,1-DCE was detected above the SWPC of 96 µg/L at 10 ft, and 20 ft, with concentrations of 210 µg/L, and 740 µg/L, respectively.

WP-99-69: Groundwater samples were collected at location WP-99-69 at 10-foot intervals from 10 ft to 30 ft bgs. 1,1-DCE was detected above the SWPC of 96 µg/L at all intervals, with concentrations of 340 µg/L, 640 µg/L, and 750 µg/L. TCE was detected slightly above the SWPC of 2,340 µg/L only at 30 ft, at a concentration of 2,500 µg/L.

WP-99-70: Groundwater samples were collected at location WP-99-70 at intervals from 10 ft to 30 ft bgs. 1,1-DCE was detected above the SWPC of 96 µg/L at 30 ft, at a concentration of 850 µg/L.

WP-99-71: Groundwater samples were collected at location WP-99-71 at intervals from 10 ft to 40 ft bgs. 1,1-DCE was detected above the SWPC of 96 µg/L at 20 ft, with a concentration of 1,200 µg/L.

WP-99-72: Groundwater samples were collected at location WP-99-72 at intervals from 10 ft to 34 ft bgs. TCE was detected above the SWPC of 2,340 µg/L at 34 ft, with a concentration of 28,000 µg/L.

Cone Penetrometer Investigation

Eighteen borings were advanced using the cone penetrometer testing method by HLA within the plant's premises to assess the potential for groundwater contamination with chlorinated solvents. The samples collected were analyzed for 1,1,1-TCA, 1,1-DCE, PCE, TCE, and vinyl chloride utilizing an on-site mobile laboratory. An examination of the analytical data for the samples revealed exceedances of the SWPC in six of the 18 cone penetrometer tests, which are overviewed in this section. In several instances, the compounds 1,1-DCE and PCE had reporting limits higher than their respective criteria due to sample dilution during laboratory analysis. Those instances are not listed in the paragraphs below. (HLA, 2000)

CP-99-06: Groundwater samples were collected at location CP-99-06 at intervals from 30 ft to 128 ft bgs. 1,1-DCE was detected above the SWPC of 96 µg/L only at a depth of 30 ft with a concentration of 760 µg/L.

CP-99-08: Groundwater samples were collected at location CP-99-08 at intervals from 34 to 158 ft bgs. 1,1,1-TCA was detected above SWPC of 62,000 µg/L at depths of 34 ft, 46 ft, 135 ft, and 158 ft, with concentrations of 380,000 µg/L, 170,000 µg/L, 160,000 µg/L, and 210,000 µg/L, respectively. 1,1-DCE was detected above the SWPC of 96 µg/L at 34 ft, 46 ft, 62 ft, 135 ft, and 158 ft, with concentrations of 9,000 µg/L, 7,500 µg/L, 710 µg/L, 2,200 µg/L, and 3,100 µg/L, respectively. TCE was detected above the SWPC of 2,340 µg/L at 34 ft, 46 ft, 135 ft, and 158 ft, with concentrations of 28,000 µg/L, 19,000 µg/L, 19,000 µg/L, and 20,000 µg/L, respectively.

CP-99-10: Groundwater samples were collected at location CP-99-10 at intervals from 29 ft to 75 ft bgs. 1,1-DCE was detected above the SWPC of 96 µg/L at 36 ft, 43 ft, and 75 ft with a concentration of 240 µg/L, 670 µg/L, and 550 µg/L, respectively. TCE was detected above the SWPC of 2,340 µg/L only at 90 ft, with a concentration of 21,000 µg/L.

CP-99-17: Groundwater samples were collected at location CP-99-17 at intervals from 20 ft to 63 ft bgs. 1,1,1-TCA was detected above the SWPC of 62,000 µg/L at 20 ft, and 32 ft, with concentrations of 93,000 µg/L, and 97,000 µg/L, respectively. 1,1-DCE was detected above the SWPC of 96 µg/L at 30 ft, and 32 ft, at concentrations of 8,900 µg/L, and 8,100 µg/L, respectively. TCE was detected above the SWPC of 2,340 µg/L at 30 ft, 32 ft, and 46 ft, with concentrations of 94,000 µg/L, 92,000 µg/L, and 10,000 µg/L, respectively.

CP-99-18: Groundwater samples were collected at location CP-99-18 at intervals from 13 ft to 148 ft bgs. 1,1-DCE was detected slightly above the SWPC of 96 µg/L at 100 ft, with a concentration of 97 µg/L.

3.4.6.3 Operable Unit 2 NCRA Investigations

The following subsections present discussions of FW/HLA pre-design activities conducted under the OU2 NCRA through the TERC contract with the USACE. Aquifer testing consisting of a stepped-rate and constant rate test were conducted by HLA.

Aquifer Testing

Aquifer testing was performed at the Chromium Plating Facility by FW/HLA in September 1999. The objectives of the tests were to determine aquifer hydraulic parameters in the vicinity of the pumping well. The following paragraphs summarize the findings of the aquifer testing.

Aquifer testing revealed that the EW-99-01 pumping zone (20 to 40 ft bgs) has high specific capacity that indicates a relatively high aquifer transmissivity. The pumping well responded largely in an unconfined manner with a quick flattening of drawdown that can be associated with either delayed drainage from an unconfined aquifer or leakage through semi-confining overlying soils. Shallow piezometers close to the pumping well exhibited relatively small drawdown responses, indicating that geologic stratification is likely restricting the vertical propagation of pumping stresses. Similar small drawdowns at more distant piezometers screened in the same zone as extraction well EW-99-01 (e.g. PZ-99-01B) suggest that delayed drainage or leakage buffered pumping stresses, so that they had not yet appeared within this three day test.

Cyclical tidal influence limited the usefulness of data from any of the observation wells, particularly because the drawdown from pumping in all observation wells was so small. Tidal variation, while only about 0.03 feet in magnitude, was generally equal to 30 percent or more of the apparent drawdown.

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

Using this approximation of T of 200 ft^2/day and the observational evidence that anisotropy (stratification) limits the flow of water from the upper portion of the saturated zone, the aquifer thickness is best approximated at 20 feet, the length of the EW-99-01 wellscreen. Using this thickness produces an estimate of the horizontal hydraulic conductivity for the pumping zone of 10 ft/day, or 7.0E-03 ft/min. This value is consistent with those observed from slug testing results in the same formation and vicinity.

The aquifer storage coefficient could not be calculated from the drawdown data given the small drawdown responses at all observation wells.

4.1 OVERVIEW

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

Section 4.2 describes the objectives of the surface water investigation, Section 4.3 reviews the methods and analyses used, and Section 4.4 describes the results obtained. The results of the remedial surface water investigation are discussed in further detail and incorporated into the ecological risk assessment and human health risk assessments in Sections 9.0 and 10.0.

4.2 OBJECTIVES

The objectives of the surface water investigation were to:

1) characterize the surface water chemistry,
2) establish a reference location to serve as a basis of comparison, and
3) determine if concentrations of site-related constituents are higher relative to the remainder of the system, and
4) determine if the constituents are present at concentrations that could pose a threat to human health or ecological receptors based on comparison to water quality criteria

The data collected during the surface water investigation were used to develop the ecological risk assessment and human health risk assessments included in Sections 9.0 and 10.0.

4.3 METHODS AND ANALYSES

The surface water investigation consisted of one activity by URSGWC:

1. Surface water sampling

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

4.3.1 Surface Water

Surface water was sampled at high and low tide in the Outfall 008/Marine Basin area at the same locations where sediments were collected to characterize the sediment chemistry. Additional samples were collected from a reference location across the Housatonic River adjacent to Nell's Island to serve as a basis for comparison. Samples collected at high tide provide input into regional surface water quality; those collected at low tide represent local contributions, including potential site-related inputs. The surface water results were used to determine if concentrations of Site-related constituents are higher relative to the remainder of the system, and whether they are present at concentrations that could pose a threat to human health or ecological receptors based on comparison to water quality criteria. Figures 4-1 and 4-2 show sampling locations for the 1999 surface water sampling effort at the SAEP Site. Table 4-1 provides a summary of the scope of work.

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

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

4.4 RESULTS

Surface water sampling was conducted to characterize the surface water chemistry. The twenty surface water samples were analyzed for VOCs, PAHs, PCBs, TAL metals, and CN for Marine Basin samples. The results were compared to the more conservative chronic value of either USEPA National Ambient Water Quality Criteria for the protection of marine life (NAWQC) or the state of Connecticut Water Quality Standards (USEPA 1992b and CTDEP 1997). The following is a summary of results.

<u>VOCs</u> Several VOCs were detected in surface water samples. Three chemicals, PCE, total 1,2-DCE, and vinyl chloride were recorded in high and low tide water samples in the Outfall 008 Drainage. Note that stormwater runoff from the Igor Sikorsky Memorial Airport also drains to the Outfall 008 Drainage. The highest concentrations were recorded during high tide at sampling station 008-01 (Figure 4-2). Of these three compounds, only total 1,2-DCE was detected in the Marine Basin, in one sample at low tide. Few other volatile chemicals were detected sporadically including methylene chloride and acetone, both common laboratory contaminants. 2-butanone and TCE were also detected during high tide in one sample in the Marine Basin and two samples in the Outfall 008 Drainage, respectively.

<u>PAHs</u> PAHs were below detection in all samples with the exception of one detection of fluoranthene during high tide in the Outfall 008 Drainage.

<u>Metals and Cyanide</u> Twenty-one metals were detected in filtered and unfiltered surface water samples. CN was below detection in all samples. Of the metals, copper, nickel, and chromium exceeded the more conservative of either State or Federal surface water quality criteria for the protection of marine life. Mercury had reporting limits that exceeded its criterion. Mercury was undetected and methyl mercury was detected at very low levels (maximum concentration of 0.000163 μg/L at station 008-01 during high tide).

With the exception of vanadium, barium, and manganese, all other metals concentrations were below NAWQC, or were present at concentrations similar to background. While vanadium, barium, and manganese were found at levels higher than background averages, they were not present at extremely high levels, and were not detected at all stations.

PCBs PCBs were below detection in all samples.

Tables 4-2 through 4-5 summarize the surface water chemistry data for Marine Basin and Outfall 008 Drainage. A complete summary of detected compounds in all environmental media is provided in Section 9.0.

7.1 OVERVIEW

This section contains a discussion of the results from the geological and hydrogeological investigations conducted by URSGWC, FW/HLA and SES at SAEP. The combined data is utilized to develop physical conceptual models for the geologic and hydrogeologic systems.

Section 7.2 discusses the results and findings of the geologic investigation; Section 7.3 presents the geologic conceptual model; Section 7.4 discusses the results and findings of the hydrogeologic investigation; and Section 7.5 presents the hydrogeological conceptual model. The data reviewed here was detailed in previous sections.

7.2 GEOLOGY

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

7.2.1 Regional Geology

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

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

The area including the Site is located on the eastern limb of the northeast-trending doubly-plunging Bridgeport Syncline. The Bridgeport Syncline is the southeastern-most fold of five major folds that comprise the synclinorium. Foliation is folded but is typically vertical to steeply southeast- and northwest-dipping (Rodgers, 1985). The bedrock of the Site lies immediately west of the staurolite (toward the east) and the kyanite (toward the west) metamorphic isograd boundary (Fritts, 1965). On the basis of geologic correlations with surrounding areas, Flint (1968) estimated bedrock to occur more than 100 ft below MSL. Based on soil borings and seismic profiling conducted during the investigations, bedrock beneath SAEP occurs at elevations ranging from 50 to 170 ft MSL.

Pleistocene glaciation, ending about 10,000 years ago, affected the southern Connecticut area. It seems likely that at least two glacial movements (possibly several more) modified the topography of the area. Glaciers advanced southward as far as Long Island, scouring the underlying bedrock as they advanced. Striations on local bedrock exposures indicate ice advancement directions between S30^{0}E and SIO^{0}W (Flint, 1968). Unconformable deposits of till and stratified sand and gravel were deposited as the glaciers retreated. These deposits are of variable thickness. As the glaciers receded further, meltwater from the ice was locally impounded, creating a freshwater lake in what is today Long Island Sound. Glacial streams entering the lake left deltaic sand deposits along coastal Connecticut, including the Site vicinity (Stone et al., 1985). Locally, the deltaic sand and gravel deposits are referred to as the Stratford Outwash sediments (Flint, 1968). The southern part of the outwash, in which a portion of the Site lies consists of sand with less than 25% gravel, most of it pebble size (Flint, 1968).

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

7.2.2 Local Geology

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

The bedrock geology underlying SAEP is reported to consist of lower Ordovician age metamorphic schists, phyllites, and paragneisses of the Oronoque Member of the Derby Hill Schist (Fritts, 1965). Flint (1968) identifies these rocks as the Orange Formation. Exposures of bedrock do not occur in the SAEP vicinity. Borings made along the Housatonic River (Flint, 1968) and borings completed on-site (ESE, 1991) reportedly encountered bedrock at depths ranging from about 60 to 160 feet below the land surface.

Recent deposition of alluvium, estuarine, tidal marsh, beach sediments, and man-placed artificial fill occur along the Housatonic River. The surficial unconsolidated sediments reported near

SAEP are Stratford Outwash, tidal marsh peat, and artificial fill (Flint, 1968; U.S. Department of Agriculture, 1981). Lordship Outwash sediments are found south of the SAEP (Flint, 1968).

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

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

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

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

7.2.3 Site-Specific Geology

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

- Supplementary Hydrogeologic Investigation Report (C.A. Rich, 1991);
- The Phase II RI Report (URSGWC, 1996);
- The results of RI activities conducted by URSGWC and HLA since 1998; and,
- The OU2 Report of HLA, (February 2000).

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

1. Fill
2. Estuarine silt
3. Peat
4. Alluvium/outwash deposits
5. Bedrock

Generalized geologic cross sections have been prepared from previous engineering reports as well as from boring logs, cone penetrometer logs, and seismic data from Phase I, Phase II, and recent investigations conducted by URSGWC and HLA. The locations of the cross-sections are shown on Figure 7-1. Figure 7-2 presents cross sections A-A' and B-B'. Figure 7-3 presents cross sections C-C' and D-D'. Figure 7-4 presents cross sections E-E', F-F', and G-G'. These descriptions are generally accurate with minor variations in the stratigraphy seen throughout SAEP. A description of these units and their distribution across SAEP is presented below (from top to bottom):

Fill

Fill material is found throughout most of the SAEP consisting of sand, gravel, and debris associated with buildings, roads, utilities, site grading, and other structures. The fill is generally about 2 to 5 feet thick, but reaches a thickness of about 20 ft adjacent to the Dike as shown on cross section G-G' (Figure 7-4). The fill is thicker near the Dike due to the emplacement of fill over existing tidal sediments to extend the shoreline of the facility in the 1940s.

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

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

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

Estuarine Silt

A thick silt deposit underlies the fill at the northern edge of the facility along the Housatonic River, as seen on cross-section A-A' (Figure 7-2). This deposit typically consists of dark organic fine silts with very fine sands, occasionally clayey and having a sulfur dioxide smell consistent with intertidal mud-flat deposits. This material is an estuarine sediment deposited by the

Housatonic River and subsequently topped with fill (both artificial and glacial fill) during enlargement of the facility property.

Cross section A-A' shows the continuity of this silt, which stretches from CP-99-05 eastward to WC2-5I at a depth of approximately 6 to 14 ft below MSL (8 to 20 feet bgs). The thickness of the estuarine silt layer varies from as much as 30 feet beneath the Causeway and adjacent to the Dike at the north end of the facility to nonexistent toward the interior of the facility (see cross sections A-A' and D-D' through G-G'). Silt deposits exist beneath the fill from the length of the Dike southwestward 150 to 250 ft to about the location of cross-section B-B' (see Figure 7-3 and 7-4). This aerial extent is consistent with the area of former intertidal mudflats, which were filled in the 1940s to extend the shoreline of the SAEP property further north and eastward toward the Housatonic River.

Peat

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

Peat material was also encountered at locations WC-8S and WC-2D at depths of 5 ft bgs and 30 ft bgs and thicknesses of 2 ft and 5 ft respectively. A 10-inch thick layer of Peat was also encountered at a depth of 64 ft bgs at MWCD-99-01B. These three peat deposits are apparently small, isolated pockets and are not a part of the continuous deposit in the lagoon area. The presence of peat at MWCD-99-01B is noteworthy because it occurs at the interface of the Estuarine Silt and the Glacial Outwash, thereby representing the earliest post-glacial estuarine deposits (see Figure 7-4, cross section F-F'). Alluvium is not present at this location.

Alluvium / Outwash Deposits

A thick zone of alluvium/outwash deposits (sand and gravel) underlies the fill, estuarine silt, and peat deposits at the Site, and unconformably overlies the bedrock. The thickness of the alluvium/outwash deposits ranges from 28 ft near WC2-2D (see cross sections A-A' and G-G') to about 156 ft thick at CP-99-08 (see cross sections E-E'). The alluvium/outwash deposits consist of sands, silty sands, and gravely sands with occasional boulders, clay stringers, and varved silt. For the conceptual model for the Site, these alluvium/outwash deposits can be grouped into two broad units: 1) Alluvium below the Fill and Estuarine silts and 2) Glacial Outwash below thealluvium.

The alluvial deposits consist of sand with trace amounts of coarser materials (sand and gravel) with clay, silt, and cobbles. This unit probably represents alluvium/outwash deposits that have been reworked and sorted by the actions of the meandering Housatonic River. The alluvium is thickest beneath the southwestern part of the Site (along Main Street, see cross-section C-C'), and thins toward the Housatonic River (see cross-sections D-D' through G-G'). The distinguishing feature of these deposits on the referenced cross-sections is the trace gravel, and loosely cemented gravel zones, although a few gravelly zones also appear within the Glacial Outwash. The bottom depth of these deposits varies between approximately 20 and 40 feet bgs.

Glacial Outwash beneath the alluvial deposits and above the bedrock surface is a fine to medium sand with some silt. The glacial outwash also contains silt/clay seams and fine silty sand lenses (see cross-sections A-A' through G-G'). The glacial outwash is generally stratified and exhibits a fining-down sequence, which has a micaceous component. Micaceous zones are observed in the northwestern area of the Site as seen in PCPT explorations CP-99-01, -02, -03, -05, and –09. These zones are mostly below 60 feet bgs.

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

Bedrock

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

Figure 2-3 presents the interpretive bedrock surface elevation based on the results. The seismic survey results show that the bedrock surface elevation is highly variable, even over relatively short horizontal distances. Therefore, HLA considers that the contours presented in Figure 2-3 are generally accurate within ±10 feet. Bedrock is deepest to the northwest along seismic Line 7 and the west end of seismic Line 2, and shallowest to the southeast along seismic Lines 3 and 5, and the east end of seismic Line 2. Site-wide, results show that the bedrock surface has a general dip direction to the northwest, with the shallowest depths to bedrock being located along Sniffens Lane and the South Parking Lot area. If the locally N35°E trending trace of the Mixville Fault (Flint, 1968) is extrapolated to the southwest (Rodgers, 1985), across the Housatonic River, it would pass directly under the SAEP. However, there is no confirmation of the existence of this fault southwest of its mapped terminus.

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

A competent amphibole-mica schist bedrock was reported at locations LW-5D, LW-9D and LW-10D at depths of 163 ft, 151.5 ft, and 103 ft below grade, respectively (CA Rich Consultants, 1991). Bedrock was encountered at three locations during the drilling of Phase II monitoring wells WC-9D2, WC-20D2 and WC-21D2; at respective depths of 156 ft, 150 ft and 148 ft below grade (see cross sections F-F' and G-G', Figures 7-4). Bedrock was encountered in post-Phase II monitoring wells WC2-1D, WC2-2D, WC5-1D, and MWCD-99-01B at respective depths of 85 ft, 62 ft, 85 ft, and 102 ft below grade.

Five feet of NX-size rock core was collected at each location from the bedrock encountered during the drilling of monitoring wells WC2-2D, WC2-3D, and WC5-1D. Core logs are included on the boring logs for these wells (Appendix B). The bedrock at WC2-3D and WC5-1D is described as a gray, strong, thickly-bedded, fine-grained phyllite. The bedrock at WC2-2D is described as a gray, strong, thickly-bedded, medium-grained schist.

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

7.3 GEOLOGIC CONCEPTUAL MODEL

This section presents a conceptual geological model of the Site based the investigations conducted by URSGWC and HLA in 1998 and 1999. The geologic model is described using subsurface cross-sections A-A’ through G-G’ and bedrock surface elevation Figure 2-3. Cross-sections A-A’, B-B’, and C-C’ are roughly parallel to the Housatonic River and the remaining cross-sections are perpendicular to the river. The conceptual geologic model has been developed, with stratigraphy descriptions on a macroscopic scale. These descriptions are typically accurate with minor variations in the stratigraphy seen throughout SAEP. The following paragraphs present the conceptual geologic model (from top to bottom):

Fill material is found throughout most of the SAEP consisting of sand, gravel, and debris associated with buildings, roads, utilities, site grading, and other structures. The fill is generally about 2 to 5 feet thick, but reaches a thickness of about 20 ft adjacent to the Dike as shown on cross section G-G' (Figure 7-4). The fill is thicker near the Dike. The composition of the fill is variable, but most of it is described as a granular material. The fill material generally consists of brown medium to fine sand and gravely sand. This fill material is most pronounced on section A-A', (but can also be observed on sections D-D', E-E', and G-G') as the sand and gravel deposit shown overlying the estuarine silt layer. The fill appears to thin somewhat with distance away from the river.

A thick silt deposit underlies the fill at the northern edge of the facility along the Housatonic River, as seen on cross-section A-A' (Figure 7-2). This deposit typically consists of dark organic fine silts with very fine sands. This material is an estuarine sediment deposited by the Housatonic River and subsequently topped with fill during enlargement of the facility property. Cross section A-A’ shows the continuity of this silt, which stretches from CP-99-05 eastward to WC2-5I at a depth of approximately 6 to 14 ft below MSL. The thickness of the estuarine silt layer varies from as much as 30 feet beneath the Causeway and adjacent to the Dike at the north end of the facility to nonexistent toward the interior of the facility (see cross sections A-A' and D-D' through G-G'). Silt deposits exist beneath the fill from the length of the Dike southwestward 150 to 250 ft to about the location of cross-section B-B' (see Figure 7-3 and 7-4).

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

A thick deposit of alluvium/outwash sediments (sand and gravel) underlies the fill, estuarine silt, and peat deposits at the Site, and unconformably overlies the bedrock. The thickness of the alluvium/outwash sediments ranges from 28 ft near WC2-2D (see cross sections A-A' and G-G') to about 156 ft thick at CP-99-08 (see cross sections E-E'). The alluvium/outwash deposits consist of sands, silty sands, and gravelly sands with occasional boulders, clay stringers, and varved silt. For the Site conceptual model, these alluvium/outwash deposits can be grouped into two broad units: 1) Alluvium below the Fill and Estuarine silts and 2) Glacial Outwash below the Alluvium.

Alluvium consists of sand with trace amounts of coarser materials (sand and gravel) with clay, silt, and cobbles. This unit is probably represents alluvium/outwash deposits that have been reworked and sorted by the actions of the meandering Housatonic River. Outwash beneath thealluvium, and above the bedrock surface is a fine to medium sand with some silt. The outwash contains silt/clay seams and fine silty sand lenses (see cross-sections A-A' through G-G'). The outwash is generally stratified, and exhibits a fining-down sequence, which has a micaceous component.

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

7.4 HYDROGEOLOGY

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

7.4.1 Local Hydrogeology

Locally, the hydraulic properties of this region are variable because of the differences in texture composition, and structural features that result from different provenance and genesis of fluvial and glaciofluvial units. A major concern at any site in the area is the heterogeneity of the subsurface materials and how this heterogeneity affects the spatial variability of hydraulic properties. The presence, extent, and continuity of fractures and lenses or layers of stratified drift are difficult to define, particularly in the deeper parts of thick deposits. The horizontal hydraulic conductivities range from $1.4\text{x}10^{-6}$ to $2.3\text{x}10^{-2}$ centimeters per second (cm/s), whereas the vertical hydraulic conductivities range from 4.7 x 10^{-6} to 3.4 x 10^{-2} cm/s. Available data, although sparse, indicate that values of porosities are commonly in the range of 20 to 35 percent (Melvin et al., 1992).

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

The climate is humid with an average annual precipitation of about 47 in. In a 30-year period (1951-80), median annual precipitation ranged from about 53 in. (135 cm) in northwestern Connecticut to 40 in. in the southwestern part of the State, and analyses of longer periods of record indicate that precipitation in any given year can be considerably greater or less than the median (Melvin et al., 1992).

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

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

7.4.2 Site-Specific Hydrogeology

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

Hydrostratigraphy

Stratigraphic borings advanced at the facility has revealed three hydrostratigraphic units: fill, estuarine silts and depositional sediments.

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

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

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

Alluvium: Sand and gravel deposits of glacial origin underlie the fill and silt deposits. The unit comprises fine-grained sand, with trace amounts of coarser sand and gravel with clay, silt, and cobbles. The working hypothesis for this unit is that glacial deposits have been reworked and sorted by the actions of the meandering Housatonic River. Thealluvium is thickest beneath the southwestern part of the Site along Main Street and thins toward the Housatonic River. The distinguishing feature of these deposits is the trace gravel, and loosely cemented gravel zones. The bottom depth of these deposits varies between approximately 20 and 40 feet bgs.

Glacial Outwash: Beneath the alluvium, and above the bedrock surface, lies a fine to medium sand with some silt, interpreted to be glacial outwash. The glacial outwash contains silt/clay seams and fine silty sand lenses. The glacial outwash is generally stratified, and exhibits a fining-down sequence. In addition, considerable inter-fingering of slit and sand occurs across the Site within the depositional sediments. The silt layers within the glacial deposits may function as semi-confining layers where they are laterally continuous and sufficiently thick.

7.4.3 Hydrodynamics

A total of 84 monitoring wells have been installed in the overburden aquifer. Twenty-four monitoring wells were installed by URSGWC during the remedial investigation and four monitoring wells were installed during HLA's investigations; the remaining were installed during previous investigations. Shallow monitoring wells (10) are screened across the water table, intermediate wells (10) are screened at depths from 30 to 50 feet , and deep wells (8) are screened just above the bedrock.

Figures 3-7, 3-8, 3-9, and 3-10 present mean groundwater elevations, and interpretive groundwater elevations, for two time periods July 20-23, 1999 and July 24-26, 1999. Mean groundwater elevations on the main portion of the facility appear to range from approximately 2.30 feet MSL beneath building B-2 to 1.00 feet MSL beneath Building B-19, located near the Dike on the eastern side of the facility.

Horizontal Groundwater Gradients

Figures 3-7 and 3-8 show the piezometric surface (water table contours) based on mean groundwater elevations recorded in shallow and intermediate monitoring wells for two time periods, July 20-23, 1999 and July 24-26, 1999. The equipotential lines indicate groundwater flow is generally from west to east and southeast toward the Housatonic River across a large portion of the facility. However, groundwater flow in the west parking lot and western portion of the study area is to the northwest toward Frash Pond. There is a large area in the center of the Site beneath Building 2 extending southwest to the west and south parking lots and Sikorsky Airport of slight groundwater mounding which may act as a groundwater divide. This feature is present during both monitoring periods. However, it is not known if this is an anomaly caused by the limited data points located in this portion of the study area. Another smaller groundwater divide is present on the north side of Building 2 where groundwater east of the divide flows toward the Housatonic River and groundwater west of the divide flows toward Frash Pond. Horizontal gradients vary widely across the main portion of the facility (Table 7-1). The horizontal gradient in the vicinity of Building 2 is relatively flat (0.0001 ft/ft). In contrast, the gradient increases dramatically near the river to the northeast (0.002 ft/ft). The estuarine silts likely act as a low permeability boundary to groundwater flow in this area resulting in the increase gradient.

Figures 3-9 and 3-10 show the piezometric surface based on mean groundwater elevations recorded in the deep monitoring wells. The equipotential lines indicate that groundwater flow in the deeper portions of the aquifer moves from west to east. However, the groundwater elevations of the deep wells generally are lower than those of the shallow wells, except for the shallow groundwater elevations adjacent to the berm. The flow pattern indicates the discharge of groundwater into the Housatonic River.

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

Vertical Groundwater Gradients

The overall dynamics of the overburden aquifer system are best shown by the cross-sectional flow diagrams (Figures 7-5 and 7-6). The diagrams clearly show the effects of the bedrock highs and the estuarine silt on the hydrodynamics at the Site. Also note that the equipotential lines tend to flatten horizontally along the berm in the eastern portion of the Site (see A-A' and D-D'). Flattened equipotentials indicate layers that have low permeability and act as confining layers. Where these low permeability layers are present there is a larger head difference between wells penetrating the shallow versus the deep portions of the over burden. The head differences indicate that these low permeability layers are acting as semi-confining units, restricting the movement of groundwater between the shallow and deep portions of the overburden unit. This results in a upward gradient in the shallow aquifer and a downward movement in the deeper aquifer (see Table 7-2). Where these low permeability layers are less extensive, the head differences between shallow and deep wells are relatively small and the equipotential lines remain essentially vertical as shown on cross-section C-C'.

Groundwater Discharge

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

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

The amount of groundwater discharge to the Housatonic River from the overburden aquifer was determined from the cross sectional flow diagrams D-D' (see Figures 7-6). The discharge rate (Q) can be obtained from flow net analysis according to the following equations:

$$Q' = \left(\frac{mKH}{n}\right)\left(\frac{dm}{ds}\right) \qquad \text{where } Q = Q'*L$$

and

m = number of flow tubes (determined from Figure 7-6)

K = hydraulic conductivity

H = total head drop across region of flow

n = number of divisions of head in flow net (determined from Figure 7-6)

dm/ds = ratio of vertical to horizontal scales

L = length of berm

The average hydraulic conductivity, determined from the results of the slug test, is approximately 63.11 ft/day for the 7 flow tubes. Substituting the relevant values into the equation yields (see calculation sheet included in Appendix Q) a discharge rate of approximately 13,709 ft^3/day (102,540 gpd) from the overburden aquifer near the berm to the Housatonic River.

Water Level Fluctuations

Unconfined aquifers with water tables near ground surface frequently exhibit diurnal fluctuations which can be attributable to evaporation and/or transpiration. Both processes cause a discharge of groundwater into the atmosphere and have nearly the same diurnal variation because of their high correlation with temperature. Appendix R contains hydrographs for several monitoring well clusters which show overall trends of the water table elevation as a result of changes in storage.

The hydrographs also indicate the regular cyclic rises and falls caused by the tidal influences on the aquifer.

Tidal fluctuations greater than seven feet were measured in surface water in the Housatonic during the monitoring period. The fluctuation of surface water levels induces the migration of surface water into and out of the aquifer. Groundwater level fluctuations were detected in several of the monitoring wells on-site as a result of the introduction of surface water during high tide or loading caused by the introduction of surface water into the aquifer system. Groundwater fluctuations of one to two were recorded in monitoring wells located within 100 feet of the berm. The fluctuation diminished proportional to the distance from the river. At the distance of 300 feet from the river, measurements detected fluctuations of less than 0.1 feet.

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

Storativity and Specific Yield

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

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

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

$$S = S_y + hS_s$$

where S_y = specific yield

h = saturated thickness

S_s = specific storage

In nearly all cases, the value of the specific yield is much larger than the value of the term hS_s, so the storativity is usually taken to be equal to the specific yield – that is, $S = S_y$.

The volume of water drained from an unconfined aquifer (ΔV) may be found from the equation:

$$\Delta V = SA\Delta h \quad = S_y A \Delta h$$

where S = storativity = specific yield (S_y)

A = surface area bounded by sides perpendicular to and parallel to the groundwater flow of interest

Δh = change (decline) in head

The specific yield of an unconfined aquifer can range from 0.01 to 0.35. The lithology encountered in borings at the Site is generally fine sand and silty sand. Based on this lithology, an initial estimate of the average specific yield at the Site is 0.2. However, using the approximate discharge rate (Q) of the aquifer (which was determined from the vertical flow analysis) will yield a more suitable estimate of the specific yield. Discharge is related to the rate of change in head of a well – the change in head over time (or Δh/Δt). The average rate of change in head was determined from hydrographs for several wells throughout the Site over a six-day period (see Table 7-3). The six-day period was to reduce the likely hood of change in storage is due to tidal fluctuations.

These data suggest that average rate of decline in head is approximately 0.019 ft/day. Little or no rainfall occurred during these periods, so the change in storage reflects discharge of the aquifer to the river.

The discharge rate (Q) is the volume of water discharged over time – that is, Q = ΔV/Δt. The discharge rate can be determined in terms of specific yield:

$$Q = \frac{\Delta V}{\Delta t} = \frac{SyA\Delta h}{\Delta t}$$

Since Δh/Δt is the rate of change in head the above equation can be arranged to solve for specific yield:

$$S_y = \frac{Q}{A\left(\Delta h / \Delta t\right)}$$

The equation can be used to determine the specific yield of the overburden aquifer at the facility based on the discharge rate (Q) calculated earlier, the surface area (A) of the Site, and the rate of change in head (Δh/Δt) averaged from measurements at six wells.

The discharge rate (Q) of the overburden aquifer to the Housatonic River through the berm area is estimated to be approximately 13,709 ft^3/day as determined using the flow net analysis described above. The area (A) of Site is about 3,062,500 ft^2. The average rate of change in head was determined to be 0.019 ft/day. Substituting these values in to the above equation results in an estimate of about 0.24 for the specific yield. This value is only an estimate, and it is based on an average discharge rate estimated from an average hydraulic conductivity from wells throughout. Also note that the average rate of decline in head determined from six wells may not be indicative of the aquifer as a whole.

Aquifer Properties and Testing

The hydraulic conductivity values obtained from the slug tests ranged from 0.343 ft/day to 270.14 ft/day. Hydraulic conductivity values for shallow screened wells ranged from a low of 0.343 ft/day at LW-3S to a high of 170.06 ft/day at WC5-3S. The wells installed within the intermediate zone ranged from a low of 0.422 ft/day at PZ-9D to a high of 270.14 ft/day at WC6-1I. The hydraulic conductivity values for the deep monitoring wells ranged from a low of 3.56 ft/day at WC2-1D to a high of 89.73 ft/day at WC2-2D. A summary table of results is presented in Table 3-8. The plots for slug tests conducted at the facility during the investigation are presented in Appendix P.

The lithology intersected by the well screen is highly variable, as shown in Table 3-8. This lithological variability across the overburden is reflected in the high variability of hydraulic conductivity values across the overburden. Due to the large range of hydraulic conductivity values (four orders of magnitude), statistical analysis of the in-situ hydraulic conductivity values across the overburden aquifer (i.e., without partitioning the data from individual stratigraphic units) was performed.

Appendix Q presents the frequency distribution of these hydraulic conductivity estimates derived from the slug tests. This figure suggests that the hydraulic conductivity of the overburden is lognormally distributed. Domenico and Schwartz (1990) reported on the work of several authors who noted that hydraulic values obtained from individual formations were lognormally distributed. The histogram of the estimated hydraulic conductivity is compared with a theoretical lognormal distribution in Appendix Q.

The effect of the large range in the variability of hydraulic conductivity can be seen on the discharge rate of groundwater from the area adjacent to the berm to the Housatonic River. The groundwater discharge rate is a function of hydraulic conductivity, the hydraulic gradient, and the cross-sectional area that is discharging groundwater (cross-section A-A'):

$$Q = f(K)iA$$

where $f(K)$ = hydraulic conductivity probability density function

i = hydraulic gradient

A = cross-sectional area

The hydraulic gradient estimate (0.0022 ft/ft) was determined from groundwater elevation data for PZ-4D and PZ-8D. The cross-sectional area of cross section A-A' was determined by multiplying the saturated thickness of the aquifer (104 feet) by the length of the area (1750 feet). Thus Q = 400f(K).

From lognormal statistical analysis of the hydraulic conductivity data, the mean value was determined to be 29.36 ft/day. Substituting this value into the above equation yields an estimate of 11,637 ft^3/day (87,045 gpd) of groundwater is discharged to the Housatonic River through the length of the berm area. This estimate agrees reasonably well with the one derived from the flow net analysis.

If it is assumed that the hydraulic conductivity is normally distributed, the data presented in Table 3-8 have a mean of 63.11 ft/day and a standard deviation of 63.10 ft/day. Using the same calculations this would indicate a discharge of 25,012 ft^3/day (187,090 gpd) from the overburden aquifer in the berm area – about 215% higher than the estimate based on a lognormal distribution.

Groundwater Flow Velocity

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

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

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

where K = hydraulic conductivity

i = hydraulic gradient

Φ = porosity

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

Based on the lithological textures and sorting observed, it is estimated that the porosity at the Site ranges from 25% for the higher-permeability deposits to 50% for the lower-permeability deposits. Assuming an average horizontal hydraulic conductivity of 63.11 ft/day, an average porosity of 35%, and that the hydraulic gradient ranges from 0.000012 to 0.0022 ft/day, it is estimated that the groundwater velocity ranges from about 1 ft/year to 145 ft/year. The higher velocities occur in the more permeable sections of the overburden deposits. These data emphasize the importance of the more permeable zones of the aquifer, especially because 1) these zones are where the discharge is occurring; and 2) these zones may provide preferential pathways for the transport of contaminants within the aquifer.

7.4.4 Hydrogeologic Conceptual Model

A conceptual model of the Site was developed based on the results from the remedial investigation and HLA's activities. Three hydrostratigraphic units were identified during the investigation: fill, estuarine silts and depositional sediments. The fill unit is the uppermost hydrostratigraphic unit and comprises sand, gravel, and debris fill. This zone contains the water table aquifer in areas near the dike where tidal influences are most apparent on groundwater. In areas further from the dike the fill unit is in the unsaturated zone. The silt is the second hydrostratigraphic unit and comprises fine silts with very fine sands, rich in organics. This unit acts as a semi-confining unit between the depositional sediments unit and the fill unit in areas along the dike where the thickness is sufficient. The low permeability of the unit also affects the movement of horizontal groundwater flow in the dike area. In areas where this unit is absent there appears to be good hydraulic connection between the fill and depositional sediment units. The depositional sediment is the lower-most hydrostratigraphic unit within the unconsolidated aquifer and overlies the bedrock. The depositional sediment consists of glaciofluvial and fluvial sediments and can be subdivided into two general units, alluvium and glacial outwash.

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

The overburden aquifer is complex both lithologically and hydrogeologically. The shallow portion of the aquifer is unconfined. The deeper portion of the overburden aquifer appears to range from being unconfined to being semi-confined, and may be fully confined locally based on the presence and thickness of the estuarine silt unit. Groundwater flow within the overburden aquifer is complex and is controlled by the estuarine silt in the shallow portion and bedrock surface elevation in the deep portions of the aquifer. In general, groundwater flows toward and discharges to the Housatonic River. Based on the findings of this investigation, 11,637 ft^3/day to 13,709 ft^3/day of groundwater is discharged to the Housatonic River through the length of the berm area. In addition, a northwesterly flow towards Frash Pond, and flow toward the drainage channel in the southern portion of SAEP have been observed. The steeper hydraulic gradient adjacent to the river along the berm can be attributed to the estuarine silt unit, which serves as a low permeability boundary to groundwater flow.

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

Vertical hydraulic gradients indicate that recharging conditions exist in the west parking lot. Vertical gradients begin to flatten in the center of the Site and reverse to discharging conditions near the Housatonic River in the shallow and intermediate wells. Deep wells indicate continued downward gradients with discharge likely into the Housatonic River beyond the intertidal mudflats. Head differences between the shallow wells and deep wells indicate that these low permeability layers are acting as semi-confining units, restricting the movement of groundwater between the shallow and deep portions of the overburden unit. This results in an upward gradient in the shallow aquifer and a downward movement in the deeper aquifer. Where these low permeability layers are less extensive, the head differences between shallow and deep wells are relatively small and the equipotential lines remain essentially vertical.

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

The hydraulic conductivity of the overburden aquifer spans four orders of magnitude (0.343 feet/day to 269.28 ft/day) with a higher frequency toward the lower range of values. The data suggest that the hydraulic conductivity is lognormally distributed. The large spatial variability in hydraulic conductivity is consistent with the complex lithology at the Site. The calculated groundwater velocity ranges from about 1 ft/year to 145 ft/year. The higher velocities occur in the more permeable sections of the overburden deposits. These data emphasize the importance of the more permeable zones of the aquifer, especially because 1) these zones are where the discharge is occurring; and 2) these zones may provide preferential pathways for the transport of contaminants within the aquifer.

8.1. OVERVIEW

This section contains a discussion of the analytical results from the geological and hydrogeological investigations conducted by URSGWC and FW/HLA at SAEP. In addition, results of SES RCRA groundwater sampling have been evaluated and incorporated into this discussion. The combined data is utilized to develop chemical conceptual models for the geologic and hydrogeologic systems.

Sub-section 8.2 discusses the results and findings of the organic and inorganic constituents in the soil; Sub-section 8.3 discusses the results and findings of the organic and inorganic constituents in the groundwater, and Sub-section 8.4 presents contaminant fate and transport calculations for organic and inorganic constituents present in the groundwater. The data reviewed here was detailed in Sections 2.0 through 6.0.

8.2. SOIL

This section discusses the analytical results from the geological investigations conducted by URSGWC and FW/HLA. Sources of data for interpretation of soil chemical characterization are from 1999 RI investigations, and 1998 through 1999 HLA investigations (including chromium-focused, OU2 NCRA, and the Causeway and dike EE/CA investigations). The organic characterization and the inorganic characterization are presented in the following subsections.

8.2.1. Organic Characterization

The major organic constituents in soil that exceed the CTDEP ICC for Direct Exposure to Soil are vinyl chloride, benzo[a]anthracene, benzo[a]pyrene, benzo[b]fluoranthene and the PCB Aroclor 1260. Figures 8-1 through 8-5 present the estimated horizontal extent of these organic compounds across the main facility at concentrations exceeding their respective ICC.

All of these organic compounds were detected on the Causeway above their respective ICC. The Causeway was initially constructed in the 1930s and additional materials, of unknown origin, were deposited along the northern edge of the Causeway during the 1950s and 1960s. The source of the fill used to construct the Causeway is unknown, but the fill contains soil, cobbles, and construction debris (e.g., wood, concrete, brick, asphalt and rebar). The depth of the fill is approximately 10 to 12 feet throughout the Causeway. It was also reported that paint solvents and wastes were burned on the Causeway as part of facility fire training operations.

Vinyl chloride was detected in soil at concentrations exceeding the ICC of 3 ppm in two locations along the berm and one on the Causeway with depths ranging from 0 to 7 feet bgs. The maximum concentration of vinyl chloride detected in soil was 36 ppm in HLA Geoprobe boring HA-99-07, located southeast of Building B-38 along the berm at a depth from 0 to 1 feet (Figure 8-1).

Benzo[a]anthracene, benzo[a]pyrene and benzo[b]fluoranthene were detected in soil at concentrations exceeding the ICC of 7.8 ppm, 1 ppm and 7.8 ppm, respectively. The exceedances were detected in areas of the South Parking Lot by Building B-41, the sump inside Building B-13 and on the Causeway. In addition, benzo[a]pyrene was detected in two samples collected in the northeast end of Building B-2 (Figures 8-2, 8-3 and 8-4). All three compounds were detected in a cluster towards the end of the Causeway with depths ranging from 0 to 10 feet bgs. Within the sample interval of 7-9 feet in HLA boring CB-99-15, the maximum concentrations for benzo[a]anthracene, benzo[a]pyrene and benzo[b]fluoranthene were detected at 1,200 ppm, 880 ppm and 940 ppm, respectively. It is probable that these concentrations on the Causeway are due to the former fire training operations.

PCB Aroclor 1260 was detected in soil at concentrations exceeding the ICC of 10 ppm in two locations along the berm and one on the Causeway. The maximum concentration of Aroclor 1260 detected in soil was 74 ppm in URSGWC sample SB09A1-4A, located southeast of Building B-37 along the berm at a depth of 0.67 to 2.67 feet (Figure 8-5).

8.2.2. Inorganic Characterization

The major inorganic constituents in soil that exceed the ICC are arsenic, beryllium, hexavalent chromium and lead. Figures 8-6 through 8-9 present the estimated horizontal extent of these inorganic compounds across the main facility at concentrations exceeding their respective ICC.

All of these inorganic compounds, except hexavalent chromium, were detected on the Causeway above their respective ICC. The source of the fill used to construct the Causeway is unknown, but the fill contains soil, cobbles, and construction debris (e.g., wood, concrete, brick, asphalt and rebar).

Arsenic appears to be the most widespread of the inorganics in soil. Arsenic was detected at concentrations exceeding the ICC of 10 ppm across the main portion of the facility and focused in the area at the end of the Causeway. The maximum concentration of arsenic detected was 3,550 ppm from URSGWC sample SB13G1-1C, that was collected between 5-7 ft from a buried pit in the north end of Building B-10 (Figure 8-6).

Beryllium exceedances appear to be concentrated on the full length of the Causeway with one additional exceedance south of Building B-81. The maximum concentration of beryllium in exceedance of the ICC of 2 ppm was detected on the Causeway in test pit TP-DEP-11 with a concentration of 13.1 ppm (Figure 8-7).

Hexavalent chromium was detected in two samples within Building B-2, beneath the former Chromium Plating Facility. The maximum concentration of hexavalent chromium in exceedance of the ICC of 100 ppm was detected in HLA sample SP-99-11 with a concentration of 640 ppm (Figure 8-8).

Lead was detected in several locations on the main portion of the facility and in three locations on the Causeway. The maximum concentration of lead in exceedance of the ICC of 1,000 ppm was detected in URSGWC sample SB09A1-4A, located downgradient of scrap metal dumpsters, at a depth of 0.67 to 2.67 feet with a concentration of 7,870 ppm (Figure 8-9).

8.3. GROUNDWATER

This section discusses the analytical results from the hydrogeological investigations conducted by URSGWC, FW/HLA, and SES. Sources of data for interpretation of groundwater chemical characterization are from 1999 RI investigations, 1998 through 1999 HLA investigations (including chromium-focused, OU2 NCRA, and the Causeway and dike EE/CA investigations), and 1999 and 2000 SES RCRA investigations. The organic characterization and the inorganic characterization are presented in the following subsections.

8.3.1. Organic Characterization

The major organic constituents in groundwater that exceed the Surface Water Protection Criteria are chlorinated solvents 1,1-DCE, PCE, and TCE. Figures 8-10, 8-11 and 8-12 present the estimated horizontal extent of these three VOCs at concentrations exceeding their respective SWPC. Figure 8-13 presents cross-sections A-A' and B-B', which show the vertical distribution of VOCs in groundwater at concentrations exceeding the SWPC. In addition, three chlorinated solvent hotspots were encountered during the groundwater investigation

1,1-Dichloroethylene

1,1-DCE appears to be the most widespread of the VOCs in groundwater. 1,1-DCE was detected in groundwater at concentrations exceeding the SWPC of 96 ppb beneath the main portion of the facility and adjacent properties (Figure 8-10). The maximum concentration of 1,1-DCE detected in groundwater is 14,000 ppb at the HLA Geoprobe exploration WP-99-48, located between Buildings B-2 and B-12. High concentrations of 1,1-DCE appear in samples with 1,1,1-TCA concentration exceedances. Under methanogenic conditions, 1,1,1-TCA reportedly transforms to 1,1-DCE, which then transforms to vinyl chloride. Given the co-location of 1,1-DCE and 1,1,1-TCA exceedances, and the potential formation of 1,1-DCE from 1,1,1-TCA, the source of 1,1-DCE may be the degradation of 1,1,1-TCA.

The estimated horizontal extent of 1,1-DCE in on-site groundwater crosses the Site in a southeastern direction extending from north of Building B-2 across Sniffens Lane to Building B-6 and from Building B-2 eastward towards the berm. There is an additional off-site source of 1,1-DCE extending from southwest of the West Parking Lot in a northeastern direction across Main Street towards Building B-2 (Figure 8-10).

The estimated vertical distribution of organics in groundwater is represented in two cross-sections which are shown in relation to the Site in Figure 7-1. As shown in Figure 8-13, cross-section A-A', the vertical distribution of 1,1-DCE exceedances in groundwater along the berm extend from DP2-2 (Building B-64) to WC2-5S (past Building B-19). The depths of the exceedances range from approximately 6 feet MSL to -34 feet MSL. In cross-section B-B', 1,1-DCE exceedances extend from WP-99-37 (the Former Chromium Plating Facility) to PZ-8D (Building B-8). The depths of the exceedances range from approximately 6 feet MSL to -84 feet MSL. The depth to bedrock beneath the former Chromium Plating Facility varies from –80 feet MSL to -110 feet MSL, dipping from east to west. Analytical results from explorations CP-99-10 (73 to 75 feet bgs sample) and WP-99-33 (76 to 80 feet bgs sample) indicate that 1,1-DCE is present at concentrations exceeding the SWPC at depths near the bedrock surface. The off-site 1,1-DCE plume is also shown on cross-section B-B' extending from WC3-1I underneath Main Street to PZ-11D. The depths of this plume range from approximately –12 feet MSL to –38 feet MSL.

Tetrachloroethylene

PCE was detected in groundwater at concentrations exceeding the SWPC of 88 ppb beneath the facility and adjacent properties (Figure 8-11). The maximum concentration of PCE detected in groundwater was 4,300 ppb in URSGWC GeoProbe DP3-2, located in the West Parking Lot.

The estimated horizontal extent of PCE in on-site groundwater is located in the area between Buildings B-2 and B-13 expanding in a southeastern direction across Sniffens Lane to the South Parking Lot and towards the berm by Buildings B-34 and B-19. An additional off-site source of PCE was encountered extending from southwest of the West Parking Lot in a northeastern direction across Main Street towards Building B-2 (Figure 8-11). This PCE plume appears unrelated to the plume located on the main site area. Borings installed within Building 2 indicate a clean zone between the two plumes as indicated in Figure 8-13. Several HLA groundwater borings (WP-99-65, WP-99-63, andWP-99-39) did not encounter PCE and 1,1-DCE. The source of the west parking lot PCE is off-site since results indicate the highest concentrations of PCE in direct push borings (DP3-2 and DP3-4) installed along the Sikorsky property boundary.

As shown in Figure 8-13, cross-section A-A', the vertical distribution of PCE exceedances in groundwater along the berm extend from DP2-8 (Building B-34) to PZ-1D (Building B-19). The depths of the exceedances range approximately from -14 feet MSL to -60 feet MSL. In cross-section B-B', PCE exceedances extend from WP-99-37 (the Former Chromium Plating Facility) to PZ-8D (Building B-8). The depths of the exceedances range from approximately 6 feet MSL to -84 feet MSL. The depth to bedrock beneath the former Chromium Plating Facility varies from –80 feet MSL to –110 feet MSL, dipping from east to west. Analytical results from explorations CP-99-10 (73 to 75 feet bgs sample) and WC2-3D (well screen 75 to 85 feet) indicate that PCE is present at concentrations exceeding the SWPC at depths near the bedrock surface. The off-site PCE plume is also shown on cross-section B-B' extending from WC-19S underneath Main Street to PZ-11D. The depths of this plume range from approximately 4 feet MSL to approximately –80 feet MSL.

Trichloroethylene

TCE was detected in groundwater at concentrations exceeding the SWPC of 2,340 ppb beneath the facility (Figure 8-12). The maximum concentration of TCE detected was 830,000 ppb in HLA GeoProbe exploration WP-99-33, located beneath the former Chromium Plating Facility in Building B-2.

The estimated horizontal extent of TCE detected in on-site groundwater extends in a southeastern direction from the northern end of Building B-2 towards the berm by Buildings B-34 and B-19. This principal TCE plume seems to end by the perimeter of the main portion of the facility before Sniffens Lane (Figure 8-12). A second TCE plume originating from an adjacent off-site property has migrated to monitoring wells (MW-19I and MW-19D) located in the west parking lot. The off-site source was confirmed with the installation of direct push borings (DP3-7 through DP3-13) in November 1999.

As shown in Figure 8-13, cross-section A-A', the vertical distribution of TCE exceedances in groundwater along the berm extend from DP2-8 (Building B-34) to PZ-1D (Building B-19). The depths of the exceedances range approximately from -14 feet MSL to -60 feet MSL. In cross-section B-B', TCE exceedances extend from WP-99-63 (Building B-2) to PZ-8D (Building B-8).

The depths of the exceedances range from approximately –10 feet MSL to -84 feet MSL. The depth to bedrock beneath the former Chromium Plating Facility varies from –60 feet MSL to –110 feet MSL, dipping from east to west. Analytical results from WC2-3D (well screen 75 to 85 feet) indicate that TCE is present at concentrations exceeding the SWPC at depths near the bedrock surface.

TCE Hotspots

Two distinct areas exist on-site where the concentrations of TCE exceed the SWPC of 2,340 ppb by at least 100,000 ppb. The first area of concern is located beneath the former Chromium Plating Facility in Building B-2 (Figure 8-12). In this area, TCE was detected in five HLA Geoprobe explorations with concentrations ranging from 110,000 ppb in exploration WP-99-08 to 830,000 ppb in exploration WP-99-33 at depths of approximately 29 feet bgs. The estimated horizontal extent of TCE in groundwater exceeding concentrations of 110,000 ppb covers the majority of the footprint of the former Chromium Plating Facility. The source of the TCE is suspected to be from degreasing conducted as part of the former Chromium Plating Facility operations.

The vertical distribution of TCE in groundwater beneath the former Chromium Plating Facility appears to be influenced by a layer of silt and very fine sand, the top of which is at an elevation of approximately -20 feet MSL (Figure 8-13). The relatively low permeability of the silt and very fine sand appear to have impeded, to a large extent, the vertical migration of the highest concentrations (>110,000 ppb) of TCE in groundwater. The conceptual model for this area is that TCE migrated from the plating facility vertically through the unsaturated zone, into and beneath the water table, pooling on the surface of the silt and very fine sand aquitard. Over time, the TCE has diffused into the silt and very fine sand aquitard. The highest concentration of TCE detected was 830,000 ppb in HLA exploration WP-99-33, immediately above the surface of the aquitard (Figure 8-13). Concentrations of TCE beneath the aquitard are generally less than the SWPC of 2,340 ppb.

The second area of concern where the concentrations of TCE exceed the SWPC of 2,340 ppb by at least 100,000 ppb is located between Buildings B-48 and B-16 (Figure 8-12). In this area, TCE was detected in the HLA Geoprobe exploration WP-99-45 at two intervals between 26 ft and 40 ft bgs, with concentrations of 264,000 ppb and 246,000 ppb, respectively.

The estimated horizontal extent of TCE in groundwater is thought to cover an area approximately 75 feet in diameter. The source of the TCE is suspected to be from degreasing operations conducted in Building B-16.

The vertical distribution of TCE in groundwater at this area appears to be influenced by the layer of sandy silt, the top of which is at an elevation of approximately –8 feet MSL (Figure 8-13). The relatively low permeability of the sandy silt appears to have impeded, to a large extent, the vertical migration of the highest concentrations (>100,000 ppb) of TCE in groundwater. The conceptual model for this area is TCE migrated from the ground surface vertically through the unsaturated zone, into and beneath the water table, pooling within the sandy silt. The TCE concentration of 264,000 ppb in exploration WP-99-45 is within the sandy silt. Concentrations of TCE beneath the sandy silt are generally less than the SWPC of 2,340 ppb. It is also apparent from the cross-section B-B’ (Figure 8-13) that the higher concentrations of TCE (>1,000 ppb) from this area have not reached the berm or the intertidal mudflats.

The depth to bedrock in the vicinity of this area varies from –90 feet MSL to –105 feet MSL, dipping from southeast to northwest. Analytical results from exploration WP-99-45 indicate that TCE concentrations do not exceed the SWPC of 2,340 ppb at a depth of 60 feet. However, the concentration of TCE detected in nearby monitoring well WC2-3D (100 feet south of WP-99-33), screened on the top of bedrock, is 3,100 ppb. This indicates that some TCE has migrated vertically to the bedrock surface, but is localized, as similar concentrations were not encountered elsewhere.

Additional chlorinated VOCs detected at concentrations exceeding SWPC in the vicinity of this area are PCE at 1,900 ppb (WP-99-45, 11-15 feet bgs), and 1,1-DCE at 4,500 ppb (WP-99-45, 11-15 feet bgs). Due to the dilutions required for analysis of samples containing high TCE concentrations, some of the results indicated other VOCs were not detected. Therefore, it is possible that the concentrations of PCE and 1,1-DCE reported above may not be the highest ones in the vicinity of exploration WP-99-45.

The concentrations of TCE in these two areas are indicative of the presence of a non-aqueous phase liquid (NAPL). The solubility of TCE in water is approximately 1,100,000 ppb. The highest concentration of TCE detected in groundwater is approximately 75 percent of TCE's solubility limit in water, which is strongly indicative of the presence of TCE NAPL at both locations. Visual observation of subsurface soil and groundwater samples did not reveal the presence of any TCE NAPL at either location.

1,1,1-Trichloroethane Hotspot

In addition to the two TCE areas of concern, the chlorinated solvent 1,1,1-TCA was detected in groundwater above the SWPC of 62,000 ppb in the center of Building B-2. The maximum concentration of 1,1,1-TCA detected was 280,000 ppb between 32 and 34 feet MSL in HLA exploration CP-99-08.

The estimated horizontal extent of 1,1,1-TCA covers an area approximately 350 feet long by 100 feet wide. The horizontal extent is based on extrapolation of concentrations from HLA explorations CP-99-08 to WP-99-48. The rationale for this interpretation is based on the historical usage of 1,1,1-TCA in degreasing operations in this portion of Building B-2. However, it is possible that the area of 1,1,1-TCA concentrations exceeding 62,000 ppb is actually two distinct areas, each focused around these explorations.

The vertical distribution of 1,1,1-TCA exceedances in exploration CP-99-08 ranges from -32 feet MSL to -62 feet MSL and then again from -133 feet MSL to -158 feet MSL. The conceptual model for this area is that 1,1,1-TCA migrated from the ground surface through the unsaturated zone, into and beneath the water table, to bedrock. The highest concentrations of 1,1,1-TCA detected were 280,000 ppb in exploration CP-99-08 at a depth of approximately –24 feet MSL and 100,000 ppb in exploration WP-99-48 at a depth of approximately –9 feet MSL. It is apparent that 1,1,1-TCA has migrated to the bedrock surface (approximately –158 feet MSL) in the vicinity of exploration CP-99-08. The extent of 1,1,1-TCA near the bedrock surface has been delineated to the southeast of CP-99-08, but is not completely delineated to the east (toward exploration WP-99-48) and northwest (toward exploration CP-99-06).

The concentrations of 1,1,1-TCA in this area are indicative of the presence of NAPL. The solubility of 1,1,1-TCA in water is approximately 1,500,000 ppb. The highest concentration of 1,1,1-TCA detected in groundwater (280,000 ppb) is approximately 19 percent of 1,1,1-TCA's solubility limit in water, which may indicate the presence of 1,1,1-TCA NAPL. Visual observation of subsurface soil and groundwater samples did not reveal the presence of any 1,1,1-TCA NAPL.

8.3.2. Inorganic Characterization

The majority of the inorganic constituents detected in groundwater at concentrations that exceed the SWPC are chromium, hexavalent chromium, arsenic, copper, cyanide, lead, and zinc. Figures 8-14 through 8-19 present the estimated horizontal extent of the metal concentrations that exceed the respective SWPC. Figure 8-20 presents cross sections A-A' and B-B', which depict the vertical distribution of inorganic concentrations in groundwater that exceed the respective SWPC.

Chromium and Hexavalent Chromium

Chromium and hexavalent chromium are present in groundwater beneath the Chromium Plating Facility at concentrations up to 950,000 ppb, which exceeds the SWPC of 1,200 ppb (the criterion for trivalent chromium was used because a criterion for total chromium has not been established) and 110 ppb, respectively (see Figure 8-14). The former plating operations conducted at the facility were the source of this contamination. Plating solutions may have leaked through cracks in the concrete floor or waste lines and migrated to the water table. It is estimated from the groundwater profile figures provided by HLA in the OU2 Report that hexavalent chromium extends to approximately 45 feet below mean sea level at concentrations that exceed the SWPC. This vertical distribution is due to the high density of the plating solutions as compared to groundwater. However, it appears that the relative differences in vertical to horizontal permeability in the fine sands and silts at approximately 30 feet bgs impeded vertical movement of the hexavalent chromium plume, causing the plume to spread horizontally. In addition, the intrusion of denser, saline water beneath the facility may provide an additional barrier to vertical migration of the hexavalent chromium. The mixing zone created by the tidal fluctuation and encroachment of surface water can create a density difference that is sufficient to impede the migration of the hexavalent chromium. The extent of the horizontal distribution is limited to the area under and surrounding the Chromium Plating Facility due to the relatively flat horizontal gradient in this area.

Arsenic

Arsenic concentrations that exceed the SWPC were detected outside the active areas of the Site at concentrations up to 67.4 ppb (see Figure 8-15). The maximum concentration was detected in WC2-4I, which is located near the berm along the Housatonic River. The arsenic contamination detected above the SWPC is widespread throughout the berm area, to the west of Building 2, and the area of the Site south of Sniffens Lane, however no direct source can be attributed. The plumes depicted in Figure 8-20 are extensive due to the number of samples with concentrations that exceed the SWPC. These concentrations may be comparable to background concentrations in this area or characteristic of the fill material.

Copper, Cyanide, and Zinc

Copper and zinc concentrations were detected above the SWPC in the direct push samples collected along the berm (see Figures 8-16 and 8-17). It is likely that the concentrations detected in the direct push samples are not indicative of groundwater quality in this area due to the method used to collect direct push samples. Soil and sediment may be collected with the groundwater in a direct push sample, which may increase the concentration detected in the sample. This assertion is further supported by the fact that these contaminants were not detected above criteria in any of the other samples collected along the berm. Copper and zinc concentrations were also detected above the SWPC in samples collected from monitoring wells located south of Sniffens Lane. River bottom sediments were reportedly dredged from the Housatonic River during the construction of Interstate 95 and then used to fill the wetlands in this area in the 1950's (Sound Environmental, 1999). These sediments most likely contained metals, which may have leached into the groundwater resulting in higher concentrations than elsewhere.

Copper, cyanide, and zinc concentrations were detected above SWPC criteria in several samples collected near the Chromium Plating Facility (see Figures 8-16, 8-17, and 8-18). The presence of these contaminants in the groundwater near the Chromium Plating Facility is likely due to the plating operations that occurred in this facility. Plating solutions may have leaked through cracks in the concrete floor or waste lines and migrated to the water table. The cyanide and zinc contamination is limited to the east side of the facility. The copper contamination is concentrated to the east of the facility, with one exceedance to the west of the facility.

Lead

Four samples collected along the berm indicate lead concentrations in groundwater that exceed the SWPC criterion (see Figure 8-19). Three of these samples were collected by direct push, therefore these results may not be indicative of groundwater quality due to the presence of soil or sediment in the sample which may have artificially increased the concentration. Lead was not detected above the SWPC criterion in the other samples collected along the berm with the exception of the groundwater sample collected from PZ-4D. The contaminant plume associated with this area is limited to the immediate vicinity of this well.. Lead was also detected in samples from monitoring wells in the South Parking Lot and West Parking Lot. The source of this contamination is unknown; these concentrations may be comparable to background concentrations in this area or characteristic of the fill material.

8.4. CONTAMINANT TRANSPORT

There are two basic processes operating to transport solutes: diffusion and advection. Diffusion is the process by which ionic and molecular species dissolved in water moved from areas of higher concentration to areas of lower concentration. Advection is the process by which moving groundwater carries contaminants. As the contaminants are carried through a porous media, the process of dispersion acts to dilute the constituents and lower its concentration and retardation is caused by physical and chemical processes so that the constituents do not move as quickly as advection rates would indicate (Fetter, 1994).

Diffusion

In porous media, diffusion cannot proceed as fast as it can in water because ions must follow longer pathways as they travel around soil grains. In addition, the diffusion can take place only through pore openings because mineral grains block many of the possible pathways. To take this into account, an effective diffusion coefficient (D*) must be used. The value of D* can be determined from the following relationship:

$$D^* = wD$$

where w is an empirical coefficient that is determined by laboratory experiments and D is the diffusion coefficient. For species that are not absorbed onto the mineral surface it has been determined that w ranges from 0.5 to 0.01. Values for D are well known for electrolytes in water. For major cations and anions, D ranges from 1×10^{-9} to 2×10^{-9} square meters pre second (m^2/s). Assuming a value of 0.01 for the empirical coefficient and using 1.1×10^{-8} ft^2/s for D, D^* is 1.1×10^{-10} ft^2/s.

It is possible for constituents to move through porous medium by diffusion, even though groundwater is not flowing. Under such conditions diffusion is more important than advection.

Seepage Velocity

Contaminants that are advecting are moving at the same rate as the average linear velocity of the groundwater, however water and contaminants can only move through the pore spaces in the soil. The effective porosity is incorporated in a calculation to determine the velocity at which water is actually moving (seepage velocity). The effective porosity is the portion of the space through which saturated flow occurs, and is assumed to be equal to the specific yield. The seepage velocity can be determined from Darcy's Law:

$$v_S = \frac{K}{n_e}\frac{dh}{dl}$$

where v_s = seepage velocity

K = hydraulic conductivity

n_e = effective porosity

$\frac{dh}{dl}$ = hydraulic gradient

Using values determined during the physical conceptual model for hydraulic conductivity (29.36 ft/day), effective porosity (0.24) and hydraulic gradient (0.00045 ft/ft), the average linear velocity is 0.055 ft/day.

Dispersivity

Mechanical dispersion is also caused by the heterogeneities in the aquifer. As groundwater flow proceeds in an aquifer, regions of greater than average hydraulic conductivity and vice versa will be encountered. The resulting variation in linear groundwater velocity results in much greater hydrodynamic dispersion than that caused by the pore-scale effects. The greater the distance over which dispersivity is measured, the greater the value that is observed. Neuman (1990) made a study of the relationship between the apparent longitudinal dynamic dispersivity and the flow length. He found that for flow paths less than 10,000 feet long, these could be related by the equation:

$$a_L = 0.0175L^{1.46}$$

where a_L = apparent longitudinal dynamic dispersivity (ft)

L =length of flow path (ft)

Since the interest of the contaminant plumes migration is whether any impacts to the surface waters may occur, the distance to the Housatonic River was used for L which ranged from approximately 437 feet to 1,437feet depending on the location of the applicable highest contaminant concentration. The apparent longitudinal dynamic dispersivity of contaminants on-site ranged from 126 feet for 1,1-DCE to 713 feet for TCE. The results of the apparent longitudinal dynamic dispersivity calculations is included in Table 8-1.

Hydrodynamic Dispersion

The process of molecular diffusion and mechanical dispersivity cannot be separated in flowing groundwater. So the coefficient of hydrodynamic dispersion (D_L) takes into account both mechanical mixing and diffusion. For one-dimensional flow it is expressed by the following equation:

$$D_L = a_L v_s + D^*$$

where D_L = longitudinal coefficient of hydrodynamic dispersion (ft²/day)

a_L = apparent longitudinal dynamic dispersivity coefficient (ft)

v_s = seepage velocity (ft/day)

D^* = molecular diffusion coefficient (ft/day)

The longitudinal coefficient of hydrodynamic dispersion of contaminants on-site ranged from approximately 40 ft²/day for TCE to 91 ft²/day for PCE. The results of the longitudinal coefficient of hydrodynamic dispersion calculations are included in Table 8-1.

Contaminant Advection Calculations

The concentration of a contaminant at some distance from the source concentration can be calculated from the following equation (Ogata, 1970):

$$C = \frac{C_0}{2}\left[erfc\left(\frac{L - v_s t}{2\sqrt{D_L t}}\right) + \exp\left(\frac{v_s L}{D_L}\right) erfc\left(\frac{L + v_s t}{2\sqrt{D_L t}}\right)\right]$$

where C = solute concentration (ug/L)

C_0 = initial solute concentration (ug/L)

L = flow path length (ft)

v_s = seepage velocity (ft/day)

t = time since release (day)

D_L = longitudinal coefficient of hydrodynamic dispersion (ft^2/day)

erfc =complimentary error function

The distance along the groundwater flowpath to the closest downgradient receptor from the highest concentration was used as L and 10 years was used as the time period. Results indicate that concentrations of 1,1-DCE, PCE, TCE, hexavalent chromium, and chromium will exceed the SWPC upon reaching this distance. Table 8-1 summarizes the results of the contaminant diffusion calculations.

Contaminant Retardation Calculations

Additional chemical process can affect the movement of contaminants in groundwater so that it may not move as quickly as the advection and diffusion only. Retardation can reduce the movement significantly depending on physical conditions of the Site and the particular contaminant. The retardation factor can be determined using the following equation:

$$R = 1 + \left(\frac{P_b * K_{oc} * foc}{\Phi} \right)$$

where R = retardation factor (dimensionless)

P_b =soil bulk density (g/cm^3)

K_{oc} =partition coefficient (ml/g)

foc =soil organic carbon content (dimensionless)

Φ = effective porosity (dimensionless)

A soil organic carbon content of 0.005, a soil bulk density of 1.45, and effective porosity of 0.24 were used for all the calculations. The partition coefficient varied based on the contaminant included in the calculation. Retardation factors for 1,1-DCE (7.69), TCE (5.69) and PCE (10.34) were calculated.

The pollutant transport rate can be calculated by dividing average seepage velocity by the retardation factor. The pollutant transport rate was calculated for 1,1-DCE (2.69 ft/yr), TCE (3.64 ft/yr), and PCE (2.00 ft/yr). Based on the above pollutant transport rates for 1,1-DCE, TCE, and PCE, it would require more than 300 years for the chlorinated solvents to contact the Housatonic River. The retardation transport calculations are included in Appendix Q and summarized in Table 8-2.

Summary of Contaminant Transport

Contaminant transport at SAEP is very complex due to the horizontal and vertical variability of the hydrostratigraphy and the nature of the contaminants present within the groundwater. In addition, the bedrock elevation appears to have an influence on the high degree of variability in hydraulic gradients found at the Site. The tidal fluctuations and surface water intrusion present along the shoreline may also influence the contaminant transport. Other influences include the presence of a groundwater divide and the differing groundwater flow direction in the western portion of the study area.

The findings of this investigation indicate that the hydrostratigraphy at SAEP is a major influence on contaminant transport. The chromium discussion in Section 8.3.2 is one example of the interaction between contaminant movement and the stratigraphy. The higher contaminant transport velocities occur in the more permeable sections of the overburden deposits. These data emphasize the importance of the more permeable zones of the aquifer, especially because 1) these zones are where the discharge is occurring; and 2) these zones may provide preferential pathways for the transport of contaminants within the aquifer.

The presence of particular chlorinated solvents and the shape and extent of the VOC plumes indicate that natural attentuation and degradation is occurring at the Site. It has been documented that chlorinated solvents including 1,1,1-TCA, PCE, TCE, and 1,1-DCE degrade through processes of natural attenuation. 1,1,1-TCA will degrade to 1,1-DCE; and PCE degrades to TCE, which degrades to DCE. As the parent product concentration decreases the daughter product concentrations will temporarily increase. DCE will also degrade to daughter products such as vinyl chloride and ethene. For instance one of the chlorinated solvent source areas is located at the former Chrome Plating Facility. The groundwater direction in this area is generally toward the northeast. Groundwater and contaminant flow may explain the migration of PCE from the Chrome Plating Facility to the edge of Building 19 (Figure 3-5) and the degradation of PCE may account for the TCE and 1,1-DCE contaminant plumes which are centrally located across the main facility.

Further evidence is found by the presence of high concentrations of 1,1-DCE in samples with 1,1,1-TCA concentration exceedances. Under methanogenic conditions, 1,1,1-TCA reportedly transforms to 1,1-DCE, which then transforms to vinyl chloride. Given the co-location of 1,1-DCE and 1,1,1-TCA exceedances, and the potential formation of 1,1-DCE from 1,1,1-TCA, the source of 1,1-DCE may be the degradation of 1,1,1-TCA. HLA reported that the groundwater temperatures outside this hotspot area ranged from 12°C to 16°C, however groundwater temperatures in this area range from approximately 21°C to 25°C (HLA, 2000). The elevated groundwater temperatures may be indicative of biological degradation in the area of the hotspot.

The hydraulic gradient has a large influence on the movement of groundwater contaminants. Horizontal gradients vary widely across the main portion of the facility (Table 7-1). The horizontal gradient in the vicinity of Building 2 is relatively flat (0.0001 ft/ft) due to bedrock lows. In contrast, the gradient increases dramatically near the river to the northeast (0.002 ft/ft). The estuarine silts likely act as a low permeability boundary to groundwater flow in this area resulting in the increased gradient. The variable hydraulic conductivity has created the shape and extent of the current groundwater plumes. The low hydraulic gradient beneath Building 2 has helped inhibit the horizontal movement of a large portion of the plume where plume migration is likely the result of diffusion rather than advective transport.

There are several unique flow conditions that were detailed in Section 7.0 that may potentially affect contaminant migration at SAEP. The tidal fluctuations caused by the Housatonic River can potentially impact contaminant transport. Studies at sites with tidal fluctuations caused by nearby surface water bodies have established that a major effect of tidal fluctuation on the migration process of a contaminant plume is the exit concentration discharging to the tidal estuary. These studies have demonstrated that concentration levels close to the bank of the estuary will be relatively low if the surface water is free of contamination. This is due to dilution from the surface water infiltrating inland during the high-tide cycles.

In general, tidal fluctuations hasten the rate of plume migration near the bank of the estuary because of the relatively high advective and dispersive fluxes induced by tides. However, tides affect migration process only within a short distance from the tidal water interface. Since the majority of the contaminant plume is located outside the interface, tidal fluctuation will not affect the rate of plume migration until an existing regional velocity brings the plume to the tidally active zone. (Chan, 1992).

The SAEP tidal study results indicate that the groundwater fluctuation is approximately one to two in monitoring wells along the berm. The fluctuations are minimal beyond 100 feet from the berm. These conditions are similar to those at the sites included in the tidal fluctuation and contaminant transport. Although the fluctuations may increase contaminant transport in localized areas, without or with tides, the rate of contaminant migration increases with higher regional hydraulic gradient. Furthermore, the effects of tidal fluctuations on the transport process become insignificant with higher regional hydraulic gradients.

Another condition possibly influencing contaminant transport is the groundwater divide located on the western portion of the Site. A large area of slight groundwater mounding encountered in the center of the Site beneath Building 2 extending southwest to the west and south parking lots and Sikorsky Airport may act as a groundwater divide. Another smaller groundwater divide is present on the north side of Building 2 where groundwater east of the divide flows toward the Housatonic River and groundwater west of the divide flows toward Frash Pond. Both groundwater divides change groundwater flow in the western portion of the Site to a northwesterly flow. The groundwater divides have affected contaminant transport at the Site. The divide extending from Building 2 to Sikorsky airport appears to have resulted in off-site contaminants impacting groundwater in the west parking lot. However, it has also aided in prohibiting contamination from the main facility from migrating to the west and south parking lots. The smaller divide has prevented on-site contamination from impacting areas to the north and west.

The results of contaminant transport calculations were conflicting between the advection/dispersion and retardation analysis. The advection/dispersion results indicated that a majority of the contaminants will reach the Housatonic River in significant concentrations within ten years. However, evidence from the remedial investigation questions the accuracy of the advection/dispersion transport results. Results of the remedial investigation indicate that the rate of predicted contaminant migration is not occurring and it is likely that any initial impacts to the groundwater occurred more than ten years ago. The advection/dispersion calculations assume that contaminant transport is the result of advection only and therefore ignores several important processes in contaminant transport which occur in nature. It appears from the advection/dispersion calculations that other physical and biological processes are important factors in analyzing contaminant transport at SAEP.

The contaminant retardation calculations do incorporate some of the physical and biological processes in the analysis. The retardation calculation results indicate significantly reduced contaminant transport values as compared to the advection/dispersion calculations. The results compared favorably to actual groundwater conditions as observed during the remedial investigation. However, the retardation calculations are limited to organic compounds only and this analysis was not performed on inorganic contaminants present in the groundwater at SAEP.

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