River Terrace RV Park Remedial Investigation/ Feasibility Study Report

River Terrace RV Park
Soldotna, Alaska

Final

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Prepared for:

Alaska Department of Environmental Conservation

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ACRONYMS AND ABBREVIATIONS

µg/kg	micrograms per kilogram
µg/L	micrograms per liter
ACL	alternative cleanup level
ADEC	Alaska Department of Environmental Conservation
ADOT	Alaska Department of Transportation
AOC	administrative order on consent
ARAR	Applicable or relevant and appropriate regulations
bgs	below ground surface
BTEX	benzene, toluene, ethylbenzene, xylenes
$CaCO_3$	calcium carbonate
CERULA	Comprehensive Environmental Response, Compensation, and Liability Act
cm^2	square centimeters
cm/sec	centimeters per second
cm^2/sec	centimeters squared per second
COCs	contaminants of concern
CSM	conceptual site models
DCE	dichloroethene
DNAPL	dense non-aqueous phase liquid
DO	dissolved oxygen
DQO	data quality objectives
DRO	diesel range organics
E&E	Ecology and Environment
EPA	Environmental Protection Agency
ET	Ecotox Thresholds
ft	feet/foot
ft/day	feet per day
ft^2/day	square feet per day
FS	Feasibility Study
g/cm^3	grams per cubic centimeter
gpm	gallons per minute
GRO	gasoline range organics
H_2O	water
HCR	hydrogen release compound
KEC	Kennard Environmental Consultants
k_{oc}	organic carbon partitioning coefficient
k_{ow}	octanol-water partitioning coefficient
KRBO	Kenai River Bridge Outfall
L/kg	liters per kilogram
L/m^3	liter per cubic meter
m	meters
mi^2	square miles
m^3-atm/mol	cubic meters X atmospheres per gram mole (units for Henry's Law constant)
MCL	maximum contaminant level
mg/kg	milligrams per kilogram
mg/L	milligrams per liter

ACRONYMNS AND ABBREVIATIONS
(continued)

MH	manhole
MP	measuring point
mV	millivolts
MW	monitoring well
NA	not available
NCP	National Contingency Plan
ND	non-detect
ºC	degrees Celsius
ºF	degrees Fahrenheit
OASIS/Bristol	OASIS Environmental Inc./Bristol Environmental Services
OH^-	hydroxide ion
OHW	ordinary high water line
ORP	oxidation-reduction potential
OSWER	Office of Solid Waste and Emergency Response
PAHs	polynuclear aromatic hydrocarbons
PCE	tetrachloroethene
PID	photo ionization detector
POTW	publicly owned treatment work
ppb	parts per billion
ppm	parts per million
RAO	remedial action objectives
RCRA	Resource Conservation & Recovery Act
RI	Remedial Investigation
RI/FS	Remedial Investigation and Feasibility Study
ROD	record of decision
ROW	right-of-way
RP	responsible party
RRO	residual range organics
RTRVP	River Terrace RV Park
RV	recreational vehicle
SB	soil boring
SESOIL	seasonal soil compartment
SHBO	Sterling Highway bridge outfall
SQB	Sediment Quality Benchmarks
SQC	sediment quality criteria
SVE	soil vapor extraction
TBD	to be determined
TCA	trichloroethane
TCE	trichloroethene
TOC	total organic carbon
torr	unit of measurement for pressure
USEPA	U.S. Environmental Protection Agency
VC	vinyl chloride
VOCs	volatile organic compounds
WQC	water quality criteria
WQS	water quality standards

EXECUTIVE SUMMARY

A remedial investigation/feasibility study (RI/FS), which is a detailed site investigation and evaluation of remedial alternatives, was conducted at the River Terrace Recreational Vehicle (RV) Park (RTRVP). RTRVP is a former dry cleaner located on the bank of the Kenai River in Soldotna, Alaska. Contamination identified at the site includes the common dry cleaning solvent tetrachloroethene (PCE), along with its degradation products trichloroethene (TCE), dichloroethene (DCE), and vinyl chloride (VC).

The overall objective of this RI/FS is to characterize the extent of contamination by PCE and its degradation products at the RTRVP, and to develop remedial alternatives to address the contamination. The RI fills in data gaps in existing site characterization data collected since early 1997. The FS identifies remedial alternatives that must ensure that regulatory requirements are met, e.g., contamination levels on site are below existing site alternative cleanup levels (ACLs) and that the contamination does not enter the Kenai River at unsafe levels (above State of Alaska Water Quality Standards [WQSs]).

Site Background

Evidence of contamination at the RTRVP was first observed by the Alaska Department of Environmental Conservation (ADEC) during investigation of a leaking drum complaint in 1992. Petroleum hydrocarbons and PCE were detected during initial assessment work in 1992. During installation of a water line on the RTRVP property in 1996, heavily contaminated soils were encountered. A soil grid sampling investigation was performed in early 1997, which indicated widespread (2.3 acres) PCE contamination in site soil, at levels up to 4,700 milligrams per kilogram (mg/kg). A series of investigations followed the 1997 soil grid sampling, and a quarterly groundwater monitoring program was initiated in summer 1997.

Several different cleanup operations have been performed at the RTRVP site to-date. Cleanup operations began with an *in situ* soil vapor extraction system to remove PCE from site soils. The system was installed and operated without an approved ADEC work plan, therefore few details are known about it. It was probably operated from early 1996 until sometime in 1997. After the water line excavation, approximately 600 cubic yards of contaminated soil from the excavation were placed into treatment cells. Laboratory analysis confirmed the presence of PCE above characteristic hazardous waste levels. In 1997 and 1998, approximately 3,300 cubic yards of soil were removed and treated on-site under the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) Removal Action.

Several different regulatory cleanup levels are relevant for the RTRVP site. The different regulatory levels are designed to protect the environment for various uses. Exceedences of any of these levels indicate potential risk to the protected users.

- ACLs were established for site soil and shallow groundwater (including all aquifers between the lower drinking water aquifer and surface grade) by the ADEC in 1997.
- Drinking water maximum contaminant levels (MCLs) apply to all water outside of the RTRVP property, including the Kenai River. The MCLs are expected to be protective of the above uses in addition to contact recreation use, agricultural use, and drinking water.
- Sediment quality benchmarks are designed to be protective of benthic organisms in river sediments. The sediment benchmarks are not regulatory criteria; they

represent screening levels for use in evaluating ecological risk. Exceedences of sediment quality benchmarks indicate need for additional evaluation (e.g., an ecological risk assessment).

The ACLs are much higher than comparable MCLs, WQSs, or ADEC soil cleanup levels. For example, the groundwater ACL for PCE is 840 micrograms per liter (µg/L) (versus the MCL and WQS of 5 µg/L), and the soil ACL for PCE is 11.5 mg/kg (versus the relevant ADEC 18 AAC 75 soil cleanup level of 0.03 mg/kg).

Even after the Removal Action, some of the remaining soil and groundwater concentrations continue to exceed the ACLs, and contamination has been detected in Kenai River surface water and sediment samples. Limited surface water sampling has indicated PCE levels up to 2.5 ug/L in the Kenai River water column adjacent to the site. Contamination levels above ecological benchmarks were also identified in Kenai River sediments.

The need for the current RI/FS is indicated by continued exceedences of the ACLs, no conclusive evidence of decreasing contaminant concentrations in groundwater, and data gaps in the site characterization data.

Remedial Investigation

The overall objective of the remedial investigation is to collect sufficient data to design and implement a remedial alternative for the site. The remedial investigation was performed in three phases between June 1999 and January 2000. The first phase focused on searching for a source area for the known groundwater plume (now termed the lower contaminant plume). During this phase, a previously unknown area of groundwater contamination was discovered in a location adjacent to the former dry cleaner building, uphill from the known spill site. The second and third phases were focused on delineating the extent of contamination in the newly discovered area, termed the upper contaminant plume.

In the RI, additional monitoring wells were installed, soil and groundwater samples were collected, and aquifer parameters (hydraulic conductivity and groundwater gradient) were measured. Data from three groundwater sampling events (July, October, and December 1999) are included in this RI. A soil gas survey was conducted to evaluate the upper contaminant plume source area. A detailed analysis was performed to evaluate the results of the investigative work. Significant conclusions are summarized below:

RI General Conclusions

- Groundwater flow and aquifer conditions at the RTRVP were better defined. Groundwater at the site was categorized into three different water-bearing zones: a deep confined aquifer, hypothesized shallow semi-confined water-bearing zones, and an upper unconfined aquifer. The upper unconfined aquifer is contained in sandy fill material across most of the site and in alluvium along the Kenai River.
- A ridge in the glacial till underlying the unconfined aquifer is interpreted to run in approximately a northwest-southeast orientation across the site, from MW-30 to MW-24 south of the former dry cleaner building.
- A groundwater divide (or ridge) is interpreted to generally follow the till ridge across the site. The divide separates site groundwater contamination into two plumes. The lower contaminant plume, which includes the area of the 1997-1998 excavation, extends south of the former dry cleaner building to the Kenai River. The upper

contaminant plume, which was discovered during RI activities, extends north of the former dry cleaner building toward the Sterling Highway.

- The source of the lower contaminant plume is interpreted to be conta0minated soil near but outside the area of the 1997-1998 excavation. Excavation perimeter soil sampling showed widespread soil contamination at levels below the ACL and several locations of soil contaminated at levels above the ACL. The 1997 grid soil sampling and excavation perimeter samples indicate that multiple source areas are likely.
- The source of the upper contaminant plume is near or under the former dry cleaner building. This source area is north of the 1997 grid sampling area and the 1997-1998 excavation area.
- The distribution of PCE soil and groundwater contamination at the site is complex. Complex contamination patterns are common at sites contaminated by PCE. PCE is heavier than water and readily sinks down through the soil. Literature research indicates that minor changes in soil type can have a major effect on the migration of PCE. The RTRVP site geology is very complex; thus a complex distribution of PCE in site soil and groundwater would be expected.
- An analysis of potential spill volume and age was performed for the site. Based on this analysis, a minimum remediation time frame of 15 years is a reasonable estimate of the amount of time needed for PCE at the site to attenuate to the RAOs, assuming the majority of PCE was spilled in the 1980s and will migrate through the lower area.

Conclusions Regarding the Lower Plume (south of the groundwater divide)

Impact of the RTRVP on the Kenai River

- A comparison of Kenai River stage information and water levels recorded in a monitoring well adjacent to the river (MW-20) during the period between September 1, 1999 and January 5, 2000 indicates that the groundwater is in direct communication with the river.
- PCE and its degradation products have been detected in Kenai River sediments adjacent to the RTRVP at levels above sediment quality benchmarks for PCE, TCE, and cis-1,2-DCE.
- The presence of PCE and its degradation products (TCE and cis-1,2-DCE) in the Kenai River water column downslope from the lower plume (e.g., between MW-6 and MW-8) is consistent with groundwater monitoring data that indicates contaminants have entered the river via groundwater contamination.
- Fish and Game Department studies indicate that water quality in the lower Kenai River has been compromised, although the cause of the decrease was not determined.
- RI sampling indicated that contamination continues to leave the RTRVP property at levels above the ACLs. The following conclusions are based on samples collected from a line of sentry wells located approximately 20 feet upslope from the Kenai River, extending from the middle of the Sterling Highway Bridge approximately 175 feet eastward across the RTRVP site.
  - In RI sampling between July 1999 and January 2000, PCE was detected at levels above its groundwater ACL (840 µg/L) in one sentry well sample (MW-6 at 980 µg/L in October 1999). PCE concentrations from two monitoring wells located within 60 feet of the river also exceeded the groundwater ACL (MW-9 at

940 µg/L in October 1999 and 1,800 µg/L in December 1999 and MW-10 at 970 µg/L in December 1999).

  - In RI sampling between July 1999 and January 2000, VC was detected at levels above its groundwater ACL (2 µg/L) in samples from two sentry wells (MW-6 at 3 µg/L in July and 3.7 µg/L in October 1999, and MW-20 at 7.6 µg/L in July and 3.3 µg/L in October 1999).
  - In RI sampling between July 1999 and January 2000, PCE has been detected above its WQS (5 µg/L) in all samples collected from all monitoring wells adjacent to the Kenai River from the middle of the Sterling Highway Bridge approximately 175 feet eastward to MW-5 (e.g., MW-13, MW-12, MW-8, MW-20, MW-6, MW-7, MW-5). In addition, PCE degradation products TCE and cis-1,2-DCE were detected in many samples from these monitoring wells at levels above their WQSs.

- Based on the distribution of PCE concentrations in samples from the line of sentry wells (MW-5 to MW-27), the western boundary of the PCE plume appears to be captured by the sentry wells. Detections of contaminants in MW-27 have been below ACLs and WQSs; therefore, MW-27 is interpreted to represent a reasonable western limit of groundwater contamination for the site. Detections of contaminants in MW-5 have been below ACLs but above WQSs; therefore, MW-5 is not interpreted to represent the eastern limit of groundwater contamination. The eastern contamination limit has not been delineated.

Evaluation of Lower Contaminant Plume Characteristics

- Between MW-5 and MW-12 (approximately 120 lineal feet), approximately 4.5 gallons per minute (gpm) of water contaminated by PCE and its degradation products is migrating off-RTRVP property into the Kenai River. This estimate is based on hydraulic gradients measured in July and November 1999, and does not consider riverbank icing effects observed in December 1999.
- Approximate groundwater and PCE travel times from MW-26 to the Kenai River ranged between almost 1 year and almost 5 years, respectively, based on measured hydraulic gradients and calculated hydraulic conductivities (slug test results). The approximate groundwater and PCE range of travel times from MW-9 to the Kenai River were 17 days and 895 days, respectively. The dissolved PCE transport velocity is slower than groundwater velocity due to the adsorption of PCE onto organic carbon in the aquifer material.
- Natural attenuation appears to be occurring in site groundwater from approximately MW-4A to the river. Natural attenuation is indicated by favorable geochemical results and the presence of PCE degradation products. However, contaminant levels in these monitoring wells remain high (i.e., above the ACLs for PCE in groundwater in some of these wells and above the WQSs for PCE in all of these wells), so natural attenuation is not degrading the PCE or its degradation products sufficiently to meet WQSs at the ordinary high water (OHW) line and protect the Kenai River in the near future.
- Visual trend analyses of PCE concentrations and total PCE degradation product concentrations were performed for nine monitoring wells across the lower contaminant plume. Three of the nine characteristic monitoring wells exhibit an apparent trend of increasing PCE concentrations over time (MW-9, MW-11, and MW-13). One monitoring well exhibits an apparent trend of decreasing PCE concentrations over time (MW-4A), although PCE concentrations detected in that

monitoring well remain considerably above the site ACL for PCE in groundwater. No visible trend is apparent in the concentrations in the remaining monitoring wells (MW-6, MW-8, MW-10, MW-14, and MW-19).

Conclusions Regarding the Upper Plume (north of the former dry cleaner building)

- Immediately north of the groundwater ridge (near the former dry cleaner building), groundwater flows in a generally northerly direction toward the Sterling Highway. However, evidence exists that this northerly flow direction may be only a local phenomenon (i.e., MW-32 has a higher water table elevation than MW-23, and MW-22 has a higher water table elevation than MW-25).
- A plume of PCE-contaminated groundwater extends from MW-16, located on the northwest side of the former dry cleaner building, northward toward MW-25 and the Sterling Highway. Four groundwater samples were collected from MW-16 between July 1999 and January 2000; all results exceeded the ACL for PCE (840 µg/L). A minimum concentration of 1,000 µg/L was detected in July 1999, and a maximum concentration of 5,500 µg/L was detected in September 1999.
- The upgradient extent of the upper plume is located near or under the former dry cleaner building. A soil gas survey conducted along the northwest and northeast sides of the former dry cleaner building encountered high (ranging from 860 parts per billion [ppb] to 13,000 ppb) PCE vapor concentrations spread relatively uniformly over the area of investigation, with the highest vapor concentrations in a sample adjacent to the former dry cleaner building.
- The downgradient extent of the upper contaminant plume is not known, but it likely extends to the Sterling Highway. The lack of PCE detections above MCLs in monitoring wells installed on the west side of the Sterling Highway (MW-28, MW-30, MW-31, and MW-34) indicates that the plume does not appear to extend beyond the Highway.
- The storm sewer backfill under the west side of the Sterling Highway is a hypothesized preferential flow path from the upper contaminant plume to the Kenai River, based on the contamination detected within the storm sewer system, the groundwater flow patterns, and the lack of contamination on the west side of the Sterling Highway. It is possible that other utility corridors or the Sterling Highway road bed are also preferential flow pathways.
- Interpretive maps of the top of the till surface and the water table contours indicate that the storm sewer system likely drains site groundwater north of the groundwater divide. The till surface was apparently excavated for the storm sewer system installation, and the backfill surrounding the storm sewer and the storm sewer itself provide a preferential flow pathway for groundwater.
- In RI sampling between September 1999 and January 2000, one of three groundwater sample results from MW-25, located approximately on the RTRVP property boundary, exceeded the ACL for PCE in groundwater (920 µg/L in September). This exceedence indicates that PCE has migrated off-RTRVP property at a level exceeding its groundwater ACL and well above the MCL of 5 µg/L.
- Natural attenuation does not appear to be occurring in the upper groundwater plume. Geochemical results were not favorable for natural attenuation, and very limited PCE degradation product concentrations have been detected in groundwater.
- Between MW-21 and MW-22 in the upper plume area (approximately 115 lineal feet) approximately 2.4 gpm of water contaminated by PCE is migrating off-property. This

estimate is based on hydraulic gradients measured in July through December 1999 and is not expected to have a significant seasonal variability.

- Approximate groundwater and PCE travel times from MW-16 to the storm sewer backfill ranged between 40 days and 200 days, respectively, estimated from measured hydraulic gradients and calculated hydraulic conductivities (slug test results). The approximate groundwater and PCE travel times from MW-25 to the storm sewer ranged between 22 days and 110 days, respectively. The dissolved PCE transport velocity is slower than groundwater velocity due to the adsorption of PCE onto organic carbon in the aquifer material.
- Based on the Johnson-Ettinger model for subsurface vapor intrusion into buildings, the carcinogenic risks posed by residential exposure to PCE vapors entering the basement of the former dry cleaner building from groundwater and subsurface soil were estimated at 4.6 X 10^{-5} and 5.2 X 10^{-6}, respectively. When risk screening indicates potential risk greater than 1 X 10^{-6}, additional evaluation of potential exposure is usually warranted.

Feasibility Study

Information gathered during the remedial investigation was used to develop the feasibility study. Seven different remedial alternatives were evaluated for both the upper contaminant plume and the lower contaminant plume. Those alternatives that met the threshold criteria of protecting human health and the environment and meeting 18 AAC 75 and other applicable state and federal regulations (hereinafter referred to as Applicable or Relevant and Appropriate Requirements) were evaluated in detail. In addition, intrinsic remediation was evaluated for each plume, although this remedial option will not meet the specified remedial action objectives (RAOs). Each alternative was scored using the five balancing criteria found in the federal National Contingency Plan (NCP), which are applicable to an FS evaluation. The scores provide a means to compare the different alternatives.

The alternatives share some common elements: all contain a long-term monitoring requirement, 5 or 15 years was used as the basis for treatment time frame, and the design criteria was achievement of WQSs in the Kenai River, as measured in the sentry wells. In practice, this means that the WQSs must be met in the sentry wells, with some allowance for attenuation between the wells and the river.

The alternatives considered for each area and their respective scores are summarized in Table E-1.

Any of the alternatives, with the exception of the “no action” alternative and “intrinsic remediation” alternative, will achieve the RAO of WQSs at the river. The treatment time frame, community concerns, plans for development, and other factors should be considered before a final remedial alternative is selected.

Table E - 1: Comparative Analysis of Remedial Alternatives

PART 1 BACKGROUND

1 INTRODUCTION

Under Contract No. 18-2-12-12, for the Alaska Department of Environmental Conservation (ADEC), OASIS Environmental, Inc. and Bristol Environmental & Engineering Services Corp. (OASIS/Bristol) performed a remedial investigation and feasibility study (RI/FS) at the River Terrace Recreational Vehicle (RV) Park (RTRVP) area in Soldotna, Alaska. Soil and groundwater contamination by tetrachloroethene (PCE) and its degradation products and petroleum products has been documented on the property commonly referred to as the RTRVP. The RTRVP property and adjacent Alaska Department of Transportation (ADOT) right of way (ROW) are referred to as the RTRVP site throughout this report. The site is located at 44761 Sterling Highway on the north bank of the Kenai River in the City of Soldotna, Alaska, Section 32, Township 5N, Range 10W, of the Seward Meridian in the Kenai Peninsula Borough.

The RI field activities were conducted in three phases. Phase I was completed in June/July 1999, Phase II in September 1999, and Phase III in November 1999 and January 2000. In addition, quarterly groundwater monitoring events were performed in conjunction with the June/July RI investigation and again in October and December, 1999. The results of all June 1999 through January 2000 site investigation activities, including the RI activities and three quarterly monitoring events, are presented in this report.

1.1 OBJECTIVES

The overall objective of this RI/FS is to characterize the extent of contamination by PCE and its degradation products at the RTRVP site, and to develop remedial alternatives to address the contamination. The remedial alternatives must ensure that regulatory requirements are met, e.g., contamination levels on site are below existing site alternative cleanup levels (ACLs) and that the contamination does not enter the Kenai River at unsafe levels (above State of Alaska Water Quality Standards [WQS]).

Contaminants of concern (COCs) for this RI/FS include PCE and its degradation products. Although petroleum products were documented in site soils during initial investigations, they were not retained as COCs. The bulk of the petroleum contamination was removed during a 1997 excavation, and petroleum products have not been detected in monitoring wells at levels exceeding ADEC groundwater cleanup levels.

1.1.1 Remedial Investigation Objectives

The primary objective of the RI was to delineate where COCs remain at the RTRVP site. The RI investigative work was performed in three phases, because each of the first two phases identified significant data gaps that were addressed in subsequent field efforts.

Phase I of the RI was designed to fill in data gaps in the existing monitoring well network and evaluate soil contamination remaining at the site. Specific objectives of the first phase RI effort are listed below.

- Install new monitoring wells (designated "MW-15" through "MW-20") and soil borings (designated "SB-01" and "SB-02") to fill data gaps in the existing monitoring network. Soil samples were used to determine whether additional soil contamination remains at the site, and the new monitoring wells were used to assist in the delineation of the dissolved-phase PCE plume.

- Collect groundwater samples from all previously-existing and newly-installed monitoring wells at the site (e.g., perform the previously-scheduled June quarterly monitoring event).
- Survey all previously-existing and newly-installed monitoring wells at the site.
- Conduct a fate and transport analysis to support an FS, using the results of the 1999 field effort in conjunction with previous monitoring results and an understanding of PCE transport and degradation properties.

Initial conclusions of the RI program were that a previously unknown area of contamination exists near the former dry cleaner building. The primary purpose of the Phase II RI was to investigate this area of contamination. Specific objectives of the Phase II RI effort are listed below.

- Install five additional monitoring wells to establish groundwater flow direction and potentially place either upgradient or downgradient boundaries on the contamination.
- Install an additional monitoring well in or near the excavation backfill in the vicinity of the soil cells to evaluate groundwater quality in this area.

After evaluating the results of the Phase II RI work, additional Phase III work was performed to confirm hydraulic containment of the PCE plume by the storm sewer structure, verify groundwater flow directions, evaluate hydrogeologic anomalies, advance a soil boring under the former dry cleaner building, and perform a soil gas investigation to assess the presence of a PCE source area.

Specific objectives of the Phase III RI effort are listed below.

- Install two additional monitoring wells near the northern site boundary and two across the Sterling Highway to the west of the site,
- Install one well point (MW-27) in backfill near the storm sewer outfall,
- Install one temporary monitoring well (MW-28) through the Sterling Highway,
- Install two angle borings/monitoring wells beneath the building,
- Install and sample 19 soil gas points,
- Conduct slug tests in six monitoring wells, and
- Install data loggers in 5 site monitoring wells (MW-15, MW-16, MW-19, MW-20, and MW-24) to better understand site groundwater behavior.

After analysis of the Phase III results, a third monitoring well (MW-34) was installed on January 13, 2000 to evaluate an area where a narrow plume could have crossed the highway. MW-34 is located across the Sterling Highway to the west of the site, between MW-31 and MW-28.

1.1.2 Feasibility Study Objectives

The primary objective of the FS was to evaluate remedial alternatives that are potentially appropriate for addressing remaining contamination at the RTRVP site. The primary goal of site remediation is to prevent, to the degree practical, migration of PCE and its degradation products into the Kenai River and away from the site.

1.2 REPORT ORGANIZATION

This report is organized into three main sections. The first section contains introductory information, describes the site setting and previous investigation results, and contains a detailed discussion of contaminant characteristics. The second section discusses the RI scope of work, findings, and conclusions. The third section is the FS. In the FS, remedial action objectives (RAOs) are defined, remedial alternatives are introduced and explained, and a detailed analysis of remedial action alternatives is presented.

The organization and content of this report are summarized below:

- Section 1:
  - Introduction: defines the purpose and organization of the RI/FS and summarizes the site setting and historical use.
  - Environmental Setting: defines the environmental setting of the RTRVP site.
  - Previous Investigation Results: summarizes the results of previous site investigations and presents groundwater analytical results in tabular form.
  - Contaminant Characteristics: provides a discussion of the chemical characteristics of the primary contaminant, PCE. PCE transport and degradation are also discussed in this section.
- Section 2: Remedial Investigation
  - Introduction: defines the purpose and scope of the RI, and discusses deviations from the work plan.
  - Findings: presents the findings of the RI field investigation.
  - Fate and Transport Analysis: presents the results of an overall analysis of the fate and transport of PCE and its degradation products at the RTRVP site.
- Section 3: Feasibility Study
  - RAOs
  - Identification and Screening of Technologies
  - Detailed Analysis of Remedial Action Alternatives

1.3 SITE SETTING

The RTRVP site is located on the north bank of the Kenai River, east of the Sterling Highway in Soldotna, Alaska (Township 5 North, Range 10 West, Section 32 Seward Meridian; approximate Latitude 60°28’ N, Longitude 151°05’ W). The RTRVP lies on three contiguous parcels of land located at the junction of the Sterling Highway and the Kenai River. The total land area of these parcels exceeds 9.5 acres. The contaminated area is predominantly located on 1 parcel less than 1 acre. The RTRVP is located immediately east of and adjacent to the ADOT Sterling Highway right-of-way (ROW). The parcel located directly adjacent to the ADOT ROW is 1.1 acres. Figure 1 is a site location map.

1.3.1 Site History

A laundry and dry cleaning facility operated on the site beginning in the late 1960s. The laundry building was constructed in 1961. There are 4 coin-operated washers and 4 coin-operated dryers still in use, but the dry cleaning facility was shut down in 1988.

PCE was the solvent used for dry cleaning at this facility, but the volume of PCE used and the disposal practices are not well documented.

Following a public complaint in 1992, the ADEC discovered 22 drums containing used oil and other substances at the site. One drum, labeled *Perchloroethylene*, was rusted badly and had several holes in it, and the surrounding soil was stained (see photographs 1 through 4 in Appendix A, which were taken during the 1992 site inspection). In 1992, ADEC verbally requested that the landowner dispose of the drums in an appropriate manner and conduct a site assessment. The verbal request was followed up in a letter dated December 10, 1993 requesting a contamination assessment plan.

A preliminary site assessment was completed in 1995. Field screening was conducted and one soil sample was submitted for laboratory analysis. This sample contained 26.1 mg/kg PCE and 14,300 mg/kg total petroleum hydrocarbons.

A work plan was submitted to ADEC on October 30, 1995 for *in situ* remediation of PCE-contaminated soils. The work plan called for the installation of a soil vapor extraction system consisting of two drive points to remediate the area of the 26.1 mg/kg PCE detection. Although the work plan was never approved, the remedial system was apparently implemented in the spring of 1996. The remediation system was removed from the site in 1997.

While installing a water line in May 1996, the landowner encountered contaminated soil. Approximately 600 cubic yards of PCE-contaminated soil were stockpiled on site. Stockpile samples were collected and analyzed.

Detailed site investigations, including soil, groundwater, surface water, and sediment sampling began in 1997. The results of these site investigations are summarized in Section 3 of this report.

During August 1997, the United States Environmental Protection Agency (USEPA) Region 10 and the site owner/responsible party (RP) entered into an administrative order on consent (AOC) for removal of contaminated soil. The purpose of the removal action was to remove all known soil that contained levels of PCE or its degradation products at levels above the soil ACLs established by the ADEC for the site. Site remediation commenced in late September 1997. Photographs 5 through 13 in Appendix A document the soil removal. The soil removal is summarized in Section 3.3 of this report.

1.3.2 Contaminants of Concern

The principal contaminant of concern at the RTRVP site is the chlorinated aliphatic compound PCE, which may be described as an ethene molecule in which all four hydrogen atoms have been replaced by chlorine. PCE is a typical solvent used in the dry cleaning industry. The chemical formula for PCE is $CCl_2{=}CCl_2$. Its chemical properties, which control the transport of PCE in the subsurface, are summarized in Table 1-1: PCE Chemical Characteristics. PCE is a dense non-aqueous phase liquid (DNAPL), which by definition is heavier than water. It is relatively insoluble and has a relatively low vapor pressure. PCE also is readily sorbed to soil organic matter.

The prevalent degradation mechanism for PCE is anaerobic degradation via dehalogenation. In dehalogenation, chlorine atoms are released, and PCE degrades sequentially to trichlorothene (TCE), cis- and trans-1,2-dichloroethene (DCE), 1,1-DCE, and VC (VC). VC eventually degrades to ethene, ethane, and carbon dioxide. Diagram 1-1 illustrates the dehalogenation pathway for PCE.

Besides PCE, other analytes detected at the site include the PCE degradation products TCE, cis- and trans-1,2-DCE, 1,1-DCE, and VC, as well as benzene, diesel-range organics (DRO), gasoline-range organics (GRO), and other petroleum hydrocarbons. Although relatively high levels of DRO and residual range organics (RRO) were detected in site soil, the areas of these detections were excavated in 1997. Only low levels of petroleum hydrocarbons remain in site soil and groundwater; therefore, they were not retained as COCs for the site.

Table 1-1: PCE Chemical Characteristics

	Value	**Source**
Molecular Weight	165.8	Pankow and Cherry, 1996
Solubility (mg/L)	150(1) 200 (2)	(1)USEPA Basics of Pump-and-Treat Ground-Water Remediation Technology (USEPA, 1990) (2)USEPA Soil Screening Guidance (USEPA, 1996a)
Diffusion Coefficient in air (cm^2/sec)	0.06968	USEPA Superfund Exposure Assessment Manual (USEPA, 1988a); value cited for 10 degrees C
Diffusion Coefficient in water (cm^2/sec)	8.2×10^{-6}	USEPA Soil Screening Guidance (USEPA, 1996a)
Henry's Law Constant (m^3-atm/mol)	0.0184	USEPA Soil Screening Guidance (USEPA, 1996a)
Vapor Pressure (torr, 25º C)	18.9	Pankow and Cherry, 1996
Adsorption Coefficient on Organic Carbon (k_{oc}) (L/kg)	265(1) 155(2) 364(3)	(1)USEPA Soil Screening Guidance (USEPA, 1996a) geometric mean value (measured) (2)USEPA Soil Screening Guidance (USEPA, 1996a) calculated value (3)Pankow and Cherry, 1996
Density (g/cm^3)	1.63	Pankow and Cherry, 1996
Relative vapor density	1.12	Pankow and Cherry, 1996 (calculated as the ratio of the density of dry air saturated with PCE to the density of dry air)

Anaerobic Tetrachloroethene Degradation Pathway

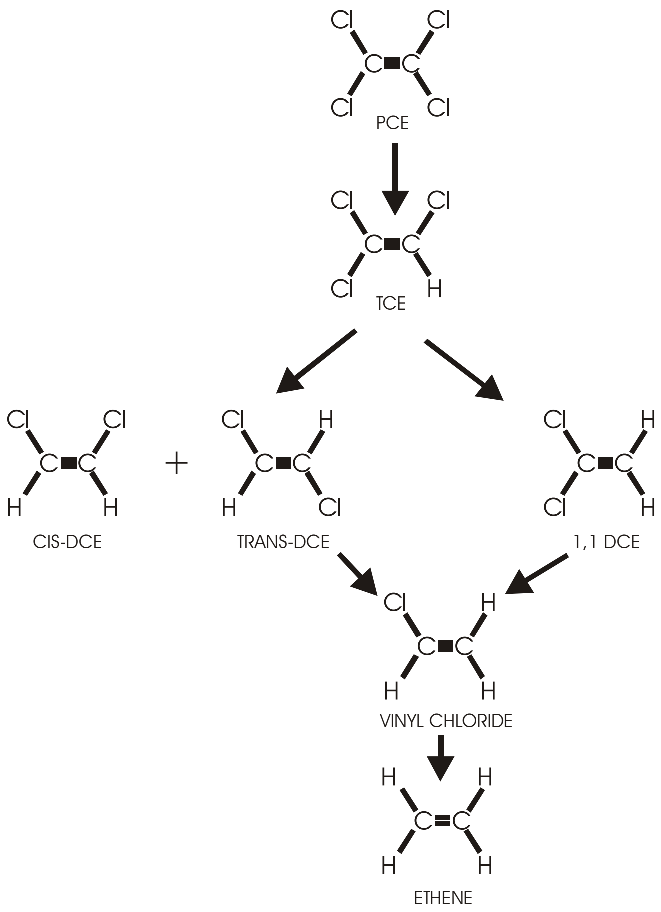

Diagram 1-1

2 GENERAL ENVIRONMENTAL SETTING

2.1 GEOGRAPHICAL SETTING

The RTRVP site is located adjacent to the Kenai River in Soldotna, which is on the Kenai Peninsula in south-central Alaska, approximately 60 air-miles southwest of Anchorage and 10 miles southeast of Kenai. The area is part of the Kenai Lowland, a glaciated coastal shelf approximately 20 to 50 miles wide and 110 miles long (Bailey and Hogan, 1995). The lowland encompasses an area of about 3,600 square miles (mi^2) and is bordered on the west by Cook Inlet, on the east by the Kenai Mountains, on the north by Turnagain Arm, and on the south by Kachemak Bay.

2.2 CLIMATOLOGY

The Soldotna area has a transitional climate influenced by the mild maritime climate of Cook Inlet and the cooler continental climate of interior Alaska (Bailey and Hogan, 1995). The mean annual temperature in Kenai is 34 degrees Fahrenheit (^oF) (Alaska Climate Summaries, 1999). Temperatures range from a July mean maximum of 62^oF to a January mean minimum of 3^oF. Mean annual precipitation is approximately 19 inches, and mean annual snowfall is 61 inches. Climatic data for Kenai are summarized in Table 2-1.

Table 2-1: Mean Monthly and Annual Climatic Data for Kenai, 1949-99

	Jan	Feb	Mar	Apr	May	June	July	Aug	Sept	Oct	Nov	Dec	Annual
Temperature (^oF)													
Mean Max.	20.5	25.9	32.6	42.3	52.4	58.2	61.7	61.5	55.0	41.7	29.2	21.9	41.9
Mean Min.	3.4	6.8	12.7	26.0	35.3	42.7	47.2	45.9	38.9	27.2	13.8	6.2	25.5
Mean	12.0	16.4	22.7	34.2	43.9	50.5	54.5	53.7	47	34.5	21.5	14.1	33.7
Precipitation (inches)													
	1.03	1.00	0.92	0.75	0.97	1.21	1.91	2.71	3.33	2.42	1.59	1.42	19.3
Snowfall (inches)													
	9.5	10.4	8.6	3.5	0.3	0	0	0	0	4.8	10.3	13.8	61.3

2.3 GEOLOGY

The Kenai Peninsula is underlain by bedrock composed of Permian-aged metamorphosed shale, greenstone, and conglomerate rocks, as well as older limestone and younger siltstone, sandstone, and coal (Bailey and Hogan, 1995). Exact depth to bedrock in the Soldotna area is unknown, but is greater than 250 feet. Surficial geologic materials include old terrace and coastal-plain deposits, alluvium, and glacial outwash and morainal deposits. Terrace, coastal-plain, and alluvial deposits are found along the Kenai River in Soldotna and generally consist of well-drained stratified silt, sand, and gravel. Glacial outwash deposits consist of well-sorted sand and gravel, and morainal belts are composed of stratified silt, sand, and gravel underlain by poorly sorted till. A mantle of wind-blown silt 3- to 6-feet thick covers most of the surficial deposits in the

Soldotna area. Permafrost has not been reported on the Kenai Peninsula (Ferrians, 1965).

2.4 SURFACE-WATER HYDROLOGY

The principal surface-water body in the project area is the Kenai River, which derives most of its discharge from the snowfields and glaciers of the Kenai Mountains. The river originates at the outlet of Kenai Lake and flows west, emptying into Cook Inlet near the City of Kenai. The Kenai River drains an area of more than 2,000 mi^2 and has an annual average runoff of approximately 2.7 cubic feet per second per square mile (Bailey and Hogan, 1995). Peak flows in the Kenai River are primarily the result of mid-summer snow melt, but have also been caused by the release of glacially impounded waters. Historical flooding along the Kenai River near Soldotna has occurred as a result of heavy precipitation, snow and ice melt, and outburst flooding from the release of glacier-dammed lakes (Federal Emergency Management Agency, 1980). When flooding occurs during the winter, the increase in flow can disrupt the river's ice cover, resulting in ice jams and subsequent backwater flooding. Surface-water runoff from the RTRVP site, which is adjacent to the Kenai River, generally drains toward the river.

2.5 GROUNDWATER HYDROLOGY

Most of the groundwater on the Kenai Peninsula is within unconsolidated deposits of glacial outwash, fluvial, and eolian (wind-blown) sediments (Bailey and Hogan, 1995). Lacustrine (lake) and eolian deposits typically are fine-grained silts and clays that are of limited use for providing water to domestic wells. Till and fluvial deposits also contain fine-grained sediments, but local zones of saturated sand and gravel are sufficiently thick to yield water in substantial quantities.

On a regional scale, two extensive confining layers of silt and clay create three distinct aquifers (Nelson, 1981). At the RTRVP site, the upper confining unit is approximately 35- to 40-feet below ground surface (bgs). Depth to the second confining layer is unknown.

3 PREVIOUS INVESTIGATION RESULTS

A detailed soil investigation in early 1997 encountered significant PCE contamination at the RTRVP site. Since then, numerous additional investigations have been conducted. Table 3-1 outlines the investigations performed between early 1997 and the present, sorted primarily by consultant/client and secondarily by date, and provides references for the site investigation reports. The reports listed in Table 3-1 are also cited in the References section of this RI/FS report.

Table 3-1: Summary of Previous Investigations at the River Terrace site

Investigation Date	***Activity Description***	***Consultant/Client***	***Report Name***
March 1997	Soil sampling on grids with grid nodes on 20-foot centers. No groundwater monitoring wells were installed.	Kennard Environmental Consultants (KEC)/Site owner	Final Site Assessment Report, May 8, 1997
October 1997	Excavation perimeter field-screening and laboratory analytical results from the October 1997 contaminated soil excavation at the RTRVP site	Site owner	Untitled, undated report submitted by site owner to Matt Carr, EPA, who provided report to ADEC on January 9, 1998 (cited as RP's interim AOC report, 1998)
May 1997	Sediment and surface water quality sampling in and near the Kenai River	Ecology and Environment (E&E)/ ADEC	Sediment and Water Quality Report May 1997 Sampling Event, River Terrace RV Park, June 1997.
June 1997	Storm water system sampling: Water samples were collected from 10 manholes and 2 system outfalls	E&E/ADEC	Stormwater Investigation Report, Sterling Highway, River Terrace RV Park, Soldotna Alaska, November 1997
July 1997	Phase I groundwater assessment to determine the presence and amount of PCE and its degradation products around the periphery of known soil contamination. Monitoring wells MW-1A and MW-1C through MW-7 and SB-1 were installed and sampled.	E&E/ADEC	Groundwater Assessment Phase 1 Final Report, River Terrace RV Park, June 1998
Various	Extensive review of existing documents and correspondence dated May 2, 1995 to June 12, 1998	E&E/ADEC	Data Review and Recommendations Final Report River Terrace RV Park Soldotna, Alaska June 1998
October 1998	Stormwater system sampling; analysis of possible groundwater connection to stormwater system; installation of soil borings and monitoring wells (MW-12, MW-13, MW-14) in ADOT ROW adjacent to RTRVP site	E&E/ADEC	Final Report Soil and Groundwater Investigation ADOT&PF Right-of-Way Sterling Highway at Kenai River Bridge No. 671 Soldotna, Alaska, June 1999
May 1999	Sediment and water sampling	E&E/ADEC	Sediment and Water Quality Report, May 1999 Sampling Event, River Terrace RV Park, Soldotna, Alaska, July 1999
October 1997	Quarterly groundwater monitoring initiated. Five monitoring wells were sampled.	Hart Crowser/Site Owner	Groundwater Sampling and Analysis, River Terrace RV Park, December 22, 1997

Investigation Date	***Activity Description***	***Consultant/Client***	***Report Name***
December 1997	Second quarterly groundwater monitoring event. Four monitoring wells were sampled.	Hart Crowser/ Site Owner	Draft Groundwater Quality Investigation December 1997 Sampling Event River Terrace RV Park, February 1998
June and August 1998	Third and fourth quarterly groundwater monitoring events.	Hart Crowser/ Site Owner	Groundwater Quality Investigation, June and August 1998 Sampling Event, River Terrace RV Park, September 14, 1998
June 1998	Removal action (approximately 4 cubic yards from the vicinity of the B9 grid)	Hart Crowser/ Site Owner	Ten Day Field Report, dated July 7, 1998
June, July and August 1998	Further investigation of soil and groundwater following excavation of the contaminated soil. Six monitoring wells (MW-3A, MW-4A, MW-8, MW-9, MW-10) were installed, and 4 soil borings were drilled	Hart Crowser/ Site Owner	River Terrace RV Park Soil and Groundwater Assessment. Soldotna, Alaska, September 16, 1998
October 1998	Fifth quarterly groundwater monitoring event	Hart Crowser/ Site Owner	Groundwater Quality Investigation October 1998 Sampling Event, River Terrace RV Park, December 22, 1998 (Preliminary)
November 1998	Soil and groundwater assessment at the building site, sewer line and area near MW-4. One monitoring well (MW-11) was installed and multiple soil borings were drilled at the site.	Hart Crowser/ Site Owner	Soil and Groundwater Assessment at the Building and Sewer Line, River Terrace RV Park. Soldotna, Alaska, February 2, 1999
December 1998	Sixth quarterly groundwater monitoring event	Hart Crowser/ Site Owner	Groundwater Quality Investigation, December 1998 Sampling Event, River Terrace RV Park, March 4, 1999
April 1999	Seventh quarterly groundwater monitoring event	Hart Crowser/ Site Owner	Groundwater Quality Investigation, April 1999 Sampling Event, River Terrace RV Park, May 28, 1999
June 1999	Soil sampling of treatment cell soils	Hart Crowser/Site Owner	Treatment Cell Closure Sampling, River Terrace RV Park, August 18, 1999

The sections below discuss each site investigation activity performed at the River Terrace RV site since early 1997. Appendix C provides a historical summary of PCE and its degradation product concentrations in soil samples collected from the site. Appendix D provides a historical summary of PCE and its degradation product concentrations in groundwater samples collected from the site.

3.1 KENNARD ENVIRONMENTAL CONSULTANTS, INC.

In late 1996, the site owner contracted Kennard Environmental Consultants (KEC) to design and implement a sampling and analysis plan to identify whether PCE contamination existed in site surface and subsurface soil. In February 1997, KEC established a grid with grid nodes on 20-foot centers across the area of suspected contamination (KEC, 1997). The longest north-northeast-south-southwest axis of the grid was 200 feet, and the longest WNW-ESE axis was 140 feet. Boreholes were drilled within this grid to depths ranging from 10 to 35 feet bgs. Numerous soil samples were collected and sent to a laboratory for analysis. PCE was detected at maximum concentrations of 4,700 milligrams of PCE per kilogram of soil (mg/kg) in surface soils and 182 mg/kg in subsurface soils (5 feet bgs) (KEC, 1997). Figure 2 illustrates the KEC grid sampling PCE results. Other detected contaminants included TCE, DCE, total petroleum hydrocarbons TPH, DRO, toluene, and xylene.

Groundwater was encountered in six boreholes drilled closest to the Kenai River and in two boreholes located on the topographic "bench" northeast of the river.

3.2 ACL DETERMINATION

In a letter dated August 27, 1997 from Lynn T. Kent of the ADEC to Gary Hinkle, site owner, ADEC established ACLs for PCE and its degradation compounds in site groundwater and soil. The letter stipulates that ACLs apply to all aquifers between surface grade and the lower drinking water aquifer, and the lower drinking water aquifer must meet drinking water standards. The groundwater ACLs established for the site are listed along with the maximum contaminant level (MCLs) in Table 3-2.

Table 3-2: River Terrace Site Upper Aquifer Groundwater ACLs

Analyte	ACL (mg/L)	MCL (mg/L)
PCE	0.84	.005
TCE	21.9	.005
cis-1,2-DCE	11.6	.07
trans-1,2-DCE	11.6	.1
1,1-DCE	.007	.007
VC	.002	.002

The soil ACLs established for the site are listed along with the ADEC Method 2 Cleanup Levels in Table 3-3. The ADEC cleanup levels were promulgated as part of 18 AAC 75 on January 22, 1999. The cleanup levels cited in Table 3-3 are the migration to groundwater pathway levels applicable for areas with under 40-inches of annual precipitation.

Table 3-3: River Terrace Site Soil ACLs

Analyte	ACL (mg/kg)	ADEC Method 2 Cleanup Level (mg/kg)
PCE	11.5	0.03
TCE	300	0.027
cis-1,2-DCE	72.1	0.2
trans-1,2-DCE	87.3	0.4
1,1-DCE	7.1	0.03
VC	2.1	0.009

3.3 OWNER/USEPA REMEDIATION ACTIVITIES

During August 1997, the USEPA Region 10 and the site owner/RP entered into an AOC for removal of contaminated soil. The purpose of the removal action was to remove all known soil that contained levels of PCE or its degradation products at levels above the soil ACLs established by the ADEC for the site.

Site remediation commenced in late September 1997. Photographs 5 through 13 (in Appendix A) show 1997 excavation activities. ADEC and E&E (1998a) reviewed the AOC interim removal report. The AOC interim report is untitled, undated, and the author is not referenced, so it is referenced herein as the RP's interim AOC report (1998). The report did not identify the quantity of soil excavated, the estimated quantity of excavated soil requiring remediation, or whether the sampling frequency met the data quality objectives (DQO). However, sampling results indicate that soil contaminated by PCE at concentrations above the site cleanup level of 11.5 mg/kg remained in place in at least three locations. The three locations are near KEC grid sampling node B7 (northeast of MW-9) at 20 feet bgs (15.5 mg/kg PCE) and 31.5 feet bgs (12.2 mg/kg PCE), and near KEC grid sampling node C6 at 17 feet bgs (18.9 mg/kg PCE). Figure 2 illustrates the KEC grid sampling results and excavation perimeter sampling results. One additional near-surface excavation perimeter sample exceeded site ACLs for PCE (not shown on Figure 2); this was addressed by Hart Crowser in June 1998 during follow-up activities (Hart Crowser, 1998a). The deeper areas of soil contamination remaining above site ACLs have not been removed.

3.4 ECOLOGY & ENVIRONMENT, INC.

3.4.1 May 1997 Sediment and Water Quality Sampling

In May 1997, E&E, under contract to ADEC, conducted a limited sampling investigation to evaluate potential PCE contamination in the Kenai River and adjacent sediments near the RTRVP site (E&E, 1997a). Surface water and sediment sampling locations are shown in Figure 22. Eight sediment samples were collected from the sediment-water interface (approximately 14 to 4 inches below the ground surface) along the river's edge, and one water sample was collected from a storm water outfall west of and beneath the Sterling Highway Bridge.

PCE and its degradation products (TCE and DCE) were detected in three of the nine sediment samples. The three contaminated sediment samples were located adjacent to the RTRVP site, downslope of the contaminated area delineated by KEC. E&E reported that cis-1,2-DCE concentrations in three sediment samples (79 micrograms per kilogram (µg/kg), 450 µg/kg, and 670 µg/kg) exceeded the Hull and Suter (1994) sediment quality benchmark of 23 µg/kg. The sediment quality benchmarks cited by E&E were not adjusted for site-specific total organic carbon data. These benchmarks have subsequently been updated by Jones et al (1997); a comprehensive discussion of sediment quality results, citing the updated benchmarks adjusted for site-specific organic carbon, is provided in Section 8 of this report. PCE, TCE, and DCE were not detected in upstream or downstream sediment samples. Two of the downstream samples, however, experienced elevated detection limits (40 µg/kg and 60 µg/kg) due to interference with high organic levels in the samples. The upriver sample collected for background purposes near the upriver property boundary showed no detections of PCE or its degradation products.

PCE and its degradation products (TCE and DCE) were detected in one water sample (collected from the stormwater outfall west of and beneath the Sterling Highway Bridge).

The storm water outfall water sample contained a PCE concentration of 23 micrograms per liter (µg/L), which exceeds the water quality standard of 5 µg/L. E&E concluded that contaminant transport towards the river may have occurred via groundwater transport and/or vapor-phase migration and infiltration (E&E, 1997a).

All historical surface water results are summarized in Table 7-2, and sediment sample results are summarized in Table 7-3 of this report.

3.4.2 June 1997 Stormwater System Investigation

In late June 1997, E&E conducted further investigations to determine whether PCE and its degradation products were entering the storm water system and discharging into the Kenai River (E&E, 1997b). Water samples were collected from two outfalls (Kobuk Street Outfall and Sterling Highway Bridge Outfall (SHBO), which is also known as the KRBO) and 10 manholes (MH) located along an approximately 3,500-foot section of the Sterling Highway.

PCE was detected in the KRBO (at 8.2 µg/L) and the MH closest to the KRBO, MH-1, (at 6.6 µg/L). No PCE or degradation products were detected in any of the other MHs. E&E hypothesized that a hydraulic connection exists between PCE-contaminated groundwater at the RTRVP site and the stormwater system. The investigation suggested that PCE contamination most likely enters the stormwater system via contaminated groundwater infiltration between MH-2 and MH-1, or at MH-1.

All historical stormwater system sampling results are summarized in Table 7-1 of this report.

3.4.3 Phase I Groundwater Assessment

A Phase I groundwater assessment was performed by E&E during July 1997 (E&E, 1998b). The overall goal of the groundwater assessment was to determine whether PCE and its degradation products have contaminated the water table aquifer, and whether groundwater contamination was migrating off-property to the Kenai River and to the adjacent ADOT ROW. E&E installed five soil borings near the known soil contamination on the RTRVP property and five in the adjacent ADOT ROW. All of these borings are located on property known as the RTRVP site in this report. Four of the borings in the ADOT ROW were completed as monitoring wells (MW-1A, MW-1C (shallow and deep), MW-2, and SB-1 (shallow and deep), and five of the borings on the RTRVP property were completed as monitoring wells (MW-3 through MW-7). Samples were analyzed by both a field laboratory and a fixed-based laboratory.

Numerous surface soil, subsurface soil, and groundwater samples were collected during the assessment. In the ADOT ROW, PCE and one or more of its degradation products were detected in 16 of 37 soil samples analyzed by the field laboratory. All detections were at trace levels (i.e., less than 50 µg/kg). On the RTRVP property, PCE and one or more of its degradation products were detected in 42 of 44 soil samples analyzed by the field laboratory and in 11 of 12 soil samples analyzed by the fixed-base laboratory. However, the fixed base laboratory MCLs were greatly elevated in the one sample without a detection, so it is possible that PCE and/or its degradation products existed in that sample. Appendix C provides a summary of all historical soil detections.

In the ADOT ROW, PCE and its degradation products were not detected in any of the four groundwater samples analyzed by the fixed-base laboratory. On the RTRVP property, PCE and one or more of its degradation products were detected in five of five groundwater samples analyzed by the fixed-base laboratory.

A comparison of PCE and its degradation products detected in groundwater versus those detected in soil for the same location indicated a consistent correlation. E&E surmised that the source of groundwater contamination was PCE-contaminated subsurface soil in direct contact with groundwater. PCE was detected above the site ACL of 840 µg/L in samples from MW-4 (1,500 µg/L) and MW-6 (1,900 µg/L). Appendix D provides a summary of all historical groundwater detections, and Table 7-4 provides a list of historical ACL exceedences.

3.4.4 ADOT ROW Soil and Groundwater Investigation

An investigation of the storm water system and soil and groundwater quality in the ADOT ROW was performed by E&E during October 1998 (E&E, 1999a). The objectives of this study were to further determine whether PCE or its degradation products have migrated from the River Terrace property to the ADOT ROW, to determine the pathway and impact of contaminated groundwater on the storm water system within the ROW, and determine whether a hydraulic connection exists between the RV Park and the ROW or the Kenai River.

The storm water system investigation included surveillance with a remotely-operated camera and a smoke test, in addition to water sample collection. Three different attempts were made to observe conditions in the storm water line between MH-2 and the SHBO using the remote camera. Rocks and other debris prevented the camera from running through most of the line between MH-2 and MH-1; however, the camera was run successfully from MH-1 most of the distance to the SHBO. The results of the camera runs provided no evidence of large volume infiltration of groundwater between MH-1 and the SHBO. In the smoke test, a smoke bomb was released in MH-1, with the storm water system blocked off. Smoke rapidly exited out the top of the lift station on the RTRVP property, indicating a direct connection between the sanitary sewer lift station and the storm water main. Water samples collected from MH-2, MH-1, and the SHBO exhibited PCE concentrations of non-detect (ND), 8.5 µg/L and 8.3 µg/L, respectively. These results were similar to the 1997 storm water sampling results. Historical stormwater analytical results are presented in Table 7-1.

Monitoring well MW-14 was installed between the sanitary lift station and MW-1 to detect contamination that may be migrating from the RTRVP site to the storm water system. Three soil borings were advanced in the ADOT ROW approximately 40 to 80 feet northeast of MW-14 to investigate whether soil contamination documented at grid nodes A1 and A2 had migrated onto the ADOT ROW property. A one-foot thick saturated interval was logged in the MW-14 boring; no saturated intervals were observed during installation of the other three soil borings. Thirteen soil samples were submitted for laboratory analysis from these four soil borings; PCE was detected in only one of these samples. The PCE detection of 160 µg/kg at 14 to 16 feet bgs in MW-14 was significantly below the site ACL of 11,500 µg/kg for soil. This sample was collected from or adjacent to the 1-foot thick saturated interval logged in the boring. No PCE degradation products were detected in any of the soil samples. PCE, TCE, and cis-1,2-DCE were detected in a groundwater sample collected from MW-14, at levels well below the site ACLs.

Three additional soil borings were drilled on the ADOT ROW near the Kenai River to determine whether contaminants were migrating toward the north bridge abutment. Two of these locations were completed as monitoring wells. Ten soil samples were submitted for laboratory analysis from these soil borings; PCE was not detected in any of these samples. Cis-1,2-DCE was detected in two soil samples (4 to 6 feet bgs and 6 to

8 feet bgs) from MW-12, at concentrations well below the site ACL. PCE, TCE, and cis-1,2-DCE were detected in soil samples from MW-12 and MW-13, at levels below the site ACLs.

Historical soil analytical results are presented in Appendix C, and historical groundwater analytical results are presented in Appendix D.

3.4.5 May 1999 Sediment and Water Quality Sampling

During May 1999, E&E collected sediment and surface water samples from the Kenai River adjacent to the RTRVP site to determine whether PCE or its degradation products continue to migrate from the site to the river (E&E, 1999b). Surface water and sediment sampling locations are shown in Figure 23.

Twelve sediment samples were collected from approximately the sediment-water interface of the Kenai River ordinary high water line (OHW), in a similar manner to the 1997 sediment sampling. Nine of the 12 sediment samples were collected at approximately 20-foot intervals from the bridge eastward along the river bank. Two samples were collected downstream of the KRBO, and one background sample was collected upstream from the site at the northeast property boundary.

Due to laboratory error, two sets of sediment samples were collected: one on May 19, 1999, and one on May 22, 1999. The sediment samples were analyzed by EPA method 5035/8260B, which requires two aliquots to be collected for each sample: one aliquot preserved in the field with methanol and one with sodium bisulfate. For the May 19, 1999 samples, the project laboratory mistakenly provided E&E with incorrect sample containers for sediment samples preserved with sodium bisulfate and could not analyze these samples. At E&E's request, the laboratory analyzed five sediment samples that were preserved with methanol. As a result of the mistake, E&E returned to the site on May 22, 1999 and recollected the 12 sediment samples. Therefore, 5 sediment samples have two results (May 19 and May 22, 1999), and 7 sediment samples have one result (May 22, 1999).

PCE and its degradation products TCE and cis-1,2-DCE were detected in 8 of the 12 sediment sample locations sampled on May 22, 1999. The detections were from sediment samples immediately adjacent to the RTRVP site, located very close to where PCE was detected in May 1997 sediment sampling. PCE and its degradation products were not detected in upstream or downstream samples. A maximum PCE concentration of 340 µg/kg was detected in the sediment samples. E&E reported that cis-1,2-DCE concentrations in two sediment samples collected on May 19, 1999 (460 µg/kg and 520 µg/kg) exceeded the Jones et al (1997) sediment quality benchmark of 400 µg/kg. The sediment quality benchmarks cited by E&E were not adjusted for site-specific total organic carbon data. A comprehensive discussion of sediment quality reports, citing adjusted benchmarks, is provided in Section 8 of this report. E&E reported that no other samples exceeded sediment quality benchmark criteria (Jones et al., 1997).

Four sediment samples were analyzed for TOC. The TOC results ranged from a minimum of ND (less than 949 µg/kg) to a maximum of 7,018 µg/kg. An average sediment TOC concentration of 3,600 µg/kg, or 0.36 percent was calculated by assuming a TOC concentration of one-half of the detection limit (275 µg/kg) for the ND sample.

Three water samples were collected from the Kenai River, and one water sample was collected from the KRBO. In the Kenai River, two samples were collected between monitoring well locations MW-6 and MW-12, and one background sample was collected upstream of the site. PCE and its degradation products were detected in three project water samples; the upstream (background) sample exhibited no detections. The PCE concentration in the KRBO sample, 17 µg/L, exceeds the surface water quality criterion of 5 µg/L. PCE concentrations in the other two samples, 0.8 µg/L (approximately downslope of MW-8) and 2.5 µg/L (approximately downslope of MW-6), did not exceed the surface water quality criterion. No degradation product results exceeded surface water quality criteria (WQC).

Historical stormwater, surface water, and sediment results are presented in Table 7-1, Table 7-2, and Table 7-3, respectively.

3.5 HART CROWSER, INC.

3.5.1 Additional Soil Removal Activities

In June 1998, Hart Crowser, Inc. (Hart Crowser) oversaw the excavation of near-surface soil located near grid node B9 (see Figure 2 for grid node location)(Hart Crowser, 1998a). Two excavation perimeter samples from October 1997 indicated PCE soil contamination remaining above the site ACL: sample 219-EX-1525, shown on Plate 2, with a detection of 27.4 mg/kg PCE and sample 217-EX-1513 with a detection of 13.9 mg/kg). The June 1998 excavation footprint was roughly circular, centered near the midpoint between grid nodes B8 and B9. Maximum excavation depth was approximately 8 feet on the eastern end of the excavation and approximately 9 feet on the western end of the excavation. Approximately 4 cubic yards of soil were removed and placed into treatment cell no. 2.

3.5.2 Quarterly Groundwater Monitoring

In October 1997, under a contract with the site owner, Hart Crowser began implementing a quarterly groundwater monitoring program at the RTRVP site. The goals of the program are listed below.

- Document existing conditions and assess changes in water quality in existing (and future) monitoring wells.
- Further evaluate evidence for hydraulic connection and the seasonal relationship between groundwater and the Kenai River.
- Through trend analysis, determine whether contaminant concentrations are changing appreciably to warrant revising the quarterly monitoring program.

Hart Crowser completed seven sampling events. Sampling events were completed in October 1997 (Hart Crowser, 1997), December 1997 (Hart Crowser, 1998b); April, June/August 1998 (Hart Crowser, 1998c), October 1998 (Hart Crowser, 1998d), December 1998 (Hart Crowser, 1999b); and April 1999 (Hart Crowser, 1999c). PCE and one of its degradation products, VC, have been detected at concentrations above the established ACLs. One or more ACLs were exceeded in samples from monitoring wells MW-4A, MW-6, MW-8, MW-9, MW-10. Appendix D provides a summary of all historical groundwater detections, and Table 7-4 provides a summary of historical ACL exceedences.

The quarterly monitoring events scheduled for July, October, and December 1999 were completed as part of this RI/FS investigation by OASIS/Bristol. A detailed discussion of these results is presented in Section 6 of this RI/FS report.

Following excavation of the contaminated soil (in 1997) Hart Crowser conducted a soil and groundwater assessment in June, July, and August of 1998 (Hart Crowser, 1998d). The assessment investigated locations where soil samples taken after soil removal showed PCE in excess of the approved ACL. Groundwater results in MW-4A and MW-10 contained PCE concentrations above the site ACL. One soil boring indicated a PCE concentration in excess of the ACL at 30.0 to 31.5 feet bgs. All historical soil sample results are provided in Appendix C of this report.

In November 1998, a soil and groundwater assessment was conducted at the former dry cleaner building sumps and sewer line to investigate possible sources for the release of petroleum hydrocarbons and dry cleaning solvents (Hart Crowser, 1999a). Additional work was conducted to assess a potential source of elevated PCE groundwater concentrations found in MW-4A (area near the AA-3 through AA-5 grid nodes). During the assessment, Hart Crowser concluded that the building sumps and sewer line contained trace amounts of PCE and low levels of DRO/RRO, but these are most likely a result of incidental contamination and are not source areas. However, Hart Crowser was unable to sample the sumps to the depths specified in the work plan (and to the water table) due to auger refusal. Soil borings installed along the AA grid nodes did not encounter soil contamination that could institute a possible source area for the elevated PCE groundwater concentrations found in MW-4A. All historical soil sample results are provided in Appendix C of this report.

4 CONTAMINANT CHARACTERISTICS

The results of an extensive literature review of the characteristics of PCE transport and degradation are summarized in this section. The physics of DNAPL transport in both the unsaturated and saturated zones is discussed. The biotic and abiotic degradation pathways of PCE are also discussed. The results of laboratory- and field-scale studies are presented to support both the transport and degradation discussions.

The contaminant characteristics discussed in this section are applied to the RTRVP site in Section 7 (Fate and Transport Analysis) of this RI/FS report.

4.1 TRANSPORT OF DNAPLS

The transport of PCE in the subsurface is controlled by its physical and chemical properties (listed in Table 1-1: PCE Chemical Characteristics). The various transport mechanisms of a PCE spill to the ground surface are briefly summarized in this paragraph and discussed in greater detail in the following subsections.

When released to the ground surface, mobile non-aqueous phase PCE will infiltrate into the subsurface by gravity and capillarity. The penetration depth and lateral spread are a function of spill volume, spill rate, and soil characteristics. The horizontal spread of a spill in the vadose zone is largely determined by capillarity and vadose zone heterogeneity (Cohen and Mercer, 1993). Residual (immobile) PCE will remain trapped in discontinuous pore spaces after free-phase PCE has migrated through an area. PCE will partition into the vapor phase, solid phase (via adsorption onto organic carbon in the soil), and aqueous phase (via dissolution into pore water). PCE dissolved into soil pore water can migrate vertically downward with infiltration water or laterally in perched water or along lower permeability contacts. Vapor-phase PCE can migrate upward or laterally via diffusion, or downward via density or pressure gradients.

4.1.1 Non-Aquesous Phase Transport

4.1.1.1 Physics of DNAPL Transport in the Unsaturated Zone

DNAPL penetration into the vadose zone is driven by gravity (downward migration) and capillarity (lateral spreading). Capillary pressure is defined as the difference between the pressure of the non-wetting fluid and the pressure of the wetting fluid. It is a function of the interfacial tension between the two fluids. On most soil surfaces, DNAPLS are wetting with respect to air but nonwetting with respect to water, which means water will occupy the pore spaces adjacent to the sand grains, air will occupy the center of the pore space, and the DNAPL will occupy the space in-between.

In an unsaturated homogeneous medium, the DNAPL will sink with minimal lateral spreading until it encounters a layer with different characteristics or until its volume is sufficiently depleted to limit further migration via gravity. The DNAPL readily displaces air due to its greater density, viscosity, and wetting characteristics. As DNAPL migrates through the soil, a significant amount is trapped in the pore spaces. This trapped DNAPL is called residual, and residual saturation is a function of the porous medium and the DNAPL. The presence of residual DNAPL in soil provides conclusive evidence that DNAPL has migrated through an area.

In the unsaturated zone, the soil column contains two fluids: air and water. The introduction of DNAPL to this system affects the relative permeability of the soil to each fluid. When more than one fluid is present in a soil column (e.g., DNAPL, water, and air), the pore space available for each fluid is diminished. Therefore, the relative permeability of the soil to each fluid is diminished in relationship to the amount of pore space occupied by the other two fluids. Relative permeability is key to unsaturated zone DNAPL transport.

DNAPL transport has been shown to be very sensitive to even slight variations in soil properties. Finer-grained soil layers in the vadose zone typically have higher water saturation than overlying coarser-grained soil. This increased water content greatly restricts the ability of the DNAPL to enter into the finer-grained soil. Therefore, DNAPL encountering a finer-grained soil layer will pool and travel laterally on top of the layer. If the DNAPL pool becomes sufficiently deep, the capillary pressure of the DNAPL will exceed the entry pressure of the soil layer, and the DNAPL will imbibe the layer.

4.1.1.2 Unsaturated Zone Field-Scale DNAPL Transport Experiment Results

Poulsen and Kueper (1992) conducted a field experiment in which six liters of dyed PCE were released to the ground surface at two different release rates; “instantaneous” and “slow drips”. The PCE was allowed to infiltrate and then was excavated. The subsurface soil was relatively homogeneous fine-to-medium grained sand (with a measured porosity of approximately 46 percent, permeability of $1\text{x}10^{-7}$ square centimeters (cm^2), and approximately 8 percent volumetric water content). Depth to groundwater was approximately 3.5 meters (m) bgs.

Poulsen and Kueper made the following conclusions regarding PCE transport through the vadose zone.

- PCE migration in sandy deposits is very sensitive to even minor changes in soil characteristics, such as permeability and capillary characteristics.
- Depth of PCE migration is a function of both release rate and soil characteristics.
- Gravity forces dominated PCE migration for approximately the top 0.5 m in the instantaneous release only; otherwise, capillary forces dominated.
- PCE migrated along soil bedding planes. Detailed measurements showed permeability differences generally less than a factor of 1.7 between adjacent sand lenses that were invaded by PCE and those that were free of PCE.
- PCE from the instantaneous release migrated to a depth of 2.1 m bgs in 90 seconds, and PCE from the drip release migrated to a depth of 3.2 m bgs in more than 100 minutes. Poulsen and Kueper indicated that the deeper penetration of the drip release was due to its smaller release area and lack of PCE ponding at the surface.
- Bulk solvent retention capacity values of 12.6 liters of solvent per cubic meter of soil (L/m^3) and 4.9 L/m^3 were calculated immediately below the source area of the instantaneous and drip releases, respectively.
- “Even relatively small volumes of spilled solvent can travel large vertical distances.”
- Poulsen and Kueper’s study area was characterized by non-horizontal laminations consisting of fine- to medium-grained beach sands. Horizontal bedding, had it been present at the site, would have “promoted a greater degree of lateral spreading within the varying permeabilty layers.”

4.1.1.3 Physics of DNAPL Transport in the Saturated Zone

In the saturated zone, DNAPL transport is controlled by capillary pressure. At a given water saturation, capillary pressure is greater in soil with lower hydraulic conductivity. In order for the DNAPL to enter the soil layer, its capillary pressure must exceed the displacement pressure of the soil layer. The displacement pressure is a function of pore size. In any subsurface environment, there are a wide variety of displacement pressures due to soil heterogeneity. The distribution of displacement pressures is a key factor in the pathway of DNAPL flow through the saturated zone.

Capillary pressure is a function of saturation. Two algebraic expressions are commonly used to represent the relationship between capillary pressure and saturation; the Brooks-Corey (1964) equation and the Van Genuchten (1980) equation. Using the Brooks-Corey equation, Kueper and Frind (1991) measured capillary pressure-saturation curves for a PCE-water system.

After mobile DNAPL travels through a soil volume, groundwater can never completely displace the DNAPL. Residual DNAPL, in the form of isolated globules within pore spaces is always left behind. These disconnected globules have been cut-off from the mobile DNAPL and are immobile, except through dissolution. Dissolution is a diffusion-driven process, which is very slow.

4.1.1.4 Laboratory-Scale Saturated Zone PCE Experiment Results

Kueper et. al., (1989) conducted a laboratory experiment to examine the effects of soil heterogeneity on PCE transport in the saturated zone. They constructed a parallel-plate laboratory cell into which four well-sorted, homogeneous silica sands were packed in horizontal and vertical layers. The sands ranged in permeability by two orders of magnitude ($5x10^{-6}$ to $8x10^{-9}$ cm^2). Dyed PCE was injected through ports at the top of the cell, and its flow pathway was photographed and measured.

Kueper et. al. made the following conclusions regarding PCE flow through the saturated zone.

- Before encountering a distinct layer in the cell, PCE traveled downward and laterally through the sand. The authors stated that the lateral PCE protrusions were probably due to minor heterogeneity within the sand type due to packing the cell. This result suggests that PCE transport is very sensitive to even subtle variations in the subsurface soils.
- When encountering a fine-grained layer, the PCE pooled, flowed laterally along the layer, and cascaded off the edges.
- If the PCE pooled to a sufficient depth to exceed the displacement pressure of a given fine-grained layer, it would then penetrate the layer.
- Small differences in permeability between two adjacent layers are capable of causing significant lateral PCE flow.

4.1.1.5 Field-Scale Saturated Zone PCE Experiment Results

Kueper et al. (1993) conducted a field experiment to evaluate the behavior of PCE in the saturated zone. In this experiment, 231 liters (approximately 61 gallons) of PCE were introduced to the water table in the Borden aquifer, Ontario, Canada. Extensive monitoring was conducted to evaluate the transport of the PCE.

Kueper et al. made the following conclusions regarding PCE transport through the saturated zone in a field setting.

- PCE migration pathways followed the horizontal bedding of the sands.
- PCE preferentially migrated through coarser-grained sand layers.
- PCE was present in certain sand layers and absent in others that visually appeared to have only subtle texture variations. The thickness of these layers ranged between a few millimeters to 5 cm.
- PCE migrated laterally and vertically.
- PCE residual content was spatially variable. PCE saturation ranged between 1 and 38 percent of pore space, and PCE saturation above approximately 15 percent represented pooled PCE. Therefore, residual content should not be considered to be a single value for a given porous media and DNAPL.
- Lateral flow occurs above a relatively finer-grained layer, regardless of whether that layer has been penetrated.
- The dominant force in saturated zone PCE migration is relative permeability. Coarse-grained layers have a higher relative permeability than fine-grained layers; therefore, PCE preferentially flows through coarse grained soils.
- In order to enter a given layer, pooling PCE must exceed the layer's displacement pressure or entry pressure.
- PCE will preferentially follow root holes or other pathways having negligible entry pressures.

4.1.1.6 Residual Saturation

Residual saturation is the saturation at which DNAPL becomes discontinuous and is immobilized by capillary forces under ambient conditions. Residual saturation is a by-product of free-phase DNAPL transport in both the vadose zone and the saturated zone.

Pure-phase PCE always leaves a trace of its transport pathway in the form of residual PCE. The primary factors affecting the volume of residual saturation include:

- Soil pore size distribution;
- Fluid wettability, viscosity, density, and interfacial tension; and
- Hydraulic gradient (in the saturated zone).

In the unsaturated zone, residual saturation and retention capacity generally increase with decreasing intrinsic permeability, effective porosity, and moisture content (Cohen and Mercer, 1993). Residual DNAPL typically occupies between 10 and 20 percent of the pore space in unconsolidated media (Cohen and Mercer, 1993). The retention capacity of the saturated zone is generally greater than that of the vadose zone. Residual DNAPL typically occupies between 15 and 50 percent of the pore space in saturated media (Cohen and Mercer, 1993).

Although residual PCE is considered immobile, it is available for dissolution into soil pore water or groundwater. Residual PCE trapped in the vadose zone also dissipates via vaporization. Therefore, the presence of residual PCE acts as a continual source for vapor phase and dissolved phase PCE. Since the drinking water MCL for PCE (0.005

mg/L) is five orders of magnitude lower than its solubility (150 mg/L), residual PCE can provide a lasting source of significant groundwater contamination.

4.1.2 Vapor-Phase Transport

Both mobile and immobile PCE will partition into the vapor phase. Dissolved phase PCE can also partition into the vapor phase, but less readily than pure phase PCE. Vapor phase PCE is transported radially outward via diffusion and vertically (up or down) via density or pressure gradients. These driving forces can transport PCE quickly away from the source area. Vapor-phase PCE is available to be sorbed onto soil particles or dissolved into soil moisture or groundwater. The transport of vapor phase PCE can significantly extend the impacted area beyond that initially contaminated by free-phase PCE.

According to criteria cited by Cohen and Mercer (1993), density-driven gas flow will likely be significant for PCE in relatively permeable (sand or gravel) homogeneous media. As with free-phase PCE flow, vapor phase migration is strongly affected by subsurface layering.

PCE has a relatively low vapor pressure of 18.9 torr at 25 degrees Celcius (^{0}C) (Pankow and Cherry, 1996). At 20 ^{0}C, the maximum vapor concentration in air is 127 milligrams per liter (mg/L). At 10 ^{0}C, a more reasonable subsurface temperature for the site, the calculated maximum vapor concentration is 64 mg/L. This value represents the maximum vapor concentration available for subsurface vapor phase transport.

4.2 PCE DEGRADATION

Degradation is the transformation of complex compounds into simpler substances, by either chemical or biological transformations. The primary pathway for PCE degradation appears to be reductive dechlorination, which is the sequential removal of chlorine atoms from the PCE molecule. Diagram 1-1 provided in Section 1.3.2 depicts the PCE degradation pathway. The final degradation products are carbon dioxide and water.

The following subsections discuss PCE degradation caused by chemical and biological transformations. Both chemical and biological transformations of PCE tend to be slow, but biological transformations appear to be generally more significant. Various PCE half-lives are given in the literature, which are listed in Table 4-1.

A site-specific PCE degradation analysis for the RTRVP site is provided in Section 7.5.

Table 4-1: PCE Half-Lives

Half-life (years)	Temperature	Calculated Half-life at 10°C* (years)	Substrate; Method	Source
0.7	20°C	2.5	Aqueous; abiotic hydrolysis	Dilling et al, 1975; Pankow & Cherry, 1996
1,300,000			Aqueous; abiotic hydrolysis	Jeffers et al., 1989
0.5-1.0	20-25°C	1.8-2.6	Soil; Scientific judgement based on aqueous aerobic biodegradation half-lives in the literature.	Howard et al., 1991; Cohen & Mercer, 1993
2.4-2.8	Unknown	--	Field-scale PCE to TCE degradation	Ellis et al., 1996 [in Wiedemeier et al., 1996]
0.096 (35 days)	Unknown	--	Microcosm Rate	Parson et al., 1984 [in Wiedemeier et al., 1996]
0.2 (0 mg/L substrate); 28 days (5 mg/L substrate)	Unknown	--	Groundwater in batch reactors by acclimated microbial consortium; Initial PCE conc. 200 µg/L in sodium acetate substrate	Ying Chih-Chui, 1997

*For each 10°C temperature decrease, the halogenated aliphatic compound reaction rate decreases by approximately 3.5 times (based on the Arrhenius equation) (Vogel et al, 1987).

4.2.1 Abiotic Degradation

PCE is susceptible to only a limited number of transformation methods, namely hydrolysis/substitution and reduction (Vogel et al., 1987). Hydrolysis occurs when water (H_2O) or the hydroxide ion (OH^-) replaces a halogen (i.e., a chlorine on the PCE molecule), resulting in a more polar, more readily degradable product. Substitution also involves a nucleophilic attack on a carbon-halogen bond, but by a compound other than H_2O or OH^-. Laboratory studies indicate that hydrolysis/substitution reactions generally are ineffective in degrading PCE. Jeffers et al. (1989) calculated an abiotic hydrolysis half-life of 1.3e+6 years for PCE (Table 4-1).

Reduction occurs when the halogenated compound acts as an electron acceptor and is reduced in the process. While PCE is susceptible to reduction reactions, these reactions rarely occur except under very reducing conditions and with assistance from anaerobic bacteria. Butler and Barker (1996) determined that dechlorination of several chlorinated solvents, including PCE, can be thermodynamically favored. PCE can sequentially dechlorinate to ethylene. The intermediate reaction products include TCE, DCE (usually the *cis*-1,2-DCE isomer), and VC, as shown in Diagram 1-1.

4.2.2 Biological Transformations

Microbiological organisms can use organic contaminants in the subsurface as a food/carbon source, producing cell mass and degradation by-products. There are many types of microorganisms that can degrade contaminants, some only under aerobic conditions and others only under anaerobic conditions. Furthermore, different contaminants may be transformed by either aerobic or anaerobic organisms. Chlorinated solvents generally are known to resist degradation by conventional biological treatment processes.

Laboratory studies conclude that PCE is resistant to aerobic degradation. Aerobic cometabolism is effective in degrading many halogenated aliphatic compounds, but not PCE. Fogel et al. (1984) observed that a methanogenic consortium of four different bacteria, which degraded ten different halogenated aliphatic compounds (including TCE), did not degrade PCE. Several other studies cited by Butler and Barker (1996) (Wilson and Wilson [1985], Nelson et al., [1987], Vannelli et al. [1990]) indicate that PCE is resistant to aerobic degradation.

The only path of PCE biodegradation appears to be reductive dechlorination in anaerobic conditions. In reductive dechlorination, the PCE molecule is used as an electron acceptor, and chlorine atoms are sequentially removed and replaced with hydrogen atoms. Additional organic species or hydrogen is required as the electron donor. The hydrogen can be produced by degradation of a primary substrate, such as benzene, ethanol, acetate, or other such compounds. Dechlorination rates are highest for the most halogenated compounds; therefore, dechlorination of PCE is favored over dechlorination of VC (one of the daughter products of PCE dechlorination). Major et al. (1991) observed that significant dechlorination of the daughter products (e.g., VC) does not begin until the PCE has disappeared.

Reductive dechlorination has been shown to degrade PCE in a variety of anaerobic systems, using a variety of electron donors. Laboratory studies indicate that reductive dechlorination is carried out by certain fermentative, acetogenic, nitrate-reducing, sulfate-reducing, and methanogenic bacteria. Methanogenic conditions appear to be the most favorable for PCE degradation. Butler and Barker (1996) cite multiple laboratory studies that indicate PCE can be transformed to TCE to *cis*-1,2-DCE to VC under highly reducing conditions. Most studies indicate that *cis*-1,2-DCE is the dominant dechlorinated species, although some cite mostly *trans*-1,2-DCE. Complete mineralization to carbon dioxide and ethylene have also been detected in some studies. Studies have shown that the less chlorinated daughter products (DCE and VC) degrade more readily under aerobic conditions than anaerobic conditions.

Limited empirical data on *in situ* PCE degradation rates are available. Most of the studies demonstrating PCE biodegradation have been performed in laboratories using either pure cultures or biological communities derived from wastewater treatment systems, not groundwater. Furthermore, the studies are generally conducted at relatively warm temperatures (often above 20^{o}C). In general, *in situ* PCE degradation is dependent upon the presence of other organic compounds. These compounds serve two functions: they act as electron donors for PCE degradation, and they are themselves degraded by subsurface bacteria, depleting available oxygen in the process.

Butler and Barker (1996) conclude that *in situ* biodegradation of PCE is difficult. In natural settings, the degradation of halogenated aliphatic compounds (including PCE) is typically very slow, with estimated half-lives on the order of 6 months to 50 years (Brubaker, 1988). Typical subsurface environments do not favor PCE degradation.

PCE is resistant to degradation in aerobic environments, and anaerobic PCE degradation results in the accumulation of daughter products that are also pollutants. To date, complete degradation of PCE is restricted to laboratory-scale, multiple stage reactors.

PART 2 REMEDIAL INVESTIGATION

5 INTRODUCTION

The field work for the RI included subsurface soil and groundwater sampling conducted across the site, a localized soil gas survey conducted on the northeast and northwest sides of the former dry cleaners building, installation and downloading of dataloggers to measure site groundwater levels, and slug testing. Appendix A (photographs 14 through 38) provide photographs taken during various stages of the RI field work.

As discussed in the introduction to this report, the RI investigative work was performed in three phases. Each of the first two phases identified significant data gaps that were addressed in subsequent field efforts. Phase I was completed in June and July 1999, Phase II was completed in September 1999, and Phase III was completed in November 1999 and January 2000. Preliminary results of the RI field investigations were provided in interim status reports (July 30, 1999; September 24, 1999; and January 10, 2000). In addition, three quarterly groundwater monitoring events were completed during this timeframe: in July, October, and December 1999. The results of the three phases of the RI and the two quarterly groundwater monitoring events are discussed in this report. Tables 5-1 and 5-2 provide a summary of sampling activities performed for this project. Exploratory borings and groundwater monitoring well locations are shown on Figure 4. Borehole logs are provided in Appendix E.

The primary objective of the RI was to support ADEC in delineating where COCs remain at the RTRVP site. Initially, the RI effort focused on filling data gaps in the center of the plume area. A comparison of the water table contours and PCE isopleths from previous studies showed an absence of monitoring wells along the inferred centerline of the PCE plume. The results of the initial investigation phase revealed an area of contaminated groundwater north of the previously-defined plume boundary. The primary goals of the second and third investigation phases were to delineate this plume, look for additional soil contamination source areas, evaluate the site groundwater flow paths, and evaluate the relationship between site groundwater plumes and the storm sewer.

5.1 RI REPORT ORGANIZATION

The RI section of this report is organized into sections. This introductory section describes the scope of work and deviations from the work plan. Following the introductory section is a “Findings” section (Section 6), which describes the overall findings of the RI. After the “Findings” section is a “Fate and Transport Analysis” section (Section 7), which contains an overall analysis of site hydrogeology and contaminant transport.

Table 5-1: Soil Sample Location, Type, and Quantity Summary

Location ID	Total Footage Drilled (feet)	EPA 8260B[1] (No. of Samples)	AK102[2] (No. of Samples)	AK103[3] (No. of Samples)	Sample Intervals (ft bgs)
Phase I: June/July 1999					
MW-15 (continuous logging)	40.5	4	--	--	12-14 / 22-24 / 30-32 / 34-36
MW-16	22	1	1	1	10-12
MW-17	35.6	4	--	--	15-17 / 20-22 / 27-29 / 34-36
MW-18 (continuous logging)	36	5	--	--	114-16 / 16-18 / 24-26 / 30-32 / 34-36
MW-19 (continuous logging)	16	2	--	--	8-10 / 12-14
MW-20 (near river)	12	2	--	--	5-7 / 10-12
SB-O/B-01	27	3	2	2	15-17 / 20-22 /
SB-O/B-02	27	3	2	2	15-17 / 20-22 /
Phase II: September 1999					
MW-21	22	1	--	--	15-17
MW-22	23	1	--	--	20-22
MW-23	23	1	--	--	20-22
MW-24	13.5	1	--	--	5-7
MW-25	22	1	--	--	20-22
MW-26	12	1	--	--	5-7
Phase III: November 1999					
MW-27 (installed by hand)	7	--	--	--	--
MW-28 (temporary)	14	1	--	--	12-14
MW-29	15	1	--	--	13-15
MW-30	20	1	--	--	11-13
MW-31	18	1	--	--	15-17
MW-32	23.5	1	--	--	20-22
MW-33 (boring only)	10	1	--	--	7
MW-34 (installed in January 2000)	21.5	1	--	--	10-12, 14-16, 19.5-21.5
Total Samples		37	5	5	

Notes:

-- No sample taken.

[1] EPA method 8260B for volatile organics

[2] Alaska method AK102 for DRO

[3] Alaska method AK 103 for RRO

Table 5-2a: Groundwater Sample Summary (for organic analyses)

	July 1999		September 1999	October 1999	November 1999	TOTAL
Location ID	**VOCs (EPA 8260B)[1]**	**PAHs (E610)[2]**	**VOCs (EPA 8260B)**	**VOCs (EPA 8260B)**	**VOCs (EPA 8260B)**	**VOCs (EPA 8260B)**
MW-1A	1	1				1
MW-1C (Deep)	1	1				1
MW-2	1	1				1
MW-3A	1	1		1		2
MW-4A	1	1		1		2
MW-5	1	1				1
MW-6	1	1		1		2
MW-7	1	1		1		2
MW-8	1	1		1		2
MW-9	1	1		1		2
MW-10	1	1		1		2
MW-11	1	1		1		2
MW-12	1	1		1		2
MW-13	1	1		1		2
MW-14	1	1		1		2
MW-15	1	1		1		2
MW-16	1	1	1	1		3
MW-17	1	1		1		2
MW-18	1	1		1		2
MW-19	1	1		1		2
MW-20	1	1		1		2
MW-21			1	1		2
MW-22			1	1		2
MW-23			1	1		2
MW-24			1	1		2
MW-25			1	1		2
MW-26			1	1		2
MW-27					1	1
MW-28					1	1
MW-29					1	1
MW-30					1	1
MW-31					1	1
MW-32					1	1
MW-34 (January 2000)						
Total Project Samples (not including QC)	21	21	7	23	6	57

Notes:

Blank indicates No sample taken.

[1] EPA method 8260B for volatile organics

[2] EPA method 610 for PAHs

Table 5-2b: Inorganic Compound Groundwater Sample Summary

	July 1999	**September 1999**	**October 1999**
Location ID	**Chloride/Sulfate *(E 300)*, TOC *(SW 9060)*, Alkalanity (E 310.1), Methane *(EPA RSK 175)*, Nitrate *(E 300)*, FE Iron (Field)**	**Chloride/Sulfate *(E 300)*, TOC *(SW 9060)*, Alkalanity (E 310.1), Methane *(EPA RSK 175)*, Nitrate *(E 300)*, FE Iron (Field)**	**Chloride/Sulfate *(E 300)*, TOC *(SW 9060)*, Alkalanity [1] (E 310.1), Methane *(EPA RSK 175)*, Nitrate *(E 300)*, FE Iron (Field)**
MW-4A	1		1
MW-6	1		
MW-8	1		1
MW-9	1		1
MW-11	1		
MW-14			1
MW-16	1		1
MW-17	1		
MW-18	1		
MW-19	1		
MW-20	1		1
MW-21		1	1
MW-22		1	
MW-23		1	1
MW-24		1	
MW-25		1	1
MW-26		1	1
TOTAL	10	6	10

Note[1]: Alkalinity was conducted on water samples collected from all 24 wells sampled in October 1999

5.2 GENERAL RI FIELD PROCEDURES

RI field activities performed at the RTRVP site generally followed the procedures outlined in the Final Work Plan for the RTRVP RI/FS (OASIS/Bristol, 1999a) and the two Work Plan Addenda (OASIS/Bristol, 1999b and 1999c), except as noted in Section 5.3. Brief descriptions of the monitoring well construction, soil sampling, and groundwater sampling procedures are outlined below.

5.2.1 Groundwater Monitoring Well Installation Procedures

The new monitoring wells were constructed with 2-inch interior diameter Schedule 40, PVC. The riser consisted of a 2-inch interior, Schedule 40, PVC (ASTM-D 1785) casing with flush-threaded joints. The well screen consisted of 10-foot sections of PVC with horizontal, 0.010-inch factory-slotted design. Screens were set so the middle of the screened interval is at or slightly below the estimated water level at the location. A sand pack consisting of U.S. No. 10-20 clean silica sand was placed around the screen, from the bottom of the borehole extending two feet above the top of the well screen. A 2-foot thick annular seal of chipped bentonite was placed above the sand in all monitoring wells. Following hydration of the chipped bentonite seal, the annulus of the monitoring well above the annular bentonite seal was grouted with a bentonite slurry. If the monitoring well was installed at a shallow depth, chipped bentonite was placed from the bentonite seal to near ground surface and then hydrated. Silica sand was added to the top approximately one foot of the boring to allow any potential surface water seeping into the flush-mount security cover to drain. All monitoring wells were fitted with lockable PVC caps.

The above procedure was modified for installation of two monitoring wells, temporary well MW-28 and monitoring well MW-27. The temporary well, MW-28, was installed in the Sterling Highway using a hydropunch. Hollow-stem augers were used to 15 feet bgs, and the hydropunch screen was driven from 15 to 17 feet bgs. The outer pipe was then retracted to expose the screen. The well was not developed and purged prior to sampling due to the slow rate of recharge and time constraints for drilling (i.e., the crew had only a 2-hour timeframe when traffic could be diverted around the drilling site). The sample was collected using a teflon bailer, provided by Hughes Drilling, that was designed specifically for the hydropunch. It was decontaminated by steam-cleaning prior to use. Monitoring well MW-27 was installed as a 2-inch diameter drive point instead of a monitoring well, because the drill rig could not access the desired location (storm sewer backfill near the Kenai River).

5.2.2 Subsurface Soil Sampling

As subsurface soil samples were retrieved from the boring, they were screened for organic vapor concentrations using a 580B photo ionization detector (PID). When the soil sampling device (split spoon sampler) was opened, the PID was used to screen the soil for detectable organic vapors that may be indicative of volatile organic compounds (VOC) contamination. An aliquot of soil from each split spoon was contained in a Ziploc® bag for analysis of organic vapor content consistent with the Underground Storage Tank Procedures Manual (18 AAC 78). Organic vapor content readings were recorded and used by the field team to decide which soil samples from each boring would be selected for analysis. The PID was also be used at drill site locations to monitor organic vapor for health and safety concerns of the drilling and sampling field crews.

Soil destined for VOCs analysis was immediately containerized to avoid volatilization before conducting lithologic descriptions of soils sampled. The remainder of the soil samples were visually classified and recorded on the boring logs (provided in Appendix E). Subsurface soil samples were placed into the appropriate sample containers, for field screening and laboratory analysis. Sample containers were pre-cleaned and certified free of contaminants by the analytical laboratory.

5.2.3 Groundwater Sampling

After installation, monitoring wells were developed by purging with either a disposable bailer or a peristaltic pump. Monitoring wells were generally developed for an hour or until approximately five wellbore volumes had been removed and water quality parameter readings (e.g., pH, conductivity, temperature) stabilized. Well development sheets are provided in Appendix F.

Monitoring well sampling consisted of the following four activities:

- Measuring the depth-to-water from the top of the well casing;
- Evacuating three wellbore volumes of water from the monitoring well (purging);
- Measuring and recording of groundwater temperature, pH, and specific conductivity after each wellbore volume has been purged;
- Measuring and recording of groundwater oxidation-reduction potential (ORP), turbidity, and dissolved oxygen (DO);
- Collecting the groundwater sample.

Groundwater sample collection procedures are described below.

A disposable polyethylene bailer was used to collect groundwater samples from monitoring wells.

When transferring water from the bailer to sample containers, care was taken to avoid agitating the sample, which promotes the loss of VOCs.

When transferring water from the bailer to sample containers, a purge water bucket (5-gallon capacity) was positioned beneath the transfer point to catch any incidental spillage of water. The incidental spillage was transferred to the purge water containers.

Groundwater samples were placed in an iced cooler immediately upon collection.

All observable physical characteristics of the groundwater (e.g., color, turbidity) were recorded in the log book and in groundwater sample data sheets (groundwater sample data sheets are provided in Appendix F).

Sample pH, temperature, specific conductivity, turbidity, and DO was measured using a Horiba U-80 Multimeter. ORP was measured using an Oakton ORP Tester. The instruments and field screening methods were calibrated and operated in accordance with the manufacturer's requirements.

Weather conditions at the time of sampling were recorded (e.g., air temperature, wind direction, recent heavy rainfall, drought condition).

Chain-of-custody procedures were used to document sample handling between sample collection and sample receipt by the analytical laboratory.

If the well did not recover quickly to permit the removal of three successive volumes, the well was bailed dry and sampled immediately following a recovery sufficient to collect a sample.

Beginning with the September 1999 field work, groundwater samples from newly-installed monitoring wells were field-screened using Quick Test™ kits, manufactured by Envirol, Inc. The test kits measure total organic chlorine concentrations, with a quantitation limit of 4 μg/L. Results from the field testing were used to guide decisions about the necessity for additional monitoring wells and their optimum locations.

5.2.4 Datalogger Installation

Levelogger transducers were placed in five monitoring wells for long-term monitoring. The transducers record water level two times per day, at 8 am and 8 pm. To evaluate groundwater level variations in different water-bearing zones and across different areas of the RTRVP site, the transducers were placed into MW-15, MW-20, MW-16, MW-19, and MW-24.

5.3 DEVIATIONS FROM WORKPLAN

Deviations from the work plan and work plan addenda were generally caused by field conditions indicating that the work plan procedures would not be possible or preferable to an alternative procedure. The goals of the RI/FS program were not compromised by any of the work plan deviations.

Deviations from each work plan are listed in this section. Methodologies described in the work plan are numbered and italicized, and the deviations are described immediately below the applicable methodologies.

5.3.1 Deviations from the 21 June 1999 Work Plan:

1) Page 4-1, Section 4.1, second paragraph. *Soil destined for VOCs analysis will be immediately containerized to avoid volatilization before conducting lithologic descriptions of soils sampled. The remainder of the soil samples will be visually classified and recorded on the boring logs. Subsurface soil samples will be placed into the appropriate sample containers, for field screening and laboratory analysis.*

Soil sample (Sample ID 99-RT-022-SL), collected from the 34 to 36 foot interval in MW-18 for VOCs, was not collected directly from the split spoon. The work plan specified that a bottom-of-hole sample from MW-18 was to be collected at a depth greater than the 34 to 36 foot interval. A silt layer was encountered at the depth interval of 34 to 36 feet and drilling was stopped to prevent penetrating the confining layer. The analytical sample was taken from the (PID) grab sample since a split spoon sample core was unavailable. Approximately 45 minutes elapsed from the time the PID grab sample was taken to the time when it was placed in the sample jar. During this time, the PID grab sample was contained in a Ziploc® bag. The analytical results could be lower than actual conditions, since some volatilization of contaminants may have occurred.

2) *Page 4-2, Section 4.2, 6th bullet. Sample temperature, pH,* DO, *ORP, and specific conductance will be measured and recorded.*

ORP measurements were not collected from monitoring wells MW-1C (deep), MW-4a, MW-5 through MW-10, MW-12, and MW-17 through MW-19, due to instrument malfunction. This deviation resulted in less ORP data available for the natural attenuation assessment; however, the natural attenuation assessment can still be made.

3) Page 5-4, Section 5.5. *A total of seven monitoring wells are proposed for installation during the Remedial Investigation…If the background well is suspected to contain contamination, an eighth well will be installed.*

Only six monitoring wells were installed. MW-21 was not installed, because its proposed drilling location was covered by one of the on site soil treatment cells. MW-22 was not installed since it was considered as an optional well to be drilled if field screening at MW-16 indicated evidence of contamination. Although laboratory results ultimately indicated high levels of contamination in the MW-16 groundwater sample, no evidence of contamination was observed during the field screening of MW-16 soil samples (the laboratory soil sample result also did not indicate significant contamination). Photograph 14 indicates Hughes Drilling beginning to drill MW-16. Although MW-21 and optional well MW-22 were not installed during the first phase of RI field work (June 1999), additional monitoring wells were installed in areas near the proposed locations for MW-21 and MW-22 during subsequent field work phases. MW-22 was installed as a background monitoring well, and MW-26 was installed near the soil treatment cell in the second phase of RI field work.

4) Page 5-4, Section 5.5.1.2. *The work plan specified that a "10-foot section of screen" was to be used for all monitoring wells.*

A 5-foot section of screen was placed in monitoring well MW-20 due to the shallow depth to the confining layer and shallow depth to the water table. No detrimental effects to the investigation as a result of this deviation are expected.

5) Page 5-4, Section 5.5.1.3. *The work plan specified that a minimum of two feet of artificial sand pack will separate the top of the well screen from the bentonite seal.*

The sand pack thickness above the top of the well screen at MW-20 was reduced to 0.5 feet due to the shallow depth of the well. No detrimental effects to the investigation as a result of this deviation are expected, because the 0.5 feet of sand pack between the top of the well screen and bentonite seal is expected to allow adequate separation between the well screen and the bentonite seal so as to not allow the bentonite into the well screen.

6) Page 5-4, Section 5.5.1.4. *The work plan specified that a minimum two-foot thickness of bentonite seal was to be installed in all the monitoring wells.*

The bentonite seal thickness was reduced to 0.5 feet in well MW-20 due to the shallow depth of the well. No detrimental effects to the investigation as a result of this deviation are expected because the 0.5 feet of bentonite seal should adequately keep surface water from migrating into the well boring.

7) Page 5-5, Section 5.5.1.5. *The annulus of the monitoring wells above the annular bentonite seal will be grouted with a lean cement bentonite mix.*

Based on the recommendation of the driller, Hughes Drilling, who has considerable drilling experience in the area including the RTRVP site, a well grout without a cement additive is less likely to frost jack and provides a better seal. The grout mixture used to seal the wells did not contain a cement additive and consisted of a bentonite powder and water slurry installed using a tremie pipe. No detrimental effects to the investigation as a result of this deviation are expected.

8) Page 5-5, Section 5.6, first paragraph. *Each newly-installed well shall be developed with a disposable bailer.*

To increase well-developing efficiency, a peristaltic pump was used in conjunction with a disposable bailer to develop monitoring wells MW-19 and MW-20. No detrimental effects to the investigation as a result of this deviation occurred, because the well was still surged with the disposable bailer and adequately developed.

9) Page 5-5, Section 5.6, first paragraph. *Development activities shall continue for a period of up to one hour or until five wellbore volumes have been purged.*

Less than five well bore volumes were removed during development at MW-15, MW-17, and MW-18 due to slow well recovery and poor yield. At MW-14, all field parameters had stabilized after 25 gallons of water was purged. No detrimental effects to the groundwater sample results are expected as a result of this deviation.

10) Table 1. *Table 1 indicates that three soil borings were planned.*

The third boring (SB-0/B-03) was not installed, because its proposed location was covered by one of the soil treatment cells on site. During the Phase II field effort, MW-26 was installed, which met the objective of the earlier planned soil boring.

5.3.2 Deviations from the 20 August 1999 RI/FS Work Plan Addendum:

1) Page 2, second bullet. *The concentrations of groundwater contamination migrating toward the river from the excavation are still unknown. In the original work plan, monitoring well MW-21 was planned to answer this question. However, this well could not be installed because one of the soil treatment cells covering the planned monitoring well location. In the proposal for this additional characterization work scope, we proposed installing this well. However, the soil treatment cells are still* on site*, and a schedule for their removal has not yet been established. Therefore, although we still recommend installation of this monitoring well, it is not included in the scope of work in the work plan addendum.*

While proposed monitoring well MW-21 was not installed due to the reasons discussed above, a monitoring well (MW-26) was installed near the proposed location of MW-21. This well is located adjacent to and on the north side of the westernmost soil pile. Monitoring well MW-26 generally met the objectives of proposed monitoring well MW-21. Note that a monitoring well numbered MW-21 has been installed at the RTRVP site; this MW-21 is not at the location discussed here.

2) Page 2, last bullet. *Install four additional monitoring wells near the northern site boundary. Three monitoring wells will be located northwest (MW-21), north (MW-22), and northeast (MW-23) of MW-16.*

The monitoring well designated as MW-21 in the addendum is now MW-25. While the well ID has changed from the addendum, the location of the monitoring well has not. A fourth additional monitoring well (MW-21) was installed to the west of MW-16. The effects of this deviation (addition of a fourth monitoring well) were beneficial to the investigation.

3) Page 4, next to last paragraph. *The monitoring wells will be installed according to the specifications of the 21 June 1999 work plan. The monitoring wells will consist of 2-inch interior diameter, PVC casing with a 10-foot section of 0.010-inch slotted PVC screen…*

A 5-foot section of screen was placed in MW-26 due to the shallow depth to the confining layer and shallow depth to the water table. No detrimental effects to the investigation as a result of this deviation are expected.

4) *The work plan specified that a minimum of two feet of artificial sand pack will separate the top of the well screen from the bentonite seal.*

The sand pack thickness above the top of the well screen at MW-24 was reduced to one foot due to the shallow depth of the well. No detrimental effects to the investigation as a result of this deviation are expected because the one-foot layer of sand pack between the top of the well screen and bentonite seal is expected to allow adequate separation between the well screen and the bentonite seal so as to not allow the bentonite into the well screen.

5) *The work plan specified that a minimum two-foot thickness of bentonite seal was to be installed in all the monitoring wells.*

The bentonite seal thickness was reduced to one foot in well MW-24 due to the shallow depth of the well. No detrimental effects to the investigation as a result of this deviation are expected because the one-foot layer of bentonite seal should adequately keep surface water from migrating into the well boring.

6) *The work plan specified that development activities shall continue for a period of up to one hour or until five wellbore volumes have been purged.*

Less than five well bore volumes were removed during development at monitoring well MW-24 due to slow well recovery and poor yield. No detrimental effects to the groundwater sample results are expected as a result of this deviation.

5.3.3 Deviations from the 29 October 1999 RI/FS Addendum:

1) Page 3, first paragraph. *To meet the objectives stated above, seven monitoring wells, 20 temporary soil gas monitoring points, and a storm sewer remediation system will be installed at the River Terrace site. Soil, groundwater, and soil vapor samples will be collected during completion of these activities. In addition, site hydrogeology will be evaluated using inorganic analytical and colorimetric dye testing techniques. Each of these activities is discussed in greater detail below.*

Seven monitoring wells, plus a possible contingency well, were to have been installed. Only five of the seven monitoring wells, plus the contingency well, were installed for a total of six monitoring wells. The reasons that two of the seven monitoring wells were not installed are described below in deviation no. 4 (MW-30) and deviation no. 5 (MW-33).

The artesian well on the north side of the former dry cleaner building was in the way of one of the 20 planned soil gas monitoring points could not be installed. Therefore, a total of 19 soil gas monitoring points were installed and sampled. Photographs 24 and 25 in Appendix A show soil gas locations.

The storm sewer remediation system has not yet been installed at the RTRVP site, but a proposal has been submitted to ADEC.

The colorimetric dye testing has not yet occurred at the RTRVP site due to the presence of ice on the Kenai River. With the ice on the river, the dye could not been seen if it were flowing into the river. The dye test is expected to occur sometime prior to May 2000.

2) Page 3, third and fourth paragraphs. *An attempt will be made to install the monitoring wells utilizing direct push methods instead of a hollow-stem-auger drilling technique. If direct push is not successful, a hollow-stem-auger drill rig will be used and the monitoring wells installed according to the specifications spelled out in the 21 June 1999 work plan.*

Monitoring well MW-27 was to be installed in the storm sewer backfill between the highway and the outfall, near the bridge abutment. Because of the steepness (not accessible for a drill rig) of the proposed monitoring well location, and in order to place the well in the sewer backfill, MW-27 was installed closer to the river than was proposed in the work plan addendum. It is just upgradient from the outfall. It was installed as a drive point by hammering a 1.5-inch diameter metal pipe with a 2-foot slotted screen into the ground using a sledgehammer (see photographs 28 and 29 in Appendix A). No detrimental effects to the investigation as a result of this deviation are expected.

3) *The work plan specified that a minimum two-foot thickness of bentonite seal was to be installed in all the monitoring wells.*

The bentonite seal thickness was reduced to one foot in well MW-29 due to the shallow depth of the well. No detrimental effects to the investigation as a result of this deviation are expected because the one-foot layer of bentonite seal is expected to adequately keep surface water from migrating into the well boring.

4) Page 4, third bullet. *Monitoring well, MW-30, will be installed between MW-22 and MW-23. This well is being installed at the request of the property owner's consultant.*

At the request of the owner's consultant, Steve Rog, this monitoring well was not installed (although Jim Gill of Hart Crowser had originally requested it of ADEC), because MW-32 was installed near the proposed location for MW-30. No detrimental effects to the investigation as a result of this deviation are expected.

5) Page 4, sixth bullet. *Monitoring well, MW-33, will be installed on the southwest side of the building. This will be installed in the angle boring mentioned below.*

Although two attempts were made to drill MW-33 and install a monitoring well, the attempts were unsuccessful because the angle of the hollow stem augers could not be maintained and the augers "walked" toward the surface as drilling progressed. Angle boring drilling attempt is shown on photograph 27 in Appendix A. However, a soil sample was obtained from the boring and the drilling/sample location is identified in this report as MW-33. As a result of this deviation, conditions under the former dry cleaner building have not been characterized.

6) Page 5, first paragraph. *If the well can be installed using direct push techniques, the monitoring well casing will be installed upon reaching groundwater or at a depth ranging between 16 and 20 feet bgs. The drive point well will be developed and purged prior to sample collection using microbailers or peristaltic pump sampling equipment.*

This temporary well, MW-28, was installed using a hydropunch. Hollow-stem augers were used to 15 feet bgs, and the hydropunch screen driven from 15 to 17 feet bgs. The outer pipe was then retracted to expose the screen. The well was not developed and purged prior to sampling due to the slow rate of recharge and time constraints for drilling. No detrimental effects to the investigation as a result of this deviation are expected.

7) Page 5, third paragraph. *If the field screening results for the soil and groundwater samples collected from this location indicate PCE is present, a contingency well MW-34 may be installed northwest to further characterize the extent of any PCE impacts.*

The contingency well identified as MW-34 in the work plan addendum was installed, but it has been designated MW-30. No detrimental effects to the investigation as a result of this deviation are expected. MW-30 is shown being drilled in photograph 30 in Appendix A.

8) Page 5, fourth paragraph. *Monitoring well MW-33 (angle boring well), will not be installed using direct push methods. Instead, this well will be installed using hollow stem auger techniques. Split spoon samples will be collected at 5-foot intervals beginning at 10 feet bgs. The split-spoon sampler will be advanced by pushing and turning since the boring will be drilled at an approximate angle of 60-degrees where the hammer on the drill rig cannot be used to drive the spoon. The split spoons used in this application are welded shut. Therefore, the soil sample will be retrieved from the spoon by tapping the spoon and allowing the*

soil to fall into sample containers. Once the desired well depth of approximately 25 feet bgs is reached, two-inch diameter PVC well casing materials will be installed in the boring and the well completed as is described in the Final Work Plan, RTRVP *RI/FS, Soldotna, Alaska prepared by OASIS/Bristol and dated 21 June 1999.*

As explained in comment 5 above, this monitoring well was not installed. Two unsuccessful attempts were made to push the split-spoon sampler to collect a soil sample. The augers were advanced, at an angle, a total of ten feet and one soil sample was collected from the auger flights from a depth of approximately seven feet. Angle boring at MW-33 is shown Appendix A photograph 27. Because the soil sample was collected from the auger flights and not from an enclosed split spoon, the analytical results may be lower than actual conditions since some volatilization of contaminants may have occurred. As a result of this deviation, conditions directly under the building could not be observed.

9) Page 5, next to last paragraph. *Soil and groundwater samples will be tested in the field for PCE/TCE using field test kits (Quick Test™) manufactured by Envirol, Incorporated.*

While groundwater samples were tested using the field test kit identified above, no soil samples were analyzed for PCE/TCE using a field test kit. Photographs 21 and 22 in Appendix A depict PCE/TCE field test kits on groundwater. Soil cannot be tested using the Envirol test kit. Soil was, instead, field-tested using a PID and Drager™ Tubes as described in the work plan addendum (see photograph 20 in Appendix A). Soil samples were also submitted to a laboratory for analysis. No detrimental effects to the investigation as a result of this deviation are expected.

10) Page 6, third paragraph. *A soil gas survey will be conducted in the vicinity of monitoring well MW-16 and the former dry cleaning building. The soil gas survey will be completed using a 15-foot grid pattern around the north and west sides of the former dry cleaning building. The locations of these borings are presented in Figure 2 (*work plan figure number, not RI/FS figure number*). Twenty temporary sample points will be installed for the collection of soil gas samples.*

Because of the presence of an artesian well on the north side of the former dry cleaning building, one of the 20 planned sample points could not be installed. The other sample points adequately covered the area on the north side of the building. No detrimental effects to the investigation as a result of this deviation are expected.

11) Page 7, second paragraph. *The* on site *artesian well and the city water (if possible) will be temporarily shut down for a period of 48 hours.*

Because of the cold air temperatures and the concern that water pipes to the mobile homes supplied by water from the artesian well at the site might freeze and burst, the artesian well was shut down for only a period of 24 hours. Photograph 31 in Appendix A shows the custody seal placed on the valve shutting water off from the RV park. A period of 24 hours should be sufficient for collecting groundwater level data. No detrimental effects to the investigation as a result of this deviation are expected.

12) Page 9, second full paragraph. *Monitoring well development and purgewater containing PCE and its degradation byproducts generated during this field effort will be removed from the site by Philip Environmental Services on November 8, 1999.*

The monitoring well development water and purgewater were removed from the site by Philip Environmental Services on November 12, 1999. No detrimental effects to the investigation as a result of this deviation are expected.

6 FINDINGS

6.1 TOPOGRAPHY

6.1.1 Survey Landmarks

OASIS/Bristol contracted Eagle River Engineering Services of Eagle River, Alaska and McLane Consulting Group of Soldotna, Alaska, both registered land surveyors with the State of Alaska, to conduct horizontal and vertical surveys of monitoring wells, soil borings, and selected landmarks. Four surveying events were conducted—one following each monitoring well installation event. The initial RI surveying event was conducted in July 1999; all existing monitoring well locations and selected landmarks were surveyed. Subsequent surveying events tied newly-installed monitoring wells into the network. The July and September surveying was performed by Eagle River Engineering Services, and the November and January surveying was performed by McLane Consulting Group. Monitoring well information is summarized on Table 6-1.

Horizontal surveys were performed to Third Order, Class 1 standards (USDC, 1978) which have a relative accuracy between directly connected adjacent points of one part in 10,000. Vertical surveys were performed using differential leveling with a relative accuracy of one part in 100 (0.01 feet). All location and elevation data are traceable to benchmark "Soldotna" located on the northern side of the Sterling Highway Bridge (shown on Figure 3). Measurements were taken at a marked measuring point (MP) for horizontal coordinates and elevation. The MP, located on top of each monitoring well casing, is a reference point of known elevation from which all water elevation measurements are taken. The results of the horizontal and vertical survey of the monitoring stations are shown in Figure 3. The surveying data are provided in Appendix B.

6.1.2 Site Topography

Surface topography at the site generally slopes toward the Kenai River. Fill is present over practically the entire site, obscuring pre-development topography. Construction of the adjacent Sterling Highway and Kenai River Bridge created an embankment along the northwest side of the site. Various filling operations apparently occurred during development of the site resulting in the creation of a "terrace" in the general vicinity of the October 1997 excavation work. Within approximately 50 to 60 feet of the normal river-stage water line, the ground slope steepens, dropping to the Kenai River along an escarpment trending parallel with the river. Photographs 18 and 33 in Appendix A show various site views of the RTRVP site. Surface elevations based on the 1999 survey results and estimated drainage flow directions are presented on Figure 3.

6.2 GEOLOGY

The Kenai Peninsula is underlain by bedrock composed of Permian-aged metamorphosed shale, greenstone, and conglomerate rocks, as well as older limestone and younger siltstone, sandstone, and coal (Bailey and Hogan, 1995). Exact depth to bedrock in the Soldotna area is unknown. However, ADOT&PF Boring No. 1 drilled in 1963 at the Kenai River Bridge reportedly encountered a one-foot thickness of coal at a depth of approximately 85 feet that may indicate bedrock at the RTRVP site (State DOH, 1963). In 1997, across the Kenai River Bridge from the RTRVP site, ADOT drilled a test hole through 185 feet of alluvial material without encountering bedrock (Barber, 1997).

Table 6-1: Monitoring Well Information

6.2.1 Soils and Stratigraphy

The shallow (less than 50 feet deep) subsurface at the RTRVP site is comprised of fill, alluvial sands and gravels near the Kenai River, tills, and glaciofluvial sands within the tills. The boring logs from all soil borings and monitoring wells installed at the RTRVP site during the RI and from previous investigations are provided in Appendix E. Figure 4 shows the locations of cross-sections that were drawn from information contained in boring logs from this investigation and from logs and excavation reports from previous site investigations. The generalized hydrogeologic cross-sections through the site are shown in Figures 5 through 10. Figures 5 (A-A') and 6 (B-B') are cross-sections through the center of the site. Figure 7 (C-C') is a cut through the Sterling Highway and central section of the site. Figures 8 (D-D') and 9 (E-E') are cuts through the northern area of the site, from MW-31 (Figure 8) and MW-34 (Figure 9) on the west side of the Highway to MW-16. Figure 10 (F-F') is a section running along the west side of the Sterling Highway. These six cross-sections show generalized soil types, the position of the water table, soil and groundwater analytical results for PCE and its most commonly-detected degradation product (cis-1,2-DCE). The November 1999 water level and groundwater contaminant data are portrayed on these figures, except for Figure 9. Figure 9 (E-E') presents December 1999/January 2000 water levels and groundwater concentrations, because MW-34, which was installed in January 2000, is included on this figure. Water levels were measured in both December 1999 and January 2000 in several wells near MW-34; the water levels indicate a stable water table over this time (elevation changes of approximately 0.01 feet); therefore, information from these two times can reasonably be shown on one figure.

Sieve analysis and total organic carbon (TOC) data are available for site soils from 1997 excavation sampling results. As identified in Table 6-2, six excavation perimeter samples were submitted for these analyses (see Figure 2 for sample locations). To put the sample locations in context with currently-existing site monitoring wells (which are not shown on Figure 2), all six samples were collected from the excavation area southwest of MW-26, southeast of MW-4A, and northeast of MW-9. The sample results are representative of the till and alluvial sands at the site. In addition, E&E (1998b) presented TOC results for two soil samples (collected from the water table aquifer in MW-5 at 4 to 6 feet bgs and collected from the till in MW-6 at 8 to 10 feet bgs). The TOC and sieve analysis data are tabulated below in Table 6-2:

6.2.1.1 Fill

Two primary areas of fill are present on the site: an extensive area of fill reportedly due to highway and utility construction activities, and backfill from the October 1997 and June 1998 excavation activities. The RTRVP owner and/or his consultants reported that highway construction activities during the 1960s resulted in up to 17 feet of fill brought onto the RTRVP property. The area of maximum fill corresponds to a pronounced topographic embankment upon which the Sterling Highway is built. In the area of the October 1997 and June 1998 excavations (shown in Figure 2 and in cross section A-A' [Figure 5]), the fill thickness varies from approximately 5 to 35 feet thick. Excavated contaminated soils are currently contained in above-grade treatment cells at the site. The 1997-1998 fill material consisted of sand and gravel (RP's interim AOC report, 1998; Hart Crowser, 1998a).

Fill placed at the RTRVP site prior to 1997 appears to have been deposited directly on organic soils in some locations. These organic soil zones are typically observed to be highly porous compared to adjacent till or fill, and contain highly organic materials that are potential pathways for contaminant migration.

Table 6-2: Soil Sieve and TOC Data

Sample ID	TOC (percent, dry weight basis)	Soil Classification	Gradation (%Gravel: %Sand:%Fines)	Estimated Hydraulic Conductivity* (ft/day)
P1-3	0.13	Silty Sand, SM	11:40:49	10^{-1} to 10
P2-16	0.33	Silty Sand, SM	27:36:37	10 to 10^{2}
P3-9	0.16	Sandy Silt, ML	13:35:52	10^{-2} to 1
P3-15	0.15	Silty Sand, SM	27:25:48	10 to 10^{2}
P5-10	0.19	Sandy Silt, ML	15:30:55	10^{-2} to 1
P5-15	0.36	Silty Sand, SM	29:24:47	10 to 10^{2}
053-SL (MW-5)	0.19	NA	NA	NA
062-SL (MW-6)	0.35	NA	NA	NA

**Reference: (USGS, 1983)*

NA: not available

6.2.1.2 Alluvial Sands and Gravels

At the RTRVP site, most boreholes drilled adjacent to the Kenai River (including one hole drilled through the deck of the bridge over the river) have penetrated approximately 6 to 12 feet of alluvial sands and gravels overlying very stiff silty till. Predevelopment aerial photographs indicated that alluvial deposits extend approximately 100 feet northwesterly from the OHW of the northern bank of the Kenai River. Site borehole logs indicate this is a reasonable interpretation.

6.2.1.3 Till and Glaciofluvial Sands within the Till

Tills at the RTRVP site are typically very stiff silt with sand, clay, gravel, and cobbles. Multiple till layers were observed during excavation activities in 1997 based on subtle variations in color and size gradation. However, most soils observed in the site contain approximately 40 to 60 percent fines. Thin layers were noted with as little as 20 percent to as high as 100 percent silt and clay. The layering observed in the till layers likely provides stratigraphic horizons for lateral migration of DNAPLs (see discussion in Section 4.1). Using borehole data, the differences between tills are subtle and difficult to distinguish and correlate across the site.

Glaciofluvial sands within the tills have been noted in several boreholes (e.g., MW-15, MW-17, and MW-18). The sands are typically fine-grained silty sands from 1 to 5 feet thick. A 2 foot core of glacial till commonly found at the RTRVP site is depicted in photograph 15 of Appendix A.

6.3 HYDROLOGY

Most of the groundwater on the Kenai Peninsula is located within unconsolidated deposits of glacial outwash, fluvial, and eolian (wind-blown) sediments (Bailey and Hogan, 1995). Lacustrine and eolian deposits typically are fine-grained silts and clays that are of limited use for providing water to domestic wells. Till and fluvial deposits also contain fine-grained sediments, but locally, zones of saturated sand and gravel are sufficiently thick to yield water to wells.

6.3.1 Site Groundwater Conditions

Based on boring logs from monitoring well installations and general hydrogeologic knowledge of the region, three water-bearing zones have been hypothesized:

- upper aquifer, which is unconfined and the most shallow of the three hydrogeologic units;
- semi-confined water-bearing zone lying below the upper aquifer; and
- lower confined aquifer.

The upper aquifer at the RTRVP site is separated from the lower confined aquifer by a dense silty till that functions as a confining unit. The semi-confined water-bearing zone consists of glaciofluvial sands within the silty till unit. The lower confined aquifer, semi-confined water-bearing zone, and upper aquifer are discussed in the following subsections, along with a discussion of the confining unit. The hypothesized communication between the water-bearing zones is included in the following discussion.

6.3.1.1 Lower Confined Aquifer.

Two test wells drilled at the site of the proposed Kenai River Bridge by ADOT in 1963 tapped a confined aquifer exhibiting flowing artesian water at a rate of up to 50 gallons per minute (State DOH, 1963). The test wells were completed at a depth of approximately 85 to 95 feet bgs, and the static hydraulic head was 18 feet above ground surface. These wells indicate that a vertically upward hydraulic gradient exists between this aquifer and shallower aquifers.

The communication between the lower confined aquifer and shallower aquifers has not been evaluated, because of insufficient data. No investigative monitoring wells have penetrated the lower confined aquifer. However, the presence of a vertically upward hydraulic gradient is significant, because it will inhibit shallow aquifer contamination from migrating into the lower confined aquifer. The lower aquifer is the assumed source of drinking water for the RTRVP property's water wells.

6.3.1.2 Confining Unit

The till layer underlies the upper aquifer at varying depths across the site (from less than 5 feet bgs to greater than 20 feet bgs). Figure 11, a contour map of the top of the till, indicates a relatively high ridge area across the center of the site, extending from MW-30 eastward to MW-29. Till was encountered in MW-30 at an elevation of approximately 61 feet, which was the highest observed elevation of the till surface at the site. Figures 5 (A-A') and 9 (E-E') illustrate cross sections through the high point of the till near MW-30. The till generally slopes down toward the river south of the high ridge, and it generally slopes northward, or away from the river, north of the ridge. A comparison of Figures 8 (D-D') and 9 (E-E') illustrates the northward-sloping till surface. Figure 11 indicates a low area in the till around MW-16 that slopes down toward MW-25 and MW-22.

Based on the presence of PCE at depths up to 30 feet bgs (RP's interim AOC report, 1998) and an understanding of PCE migration (see discussion in Section 4.1.1), the till should not be regarded as impermeable. The till probably contains fractures, joints, root tubules, and stratigraphic layering or thin, permeable interbedding that allows flow between hydrogeologic units. The locations, orientations, and hydraulic conductivities of these more permeable zones are largely unknown.

6.3.1.3 Semi-Confined Water-Bearing Zone.

This intermediate flow regime consists of thin discontinuous semi-confined sand and silty sand water-bearing zones within the till. Several monitoring wells (MW-15, MW-17, and, MW-18) tap the water-bearing zone shown in Figure 6 (cross section B-B'). Groundwater in this unit is confined above and below by fine-grained till. It is probable that multiple, discontinuous water-bearing zones exist within the till at the site.

The connection between the semi-confined water-bearing zone and the underlying confined aquifer has not been evaluated, due to insufficient data. However, there is evidence that the upward gradient observed in the lower confined aquifer continues upward some distance into the till. An upward gradient is shown by the July, 1999 water level measurements in the nested well pair SB1-Deep/SB1-Shallow. The water level in SB1-Deep (screened from 44 to 49 feet bgs) was 5.8 feet higher than the water level in well SB1-Shallow (screened from 29 to 34 feet bgs) during July 1999 (see Figure 10 [F-F'] and discussion in Section 0). The upward trend was also observed in the September and October groundwater sampling. SB-1 Shallow and SB1-Deep are probably completed in two of several saturated zones present in the till.

The connection between the semi-confined water-bearing zone and the overlying water table aquifer is uncertain. At certain times of the year, there is apparently an upward gradient between the semi-confined water-bearing zone (tapped by MW-15, MW-17, and MW-18) and the water table aquifer, whereas at other times, the vertical gradient is apparently downward. As shown in Figure 6 (B-B'), the potentiometric surface for the semi-confined water-bearing zone is below the November 1999 water table, indicating a downward vertical gradient. However, in July the potentiometric surface was above the water table, indicating an upward vertical gradient. The water table measurements are further discussed in Section 0.

6.3.1.4 Upper Aquifer.

The upper aquifer lies within the fill or alluvium (consisting of silty sand and gravel material) above the till unit. The upper aquifer is likely recharged during precipitation events and snowmelt, and discharges to the Kenai River, although temporary high flow stages of the river create the potential for temporary recharge from the river as bank storage. This aquifer may be discontinuous across the site's upper (more elevated) region, and the groundwater flow in the upper regions may be through preferential pathways created from the topography of the underlying till. Near and along the river at the lower elevations of the site, alluvial sands and gravel are predominant and groundwater flow is likely more uniform. As discussed in Section 0, the upper aquifer is apparently hydraulically connected to the Kenai River.

The presence and movement of groundwater in the upper aquifer is affected by the topography of the till unit that it overlies. Figure 11 presents the interpreted elevation of the top of the till. There is an apparent groundwater divide in the upper aquifer, which coincides with a topographic ridge in the top of the till surface. The ridge is interpreted to be present in the upper portion of the site, and lies in an east-west direction along the southern edge of the site's former dry cleaner building. The ridge causes the upper aquifer to flow northward immediately north of the ridge (north of the former dry cleaner building), and southward from areas south of the former dry cleaner building. Figures 5 and 6 show the water table in cross section. On Figure 5 (A-A'), water northeast of the area near MW-14 flows to the northeast, and water southwest of MW-14 flows to the southwest. Figure 6 (B-B'), illustrates the groundwater high between MW-17 and MW-18, sloping off in both directions.

The groundwater flow is also affected by manmade changes to the subsurface associated with road and underground utility excavations. Figure 12 shows the approximate location of the utilities. Available ADOT design drawings from 1971 (see Diagram 6-1 below) show that a storm sewer drainage system is present in the Sterling Highway ROW (Figure 12). The storm sewer discharges just below the water level of the Kenai River downstream of the Kenai River bridge at the KRBO. The storm sewer slopes upward from its discharge point to a MH-1 in the Sterling Highway at the northwest end of the cross-connect shown in Figure 12 and Diagram 6-1. The storm sewer continues northeast under the Sterling Highway from the MH. The cross-connect is a concrete conduit extending from the lift station of the sanitary sewer system on the RTRVP property to the storm sewer line (see photograph 19 in Appendix A). It is identified as a "sewer lift station emergency overflow" on the design drawings. The cross-connect rises at a slope of 2 percent from the MH to the lift station shown in Figure 12 and Diagram 6-1.

This storm sewer system apparently has a local influence on groundwater flow directions in the upper aquifer (see Figures 5, 7, and 8). Plate 1 is an idealized isometric cross-section along the storm sewer. The elevation of the storm sewer invert at MH-1 is shown as approximately 53 feet above sea level in Diagram 6-1. This is approximately five feet lower than the water table elevation in wells MW-2 and MW-14, which are the nearest monitoring wells to the MH on each side of the Sterling Highway. Figures 5 (A-A') and 7 (C-C') show cross-sections depicting the relative locations of the storm sewer, top of till, and groundwater. The backfill for the storm sewer may be more permeable than surrounding soils and is certainly more permeable than the glacial till. The storm sewer/backfill structure is inferred to function as a local drain of groundwater near the highway (Figure 14). E&E (1999b) observed water flowing from the storm sewer into the Kenai River during non-storm periods (they observed clear storm sewer water, which was readily distinguishable from silty Kenai River water). This observation indicates that the storm sewer/backfill structure likely drains groundwater somewhere in the area. Figure 8 is a cross section showing the relative position of the storm sewer and the water table between monitoring wells MW-25 and MW-31, in the northern section of the site. As illustrated in this cross section, the potential exists for migration of contaminants into the storm sewer backfill and then down through the backfill toward the Kenai River. A comparison of Figure 9, which cuts through the storm sewer approximately 50 feet north of Figure 8, with Figure 8, indicates that the till surface is sloping northward in this area. The storm sewer backfill would not be trenched through till near MW-34, as it was near MW-31. Therefore, the storm sewer effects on the groundwater flow are expected to lessen in a northern direction. The storm sewer is located approximately at or below the water table in the vicinity of the RTRVP (from approximately MH-2 to the Kenai River). Conclusions regarding the effect of the storm sewer system on groundwater flow from the RTRVP site are provided in Section 7.3.1.5.

The city sewer and water lines are not anticipated to represent preferential flow paths for groundwater at the RTRVP site. Backfill associated with private sewer and water lines on the RTRVP property probably represent localized preferential flow paths, but they do not appear to significantly alter the groundwater flow pathways.

A buried sanitary sewer manhole exists on the site. The manhole is located north to northwest of the telephone box in line with MW-14. The sewer line from the RTRVP property, the sewer line traveling under the Soldotna Bridge, and the sewer line running south down the Sterling Highway all converge at the buried man hole. From there, the waste travels northwest along Riverside Drive.

Diagram 6-1: ADOT Design Diagram Showing Utility Locations (1971)

As-built drawings were not available for the sanitary sewer lift station located on the RTRVP property. Conclusions were drawn concerning the configuration of the lift station from information provided by the City of Soldotna. The pipe that is shown flanged off in photograph #19 (Appendix A) is the sewer line that continues on to the buried manhole transporting sanitary waste from the RTRVP property. According to the City of Soldotna, the waste that enters into the lift station is pumped into the flanged off pipe connecting into it outside of the lift station rings. The pipe is flanged off so the maintenance crews can access the pipe and clean it out if needed. The City of Soldotna also confirmed that the metal ring shown in Photograph #19 is indeed the abandoned cross connect pipe.

Invert elevations of the sanitary sewer line range in elevation from 68.1 to 65.2 feet above sea level. The water line ranges from 68.1 to 61.2 feet above sea level. All of these elevations are above the maximum December 1999 water table elevations of 59.24 feet and above the maximum top of till elevation of 60.8 feet (Figure 11). If site water table elevations in the vicinity of the water line increase by 1 to 2 feet, the water line backfill could potentially act as a preferential flow path on the west side of the Sterling Highway, but this should not impact GW contaminant flow from the RTRVP site.

The City of Soldotna indicated that information on the private water and sewer line at the RTRVP property are not available at the City of Soldotna and must be provided by the property owner. However, we reviewed the DOWL Engineers design drawing (dated 5/4/95) for the utilities on the RTRVP property to evaluate the potential impacts of these lines on groundwater flow.

According to the DOWL drawing, the water line elevation adjacent to the former dry cleaner building (in the vicinity of the sign) is approximately 64 feet. The design elevation of the water line “tee” is approximately 58 feet. When this information is compared to the top of till (March 2000 Draft RI/FS Figure 11), it indicates that the water line in the vicinity of the “tee” is likely excavated into the till. Specifically, the water line extending northward from the tee to approximately MW-15 is likely excavated into the till. This location is near of the groundwater divide and could provide a conduit for groundwater from north of the divide to flow south of the divide. However, since the probable excavation into the till is fairly short (less than 30 feet long), this effect would be expected to be only a localized phenomenon.

According to the DOWL drawing, the influent sewer line elevation at the lift station is approximately 51.25 feet, and the effluent sewer line elevation is approximately 65.7 feet. The DOWL map indicates sewer line elevations of 51.1 at the manhole adjacent to the soil cell. These elevations are lower than the influent sewer line elevation at the lift station, which does not make sense. Based on other information (the elevation of the cross-connect pipe surveyed in July 1999 at 57.25 feet and the sewer crossing invert elevation of 65.21 feet at the buried manhole shown on one of the as-builts), the influent and effluent sewer line elevations seem reasonable, so although there is apparently an error in either the influent sewer line elevation at the lift station or the sewer line elevations at the manhole, the elevations are close enough for this interpretation, the information on the DOWL drawing is adequate for interpreting the effect of the sewer lines on the groundwater flow pathways. When the sewer line elevation information is compared to the top of till it indicates that the sewer lines north and northeast of the lift station are significantly above the top of till and should not significantly affect groundwater flow patterns. South of the lift station, the influent sewer line is very likely excavated into the glacial till. The backfill from the influent sewer line could provide a conduit for groundwater flow from the lift station toward the manhole located adjacent to the western soil cell. However, this is also probably a localized effect.

Other utilities buried on site, including electric (Homer Electric), gas (Enstar), cable (PTI), and telephone (GCI), at the RTRVP property do not represent preferential flow paths for groundwater at the site. These utilities are buried no deeper than 3-5 feet bgs. Approximate utilitity locations at the RTRVP site are represented on Figure 12.

With the exception of the storm sewer, most of the utilities in the ROW are probably buried too shallow to directly affect groundwater flow, although all of the burial depths are not known. Diagram 6-1 shows the sewer line at an elevation of 65 to 66 feet and a water line at an elevation of approximately 60 feet in the plane of the storm sewer. The maximum elevation of the top of the till surface is approximately 60 feet along the top of the till ridge (Figure 11). Based on these relative elevations, it is unlikely that any trenching into the till was necessary for burial of the sewer line. Depending on the unevenness of the till surface, some trenching into the till may have been necessary for burial of the water line. With the limited burial depth information, the potential effects of the water line on groundwater flow cannot be conclusively determined.

6.3.2 Water Elevations and Groundwater Flow Directions

During each RI field work phase and each quarterly groundwater monitoring event, water levels in the monitoring wells were measured using procedures detailed in the work plans. RI field activities were performed in accordance with the Final Work Plan for the RTRVP RI/FS (OASIS/Bristol, 1999a) and the Work Plan Addenda (OASIS/Bristol, 1999b and 1999c), and quarterly monitoring activities were performed in accordance with the Final Work Plan for Quarterly Groundwater Monitoring (OASIS/Bristol, 1999f).

Table 6-1 presents a summary of groundwater level elevations from July through January 2000. For each monitoring well, the water-bearing zone or aquifer within which it is interpreted to be completed is also provided in Table 6-1. The water level measurements in Table 6-1 need to be viewed with caution. Many of the wells at the site are subject to slow recharge following development and sampling due to the low permeability of the soils they are screened in, and water level measurements following these activities may not represent the true water level. In particular, water levels for SB-1 (shallow) after October 25, 1999, are believed to reflect a slowly recovering water level in the well, and not the static water table. The water levels measured in MW-30 (November 12, 1999 and December 13, 1999) are also interpreted to represent a slowly recovering water level and not the water table elevation.

Two contour maps were prepared to illustrate the groundwater flow regime in the site's upper (water table) aquifer. Figure 13 presents a potentiometric surface map based on water level elevations measured on July 7, 1999, and Figure 14 presents a potentiometric surface map based on water level elevations measured on December 13, 1999. Only the water levels indicated in Table 6-1 as representing the upper aquifer were used in these maps. Water level information from monitoring wells completed in the semi-confined water-bearing zone does not represent the true water table position in the upper aquifer. The isopleths are marked with question marks in areas where the interpretation is questionable because of a lack of data. The groundwater flow directions are depicted as blue arrows.

Figure 13 represents the interpreted potentiometric surface for July 7, 1999. The water table aquifer monitoring wells used to develop the contour map are those completed in the upper aquifer (prior to July 7, 1999); this interpretation has not been revised to reflect data collected after July 1999. In July 1999, MW-11 was interpreted to be screened in the semi-confined water-bearing zone, because its water elevation did not correspond well with the interpreted water table, and the completion formation was not clear from the boring log MW-11. It should be noted that the contours shown on the western portion of the Figure 13 are for illustrative purposes only to depict the concept of preferential flow through backfill material associated with underground utilities. The data necessary to correctly contour the water table under the Sterling Highway were not available in July.

Figure 14 represents the upper aquifer water table potentiometric surface isopleths interpreted from all upper aquifer monitoring wells, except MW-30. The water level measured in MW-30 is not interpreted to represent the water table level. Review of the boring log and well construction show that MW-30 is either capturing water from sandy stringers within the glacial till, or is acting as a sump for a very thin layer of water over the till.

The water level from MW-11 is included in Figure 14, because it is now interpreted to be completed in the upper water table aquifer. This interpretation is based primarily on two things: (1) The water level measured in MW-11 fits in with the groundwater ridge interpretation (which is based primarily on data from monitoring wells installed after July 1999), and (2) An analysis of common cations and anions indicates that the water in MW-11 is representative of the water table aquifer (see discussion in Section 6.4.2.2).

A comparison of Figure 13 and Figure 14 indicates a similar flow regime in the lower part of the site during these two time periods (July and December 1999). In both cases, water flows southward from the till ridge in the middle of the site toward the Kenai River. The horizontal gradient in the southern area is approximately 0.12, with a lower gradient (approximately 0.02) near the river. The only significant difference between Figures 13 and 14 in the lower part of the site is a groundwater reversal noted near the river in Figure 14 that is not present in Figure 13. This reversal was also not noted during the November 1999 groundwater measurements (Table 6-1). In December, the groundwater elevations adjacent to the river (e.g., MW-6) are greater than the elevations further back from the river (e.g., MW-9). This reversal is restricted to the wells near the river and probably reflects groundwater mounding due to river bank ice conditions. Photograph 32 Appendix A, shows the Kenai River covered by ice. Hart Crowser (1999c) also observed a groundwater reversal in December 1998, which they attributed to a rise in the river.

Significant differences in Figures 13 and 14 are observed in the northern (upper) part of the site. The horizontal hydraulic gradient north of the till ridge is approximately 0.01 feet per foot. The significant differences between Figure 13 and Figure 14 are indicated below:

- Figure 14, which was constructed using more data than Figure 13, indicates a groundwater divide running across the site east of MW-30 toward MW-29. This divide generally corresponds to the ridge in the till indicated on Figure 13. Immediately north of the ridge, groundwater flows in a generally northerly direction. However, evidence exists that this northerly flow direction may be only a local phenomenon (i.e., MW-32 has a higher water table elevation than MW-23, and MW-22 has a higher water table elevation than MW-25). Therefore, groundwater flow in this area is interpreted to be in a generally southerly direction, with localized northwesterly flow due to the till ridge. The general southerly flow direction interpretation is based on sparse data.
- An interpretation of the effects of the till ridge is incorporated into Figure 14. The shaded area on Figure 14 corresponds to an area of little or no appreciable groundwater on top of the till layer. In the vicinity of MW-30, only a very thin layer of water is interpreted to be present. Soil borings across the Sterling Highway from MW-30 indicate that the till elevation is similar to that encountered in MW-30. Based on the till elevation and expected groundwater elevation, the area of little or no

appreciable groundwater on top of the till layer is expected to continue across the Sterling Highway from MW-30.

- An interpretation of the effects of the storm sewer backfill on the water table is depicted on Figures 8, 9, 10, and 14. As shown in Figure 14, the storm sewer is interpreted to be draining water from most of the northern section of the site (everything north of the groundwater divide). The till ridge controls the unconfined aquifer groundwater flow in this area; the water from the vicinity of MW-21 is forced in a generally northward direction by the till ridge to the west of MW-21. The groundwater from the MW-21 vicinity is forced down the storm sewer backfill trench to the Kenai River, because the backfill trench is cut down into the till in this area. As discussed in Section 6.3.1.4, the effects of the storm sewer on groundwater flow are expected to diminish in the vicinity of MW-34. Conclusions regarding the effect of the storm sewer system on groundwater flow from the RTRVP site are provided in Section 7.3.1.5.

For purposes of discussion, the RTRVP site has been divided into two areas: the lower (more southerly) area, and the upper (more northerly) area. These two areas are separated by the groundwater divide. The lower area extends south from the former dry cleaner building to the Kenai River, and the upper area extends from the former dry cleaner building north to the Sterling Highway. Because the groundwater contamination present in each of these areas migrates in different directions, it is convenient to discuss them separately. The upper and lower plumes are discussed separately throughout this RI/FS report.

The monitoring wells can be grouped by their relative water level behavior, as reflected by the water elevations provided in Table 6-1. The following discussion focuses on monitoring wells MW-1 through MW-20 (the wells installed prior to July 1999), because the monitoring wells installed after July 1999 have little data available on which to base a trend discussion.

- <u>Unconfined aquifer monitoring wells primarily influenced by the river stage</u>
    - The monitoring wells completed in the uppermost aquifer near the river (MW-5, MW-6, MW-7, MW-8, MW-12, MW-13, and MW-20) behave similarly. Water levels in most of the wells near the Kenai River generally decreased by approximately 2 feet between July and October/November 1999. The December water levels were slightly higher than the July water levels, probably reflecting groundwater backing up due to river edge icing.
    - During the period from July through November 1999, water levels in monitoring wells MW-9 and MW-10 behaved similarly to those adjacent to the river, but the water level changes were dampened, presumably due to their increased distance from the river.
    - During the December 1999 monitoring, water levels in the monitoring wells adjacent to the river (MW-5, MW-6, MW-7, MW-8, MW-12, MW-13, and MW-20) were greater than water levels in MW-9 and MW-10. Although water levels in all of these monitoring wells increased between November and December 1999, the increase was most pronounced in the wells adjacent to the river, presumably because of the river edge icing effect.
- <u>Unconfined aquifer water levels not directly affected by river stage</u>
    - Water levels in MW-16 and MW-3A, screened in the unconfined aquifer further away from the river, behaved similarly. The water levels generally

  increased by approximately 0.5-feet between July and October 1999, and the December 1999 water levels were between the July 1999 and October 1999 levels. This water level behavior is opposite in direction from the water level behavior observed in the monitoring wells near the river.

  - Water levels in MW-4A and MW-19 behaved very differently from MW-9 and MW-10, which are located nearby (see Figure 14). The water levels in MW-4A and MW-19 increased by over a foot between July and October 1999, and then decreased again by December 1999, to levels below the July 1999 elevations.
  - The water levels in monitoring wells MW-11 and MW-14, interpreted as being screened in the upper aquifer, behaved similarly to the water levels in MW-4A and MW-19. They increased by over a foot between July and October/November 1999, and then decreased by approximately 0.5-feet between October and December 1999.
  - Water levels in monitoring wells MW-1A and MW-2 don't appear to follow the patterns noted in any other monitoring well groups.
- <u>Semi-confined water-bearing zone water levels</u>
  - Water levels in the wells screened in the semi-confined water-bearing zone (MW-01C (deep), MW-15, MW-17, and MW-18) generally increased by approximately 1-foot between July and October/November 1999, and then decreased back to approximately the July elevations by December 1999.
  - The relationship between the semi-confined water-bearing zone and the upper unconfined aquifer apparently varies over time. Water level data collected between July and November 1999 in the upper aquifer and the semi-confined water-bearing zone shows that upper aquifer water level elevations are relatively greater than the water level elevations in the semi-confined water-bearing zone. However, the December 1999 upper aquifer water level elevations are less than those of the semi-confined water-bearing zone. Due to lack of recharge during the winter months, the upper aquifer water level elevations apparently decline and become less than the pressure head of the semi-confined water-bearing zone.

In addition to the manual water level measurements, dataloggers installed in five site monitoring wells provide a record of site groundwater levels. The dataloggers record the water level twice per day, at 8 am and 8 pm. Data from the dataloggers in MW-16, MW-20, MW-24, and MW-19 from November 9, 1999 through December 13, 1999 are plotted in Figure 15. The datalogger in MW-15 was initially installed too deep in the monitoring well (but has been subsequently fixed); therefore, no data are currently available from that well. Review of Figure 15 indicates similar groundwater behavior in MW-16, MW-19, and MW-24 over this time period. The water level in MW-20 behaved similarly to the water levels in the other wells, until December 11, 1999. As was discussed briefly earlier in this section and as will be discussed in more detail in Section 6.3.5, this effect is believed to reflect river icing.

6.3.3 Slug Tests

Slug tests were performed at monitoring wells MW-4A, MW-9, MW-18, MW-19, and MW-20 on July 9, 1999, and a second round of slug tests were performed at monitoring wells MW-6, MW-12, MW-15, MW-16, MW-24, and MW-25 between October 28 and November 1, 1999. Although a slug test for MW-18 was attempted, the water level did not recover during the time that the data logger was in the monitoring well. Field collection of the slug test data included the following steps:

1) A data logger was lowered into the well at a measured depth.
2) A metal slug was then lowered into the well.
3) After the water level was allowed to stabilize, the slug was removed.
4) The data logger, with the recorded data, was removed from the well after water level has reached its starting elevation.

Analysis of the slug test data was performed using Super Slug ™ v 3.1 software by Starpoint Software (1998). The hydraulic conductivity was estimated by several different methods: the Bouwer and Rice graphical method, the Bouwer and Rice automatic parameter estimation method, and a sensitivity analysis method. In addition, selected slug test results were analyzed manually using EXCEL spreadsheets. The results of each of these analyses are presented, along with the parameter estimation assumptions, in Appendix G, Table 1. Appendix G also contains graphs of the slug test data for each well tested (Appendix G, Figures 1 through 11).

Table 6-3 presents a summary of the slug test results. The geometric mean of the various results was used as the "selected result" for each well. As indicated in Table 6-3, aquifer hydraulic conductivity and transmissivity were estimated for each of the major hydrogeologic units identified at the site: glacial till or silt, alluvial material, and the upper plume area (either fill or alluvial material). Table 6-3 also presents information regarding the confidence in the slug test result. Slug test interpretation involves estimating trend lines through time versus drawdown data. The interpretation of some of these trend lines was relatively straightforward; these are indicated by “good” confidence in the result. The interpretation of other trend lines was more problematic, as indicated by “average” or “below average” confidences.

A review of the information in Table 6-3 indicates that slug testing results were grouped together to estimate hydraulic conductivities for the various water-bearing zones present at the site.

- The minimum hydraulic conductivity (3.3E-05 [0.000033] centimeters per second [cm/sec]) was measured in the slug test from MW-15, which is interpreted to be completed in the semi-confined water-bearing zone within the till. The geometric mean of the MW-15 and MW-4A results (4.6 E-04 cm/sec) is interpreted as the best estimate of the till’s hydraulic conductivity.
- The maximum hydraulic conductivity (1.6E-01 cm/sec) was measured in the slug test from MW-25, which is located in the upper plume. The geometric mean of the MW-25 and MW-16 results (3.1E-02 cm/sec) is interpreted as the best estimate of the hydraulic conductivity in the upper plume area.
- Slug tests were performed in five monitoring wells interpreted to be completed in alluvial material near the Kenai River. The geometric mean of the results from these five wells (1.9E-02 cm/sec) is interpreted as the best estimate of the hydraulic conductivity in the alluvial material near the Kenai River.
- Very similar hydraulic conductivities were calculated for the alluvial material, fill, and upper plume lithology. Each of these areas has a hydraulic conductivity of approximately 2E-02 cm/sec, or 50 ft/day.

Table 6-3: Estimated Hydraulic Conductivities and Tranmissivities for River Terrace Site Soils, from Slug Testing

Hydraulic Conductivity - Bouwer and Rice Graphical Method

	Till		**Alluvial Material**					**Fill (1)**	**Upper Plume**	
	MW-15	**MW-4A**	**MW-19**	**MW-9**	**MW-6**	**MW-12**	**MW-20**	**MW-24**	**MW-16**	**MW-25**
Hydraulic Conductivity (cm/sec)	3.3E-05	6.3E-03	3.3E-02	5.3E-02	1.4E-02	2.0E-02	5.1E-03	1.0E-02	6.2E-03	1.6E-01
Confidence in Result	Below Avg	Average	Below Avg	Good	Good	Good	Average	Average	Below Avg	Good
Soil Classification	Till	Silt	Silty Sand	Sand	Sand	Sand and Gravel	Silty Sand	Sand	Gravel w/ fine sand	Cobbles
Notes:			Bottom 4 ft in Silt							
Geometric Mean for Unit	**4.6E-04**		**1.9E-02**					**1.0E-02**	**3.1E-02**	
(K in cm/sec)										
Hydraulic Conductivity in ft/day)	**1.3**		**54**					**29**	**88**	
Transmissivity (ft^2/day)	**Not Calculated (2)**		**307**					**90**	**464**	
Classification	**Till**		**Alluvial Material**					**Fill (1)**	**Upper Plume**	

NOTE: (1) This material may either be alluvial deposits on top of the till or fill material.
(2) Not calculated due to variable aquifer thickness

The slug test results presented herein differ from those previously reported (in the January 10, 2000 interim status report). The results presented in the interim status report represented only initial results generated by the Bouwer and Rice automatic parameter estimation method, without additional interpretation and verification by other analysis tools.

Uncertainty and result variability is expected in slug testing. Slug tests, by their nature, evaluate aquifer parameters only within a very short distance from the wellbore. Several of the factors that introduce variability into slug test results are listed below:

- variation in testing procedures,
- variability in the material compaction (disturbed versus native soil conditions),
- completeness of well development,
- differences in aquifer materials due to depositional environment, and
- equipment calibration error.

It is recommended that future slug testing should include at least three electronic water level measurements during each test, preferably during the beginning, middle, and end of the test. Electronic water levels taken during the test will provide calibration points to verify the depth range measured by the data logger or can be used to calibrate the data set. In addition, it is beneficial to run three slug tests at each well during a slug testing episode. Results from three tests can be compared to increase the confidence in the slug testing conclusions.

6.3.4 Calculated Aquifer Parameters

Based on the slug testing results, measured groundwater gradient, and observed water table depth, groundwater flow parameters were calculated for the site. These parameters are summarized in Table 6-4. In particular, low, average, and high groundwater flow velocities, contaminant transport velocities, and unit discharge were calculated for three sections of the site: the alluvium near and adjacent to the river, the “lower contaminant plume” area, and the “upper contaminant plume” area. The “lower contaminant plume” area is represented by data from monitoring wells located in the central area of the lower contaminant plume, whereas the Kenai River alluvium is represented by data from monitoring wells located along the river. Higher groundwater velocities and unit discharge were calculated for Kenai River alluvium, because the monitoring wells adjacent to the river measured higher hydraulic conductivities in slug testing, and a higher hydraulic gradient is measured close to the river.

The second set of columns in Table 6-4 represents contaminant transport velocity calculations. Contaminant transport velocity is retarded by contaminant adsorption on organic carbon in the soil. Two sets of numbers are provided for each primary contaminant (PCE, TCE, and DCE)—“R” and “v”. “R” refers to the retardation factor, which is a factor indicating how much slower than groundwater the contaminant travels, and “v” refers to the contaminant transport velocity.

Table 6-4: Calculated Aquifer Parameters in the Uppermost Aquifer at the River Terrace Site

RIVER TERRACE SITE:
Kenai River Alluvium

Groundwater Velocity and Unit Discharge Calculations ***Contaminant Transport Velocity Calculations***

Estimates	k	I	n	D	L	Hw	A	U	v	Q	Q	R (PCE)	v (PCE)	R (TCE)	v (TCE)	R (DCE)	v (DCE)
(units)	**(ft/day)**	**(ft/ft)**	**-**	**(ft)**	**(ft)**	**(ft)**	**(ft^2)**	**(ft/day)**	**(ft/day)**	**(ft^3/day)**	**(gpm)**	**(ft/day)**	**(ft/day)**	**(ft/day)**	**(ft/day)**	**(ft/day)**	**(ft/day)**
Low	14	0.01	0.3	10	140	6	840	0.14	0.47	118	0.61	6.98	0.07	3.07	0.15	2.41	0.19
Average	54	0.02	0.3	10	200	7	1,400	1.08	3.60	1512	7.85	5.04	0.71	2.43	1.48	1.55	2.32
High	150	0.03	0.3	10	220	8	1,760	4.50	15.0	7920	41.1	3.18	4.72	2.32	6.47	1.51	9.93

RIVER TERRACE SITE:
Lower Contaminant Plume

Estimates	k	I	n	D	L	Hw	A	U	v	Q	Q	R (PCE)	v (PCE)	R (TCE)	v (TCE)	R (DCE)	v (DCE)
(units)	**(ft/day)**	**(ft/ft)**	**-**	**(ft)**	**(ft)**	**(ft)**	**(ft^2)**	**(ft/day)**	**(ft/day)**	**(ft^3/day)**	**(gpm)**	**(ft/day)**	**(ft/day)**	**(ft/day)**	**(ft/day)**	**(ft/day)**	**(ft/day)**
Low	0.09	0.06	0.3	14	140	5	700	0.01	0.02	3.8	0.02	6.98	0.00	3.07	0.01	2.41	0.01
Average	1.3	0.1	0.3	17	200	7	1,400	0.13	0.43	182	0.94	5.04	0.09	2.43	0.18	1.55	0.28
High	28	0.14	0.3	20	220	8	1,760	3.92	13.1	6899	35.8	3.18	4.11	2.32	5.63	1.51	8.65

RIVER TERRACE SITE:
Upper Contaminant Plume

Estimates	k	I	n	D	L	Hw	A	U	v	Q	Q	R (PCE)	v (PCE)	R (TCE)	v (TCE)	R (DCE)	v (DCE)
(units)	**(ft/day)**	**(ft/ft)**	**-**	**(ft)**	**(ft)**	**(ft)**	**(ft^2)**	**(ft/day)**	**(ft/day)**	**(ft^3/day)**	**(gpm)**	**(ft/day)**	**(ft/day)**	**(ft/day)**	**(ft/day)**	**(ft/day)**	**(ft/day)**
Low	18	0.01	0.3	18	80	3	240	0.18	0.60	43	0.22	6.98	0.09	3.07	0.20	2.41	0.25
Average	88	0.01	0.3	20	115	4.5	518	0.88	2.93	455	2.36	5.04	0.58	2.43	1.21	1.55	1.89
High	454	0.02	0.3	22	150	6	900	9.08	30.3	8172	42.4	3.18	9.52	2.32	13.05	1.51	20.04

SYMBOLS

k=hydraulic conductivity
i=gradient
n=porosity
L=length of plume
Hw=height of water table
D=depth from ground surface to till
A=cross sectional area
R=groundwater transport retardation

CALCULATIONS

u=Darcy velocity
v=groundwater velocity=ki/n
Q=flow=kiA

Fc=flux (mass/day) of contaminants in 1-ft^3 of water
Mc=mass of contaminants water flowing through the site

UNITS

ft^2=square feet
gpm = gallons per minute

The information provided in Table 6-4 can be used to estimate the flux of contaminated water leaving the site. The average numbers were used to draw the following conclusions regarding the contaminated groundwater flux.

- Between MW-5 and MW-12 (approximately 120 lineal feet), approximately 4.5 gallons per minute gpm of water contaminated by PCE and its degradation products is migrating off-property into the Kenai River. This estimate is based on hydraulic gradients measured in July and November 1999, and does not consider riverbank icing effects observed in December 1999.
- Between MW-21 and MW-22 in the upper plume area (approximately 115 lineal feet) approximately 2.4 gpm of water contaminated by PCE is migrating off-property. This estimate is based on hydraulic gradients measured in July through December 1999 and is not expected to have a significant seasonal variability.

The data provided in Table 6-4 can also be used to estimate the travel time of groundwater and contaminants at the site. Travel time between various monitoring wells and the Kenai River (for the lower plume), or the storm sewer (for the upper plume) is provided in the table below. The travel times indicated in Table 6-5 are based on the average groundwater and PCE velocity values given in Table 6-4. Low and high travel times could be calculated by dividing the relevant distance (provided in Table 6-5) by the appropriate low and high groundwater and PCE velocities (provided in Table 6-4).

Table 6-5: Average Groundwater and PCE Travel Times, River Terrace Site

Lower Plume	
Monitoring Well (approximate distance to the OHW line of the Kenai River in feet)	Groundwater/PCE Travel Time from Specified Monitoring Well to Kenai River (days)
MW-24 (220 feet)[1]	510 days/2,450 days
MW-26 (140 feet)[1]	325 days/1,500 days
MW-4A (100 feet)[2]	28 days/140 days
MW-9 (60 feet)[2]	17 days/85 days
Upper Plume	
Monitoring Well (approximate distance to storm sewer in feet)	Groundwater/PCE Travel Time from Specified Monitoring Well to Storm sewer (days)
MW-16 (115 feet)[3]	40 days/200 days
MW-25 (65 feet)[3]	22 days/110 days

Notes:

[1] Travel time based on average “lower contaminant plume” velocity values (0.43 ft/day for groundwater, 0.09 ft/day for PCE [due to adsorption to organic carbon, as discussed above])

[2] Travel time based on average “Kenai River Alluvium” velocity values (3.6 ft/day for groundwater, 0.71 ft/day for PCE)

[3] Travel time based on average “upper contaminant plume” velocity values (2.93 ft/day for groundwater, 0.58 ft/day for PCE)

6.3.5 Surface Water Influence/Effects on Groundwater

Groundwater and surface water gauging conducted by E&E in 1997 and by OASIS/Bristol in 1999 support the conclusion that the Kenai River is hydraulically connected to site groundwater. At low to normal river stage, the Kenai River is expected to be a gaining stream, as groundwater flows into the Kenai River. As a result of the hydraulic gradient toward the river, pore water in the coarse-grained soils of the riverbank will move toward the Kenai River. At high or flood stage levels, groundwater flow can temporarily reverse direction, and the Kenai River can become a losing stream. Due to river icing conditions, groundwater flow can also temporarily reverse direction, with groundwater “backing up” adjacent to relatively impermeable river bank ice. Gradient reversal in the aquifer adjacent to the Kenai River edge, however, will probably not result in appreciable flow reversal more than 25 to 50 feet laterally from the OHW line. The limited extent of expected flow reversal may be due to any of the following conditions:

- the hydraulic conductivity of the soils is substantially lower on the terraced portion of the site compared to the bank deposits adjacent to the river water line,
- flood events are typically short-term events, and
- hydraulic gradients toward the river are relatively steep compared to river level fluctuations.

Table 6-6 presents a summary of Kenai River mean monthly discharges and stages at Soldotna (USGS gaging station 15266300) over 30 years. This gaging station is located near the center of the span on the downstream side of the Sterling Highway bridge adjacent to the RTRVP site. The mean monthly stage varies from a low in March of 5.21 feet to a high in August of 9.90 feet. The stage datum is an elevation of 35.34 feet.

Table 6-6: Mean Monthly Discharges and Stages at USGS Kenai River at Soldotna Stream-Gaging Station (1965-1995)

OCT	NOV	DEC	JAN	FEB	MAR	APR	MAY	JUN	JUL	AUG	SEP
Discharge (cubic feet per second),											
7,158	3,480	2,283	1,889	1,669	1,367	1,557	3,171	8,428	13,310	14,660	12,010
Stage (feet)											
7.94	6.57	5.98	5.75	5.60	5.21	5.48	6.43	8.33	9.60	9.90	9.29

Source: Bailey, B.J. and E.V. Hogan, 1995. Overview of Environmental and Hydrogeologic Conditions near Kenai, Alaska. USGS Open-File Report 95-410.

A 4-month record of the river stage at the Soldotna Stream-Gaging Station on the Kenai River was obtained from the USGS (1999). The mean daily river stage from September 1, 1999 to December 13, 1999 is plotted along with the groundwater level data from MW-20 in Figure 16. This graph strongly indicates that the groundwater level in MW-20 is tied to the surface water level.

6.4 ANALYTICAL DATA

The RI/FS sampling plan was designed to better understand the distribution of the contaminant plume and to collect basic information necessary to evaluate remedial alternatives such as intrinsic remediation, *in situ* biological treatment, zero valence iron, and other treatment methods. Locations and types of samples collected are shown in Table 5-1 and Table 5-2.

The known contaminants at the site are VOCs (PCE, its associated breakdown compounds, and aromatic petroleum hydrocarbons), DRO, and RRO. PCE and its breakdown products are the primary contaminants of concern. For the RI activities, primary laboratory analysis was performed by Columbia Analytical Services and

MultiChem. Quality assurance samples collected during the June and July 1999 soil and groundwater sample collection were submitted to Quanterra for analysis. Only low levels of petroleum hydrocarbons have been detected in site groundwater, and the soil with higher levels of petroleum hydrocarbons was excavated in 1997; therefore, petroleum hydrocarbons have not been retained as COCs.

Tabulated analytical results are presented in this section of the RI report. To aid readability of this report, these long tables are presented in a group at the end of this section instead of interspersed within the text, as the other tables are presented in this report.

6.4.1 Soil Analytical Program

All soil samples selected for laboratory analysis were analyzed for VOCs (using USEPA method 5035/8260B). Soil samples from areas where petroleum contamination is possible were also analyzed for DRO (AK 102), and RRO (AK103). Only soil samples collected during the June 1999 sampling event were analyzed for DRO and RRO. Monitoring wells MW-15, MW-16, MW-17, MW-18, MW-19, MW-20 and soil borings SB-1 and SB-2 were installed in June 1999. Subsequent soil samples were not analyzed for DRO or RRO, because DRO and RRO levels were consistently below the site ACLs stated in the August 27, 1997 letter and because VOCs have been shown to be the primary constituents of concern at the site. The number of samples was based on field conditions and is shown in Table 5-1. Soil sample results are shown in Table 6-8 and Plate 5. Appendix C presents a summary of historical soil analytical data. PCE is known to migrate along stratigraphic layers or contacts, particularly where a more permeable layer overlies a less permeable layer. Sampling targeted these layers whenever possible. However, many small-scale heterogeneities are present in the site subsurface that are impossible to predict or target.

6.4.2 Groundwater Analytical Program

6.4.2.1 Organic Contaminant Analytical Program

Groundwater samples were collected from various monitoring well locations from the time of the initial RI sampling in July 1999 through the November 1999 sampling event. Monitoring wells MW-1A, MW-1C, MW-2, SB-1 Shallow and SB-1 Deep were removed from the sampling list following the July 1999 sampling, because no COCs have been detected in groundwater samples from these monitoring wells (see Appendix D). MW-5 was not sampled in November 1999, because it was dry. MW-5 has a history of going dry.

Field techniques were used to evaluate DO, ferrous iron, ORP, pH, temperature, and conductivity from all the wells monitored between July 1999 and January 2000. The groundwater sample data sheets are provided in Appendix F, and the field parameter results are presented in Table 6-9.

Groundwater samples were analyzed for VOCs (USEPA Method 8260B) and polynuclear aromatic hydrocarbons (PAHs) (USEPA Method 610) during the July 1999 sampling event. Only PCE, its degradation products, and benzene have historically been detected in site groundwater at levels near or above their drinking water MCLs. The analytical results for these analytes are provided in Table 6-10 and Plate 5. During subsequent sampling events, groundwater samples were not analyzed for PAHs based on historical analytical data that indicates these compounds are not present in the groundwater at concentrations near or above ADEC WQS specified in 18 AAC 75 or 18 AAC 70 (i.e., 15 μg/L total aqueous hydrocarbon standard for groundwater connected to surface water).

Selected groundwater samples were analyzed for natural attenuation (intrinsic remediation) parameters, as defined by the Air Force Center for Environmental Excellence Technical Protocol for Evaluating Natural Attenuation of Chlorinated Solvents in Groundwater (Wiedemeier, et al., 1996). These samples were collected from monitoring wells that are located upgradient of or near the known PCE plume and were analyzed for chloride, sulfate, nitrate, methane, total organic carbon, iron (dissolved) and alkalinity. The monitoring wells sampled for natural attenuation parameters, which varied by sampling event, are provided along with the analytical results for these samples are provided in Table 6-11 and Plate 7. Photographs 34 and 35 in Appendix A show groundwater purging and sampling activities at the RTRVP site.

6.4.2.2 Cation/Anion Balance Analysis

Samples were also collected from the artesian well located adjacent to the former dry cleaner building and selected monitoring wells in order to perform an anion/cation balance. Most of the anion/cation analyses were performed on samples collected during the October/November 1999 sampling event. In December 1999, three additional samples were collected: a second sample from MW-24, a sample from MW-11, and a sample from the Credit Union One building (located across the Sterling Highway from the RTRVP). The water sample from the Credit Union One building was collected by an OASIS/Bristol sampler with input from the city of Soldotna. The water samples were submitted for laboratory analysis by EPA methods 6010, 300.0 and 310.1.

The purpose of the cation/anion balance analysis is to differentiate between different water-bearing zones on the site. Waters with similar anion/cation profiles likely originate from the same source. Consequently, the data can be used to infer that a group of monitoring wells is from a connected hydrogeologic unit. Conversely, if the anion/cation profiles are very different, it is possible that the waters either originate from different hydrogeologic units or water from one of the wells has been influenced by some solute that has not affected the other water. One specific question that the cation/anion balance was used to answer was whether the groundwater mound observed in MW-24 was possibly caused by a leaking artesian well near the former dry cleaner building (Figure 14), or whether it reflected "natural" groundwater conditions. The analytical results for these samples are provided in Table 6-12.

The anion/cation analyses were performed on samples from the following wells:

- MW-4A
- MW-9
- MW-11
- MW-14
- MW-15
- MW-16
- MW-20
- MW-22
- MW-24 (two samples)
- MW-26
- MW-28
- MW-31
- SB-1(S)
- SB-1(D)
- Artesian Well (located adjacent to the former dry cleaning building)
- City water supply (collected from the Credit Union One building across from the site)

The anion/cation data were evaluated using multivarient plots and a Collin's bar graph (Appendix H). General conclusions from the anion/cation balance analysis are listed below:

- The water measured in MW-24 is interpreted to be consistent with other groundwater measured at the site; therefore, the groundwater mound is interpreted to be a natural phenomenon. Two samples were collected from MW-24 because the first analysis indicated that the water was radically different from all other water at the site. The second sample indicated that water in MW-24 has substantially more bicarbonate than other waters; however, the ratios are similar to other water table waters on the site. Inspection of the multivarient plot (Appendix H) reveals that MW-24 water has the same general characteristic shape as waters from MW-11, MW-4a, and other monitoring wells.

- The sample from MW-24 did not resemble the samples from either the site artesian well or the city water supply, suggesting that the water in this monitoring well is not the result of a water line leak. The city water supply and the site artesian well were similar; with a strong bicarbonate spike and relatively similar levels of the cations (the city water supply also had a chlorine spike, as would be expected from a city water supply).
- MW-11 more closely resembles the unconfined aquifer samples (particularly MW-24 and MW-4A) than the semi-confined water bearing zone samples from MW-15 and SB-1 (D), suggesting that the water in this monitoring well comes from the unconfined aquifer.
- SB-1(D), SB-1(S) and MW-15 were similar, with cations dominated by the sum of sodium and potassium, and anions dominated by bicarbonate. The different character of these samples is added evidence that these wells tap a different water bearing zone than the wells completed in the water table aquifer.
- In general, the anions of most samples were dominated by bicarbonate. The only exceptions to this trend were samples from MW-14, MW-16, and MW-31, which also indicate a relatively high sum of chloride and nitrate ions.
- Water in the vicinity of MW-16 is similar to water in the right-of-way, e.g. MW-31.
- Most of the water table aquifer wells sampled are similar to each other. The Collins graph is the best depiction of similarity of water between wells.
- The anion/cation data support the conceptual model that at least three water bearing zones are present at the site: the water table aquifer, the confined artesian aquifer, and at least one semi-confined water bearing zone beneath the water table aquifer.

6.4.2.3 Drinking Water Sampling Program

A drinking water sampling program was performed to determine whether the contamination detected in RTRVP site groundwater had impacted any off site drinking water wells. There are two known drinking water wells located on the RTRVP site; the ADEC performed a well search to determine whether any additional private drinking water wells were potentially downgradient (in a northwesterly direction) within ¼-mile of the site. Four wells were found. On 18 August 1999, attempts were made to sample all wells identified by the well search. Well locations and associated sample identification numbers are summarized in Table 6-7.

Table 6-7: Drinking Water Sample Summary

Location of Drinking Water Well	Sample Number
Perfect Nails at 44712 Sterling Highway	KRT81899-NAIL
Private Residence at 355 Riverside Drive	KRT81899-SAB & KRT81899-DSAB
Private Residence at 451 Kobuk	None
Tail Waggers at 44720 Sterling Highway	KRT81899-GROOM

The well at 451 Kobuk had not been used for several years and was completely rusted shut. No attempt was made to sample the well because it was feared that any attempt to remove the rusty bolts could result in metal failure and uncontrolled release of the artesian water. The remaining three wells were all sampled and subsequently analyzed by EPA method 524.2. One duplicate sample was also collected. Columbia Analytical Services, Inc., of Anchorage, performed the analyses.

Neither PCE nor any of its breakdown products were detected in any of the samples. There is no indication that contamination from the RTRVP site has impacted any of the drinking water wells sampled. The duplicate sample from the private residence at 355 Riverside Drive contained the trihalomethanes chloroform at 12 µg/L and bromodichloromethane at 1 µg/L. No contaminants were detected in the project sample. The MCL for both of these trihalomethanes in 100 µg/L, which is one to two orders of magnitude above the levels detected. The source of the contamination in the duplicate is unknown. However, trihalomethanes can occur naturally and are commonly produced during the chlorination of water.

6.4.3 Soil Gas Analytical Program

A soil gas sampling program was conducted to delineate the source of the groundwater contamination detected in MW-16. Soil gas samples were collected from 19 temporary soil gas monitoring points installed in November 1999 to the north and northwest of the former dry-cleaner building, between MW-16 and MW-23 (see photographs 24 and 25 in Appendix A). Soil gas points were driven to approximately 15 feet bgs (groundwater was encountered at approximately 18 feet bgs in MW-16 and MW-23). Soil gas samples were analyzed for VOCs using EPA method TO-14, and the analytical results are shown on Figure 17. Soil samples collected from selected soil gas borings were collected and analyzed for VOCs (USEPA method 8260B) for waste characterization. The analytical results for soil samples are provided in Table 6-8.

As indicated on Figure 17, PCE was detected in every soil gas sample at concentrations between 860 parts per billion (ppb) and 13,000 ppb. The highest soil gas reading of 13,000 ppb was observed at SG-18, near the doorway into the building. PCE concentrations in the four soil gas points surrounding MW-16 were between 2,400 and 3,800 ppb. Only relatively minor concentrations of cis-1,2-DCE were detected in the soil gas samples (cis-1,2-DCE was not detected in most samples, and the maximum detected concentration was 140 ppb).

The soil gas survey encountered relatively high PCE vapor concentrations spread relatively uniformly over the area of investigation. The soil gas survey did not conclusively indicate a source area, but rather indicated that the entire soil gas survey area is contaminated. The source of the soil gas vapors is assumed to be contaminated groundwater.

6.4.4 Data Validation

Analytical results from soil and groundwater samples collected during the RI/FS have been reviewed and validated to determine their suitability for use. An explanation of DQO established for the RI/FS, an explanation of the review and validation process, and a detailed description of the results are provided in the Quality Assurance Report presented in Appendix I. Data review and validation procedures were completed in accordance with applicable EPA procedural guidance. In general, the analytical data is usable for its intended RI/FS purpose. A small percentage of the laboratory analytical results were qualified as estimated due to holding time exceedences and discrepancies associated with specific internal and external calibration compounds. This data has been flagged in the data tables to indicate the potential bias associated with these results.

Table 6-8: 1999 Soil Analytical Data Summary

Table 6-9: Field Measured Parameters, July-November 1999

Table 6-10: 1999 Groundwater Analytical Data Summary

Table 6-11: Groundwater Intrinsic Remediation Parameters

Table 6-12: Groundwater Cation/Anion Analytical Data

7 FATE AND TRANSPORT ANALYSIS

7.1 EXTENT OF SURFACE WATER AND SEDIMENT CONTAMINATION

7.1.1 Storm Water Contamination

As discussed in Sections 3.4.2 and 3.4.5, E&E collected samples from the storm sewer system adjacent to the RTRVP site in 1997 and 1999 (E&E, 1997b and 1999a). Several samples have been collected from the KRBO, and the 10 MHs along the stormwater system, located along an approximately 3,500-foot section of the southbound lane of the Sterling Highway, were sampled in June 1997. Analytical results are summarized below in Table 7-1.

Table 7-1: Historical Storm Water Sample Results, RTRVP, Soldotna, Alaska

Location	*Sample Date*	*Sampled By*	*cis-1,2-DCE (μg/L)*	*TCE (μg/L)*	*PCE (μg/L)*
MH-1	June 1997	E&E	<3	<3	6.6
MH-1	June 1997	E&E	<3	<3	6.9
MH-1	October 1998	E&E	<1	<1	8.5
MH-2	June 1997	E&E	<3	<3	<3
MH-2	October 1998	E&E	<1	<1	<1
MH-3	June 1997	E&E	<3	<3	<3
MH-4	June 1997	E&E	<3	<3	<3
MH-5	June 1997	E&E	<3	<3	<3
MH-6	June 1997	E&E	<3	<3	<3
MH-7	June 1997	E&E	<3	<3	<3
MH-10	June 1997	E&E	<3	<3	<3
KSO (Kobuk Street Outfall)	June 1997	E&E	<3	<3	<3
KRBO (also called SHBO)	May 1997	E&E	<0.5	0.6	23
KRBO	June 1997	E&E	<3	<3	8.2
KRBO	October 1998	E&E	<1	<1	8.3
KRBO	May 1999	E&E	0.18	0.33	17
KRBO	May 1999	E&E	0.18	0.27	16
KRBO	November 1999	Hart Crowser	<1	<1	6.69

As indicated in Table 7-1, PCE has been detected in samples from (MH-1), located adjacent to the RTRVP site and the KRBO (see Figure 14). PCE was not detected in samples collected from the other storm sewer MHs. This distribution of PCE detections depicts that PCE is entering the storm sewer system between MH-2 and MH-1, which is adjacent to the RTRVP site.

Three of the four KRBO sample results exceeded the water quality standard of 5 μg/L for PCE.

7.1.2 Surface Water Contamination

As discussed in Sections 3.4.1 and 3.4.5, two surface water sampling events were performed by E&E (1997a and 1999b). Surface water analytical results are summarized below in Table 7-2, and surface water sampling locations are shown in Figure 23.

Table 7-2: Historical Surface Water (Kenai River) Analytical Result Summary, RTRVP, Soldotna, Alaska

Location	*Sample ID*	*Date Sampled*	VC (*μg/L)*	*cis-1,2-DCE (μg/L)*	*trans-1,2-DCE (μg/L)*	*TCE (μg/L)*	*PCE (μg/L)*
SW-2 (downslope of MW-8)	99RTRVP003SW	5/19/99	<0.02	15	0.081	1.9	2.5
SW-3 (downslope of MW-6)	99RTRVP004SW	5/19/99	<0.02	2.8	<0.02	0.26	0.81
Background (near upriver site boundary)	99RTRVP005SW	5/19/99	<0.02	<0.02	<0.02	<0.02	<0.02
Background (near center of river bend)	202WA	May-97				<0.5	<0.5

As indicated in Table 7-2, PCE has been detected in samples from the Kenai River adjacent to the RTRVP site and downslope of the contaminated groundwater plume. Although the levels of PCE detected in the Kenai River water samples did not exceed the water quality standard of 5 μg/L for PCE, these samples represent the result of mixing (dilution) with Kenai River water.

The significance of the surface water contamination with respect to ecological risk is discussed in Section 8.

7.1.3 Sediment Contamination

As discussed in Sections 3.4.1 and 3.4.5, two surface water and sediment sampling events were performed by E&E (1997a and 1999b). Sediment sample locations are depicted on Figure 23, and analytical results are summarized below in Table 7-3.

As indicated in Table 7-3, PCE and its degradation products (TCE, cis-1,2-DCE, trans-1,2-DCE, and 1,1-DCE) have been detected in Kenai River sediments adjacent to the RTRVP site. The significance of the sediment contamination with respect to ecological risk is discussed in Section 8.

7.2 EXTENT OF SOIL CONTAMINATION

Figure 2 presents historical soil contamination information. The information on Figure 2 is based on 1997 grid sampling and excavation perimeter sampling. Plates 2 through 4 are 1997 cross-sections indicating historical soil contamination information in cross-section view. These plates were compiled primarily from the KEC soil grid samples (KEC, 1997). Excavation perimeter soil samples (RP's interim AOC report, 1998) and soil information from E&E (1998b) are also incorporated into these figures. These figures represent the interpreted soil contamination at the RTRVP site prior to the 1997 and 1998 excavations performed as part of the AOC between EPA and the RTRVP property owners.

Table 7-3: Historical Sediment Sample Results, RTRVP, Soldotna, Alaska

Plate 5 presents the known soil contamination remaining on site after the 1997 and 1998 excavations. Soil sample results from the monitoring wells, soil borings, and test pits at the site, are presented on Plate 5 along with the three excavation perimeter soil samples that indicated PCE remaining above the ACL. Soil sample PCE results above the site ACL are highlighted in red, and soil sample results above the ADEC Method 2 migration to groundwater soil cleanup level for PCE are highlighted in blue. The highlighting is limited to PCE results, because only PCE has been detected at levels above the soil ACL in site soil samples. The ADEC Method 2 levels are indicated on this plate for comparison purposes. Although they do not apply to the RTRVP property (because the ADEC established ACLs for the site), they represent the level of impact generally considered to be of concern at other contaminated sites. The Method 2 soil cleanup level (for migration to groundwater in the “under 40 inch” annual precipitation zone) for PCE is 0.02 mg/kg (the Method 2 cleanup levels for all PCE degradation products are presented in Section 3.2). Note that excavation perimeter samples that detected PCE above the ADEC Method 2 levels but below the site ACLs are NOT depicted on Plate 5. This information is available on Figure 2. A soil sampling spoon driven at the RTRVP site is shown in photograph 15 in Appendix A.

7.2.1 Likely PCE Release Areas

As discussed in Section 3-1, a maximum PCE soil concentration of 4,700 mg/kg was detected in a surface soil sample from the site (node B7 on Figure 2 and Plate 2). This concentration indicates a probable PCE spill location. Based on solvent retention values reported by Poulsen and Keuper (1992) (discussed in 4.1.1.2), the likely range of PCE soil concentrations resulting from a surface release of pure PCE product is estimated to be between 3,500 mg/kg and 10,000 mg/kg. The calculation is shown below.

$$\left(\frac{12.6LPCE}{m^3soil}\right)\left(\frac{1m^3}{35.3ft^3}\right)\left(\frac{ft^3}{130lb}\right)\left(\frac{2.2lb}{kg}\right)\left(\frac{1.623X10^6mgPCE}{L}\right)\approx 10{,}000mg/kg$$

$$\left(\frac{4.9LPCE}{m^3soil}\right)\left(\frac{1m^3}{35.3ft^3}\right)\left(\frac{ft^3}{140lb}\right)\left(\frac{2.2lb}{1kg}\right)\left(\frac{1.623X10^6mgPCE}{L}\right)\approx 3{,}500mg/kg$$

This calculation is valid for a recent spill that has not undergone significant leaching or degradation. Lower PCE concentrations in near-surface soils may also indicate release points that have undergone more degradation. PCE concentrations of 177 mg/kg (5 feet bgs) at the A1 node, 114 mg/kg (surface) at the B3 node, and approximately 20 mg/kg in surface samples at the A8 and AA8 nodes indicate other likely release areas (see Figure 2 and Plates 2 and 4).

The soil surrounding the maximum PCE detection was excavated in 1997 (RP’s interim AOC report, 1998) (see photographs 5 through 13 in Appendix A). Additional surface soil was excavated in June 1998 (Hart Crowser, 1998a). The following subsections discuss soil contamination remaining in-place, as detected during pre-RI activities and as detected by RI soil sampling. As discussed above, Plate 5 presents a compilation of the known soil contamination remaining on site, as of the date of this report.

7.2.2 1999 RI Soil Results

PCE and its degradation products were not detected at concentrations above the site ACLs in any of the soil samples collected during 1999 and January 2000 RI activities (see Table 6-8). The maximum PCE concentration detected in the 1999-2000 RI activities was 2,000 µg/kg, which was detected in a soil sample from 8 to 10 feet bgs in MW-19. In the upper (northern) plume area, a maximum PCE concentration of 190 µg/kg was detected in a soil sample from MW-25. The maximum degradation product detections were at least two orders of magnitude lower than the ACLs.

7.2.3 Soil Contamination Remaining In-place

The June 1999 through January 2000 RI soil sampling results can be interpreted with previous soil sampling results to estimate the remaining extent of soil contamination. The following discussion is based only on available analytical results. It is possible that contamination exists, in addition to what is discussed below, at the site. In a letter dated February 10, 1998 to the EPA, the ADEC discusses that more contaminated areas may exist than specified in the RP's interim AOC report (1998), because the report failed to discuss how the field screening was performed and QA/QC procedures. Based on the expected pattern of free-phase PCE transport in the vadose zone (as discussed in Section 4.1.1), it is likely that not all of the PCE-contaminated soil was discovered by the grid sampling and subsequent excavation.

7.2.3.1 Upper Contaminant Plume

The 1997 KEC grid sampling and other investigations prior to the 1999 RI did not extend to the upper contaminant plume area. Therefore, only the 1999 RI results are available to assess soil contamination in this area. As indicated in Table 6-8, no 1999 soil results exceeded site ACLs, and the maximum PCE concentration detected in upper contaminant plume soils was 190 µg/kg. However, significant PCE contamination is indicated near the former dry cleaner building based on the 1999 soil gas survey results and groundwater contamination detected in samples from MW-16 and MW-25 (see Figure 17 and Table 6-10).

7.2.3.2 Lower Contaminant Plume

For ease of discussion purposes, the lower contaminant plume is divided into two sections (a northern section and a southern section) for this soil contamination discussion.

<u>Northern Section of the Lower Contaminant Plume (North of Sewer Lift Station and West of the Building):</u>

Results of soil sampling in this area have not identified any remaining soil with PCE levels above the ACL. However, soil samples from the perimeter of the 1997 excavation and from subsequent investigations indicate soil impact remaining above the ADEC Method 2 (migration to groundwater, in the under 40-inch precipitation zone) soil cleanup level (0.02 mg/kg).

Review of analytical results for soil samples collected from the 1997 excavation indicated PCE contamination remains in-place in the northern section of the site at levels below the site ACL of 11.5 mg/kg (RP's interim AOC report, 1998) (see Figure 2). Excavation floor samples in this part of the site indicated PCE concentrations ranging from below detection levels to approximately 4 mg/kg.

Soil borings advanced by E&E (1999a) and Hart Crowser (1999a) did not encounter soil contamination in this northern section. Plate 5 shows the approximate locations of these soil borings (the boring locations were not surveyed), and the soil sample results are provided in Appendix C. Three soil borings were advanced by E&E in the ADOT ROW approximately 40 to 80 feet northeast of MW-14. A maximum depth of 30 feet bgs was reached by two of these borings (E&E-SB-4 and 6). Thirteen soil samples were submitted for laboratory analysis from the three soil borings and the MW-14 boring.

PCE was detected in only one of the samples (in the boring for MW-14 at a concentration of 0.16 mg/kg, which is well below the site ACL), and no PCE degradation products were detected in any of the soil samples (E&E, 1999a). PCE was not detected in soil samples collected from approximately 6 feet bgs in Hart Crowser's soil borings SB-6 and MW-11 (Hart Crowser, 1999a, and Appendix C).

Southern Section of the Lower Contaminant Plume (South of Sewer Lift Station):
Results of soil sampling in this area have indicated PCE contamination remains above the ACL at the base of the excavation near node B7 (northeast of MW-9) and in soil boring SB-3A, which is located in the same general vicinity. Soil contamination below the ACL but above the ADEC Method 2 (migration to groundwater, in the under 40-inch precipitation zone) soil cleanup level (0.02 mg/kg) is widespread in this section of the site. Some contamination was detected in all samples collected within 20 feet of the B7 node.

Soil samples collected from the 1997 excavation indicated PCE contamination remaining in-place in the southern section of the site. As indicated on Plate 5, PCE was detected in three excavation perimeter soil samples (near node B7, north of MW-9) at levels above the ACL (between 12.2 mg/kg and 18.9 mg/kg). Other excavation base samples in the southern section of the site indicated PCE concentrations ranging from below detection levels to approximately 9.5 mg/kg (RP's interim AOC report, 1998) (Figure 2).

Soil borings advanced by E&E (1999a) and Hart Crowser (1998b) also encountered soil contamination in the southern section of the site in 1998 (Plate 5). E&E advanced three soil borings (MW-12, MW-13, and E&E-MW-15) in the ADOT ROW near the Kenai River. A maximum depth of approximately 12 feet bgs was reached by one of these borings (MW-12). PCE was not detected in any of ten soil samples submitted for laboratory analysis (see Appendix C). Cis-1,2-DCE was detected at concentrations of 0.38 mg/kg and 0.33 mg/kg in samples from 4 to 6 feet bgs and 6 to 8 feet bgs, respectively, from MW-12. These concentrations are well below the site ACL but above the Method 2 cleanup levels. Hart Crowser's soil borings SB-2A/SB-2B reportedly encountered nearly 30 feet of sandy soil, interpreted as fill, underlain by silt. PCE was detected in soil samples collected from 30- and 35-feet bgs (from the silt) at concentrations between 0.1 mg/kg and 0.72 mg/kg, which are below the site ACL but above the Method 2 cleanup levels (Appendix C). Hart Crowser's soil borings SB-3A/SB-3B reportedly encountered nearly 20 feet of sandy fill underlain by silt. PCE was detected at levels up to 20 mg/kg (exceeding the site ACL of 11.5 mg/kg) in samples from 25- to 30-feet bgs in boring SB-3A. Hart Crowser's soil boring for MW-4A encountered approximately 7 feet of fill underlain by silt. PCE was detected in a soil sample from MW-4A at approximately 10 feet bgs at a concentration of 0.025 mg/kg, which is below the site ACL but above the Method 2 cleanup level.

7.3 EXTENT OF GROUNDWATER CONTAMINATION

Sampling of site groundwater monitoring wells in 1999 indicated a widespread area of PCE contamination in the water table aquifer. Groundwater contamination at the RTRVP site is discussed in terms of two plumes: a lower (southern) plume and an upper (northern) plume. Both plumes are located in the upper, or water table aquifer, and they are separated by the groundwater divide running across the site from approximately MW-29 to MW-24 to MW-30 (Figure 14). Monitoring wells north of this divide (e.g., MW-16, MW-21, MW-22, and MW-25) are interpreted to belong to the upper plume, and monitoring wells south of this divide (from MW-4A to MW-26 to MW-3A, extending south to the Kenai River) are interpreted to measure the lower plume. MW-11 and MW-14 are also interpreted to belong to the lower plume, although their characteristics are somewhat different from other monitoring wells in this plume (see discussion in Sections 7.3.1, 7.5, and 7.6.3). It is uncertain which plume is tapped by monitoring wells near the crest of the divide (MW-29, MW-24, and MW-30). Groundwater purging and sampling activities are presented in photograph 34 and 35 of Appendix A.

Table 6-10 presents the 1999 groundwater analytical results for PCE, its degradation products (TCE, DCE, and VC), and benzene, and Plate 6 illustrates the July 1999 through November 1999 groundwater analytical results. On Plate 6, ACL exceedences are highlighted in red, and MCL exceedences are highlighted in blue. The historical ACL exceedences are summarized in Table 7-4. All historical groundwater sample results are provided in Appendix D.

In the upper plume area, samples from MW-16 and MW-25 have exceeded the groundwater ACL for PCE. All samples collected from MW-16 have exceeded the ACL, whereas only the September 1999 sample from MW-25 exceeded the ACL.

In the lower plume area, the ACL for PCE and VC have been exceeded in samples from several monitoring wells. Samples from MW-4A have historically always exceeded the groundwater ACL for PCE. In MW-6, MW-8, and MW-10 (sentry wells), the groundwater ACL for PCE has been exceeded sporadically. In MW-9, the ACL for PCE has been exceeded in three of the last four sampling events. The ACL for VC has been exceeded in many samples from MW-6, and some samples from MW-9 and MW-20.

The MCL for benzene was slightly exceeded in several samples from MW-4A, MW-6, and MW-9; however, no benzene exceedences have been measured between December 28, 1998 and the most recent sampling event incorporated into this report (December 13, 1999).

Table 7-4: Groundwater ACL Exceedences Summary (also including benzene MCL exceedences) from July 1997 through December 1999, RTRVP, Soldotna, Alaska

Location	Date Sampled	VC (μg/L)	PCE (μg/L)	Benzene (μg/L)
MW-4	7/22/97		1500	
MW-4A	8/1/98		3540	7.56
MW-4A	10/20/98		2500	
MW-4A	12/29/98		3000	7
MW-4A	4/15/99		2400	
MW-4A	5/10/99		2300	
MW-4A	7/9/99		1900	
MW-4A	10/28/99		1600 J,H	
MW-4A	12/14/99		1200	
MW-6	7/22/97		1900	
MW-6	10/28/97	4.5		5.8
MW-6	12/31/97	3.07		5.61
MW-6	4/23/98	2.13		
MW-6	4/15/99	2.85		
MW-6	7/8/99	3 J,H		
MW-6	10/26/99	3.7	980	
MW-8	10/21/98		960 J,H	
MW-9	12/28/98	6		6
MW-9	4/15/99	2.49	910	
MW-9	7/9/99	3		
MW-9	10/27/99	2.6	940 J,H	
MW-9	12/14/99		1800	
MW-10	8/2/98		958	
MW-10	10/21/98		1200 J,H	
MW-10	12/15/99		970	
MW-16	7/7/99		1000	
MW-16	9/2/99		5500	
MW-16	10/27/99		2500	
MW-16	12/14/99		2700	
MW-20	7/8/99	7.6 J,H		
MW-20	10/27/99	3.3		
MW-25	9/3/99		920	

7.3.1 Spatial Discussion of Dissolved Phase Contamination Trends

In this subsection, the dissolved phase contamination trends are discussed with respect to their location in space, e.g., spatially. Figure 18 presents the approximate extent and concentrations of dissolved PCE in groundwater based on the July 1999 through November 1999 sampling results, and Figure 19 presents the same information based on the July 1999 through January 2000 sampling results. Both figures show two areas of high PCE contamination, a northerly, or upper plume centered on MW-16, and a southerly, or lower, plume near MW-4A. In the lower plume, Figure 19 also indicates a third area of high PCE concentrations centered on MW-9. Review of the groundwater flow directions depicted on Figure 14 indicates that the northerly plume is migrating in a north-northwesterly direction, whereas the southern plume is migrating in a southwesterly direction. These two areas are discussed in greater detail in the following paragraphs.

7.3.1.1 PCE Plume Extending from MW-4A to the Kenai River.

Monitoring wells completed in the fill and alluvium adjacent to the Kenai River include MW-4A, MW-19, MW-9, MW-10, and the row of sentry wells spaced along the shore of the Kenai River (MW-6, MW-7, MW-8, MW-12, MW-13, MW-20, and MW-27). All of these wells are completed in the upper (i.e., water table) aquifer. The distinguishing characteristic of the groundwater in this area is that it contains relatively high concentrations of PCE degradation products, in addition to relatively high PCE concentrations (see Appendix C). This is a strong indication of natural attenuation, since there is no other reasonable source for the PCE degradation products at this site. PCE degradation was discussed in general terms in Section 4.2, and a RTRVP site-specific discussion will be provided in Section 7.6.2.

Figure 19 indicates two areas of greatest PCE concentrations—one area is centered on MW-4A, and the other is centered on MW-9. During the July and November 1999 sampling, only MW-4A samples exceeded 1 mg/L PCE (Figure 18); however, the December 1999 sample from MW-9 also exceeded 1 mg/L PCE. It is inferred that a flowpath extends from MW-4A toward MW-10 and MW-8. It is also inferred that the PCE in the second area of high PCE concentrations (from MW-9 to MW-6 and MW-20) is coming from the former excavation backfill. Monitoring well MW-19, located between MW-4A and MW-9, has anomalously low PCE concentrations, relative to MW-4A and MW-9, but relatively high degradation product concentrations. Based on sampling data from 1999, the sentry wells appear to intercept the center of the contamination plume based on the decreasing contaminant concentration from the center (MW-20 and MW-6 – up to 980 µg/L PCE), to the eastern (MW-5 – up to 31 µg/L PCE), and western (MW-13 – up to 90 µg/L PCE) ends of the well line (Figures 18 and 19).

7.3.1.2 PCE Plume centered on MW-16.

Groundwater from MW-16 contained the highest PCE concentrations detected on the site since monitoring wells were installed in July 1997. This plume, located in the upper water table aquifer, is flowing in a generally northward direction toward MW-25 (see Figure 14). This plume is distinguished by high PCE concentrations and a near absence of PCE degradation products, indicating little or no natural attenuation has occurred.

7.3.1.3 Semi-Confined Water-Bearing Zone.

Monitoring wells completed in the semi-confined water-bearing zone include MW-15, MW-17, and MW-18. In addition, MW-1C (deep) and SB-1 (deep and shallow) are inferred to be completed in semi-confined water-bearing zones. It is unknown whether there is a connection between these water-bearing zones. No contamination has been detected in samples from any of these monitoring wells (see Appendix C).

After the July 1999 sampling event, it was uncertain whether MW-11 was possibly completed in the semi-confined water-bearing zone. However, based on interpretations of subsequent data (additional well data as discussed in Section 0 and cation/anion balance data as discussed in Section 6.4.2.2) it is now interpreted to be completed in the upper aquifer.

7.3.1.4 West Side of the Sterling Highway.

Monitoring wells installed on the west side of the Sterling Highway (MW-27, MW-2, SB-1 (shallow and deep), MW-30, MW-31, MW-34, and temporary well MW-28 installed in the highway) have not encountered PCE contamination at concentrations greater than 2 µg/L (compared to the MCL of 5 µg/L). All of these monitoring wells, except SB-1 (shallow and deep), were completed in the upper water table aquifer. SB-1 (shallow and deep) are apparently completed in two water-bearing intervals within the semi-confined water-bearing zone. The nearly complete lack of contamination in this area is inferred to be attributable to groundwater capture prior to reaching these wells (i.e., by the storm sewer system or another feature).

7.3.1.5 Storm Sewer System.

As discussed previously in Section 6.3.1.4 and as indicated in Figures 7, 8, and 12, the storm sewer system running along the Sterling Highway is interpreted to capture groundwater north of the groundwater divide. The evidence supporting this conclusion includes a groundwater table interpretation indicating flow from the site to the storm sewer system and detections of PCE and its degradation products indicating that contamination is entering the storm sewer system between MH-2 and MH-1. Both of these points are discussed in detail below, along with an evaluation of the cross-connect between the RTRVP site and the storm sewer system (another potential groundwater flow conduit between the RTRVP site and the storm sewer system).

<u>Groundwater Table Interpretation:</u> The relationship between the water table, till surface, and storm sewer system elevation indicates that groundwater would be likely to flow from the northern portion of the site down the storm sewer/storm sewer backfill. Diagram 6-1 provides an as-built drawing of the storm sewer system adjacent to the RTRVP site. Due to the presence of the till ridge extending southeastward from MW-30, site groundwater northward from the vicinity of MW-21 flows in a generally northerly direction toward the Sterling Highway and the storm sewer system (see Figures 8 and 12). This interpretation is based on the relative elevations of key features: the storm sewer elevation (invert elevation of approximately 54 feet), the elevation of the top of till near MW-30 (approximately 60 feet), and the water table elevation (between 56 and 57 feet in MW-16, MW-21, and MW-25). The storm sewer backfill is inferred to provide a zone of relatively high permeability for site groundwater to enter and flow toward the Kenai River. The relative amount of flow into the actual storm sewer versus the flow in the backfill is presently unknown.

<u>Contamination in the Storm Sewer System:</u> Detections of PCE and its degradation products in samples from MH-1 and the KRBO indicate that contamination is entering the storm sewer system between MH-2 and MH-1. Historical sampling by E&E (1997b, 1999a, and 1999b) and Hart Crowser (1999e) has indicated PCE and several of its degradation products in the storm sewer system adjacent to the RTRVP site but not in the rest of the storm sewer system (Table 7-1). PCE has been detected in all five samples collected from the KRBO and in both samples collected from MH-1at levels above the WQS. MH-2 has been sampled twice; PCE and its degradation products were not detected in either sample. MH-2 is shown in Appendix A photograph 26. In an attempt to locate the area where groundwater was entering the storm sewer, E&E performed a remote camera surveillance of the storm sewer system in the vicinity of the RTRVP site in fall 1998. Rocks and other debris in the storm sewer between MH-2 and MH-1 prevented evaluation of this section of pipe using the remote camera, but no evidence of large volume infiltration of groundwater was seen between MH-1 and the SHBO.

<u>Influence of the Cross-Connect:</u> Backfill for the cross-connect line running from the sewer lift station on the RTRVP property to MH-1 in the storm sewer system provides a potential contamination pathway to the storm sewer system. A smoke test performed by E&E in the fall of 1998 (E&E, 1999a) indicated a direct connection between the cross-connect line and the storm sewer. Photograph 19 in Appendix A shows the inside of the

MH with the cross-connect line. This cross-connect line has since been plugged. Survey data collected during the July RI effort indicates that the cross-connect is located at an elevation of 57.75 feet at the lift station, and utility drawings indicate the cross-connect enters the storm sewer system at an elevation of approximately 56 feet (Diagram 6-1). This elevation information is projected onto Figure 5 (Cross Section A-A') and Figure 6 (Cross Section C-C'). Comparison of this information with the top of till contour map (Figure 11) indicates that the cross-connect at the lift station is interpreted to be above the top of the till, but the cross-connect at the MH is interpreted to be at or below the pre-storm sewer till surface. Therefore, it is interpreted that the cross-connect backfill could be a conduit for groundwater flow from the RTRVP site to the storm sewer system backfill. However, the groundwater contamination levels in this section of the site are relatively low (maximum PCE concentration of 87 μg/L in MW-14—see Appendix D) with respect to other groundwater contamination levels at the RTRVP, so the impact of this flow conduit is not expected to be as significant as flow into the storm sewer backfill from the northern portion of the site (e.g., maximum PCE concentration of 920 μg/L detected in MW-25).

E&E speculated that MW-14 was installed in the cross-connect backfill. Using the comprehensive site survey data collected during the RI, MW-14 appears to be offset by approximately 5 feet from the cross-connect center line (see Figure 14). Consequently, the monitoring well is near to, but probably not completed within the cross-connect backfill.

<u>MW-27 Sample Results:</u> During the November 1999 RI field effort, a monitoring well (MW-27) was installed in the storm sewer backfill near the Kenai River (see Figure 18). Very low PCE concentrations have been detected in this well (maximum concentration of 1.1 μg/L, see Appendix D). This result was not expected, given the higher concentrations detected in the KRBO itself. It is unknown why such low detections were measured in this well. The most likely explanation is that the low detections result from the contaminated groundwater in the storm sewer system dispersing away from the storm sewer backfill before reaching MW-27. The storm sewer is interpreted to emerge above the top of the till surface somewhere between SB-1 and MW-27 (see Figure 10 [F-F']). Contaminated groundwater flowing down the storm sewer backfill could, and likely would, disperse away from the storm sewer backfill when it emerges above the top of the till surface. Other possible explanations are listed below:

(1) MW-27 may not actually be installed in the backfill,

(2) Clean groundwater draining into the storm sewer backfill from the area west of the Sterling Highway may be diluting the PCE contamination in the backfill. The concentrations detected in MW-27 are lower than the concentrations detected in samples from the KRBO; this result would indicate that holes in the storm sewer or other means by which groundwater can enter the storm sewer are more prevalent in the area north of approximately MH-1 than in the area south of approximately MH-1 (see sample results in Table 7-1). Therefore, the contamination in the storm sewer is undergoing less dilution than the contamination in the storm sewer backfill.

<u>Storm Sewer System Conclusions:</u> Although evidence suggests that contamination from the northern plume of the RTRVP site, possibly along with contamination from the vicinity of MW-14 via the cross-connect, is flowing from the site to the Kenai River via the storm sewer system, alternative explanations are possible. It is possible that a combination of preferential pathways from other buried utilities or natural site heterogeneities could provide conduits for contaminated groundwater to flow from the RTRVP site to the storm sewer, and ultimately to the Kenai River. Backfill from burial of other utilities (i.e., sanitary sewer, water, electric, cable, and gas), whose burial depths

are unknown, located in the Sterling Highway ROW (see approximate locations in Figure 12) could provide conduits for groundwater flow. The cross-connect between the RTRVP property and the storm sewer system provides another potential flow conduit. Installation of approximately three new monitoring wells in the Sterling Highway ROW would help determine the groundwater flow path or paths from the RTRVP site to the storm sewer system. Additional monitoring wells would also provide additional evidence to either support or dispute the conclusion that the northerly groundwater flow direction north of the divide is only a local phenomenon, i.e., that the prevalent groundwater flow direction is toward the south or southwest.

7.3.2 Temporal Discussion of Dissolved Solvent Contamination Trends

In this subsection, the dissolved phase contamination trends are discussed with respect to their changes over time, e.g., temporally.

7.3.2.1 Trend Chart Evaluation

Figure 20 presents graphs illustrating the changes in the concentrations of PCE and its by-products over time for ten of the site's monitoring wells, which were chosen to be characteristic of the different RTRVP water-bearing zones and PCE concentrations (trend curves for all of the site monitoring wells are provided in Appendix J). In Figure 20, two curves are presented for each monitoring well: one shows the PCE concentrations on a molar basis, and the other shows the sum of the PCE daughter product concentrations, also on a molar basis (TCE+DCE+VC). Concentrations were converted from a mass basis (mg/L) to a molar basis (mol/L) so that accurate comparisons could be made between the curves.

The concentration trends and relative concentrations of PCE and its daughter products are dependent on several factors, including:

distance of the sample location from the contaminant source or sources,

presence of multiple source areas,

occurrence of natural attenuation processes,

seasonal variations in groundwater recharge, and

changes to the soils in the area (e.g. remedial excavation and back-filling).

Due to the small number of sample points, a regression analysis has not been performed on the data shown in Figure 20. As additional data are collected, regression analysis will become a useful tool for evaluating PCE trends. In the meanwhile, only a visual evaluation of the potential data trends has been performed. By inspection, three of the ten characteristic monitoring wells exhibit an apparent trend of increasing PCE concentrations over time (MW-9, MW-11, and MW-13). One monitoring well exhibits an apparent trend of decreasing PCE concentrations over time (MW-4A), although PCE concentrations detected in that monitoring well, which have never been below the site ACL for groundwater, remain significantly above this ACL. No visible trend is apparent in the remaining graphs.

The monitoring wells depicted in Figure 20 can be sorted into two categories: 1.) mostly PCE, and 2.) mostly PCE breakdown products. Monitoring wells MW-4A, MW-6, MW-8, MW-9, MW-10, and MW-19 have greater concentrations of daughter products than PCE. The remaining four wells (MW-11, MW-13, MW-14, and MW-16) have higher concentrations of PCE than daughter products. Figure 20 indicates that greater levels of biodegradation have occurred in the groundwater measured in the lower portion of the site than in the other site monitoring wells. MW-19 exhibits the greatest level of biodegradation of the ten wells evaluated. A detailed discussion of biodegradation at the RTRVP site is provided in Section 7.5.

7.3.2.2 Trends in ACL Exceedences

As discussed previously, VC and PCE have been detected in site groundwater at levels above ACLs (see Table 7-4). The PCE exceedences were graphed and discussed in the previous section; the VC exceedences are discussed in this section.

In the lower plume area, the ACL for VC (which is equal to the WQS of 2 μg/L) has been exceeded in samples from several monitoring wells. The ACL for VC has been exceeded in many samples from MW-6, and some samples from MW-9 and MW-20. All detections are less than 10 μg/L, and no trend of increasing or decreasing VC concentrations is apparent in any of these monitoring wells. The ACL for VC was exceeded in three of four samples collected from MW-6 in 1999 (2.85 μg/L in April, 3 μg/L in July, and 3.7 μg/L in October). The ACL for VC was also exceeded in three of four samples collected from MW-9 in 1999 (2.49 μg/L in April, 3 μg/L in July, and 2.6 μg/L in October). The ACL for VC was also exceeded in two of three samples collected from MW-20 in 1999 (7.6 μg/L in July, and 3.3 μg/L in October). VC was not detected in any of the samples collected in December 1999 (and the detection limits were similar to previous 1999 detection limits).

7.4 EFFECT OF SOIL LEACHING ON GROUNDWATER QUALITY

Leaching from site soil contaminated by PCE at levels below the ACL can result in groundwater impact. The impact on groundwater of soil leaching at its PCE ACL (11.5 mg/kg) was evaluated using equations provided by the ADEC in 18 AAC 75 (Oil and Hazardous Pollution Control Regulations). The "migration to groundwater" equations are applicable for calculating site-specific soil cleanup values based on targeted groundwater concentrations, or conversely, calculating groundwater concentrations from targeted soil concentrations. The migration to groundwater equations consider contaminant volatilization and adsorption to organic carbon, but do not consider contaminant degradation. The migration to groundwater equations assume that the contaminated soil lies directly over the groundwater; no allowance is made for attenuation during downward migration through the vadose zone. Site-specific infiltration rate, soil parameters (e.g., TOC, porosity, hydraulic conductivity and gradient, aquifer thickness, and source length), and chemical parameters (e.g., k_{oc} and Henry's Law Constant) may be input into the migration to groundwater equations. Note that the most recent version of the 18 AAC 75 regulations, which contain the referenced migration to groundwater equations, had not yet been promulgated when ACLs were established for this site in 1997. This issue is discussed further in Section 9.5.

The migration to groundwater calculations were used, along with a range of reasonable input parameters, to estimate the likely effect of soil leaching at the ACL. Six scenarios were evaluated: minimum, average, and maximum groundwater parameters at the lower plume, and minimum, average, and maximum groundwater parameters at the upper plume area. The minimum, average, and maximum groundwater parameters refer to the minimum, average, and maximum site hydraulic conductivity and gradient listed in Table 6-4. Each of the six scenarios was evaluated over the range of reported PCE organic carbon/soil partitioning coefficient (k_{oc}) values. The spreadsheets are provided in Appendix K, and general conclusions are listed below:

- Only one scenario, the "minimum" groundwater parameter scenario at the lower plume site, predicted that PCE leaching from soil contaminated at the

ACL (11.5 mg/kg) would cause groundwater contamination above the ACL (0.84 mg/L).

- All of the scenarios predict that PCE leaching from soil contaminated at the ACL would cause groundwater contamination above the MCL (0.005 mg/L).
- The "average" set of groundwater parameters at the lower plume predicts that PCE leaching from soil contaminated at the ACL would result in a groundwater concentration of 0.6 mg/L. The extreme range of predicted groundwater concentrations from soil leaching at the ACL (represented by maximum groundwater parameters and maximum k_{oc}, and minimum groundwater parameters with minimum k_{oc}) is 1.58 mg/L to 0.03 mg/L.
- Based on the observed groundwater PCE concentrations in the lower plume, it is likely that soil containing PCE at levels above the 11.5 mg/kg ACL remains on site upgradient of MW-4A and possibly upgradient of MW-9. The maximum predicted groundwater concentration resulting from PCE leaching at 11.5 mg/kg is 1.58 mg/L, which has been exceeded in samples from both MW-4A and MW-9.
- The "average" set of groundwater parameters at the upper plume predicts that PCE leaching from soil contaminated at the ACL would result in a groundwater concentration of 0.15 mg/L. The extreme range of predicted groundwater concentrations from soil leaching at the ACL (represented by maximum groundwater parameters and maximum k_{oc}, and minimum groundwater parameters with minimum k_{oc}) is 0.67 mg/L to 0.01 mg/L.
- Based on the observed groundwater PCE concentrations in the upper plume, it is likely that soil containing PCE at levels above the 11.5 mg/kg ACL remains on site upgradient of MW-16. The maximum predicted groundwater concentration resulting from PCE leaching at 11.5 mg/kg is 0.67 mg/L, which has been exceeded in all samples from MW-16.

7.5 PCE LEACHING FROM SPREAD TREATMENT CELL SOILS

Approximately 125 cubic feet of investigation-derived wastes (IDW) contaminated soil stored in drums and approximately 100,000 cubic feet of soil contaminated by PCE and its degradation products stored in two on site treatment cells are presently located on the RTRVP property. To address the disposal of these soils, a "Contained-in" determination was made (the "Contained-in" determination is included in Appendix R and discussed further in Section 9.5). Modeling activities were performed to support the "Contained-in" determination for disposal of the excavated soils and IDW generated at the RTRVP site. Specifically, the purpose of the modeling was to evaluate the impact of PCE leaching from PCE-impacted soils spread on the site at a range of potential lift thicknesses.

Modeling was performed using the Seasonal Soil Compartment (SESOIL) model to simulate leaching through the vadose zone, and the Analytical Transient 1,2,3-Dimensional (AT 1,2,3-D) model to simulate groundwater transport. In general, input parameters were chosen to be conservative and over-predict leaching. A soil concentration of 11.5 mg/kg PCE was input into the model to represent the spread soils. This input concentration was based on the maximum allowable PCE concentration established for site soils in an August 27, 1997 letter from Lynn Kent of the ADEC to Gary Hinkle, site owner. The approximately 100,000 cubic feet of contaminated soil at the site was divided into three theoretical lift thicknesses (5-foot lift, 2-foot lift, and 1-foot lift) for modeling purposes. The conclusion of modeling activities is that the stockpile soils can be spread on the site subject to the following conditions:

- Soil spreading should occur in locations where the minimum depth to groundwater (measured from the original, or pre-spreading, land surface) is 5 feet. A 5-foot depth to groundwater is a minimum depth observed in site monitoring wells located on RTRVP property, excluding the sentry wells located along the Kenai River. The 5-foot depth to groundwater was used in the modeling simulations.
- Soil spreading should occur no closer than approximately 100-feet from the Kenai River. This distance is based on the minimum depth to groundwater distance (discussed in the previous condition) and practical site considerations (e.g., it would be impractical to spread soils on the embankment sloping toward the Kenai River).
- Soil spreading can occur in a lift thickness up to 2-feet without adversely impacting groundwater quality.
- Soil spreading should occur during late spring/early summer timeframe, after spring breakup, but as early in summer as possible. This will allow maximum time for PCE volatilization after spring breakup and before fall rains maximize water infiltration.
- Soil spreading should occur during periods of no precipitation to minimize runoff concerns and maximize volatilization of the contaminants. If practical, spreading in lifts of approximately 6 inches at a time will further increase volatilization.
- Snow covering the spread soils should be removed prior to breakup for a period of 5 years. This will minimize the slug of water infiltration occurring as a result of spring melting.
- Spread soils should be re-vegetated to increase evapotranspiration, increase stability, and enhance natural attenuation.
- Engineering controls must be utilized to ensure that no surface runoff occurs.

A detailed modeling discussion is provided in Appendix L. Appendix L includes a discussion of model input parameters and calibration, modeling results, and a limited sensitivity analysis. The sensitivity analysis evaluated the effect of a different soil input concentration (the average soil treatment cell concentration, as reported by (Hart Crowser, 1999d), increased precipitation, and different soil properties. Figure 1 in Appendix L provides a graph of predicted PCE concentrations for the 1-foot, 2-foot, and 5-foot lift soil models. Also in Appendix L is the SESOIL output file for the 2-foot lift base case.

7.6 INTRINSIC REMEDIATION EVALUATION

Intrinsic remediation, also commonly referred to as natural attenuation, includes the natural chemical, physical, or biological processes that reduce or eliminate contaminant concentrations in soil, surface water, or groundwater. Intrinsic remediation results from several subsurface attenuation mechanisms that are either destructive or nondestructive to the contaminant. Destructive attenuation removes contaminant mass from the soil or water. Biodegradation is the most important destructive attenuation mechanism (Wiedemeier, et al., 1996). Nondestructive attenuation mechanisms include sorption, dispersion, dilution, and volatilization.

7.6.1 Intrinsic Remediation of Chlorinated Solvents

A detailed discussion of PCE biodegradation is provided in Section 4.2. The basic concepts are reiterated here, and a site-specific intrinsic remediation evaluation is performed for the RTRVP site.

The lines of evidence used to document intrinsic remediation of chlorinated solvents are based on evidence of biodegradation through three different pathways: use as an electron acceptor, use as an electron donor, or cometabolism. At a given site, one or all of these processes may be operating, although the use of chlorinated hydrocarbons as electron acceptors appears to be the most important process under natural conditions. In this case, biodegradation of chlorinated hydrocarbons will be an electron-donor-limited process (i.e., the process is limited by the availability of a carbon source).

The most important process for the natural biodegradation of the more highly chlorinated solvents (i.e., PCE) is reductive dechlorination. During this process, the chlorinated hydrocarbon is used as an electron acceptor and a chlorine atom is removed and replaced with a hydrogen atom. In general, reductive dechlorination occurs by sequential dechlorination from PCE to TCE to DCE to VC to ethane (see Diagram 1-1 in Section 1.3.2). Reductive dechlorination has been demonstrated to occur under nitrate- and iron-reducing conditions, but the most rapid rates occur under sulfate-reducing and methanogenic conditions (Wiedemeier, et al., 1996). Because chlorinated hydrocarbons are used as electron acceptors during reductive dechlorination, there must be an appropriate source of carbon for microbial growth in order for this process to occur. Potential carbon sources include natural organic matter, fuel hydrocarbons, or other anthropogenic organic compounds. Reductive dehalogenation reactions occur primarily under anaerobic conditions.

It is generally believed that microorganisms are incapable of growth using PCE and TCE as a primary substrate (electron donor). The less oxidized chlorinated hydrocarbons (e.g., VC), however, can be used as the primary substrate in biologically mediated oxidation-reduction reactions (Wiedemeier, et al., 1996). These reactions occur under aerobic and some anaerobic conditions.

When a chlorinated hydrocarbon is biodegraded by cometabolism, the degradation is catalyzed by an enzyme or cofactor that is fortuitously produced by the organisms for other purposes. During cometabolism, bacteria indirectly transform the chlorinated hydrocarbon as they use benzene, toluene, ethylbenzene, and total xylenes (BTEX) or another substrate to meet their energy requirements.

Three types of plume behavior can be expected for a chlorinated solvent plume depending on the amount of solvent, amount of organic substrate, distribution and concentration of natural electron acceptors, and the types of electron acceptors being utilized. A discussion of each of the three plume behaviors as defined by Wiedemeier et al., (1996) is provided here.

Type 1-plume behavior occurs when the primary substrate is an anthropogenic carbon (e.g., BTEX), and microbial degradation of this anthropogenic carbon drives reductive dechlorination of the chlorinated hydrocarbon. This reaction only occurs under strongly reducing conditions.

Type 2-plume behavior occurs in areas that are characterized by relatively high concentrations of naturally-occurring dissolved organic matter, such as occurs in a wetland environment. The Type 2 plume is the same as a Type 1 plume except for the source of the biologically available carbon substrate.

Type 3-plume behavior occurs in areas that are characterized by inadequate concentrations of natural or anthropogenic carbon, and concentrations of DO that are greater than 1 to 2 mg/L. Under these relatively aerobic conditions reductive dehalogenation will not occur. However, VC and other less chlorinated compounds produced under Type 1- or Type 2-plume behavior can be rapidly biodegraded, acting as a primary substrate (electron donor), under these aerobic conditions.

7.6.2 Intrinsic Remediation in Upper and Lower Groundwater Plumes at the RTRVP site

Selected monitoring wells were sampled for natural attenuation parameters in the July, September, October, and December 1999 sampling events. The wells sampled and the associated analytical results are presented in Table 6-11. Plate 7 illustrates the natural attenuation parameter results for July through October 1999. The following sections discuss the five geochemical parameters that were used to evaluate the effectiveness of intrinsic remediation. DO is the most thermodynamically favored election acceptor used by microbes for the biodegradation of organic carbon, whether natural or anthropogenic. During aerobic respiration, DO concentrations decrease. After depletion of DO, anaerobic microbes will use nitrate as an electron acceptor, followed by iron (III), then sulfate, and finally carbon dioxide (methanogenesis). Each sequential reaction drives the ORP of the groundwater downward into the range within which reductive dechlorination can occur. Kastner (1991) cites an ORP range of -150 to -210 millivolts (mV) for reductive dechlorination of PCE to occur.

7.6.2.1 DO

As discussed above, DO is the most thermodyamically favored electron acceptor for biodegradation. DO was measured in the field, and the results are provided on Table 6-9 and Plate 7.

MW-24, MW-29, and MW-30, which are not clearly in either the upper or lower plume, exhibited DO levels between 3.7 mg/L (October 1999 level in MW-24) and 6.11 mg/L (September 1999 level in MW-24). MW-32, the background well, exhibited a DO level of 4.72 mg/L.

Semi-Confined Water-Bearing Zone

The July 1999 through January 2000 sampling results for the semi-confined water-bearing zone (e.g., MW-15, MW-17, MW-18, SB-1 (deep and shallow), and MW-1C (deep)) indicate DO levels between 0.38 mg/L (MW-15) and 5.56 (MW-18).

Upper Plume

The July 1999 through January 2000 sampling results from monitoring wells in the upper plume and nearby wells outside the plume (e.g., MW-16, MW-21, MW-22, MW-23, MW-25, MW-31, and MW-32) indicate that this section of the site has relatively high DO concentrations (2.5 mg/L to 8.01 mg/L). The DO readings in most of these monitoring wells were between 4 mg/L and 7 mg/L. These DO levels are indicative of aerobic groundwater conditions.

Lower Plume

The July 1999 through January 2000 sampling results from monitoring wells in the lower plume and nearby wells outside the plume indicate that this section of the site has a wide range in DO concentrations (0.85 mg/L to 7.75 mg/L). The DO readings in most of these monitoring wells were between 1 mg/L and 5 mg/L. The DO readings in the lower plume were generally lower than the DO readings in the upper plume.

There is a discrepancy between the July 1999 DO levels and the October through December 1999 DO levels in the lower plume area. In July, the DO concentrations in MW-3A, MW-4A, MW-6, MW-8, MW-9, MW-10, MW-12, MW-14, MW-19, and MW-20 ranged between 0.85 mg/L and 1.92 mg/L. In October and December 1999, the DO concentrations in these monitoring wells ranged from 2.64 mg/L to 5.69 mg/L. The reason for this discrepancy is unknown, but it could possibly be due to inaccurate DO readings or seasonal variability. The reason for the discrepancy between the DO levels observed in July 1999 and those measured later is unknown, but it may be due to several factors, including the fact that accurate measurement of low DO concentrations is difficult. Alternatively, the DO levels could experience seasonal variability (although other geochemical parameters, discussed below, indicate anaerobic conditions).

Depressed DO concentrations in the lower portion of the site (based on the July 1999 sampling) indicate that some type of aerobic degradation process has occurred on the site to utilize the DO. The reason for the degradation is unknown, but it is likely that degradation of naturally-occurring organic carbon or possibly fuel hydrocarbons (although only low levels of fuel hydrocarbons have been detected in a few monitoring locations on the site) has occurred.

Wiedemeier et al., (1996) report that anaerobic bacteria generally cannot function at DO concentrations greater than about 0.5 mg/L. As noted above, all site monitoring wells exhibited DO levels greater than 0.5 mg/L (except MW-15 in July 1999). While the presence of oxygen precludes the anaerobic degradation, it is apparent from the presence of PCE daughter products that some parts of the site are anaerobic, or have previously been anaerobic. Other geochemical analyses (e.g., the presence of methane and ferrous iron, see discussion below) provide further evidence of anaerobic groundwater conditions at the site.

The site appears to have complex groundwater flow patterns, which could result in “pockets” of aerobic and anaerobic groundwater conditions that are not necessarily representative of the overall aquifer conditions. Examples are crevices in rocks, discontinuous geologic strata, and areas of degradable total organic carbon.

Overall, although the DO data are difficult to interpret, there is no indication of anaerobic conditions in the upper plume area, but there are indications of anaerobic conditions in the lower plume area. Therefore, reductive dechlorination of PCE would be more likely in the lower plume area.

7.6.2.2 Nitrate-Nitrite

After DO has been depleted in the microbiological treatment zone, nitrate may be used as an electron acceptor for anaerobic biodegradation of organic carbon via denitrification. In order for reductive dechlorination to occur, nitrate concentrations in the contaminated portion of the aquifer must be less than 1.0 mg/L (Wiedemeier, et al., 1996).

MW-24, which is not clearly in either the upper or lower plume, exhibited a low nitrate concentration, 0.1 mg/L, in September 1999. MW-32, the background monitoring well, exhibited a nitrate concentration of 4 mg/L in December 1999.

Semi-Confined Water-Bearing Zone

Nitrate-nitrite was not evaluated in any monitoring wells completed in the semi-confined water-bearing zone.

Upper Plume

Groundwater samples collected from the upper plume area between July and December 1999 had nitrate concentrations between 0.54 mg/L and 18.0 mg/L (Table 6-11 and Plate 7). Four monitoring wells in the upper plume area (MW-21, MW-22, MW-23, and MW-25) exhibited nitrate-nitrite concentrations above 1 mg/L (see Table 6-11 and Plate 7). One monitoring well, MW-21, exhibited nitrate-nitrite concentrations of 12 mg/L

(September 1999) and 18 mg/L (October 1999), which exceed the MCL for nitrate. The source of nitrate-nitrite in this monitoring well is unknown. Comparison of the July through December 1999 data indicates similar nitrate-nitrite concentrations for these time periods.

In summary, the laboratory analytical data indicate that the groundwater nitrate levels across much of the upper plume area are too high for reductive dechlorination to occur.

Lower Plume

Groundwater samples collected from the lower plume area between July and December 1999 had nitrate concentrations between ND and 7.0 mg/L (Table 6-11 and Plate 7). One monitoring well in the lower plume (MW-14) exhibited nitrate-nitrite concentrations above 1 mg/L (7 mg/L in October 1999 and 3.5 mg/L in December 1999). Nitrate-nitrite was only detected in three other lower plume monitoring wells: MW-4A (0.6 mg/L in October 1999), MW-26 (0.3 mg/L in September 1999) and MW-24 (0.1 mg/L in September 1999). Comparison of the July through December 1999 data indicates similar nitrate-nitrite concentrations for these time periods.

In summary, the laboratory analytical data indicate that the groundwater nitrate levels across the lower plume area are generally low enough for reductive dechlorination to occur.

7.6.2.3 Iron (II)

Iron [III] (or ferric iron) can be used as an electron acceptor during anaerobic biodegradation of organic carbon. During this process, iron (III) is reduced to iron (II), which may be soluble in water. Iron (II) concentrations can be used as an indicator of anaerobic degradation of organic carbon, fuel compounds and VC. Generally, iron (II) concentrations greater than 1.0 mg/L can be used as an indicator that anaerobic degradation is occurring (Wiedemeier, et al, 1996).

Ferrous iron measurements were obtained from site monitoring wells using both laboratory analysis and field testing. For the laboratory analysis, groundwater samples were collected, filtered in the field, and analyzed for dissolved iron. Since only ferrous iron is soluble in water, the laboratory results reflect ferrous iron. For the field analysis, a Hach™ field test kit was used to test water for ferrous iron. Both the laboratory and field test kit results are discussed in this section.

MW-24, which is not clearly in either the upper or lower plume, did not exhibited a ferrous iron detection in September 1999. Ferrous iron was also not detected in MW-32, the background well.

Semi-Confined Water-Bearing Zone

Iron was evaluated in two semi-confined water-bearing zone monitoring wells (MW-15 and MW-17) using field testing. Iron was not detected in samples from either of these wells.

Upper Plume

In the upper plume area, only one groundwater sample was submitted for laboratory analysis for iron (July 1999 sample from MW-16) (see Table 6-11 and Plate 7). A very low dissolved iron concentration (0.035 mg/L) was detected in this well. Field ferrous iron measurements (using a Hach™ field test kit) were taken in upper plume monitoring wells MW-16, MW-21, MW-22, MW-23, and MW-25. A maximum iron concentration of

0.6 mg/L was detected using the field kit (in a September 1999 sample from MW-21). These results indicate very low ferrous iron concentrations, which indicate that anaerobic degradation is not occurring in the upper plume.

Lower Plume

In the lower plume area, groundwater samples from MW-6, MW-9, MW-11, and MW-19 were submitted for laboratory analysis for iron (see Table 6-11 and Plate 7). Laboratory results indicated ferrous iron concentrations greater than 1 mg/L in monitoring wells MW-6, MW-9, and MW-19, which also exhibited low DO and elevated methane concentrations.

Field measurements of ferrous iron (using a Hach™ field test kit) indicated iron concentrations exceeded 1 mg/L in monitoring wells MW-6, MW-9, MW-19, and MW-20 (Table 6-11), and iron was not detected in monitoring wells MW-4A, MW-11, MW-14, and MW-26. Ferrous iron concentrations between 0.6 mg/L and 0.4 mg/L were detected in MW-7 and MW-8. Where comparison is possible, the field measurements generally agree with the laboratory analytical results.

The iron results indicate that anaerobic degradation is occurring in the vicinity of MW-6, MW-9, MW-19, and MW-20.

7.6.2.4 Sulfate

After DO and nitrate have been depleted in the microbiological treatment zone, sulfate may be used as an electron acceptor for anaerobic biodegradation. This process is termed sulfate reduction and results in the production of sulfide. Concentrations of sulfate greater than 20 mg/L may cause competitive exclusion of dechlorination (Wiedemeier et al, 1996).

MW-24, which is not clearly in either the upper or lower plume, exhibited the highest sulfate concentration detected in site monitoring wells, 72.1 mg/L. MW-32, the background well, exhibited a sulfate level of 9.6 mg/L.

Semi-Confined Water-Bearing Zone

Sulfate was not evaluated in any monitoring wells completed in the semi-confined water-bearing zone.

Upper Plume

Groundwater samples collected from the upper plume area between July and December 1999 had sulfate concentrations between 6.4 mg/L and 28.0 mg/L (Table 6-11 and Plate 7). Sulfate concentrations in MW-21 were above 20 mg/L, and sulfate concentrations in MW-16 were between 11.6 mg/L and 19.1 mg/L. Sulfate concentrations in monitoring wells MW-22, MW-23, MW-25, and MW-32 were less than 14 mg/L.

Laboratory analytical data indicate that sulfate concentrations may not be low enough for reductive dechlorination to occur in the most contaminated areas of the upper plume (near MW-16).

Lower Plume

Groundwater samples collected from the lower plume area had sulfate concentrations between 3.0 mg/L (in MW-9) and 15.8 mg/L (in MW-6) (Table 6-11 and Plate 7). No monitoring wells in the lower plume exhibited sulfate concentrations greater than 20 mg/L. Laboratory analytical data indicate that sulfate concentrations are low enough for reductive dechlorination to occur in the lower plume.

In general, the monitoring wells in the lower portion of the site (except MW-4A) exhibited lower sulfate concentrations (less than 10 mg/L) than the monitoring wells in the upper portion of the site. These results suggest that sulfate reduction may be occurring in the lower portion of the site (the anaerobic portion).

7.6.2.5 Methane

During methanogenesis acetate is split to form carbon dioxide and methane, or carbon dioxide is used as an electron acceptor, and is reduced to methane. Methanogenesis generally occurs after oxygen, nitrate, and sulfate have been depleted in the treatment zone. The presence of methane in groundwater is indicative of strongly reducing conditions. Methane concentrations greater than 0.5 mg/L may indicate degradation of PCE (Wiedemeier et al, 1997).

MW-24, which is not clearly in either the upper or lower plume, exhibited a low methane detection, 0.02 mg/L in September 1999. Methane was not detected in MW-32, the background well.

Semi-Confined Water-Bearing Zone

Methane was not evaluated in any monitoring wells completed in the semi-confined water-bearing zone.

Upper Plume

Groundwater samples collected from the upper plume area between July and December 1999 exhibited low methane concentrations (Table 6-11 and Plate 7). A maximum methane concentration of 0.021 mg/L was detected in a sample from MW-22 in September 1999. Comparison of the July through December 1999 data indicate generally similar methane concentrations for these time periods.

The methane results do not indicate that methanogenesis is occurring in upper plume groundwater.

Lower Plume

Groundwater samples collected from the lower plume area between July and December 1999 had methane concentrations from below method detection limits to 5.4 mg/L (Table 6-11 and Plate 7). Monitoring wells MW-4A, MW-7, MW-11, MW-14, and MW-26 exhibited methane levels less than 0.5 mg/L. Except for MW-4A, these locations correspond to relatively high DO concentrations and low contaminant (or low PCE daughter product) concentrations. MW-7, which has only been sampled once for methane (in December 1999), exhibited a methane concentration of 0.3 mg/L. Monitoring wells MW-6, MW-8, MW-9, MW-19, and MW-20 exhibited methane levels above 0.5 mg/L.

Comparison of the July through December 1999 data indicates generally similar methane concentrations for these time periods. The most significant difference was noted in MW-8. Methane was not detected in MW-8 in October, but it was detected at relatively high concentrations in July (0.68 mg/L) and December (0.737 mg/L).

The methane results indicate that methanogenesis is occurring in lower plume groundwater. Sample locations with the greatest concentrations of PCE daughter products generally have the greatest concentrations of methane (e.g., MW-9 and MW-19, see Plate 7).

7.6.2.6 Oxidation Reduction Potential

ORP is an indicator of the relative tendency of a solution to accept or transfer electons. Wiedemeier et al. (1996) state that the reductive pathway is possible for ORP values less than 50 mV and is likely at ORP values less than –100 mV.

Between July 1999 and January 2000, the ORP was measured in various site monitoring wells. Due to ORP meter malfunction during the July 1999 sampling event, data were only collected from six, mostly uncontaminated, monitoring wells. In October 1999, the ORP was measured in most monitoring wells using field instrumentation. ORP measurements ranged from -113 mV to 43.6 mV (Table 6-9). During December 1999, the ORP was measured in only nine site monitoring wells (MW-21, MW-22, MW-23, MW-27, MW-29, and MW-30). The ORP instrument malfunctioned due to the extreme cold, and measurements could not be made in the remaining monitoring wells.

MW-24, MW-29, and MW-30, which are not clearly in either the upper or lower plume, exhibited ORP values between –64 mV (in MW-30) and 20 mV (in MW-24). MW-32, the background well, exhibited an ORP value of 46.1 mV.

Semi-Confined Water-Bearing Zone

In July and October 1999, the ORP in the semi-confined water-bearing zone ranged from –67.3 mV to –114 mV.

Upper Plume

Between July 1999 and January 2000, the ORP in upper plume monitoring wells ranged from –25 mV (in MW-23) to 180 mV (in MW-34) (see Table 6-11 and Plate 7). In most upper plume monitoring wells, the ORP ranged between approximately 5 mV and 50 mV. Only MW-16 (in July and September, but not in October), MW-21 (in September 1999, but not in October) and MW-34 registered ORP values above 50 mV. In comparing the July, September, October, and December ORP values, no pattern is clear.

Based on these ORP results, the reductive pathway is possible but not likely at much of the upper plume area. However, the center of the contamination plume, at MW-16, is in an area of relatively elevated ORP readings, where the reductive pathway is not possible.

Lower Plume

Between July 1999 and January 2000, the ORP in lower plume monitoring wells ranged from –36.2 mV (in MW-3A) to 190 mV (in MW-11) (see Table 6-11 and Plate 7). Due to the various instrument malfunctions discussed above, very limited ORP data were available for the lower plume monitoring wells. Based on the limited ORP results, the reductive pathway is possible but not likely at much of the lower plume area.

7.6.2.7 Chloride

During biodegradation of chlorinated hydrocarbons dissolved in groundwater, chloride is released into the groundwater. This process results in chloride concentrations in groundwater in the contaminant plume that are elevated relative to background concentrations. Because of the neutral chemical behavior of chloride, it can be used as a conservative tracer to estimate biodegradation rates.

MW-24, which is not clearly in either the upper or lower plume, exhibited a chloride value of 24.5 mg/L. MW-32, the background well, exhibited a chloride value of 27 mg/L.

Semi-Confined Water-Bearing Zone

In July 1999, chloride concentrations in the semi-confined water-bearing zone ranged from 19.2 mg/L (in MW-17) to 33 mg/L (in MW-18) (see Table 6-11 and Plate 7). No other semi-confined water-bearing zone monitoring wells were sampled for chloride during the 1999-2000 RI activities.

Upper Plume

Between July 1999 and January 2000, chloride concentrations in upper plume monitoring wells ranged from 15.9 mg/L (in MW-16) and 99.7 mg/L (in MW-25) (see Table 6-11 and Plate 7).

Lower Plume

Between July 1999 and January 2000, chloride concentrations in lower plume monitoring wells ranged from 1 mg/L (in MW-26) and 30 mg/L (in MW-14) (see Table 6-11 and Plate 7).

Areas of highest chloride concentrations correspond to the upper portion of the site (e.g., MW-16 and MW-25). These data do not appear to correlate with areas of chlorinated compound degradation. Elevated chloride levels can also be derived from other sources such as road deicers or septic effluent.

7.6.2.8 Total Organic Carbon (TOC)

TOC is a carbon and energy source that drives dechlorination. Wiedemeier et al (1996) state that TOC concentrations greater than 20 mg/L indicate adequate carbon source for dechlorination.

MW-24, which is not clearly in either the upper or lower plume, exhibited a TOC value of 24.3 mg/L. MW-32, the background well, exhibited a TOC value of 10 mg/L.

Semi-Confined Water-Bearing Zone

TOC was not evaluated in any monitoring wells completed in the semi-confined water-bearing zone.

Upper Plume

Between July 1999 and January 2000, TOC concentrations in upper plume monitoring wells ranged from 2.8 mg/L (in MW-16) and 25 mg/L (in MW-22) (see Table 6-11 and Plate 7). In the upper plume area, only MW-22 exhibited a TOC concentration over 20 mg/L. Comparison of the July 1999 through January 2000 upper plume data indicate similar TOC concentrations for these time periods.

In general, the distribution of TOC across the upper plume suggests that TOC is insufficient to drive dechlorination.

Lower Plume

Between July 1999 and January 2000, TOC concentrations in lower plume monitoring wells ranged from 3.9 mg/L (in MW-14) and 42.6 mg/L (in MW-9) (see Table 6-11 and Plate 7). A total organic carbon concentration of 96.5 mg/L detected in the October sample from MW-26 is believed to be due to organic material suspended in the water, which was very highly turbid.

Monitoring wells MW-4A, MW-6, MW-7, MW-9, MW-19, and MW-20 exhibited TOC concentrations over 20 mg/L. Comparison of the July 1999 through January 2000 data indicate similar TOC concentrations for these time periods, except in MW-19, where the December level (11 mg/L) was approximately one-third of the July level (31.7 mg/L).

The distribution of TOC across the site suggests that TOC can provide an adequate carbon source for dechlorination in much of the lower PCE plume.

7.6.2.9 General Inorganic Groundwater Parameters

7.6.2.9.1 Alkalinity

Alkalinity in groundwater is a measure of the water's capacity to neutralize acid. Alkalinity is important in the maintenance of groundwater pH because it buffers the groundwater system against acids generated through both aerobic and anaerobic biodegradation processes. Alkalinity is generally expressed in terms of calcium carbonate ($CaCO_3$).

MW-24, which is not clearly in either the upper or lower plume, exhibited alkalinity values between 517 mg/L and 1,470 mg/L in September, October, and December 1999 sampling. These were the three highest alkalinity measurements measured at the site. Alkalinity was not measured in MW-32, the background well.

Semi-Confined Water-Bearing Zone

In July 1999 through January 2000, alkalinity levels in the semi-confined water-bearing zone ranged from 289 mg/L (in MW-15) to 345 mg/L (in SB-1 (shallow)) (see Table 6-11 and Plate 7).

Upper Plume

Between July 1999 and January 2000, alkalinity levels in upper plume monitoring wells ranged from 68 mg/L (in MW-16) and 240 mg/L (in MW-22) (see Table 6-11 and Plate 7). Only MW-22 exhibited an alkalinity above 120 mg/L in the upper plume area.

Lower Plume

Between July 1999 and January 2000, alkalinity levels in lower plume monitoring wells ranged from 61 mg/L (in MW-19) and 184 mg/L (in MW-3A) (see Table 6-11 and Plate 7).

Groundwater alkalinity data supports the geologic log and water level measurement interpretation that the site groundwater samples appear to be from two different hydrogeologic units. In general, wells from the semi-confined water-bearing zone tend to have higher alkalinity (close to or greater than 300 mg/L) than wells from the water table aquifer (less than 150 mg/L). One exception to this generalization is MW-24, which is in the water table aquifer but had alkalinity measurements between 510 mg/L and 1,470 mg/L.

7.6.2.9.2 Temperature

Groundwater temperature directly affects the solubility of oxygen in water and the metabolic activity rate of bacteria. Oxygen is more soluble in colder water temperatures.

Groundwater temperatures less than 5°C tend to inhibit biodegradation rates, and the biodegradation rate typically doubles for every 10°C increase in the water temperature.

Groundwater temperatures measured in monitoring wells from the RTRVP site ranged from 0.1°C (in MW-29) to 12.1°C (in MW-26) during the period between July 1999 and January 2000 (see Table 6-9). The highest temperatures were recorded in MW-24 and MW-26 in September 1999. The water table is very shallow in these wells, and the water temperature appears to be strongly affected by the outside air temperature. In MW-26, the water temperature dropped from 12.1°C in September to 6°C in October to 4.1 °C in December, and MW-24 behaved similarly. Groundwater temperature measured in monitoring wells containing very little water (e.g., MW-5, MW-27 and MW-29) is also strongly affected by outside air temperature. For example, MW-29 dropped from 6.7 °C in November 1999 (when it was installed) to 0.1 °C in December 1999, and MW-27 behaved similarly. Most of the remaining monitoring wells experienced lower water temperature variation during this time period, with temperatures generally between 4°C and 8°C.

Overall, low ambient water temperatures may limit biodegradation rates in both the upper and lower plumes.

7.6.2.9.3 pH

Groundwater pH is an environmental indicator that has an effect on the presence and activity of microbial populations. This is especially true for methanogens. Bacteria capable of degrading petroleum hydrocarbons typically prefer pH values between 5 and 9 standard units (Wiedemeier et al, 1997). Measurements taken from the monitoring wells between July 1999 and January 2000 ranged from 5.5 to 9 pH units (Table 6-9).

Overall, the pH of site groundwater indicates favorable conditions for bacteria growth in both the upper and lower plumes.

7.6.3 PCE Degradation Product Analysis

The presence of PCE degradation products in site groundwater provides another line of evidence for the degradation of PCE. The percentages of PCE and its degradation products relative to the total halocarbon concentrations detected in all site monitoring wells between July 1997 and November 1999 were calculated to determine whether any biodegradation trends exist at the RTRVP site.

The analysis involved calculating the percentages of the total halocarbons detected in each groundwater sample comprised of PCE, TCE, DCE, and VC (on a molar basis). At each well, the percentages of these compounds were averaged to provide an overall characterization of the site groundwater. Average PCE, TCE, DCE, and VC percentages are summarized in Table 7-5. As indicated in Table 7-5, the average percentage of PCE varies from 100 percent of total halocarbons in MW-23 to 3 percent of total halocarbons in MW-19. This result indicates an absence of biodegradation in MW-23, and near total PCE degradation in MW-19. Due to the difficulty in obtaining meaningful results in samples with very low detections, only monitoring wells with total halocarbon concentrations greater than 0.005 mg/L were included in the analysis.

Figure 21 was prepared to graphically illustrate the results of the degradation product analysis provided in Table 7-5. The degradation product analysis suggests two PCE plumes, as depicted on Figure 21. The primary distinguishing characteristic between the two plumes is the percentage of PCE versus the percentage of PCE degradation products in the groundwater. PCE comprises greater than 60 percent of the total halocarbons in the northern plume, whereas PCE comprises less than 40 percent of the

Table 7-5: Average PCE, TCE, DCE, and VC Percentages

total dissolved halocarbons in the southern plume. As illustrated in Figure 21, the northern plume migrates in a northerly direction and the southern plume migrates in a southwesterly direction. The two plumes may actually represent commingled plumes from multiple release events; however, they have sufficiently different groundwater geochemistry and contaminant characteristics to allow simplification to two PCE plumes for discussion purposes.

The two plumes illustrated in Figure 21 are similar to, but not exactly equal to, the upper and lower plumes discussed throughout this RI report (definitions in Section 7.3). The upper and lower plumes discussed throughout this RI report (definitions in Section 7.3) are separated by the groundwater divide (Figure 21, Figure 14). Monitoring wells located approximately on top of the divide (MW-30, MW-24, and MW-29) are not clearly part of either the upper or the lower plume. By contrast, the two plumes illustrated in Figure 21, which are based on average PCE percentage, have slightly different boundaries. In Figure 21, MW-24 and MW-21 are part of the southerly plume, and MW-11, MW-13, and MW-14 are not part of either plume. MW-11, MW-13, and MW-14 have the PCE percentages similar to the northerly plume but are located physically closer to the southerly plume. They are not depicted as belonging to either plume, because it is interpreted that localized geochemical conditions have caused a lower degradation rate than would otherwise be expected in these monitoring wells.

7.6.4 Intrinsic Remediation Conclusions

The conclusions drawn from the intrinsic remediation analysis are illustrated in Figure 21. In simplified terms, the site appears to contain two PCE plumes: one in the northern area of the site (located adjacent to and north of the former dry cleaning building), and one in the southern area of the site (located to the south of the former dry cleaning building). The groundwater geochemistry and the presence of PCE degradation products both indicate that biodegradation is occurring in the lower (southern) PCE plume. Conversely, groundwater geochemistry and the lack of PCE degradation products indicate that biodegradation is not occurring in the upper (northern) PCE plume.

Overall, site groundwater geochemistry indicated the occurrence of anaerobic degradation in the area outlined in purple on Figure 21. Figure 21 also provides average DO and methane concentrations for all of the site monitoring wells. The area outlined in orange on Figure 21 is characterized by relatively low DO content, relatively high methane concentrations, and relatively high PCE daughter product concentrations.

The remaining intrinsic remediation parameter data also support the conclusions presented in Figure 21. The general groundwater geochemistry is favorable for reductive dechlorination (e.g, anaerobic degradation) in the lower plume. Ferrous iron and sulfate results generally support the conclusion that anaerobic degradation is occurring in the lower plume but not in the upper plume. TOC data indicate adequate TOC concentrations to support biological activity in the lower plume but not the upper plume. Presumably, naturally-derived organic carbon sources within the aquifer act as the carbon substrate for the reductive dechlorination reaction to occur. The data for the lower plume demonstrate Type 2-plume behavior as discussed in Section 7.6.1. The data for the upper plume do not indicate significant reductive dechlorination.

7.7 RIVER TERRACE PCE SPILL VOLUME ANALYSIS

The spill volume was evaluated to estimate the rate PCE is attenuating from the site, the length of time remediation will be required, and to assess the likelihood that significant source areas remain on RTRVP property. This information is needed to estimate the amount of time that will be required to operate a treatment system and the amount of contaminant that discharges into the river.

The complete spill volume analysis is contained in Appendix N. Only the introduction and results of the analysis are presented in this section. The spill volume analysis was performed using the following procedure:

- Media and the physical state of PCE were identified and presented in Section M.2.
- The likely volume of spill was assessed based on expected waste generation rates for a small dry cleaner, field reports, and literature information. Development of the volume estimate from literature sources and reports is contained in Section M.3 Volume Calculation Based on Literature Review.
- The volume of contamination volatilized, migrated off RTRVP property, excavated, and still present (based on site data) was calculated. This volume is termed the "accounted volume" in this evaluation. Development of the volume estimate based on site data is contained in Section M.4 Volume Calculation Based on Site Data. Contaminant migration and attenuation rates were developed for use in Section M.4 from site data and field tests presented in the remedial investigation.
- In Section M.5, Comparison of Volume Estimates with Spill Age, the possible spill volumes assessed in Section M.3 are compared to the "accounted volume" and attenuation rate data developed in Section M.4. A mass balance approach was used to determine what combinations of spill age and volume were credible given site data, attenuation rates, and literature information. Each mass balance calculation resulted in an "unaccounted volume" for each spill age and volume scenario. This "unaccounted volume" was considered potentially still present on site and the probable primary source of continued contamination in ground water. Only certain combinations of spill volume and age were deemed credible after completion of the mass balances. The mass balance identified several factors that further limited the possible ages and volumes of spills:
    - A range of attenuation rate was calculated from site data. Based on this attenuation rate, a predictable range of contaminant volume migrates off RTRVP property each year. Multiplying the attenuation rate by an assumed number of years results in the minimum spill size necessary for any age of spill. This calculation also indicates the maximum possible spill size for any age of spill.
    - Small dry cleaners generate predictable amounts of waste each year. Multiplying the years of operation by the expected waste generation rate results in a maximum spill volume at any particular time. The more years of operation, the more spill volume possible.
    - To be considered a credible scenario the "accounted volume" needed to be less than the assumed total spill volume.
    - Site data strongly suggests that significant (gallons) of PCE and its breakdown products migrate off RTRVP property each year; consequently, scenarios with very small spill volumes or very old spills are unlikely because these scenarios do not contain sufficient volume of PCE to sustain the off RTRVP migration currently ongoing.
- Probable attenuation time frame was estimated in Section M.6. For credible scenarios, the amount of time the site will continue to significantly impact water quality was estimated by computing the difference between the likely spill volume and the "accounted volume" and dividing this sum by the calculated attenuation rate.

The results of the analysis indicated that the likely spill sizes ranged from 500 to 1,000 gallons and most likely spill date was in the 1980's. Significant concentrations of groundwater contamination are predicted to emanate from the lower area for approximately 15 years. A predicted length of time for contamination to continue to emanate from the upper area could not be assessed; however, the length of time could exceed 15 years because the attenuation rate from the upper area is slower than the lower area.

Important to understanding this analysis is the basic premise that the distribution and degradation of PCE could have been caused by several different spills of differing magnitude over time.

7.8 RI SUMMARY AND CONCLUSIONS

Results of the OASIS/Bristol 1999 remedial investigation activities further defined site conditions and refined the estimated extent of impact at the RTRVP site. Key highlights are summarized below.

RI General Conclusions

- Groundwater flow and aquifer conditions at the RTRVP site were better defined. Groundwater at the site was categorized into three different water-bearing zones: a deep confined aquifer, a hypothesized shallow semi-confined water-bearing zone, and an upper unconfined aquifer. The upper unconfined aquifer is contained in sandy fill material across most of the site and in alluvium along the Kenai River. The depth to groundwater in the upper unconfined aquifer is a minimum of approximately 8 feet bgs under the former dry cleaner building. This depth is inferred from the depth to groundwater measured in monitoring wells MW-16 and MW-23 (approximately 18 feet bgs), and the fact that the bottom floor of the former dry cleaner building is approximately 10 feet bgs of MW-16 and MW-23 (e.g., the building is built into a slope).
- An analysis of common anions and cations from selected site monitoring wells supports the conceptual model that at least three water bearing zones are present at the site: the water table aquifer, the confined artesian aquifer, and at least one semi-confined water bearing zone beneath the water table aquifer.
- A ridge in the glacial till underlying the unconfined aquifer is interpreted to run in approximately a northwest-southeast orientation across the site, from MW-30 to MW-24 south of the former dry cleaner building.
- A groundwater divide (or ridge) is interpreted to generally follow the till ridge across the site. The divide separates site groundwater contamination into two plumes. The groundwater divide was initially interpreted as a mound centered on MW-24. Installation of monitoring well MW-29 clarified that it is a ridge following the till ridge.
- Soil samples collected in 1999 did not document any additional areas of soil contamination exceeding ACLs for RTRVP.
- The distribution of PCE soil and groundwater contamination at the site is complex. The 1997 grid soil sampling and excavation perimeter samples indicate that multiple source areas are likely. Soil gas and groundwater contamination near the former dry cleaner building indicate another source area, although minimal soil investigation work in the area has revealed no significant soil contamination.
- Based on an understanding of free-phase PCE migration and the heterogeneous nature of site geology, a widespread area of PCE soil contamination would be

expected. As literature research indicates, even small-scale heterogeneities in site characteristics have a significant effect on the migration of PCE. The hydrogeology of this site is very complex, with both small and large-scale heterogenities (layers of significantly different soil types and man-made preferential flow paths, such as the storm sewer corridor).

- An analysis of potential spill volume and age was performed for the site. Based on this analysis, a minimum remediation time frame of 15 years is a reasonable estimate of the amount of time needed for PCE at the site to attenuate to the RAOs, assuming the majority of PCE was spilled in the 1980s and will migrate through the lower area.

Conclusions Regarding the Lower Plume (south of the groundwater divide)

Impact of the RTRVP site on the Kenai River

- A comparison of Kenai River stage information and water levels recorded in a monitoring well adjacent to the river (MW-20) during the period between September 1, 1999 and January 5, 1999 indicates that the river is in direct communication with groundwater.
- PCE and its degradation products have been detected in Kenai River sediments adjacent to the site at levels above sediment quality benchmarks for PCE, TCE, and cis-1,2-DCE.
- The presence of PCE and its degradation products (TCE and cis-1,2-DCE) in the Kenai River water column (E&E, 1999b) downslope from the lower plume (e.g., between MW-6 and MW-8) is consistent with groundwater monitoring data indicating contaminants have entered the river via contaminated groundwater migration.
- Results of additional monitoring well installations and sampling along the Kenai River bank further defined the levels and extent of groundwater impact in the alluvial water table aquifer adjacent to the Kenai River. The following conclusions are based on samples collected from a line of sentry wells located approximately 20 feet upslope from the Kenai River, extending from the middle of the Sterling Highway Bridge approximately 175 feet eastward across the RTRVP site.
    - In RI sampling between July 1999 and January 2000, PCE was detected at levels above its ACL (840 µg/L) in samples from two sentry wells (MW-6 at 980 µg/L in October 1999 and MW-10 at 970 µg/L in December 1999). These results indicate that PCE continues to leave the RTRVP property at concentrations above the ACL for PCE in groundwater.
    - In RI sampling between July 1999 and January 2000, VC was detected at levels above its ACL (2 µg/L) in samples from two sentry wells (MW-6 at 3 µg/L in July and 3.7 µg/L in October 1999, and MW-20 at 7.6 µg/L in July and 3.3 µg/L in October 1999). These results indicate that VC continues to leave the RTRVP property at concentrations above the ACL for VC in groundwater.
    - In RI sampling between July 1999 and January 2000, PCE was detected above its WQS (5 µg/L) in all samples collected from all monitoring wells adjacent to the Kenai River from the middle of the Sterling Highway Bridge approximately 175 feet eastward to MW-5 (e.g., MW-13, MW-12, MW-8, MW-20, MW-6, MW-7, and MW-5). In addition, PCE degradation products TCE and cis-1,2-DCE were detected in many samples from these monitoring wells at levels above their WQSs.

- Based on the distribution of PCE concentrations in samples from the line of sentry wells (MW-5 to MW-27), the central part of the PCE plume appears to be intercepted by the sentry wells. Detections of contaminants in MW-27 have been below ACLs and WQSs; therefore, MW-27 is interpreted to represent a reasonable western limit of groundwater contamination for the site. Detections of contaminants in MW-5 have been below ACLs but above WQSs; therefore, MW-5 is not interpreted to represent the eastern limit of groundwater contamination. However, there is no reason to believe that contaminant concentrations will increase significantly east of MW-5.

Evaluation of Lower Contaminant Plume Characteristics

- Between MW-5 and MW-12 (approximately 120 lineal feet), approximately 4.5 gpm of water contaminated by PCE and its degradation products is migrating off-RTRVP property into the Kenai River. This estimate is based on hydraulic gradients measured in July and November 1999, and does not consider riverbank icing effects observed in December 1999.
- Approximate groundwater and PCE travel times from MW-26 to the Kenai River ranged between almost 1 year and almost 5 years, respectively, based on measured hydraulic gradients and calculated hydraulic conductivities (slug test results). The approximate groundwater and PCE travel times ranged from MW-9 to the Kenai River were 17 days and 85 days, respectively. The dissolved PCE transport velocity is slower than groundwater velocity due to the adsorption of PCE onto organic carbon in the aquifer material.
- Natural attenuation appears to be occurring in site groundwater from approximately MW-4A toward the river. Natural attenuation is indicated by favorable geochemical results and the presence of PCE degradation products. On a molar basis, the sum of PCE degradation product concentrations exceeds the PCE concentrations in monitoring wells MW-4A, MW-6, MW-8, MW-9, MW-10, and MW-19), which indicates that significant PCE degradation has occurred. However, contaminant levels in these monitoring wells remain high (i.e., above the ACLs for PCE in groundwater in some of these wells and above the WQSs for PCE in all of these wells), so natural attenuation is not degrading the PCE or its degradation products sufficiently to meet WQSs at the OHW line and protect the Kenai River in the near future.
- Visual trend analyses of PCE concentrations and total PCE degradation product concentrations were performed for nine monitoring wells across the lower contaminant plume. Three of the nine characteristic monitoring wells exhibit an apparent trend of increasing PCE concentrations over time (MW-9, MW-11, and MW-13). One monitoring well exhibits an apparent trend of decreasing PCE concentrations over time (MW-4A), although PCE concentrations detected in that monitoring well remain considerably above the site ACL for PCE in groundwater. No visible trend is apparent in the concentrations in the remaining monitoring wells (MW-6, MW-8, MW-10, MW-14, and MW-19).

Conclusions Regarding the Upper Plume (north of the former dry cleaner building)

- Immediately north of the groundwater ridge (near the former dry cleaner building), groundwater flows in a generally northerly direction toward the Sterling Highway. However, evidence exists that this northerly flow direction may be only a local phenomenon (i.e., MW-32 has a higher water table elevation than MW-23, and MW-22 has a higher water table elevation than MW-25).
- A plume of PCE-contaminated groundwater extends from MW-16, located on the northwest side of the former dry cleaner building, northward toward MW-25 and the

Sterling Highway. Four groundwater samples were collected from MW-16 between July 1999 and January 2000; all results exceeded the ACL for PCE (840 µg/L). A minimum concentration of 1,000 µg/L was detected in July 1999, and a maximum concentration of 5,500 µg/L was detected in September 1999.

- The upgradient extent of the upper plume is unknown but is probably located under the former dry cleaner building. Soil samples collected from the monitoring well borings did not indicate significant soil contamination. A soil gas survey conducted along the northwest and northeast sides of the former dry cleaner building encountered relatively high PCE vapor concentrations spread relatively uniformly over the area of investigation. The soil gas survey did not conclusively indicate a source area, but rather indicated that the entire soil gas survey area is contaminated. The source of the soil gas vapors is assumed to be contaminated groundwater.
- The downgradient extent of the upper contaminant plume is not known, but it likely extends to the Sterling Highway. The lack of PCE detections in monitoring wells installed on the west side of the Sterling Highway (MW-28 and MW-34) indicates that the plume does not appear to extend beyond the Highway.
- The storm sewer backfill under the west side of the Sterling Highway is a hypothesized preferential flow path from the upper contaminant plume to the Kenai River, based on the contamination detected within the storm sewer system, the groundwater flow patterns, and the lack of contamination on the west side of the Sterling Highway. It is possible that other utility corridors or the Sterling Highway road bed are also preferential flow pathways.
- In RI sampling between September 1999 and January 2000, one of three groundwater sample results from MW-25, located approximately on the RTRVP property boundary, exceeded the ACL for PCE in groundwater (920 µg/L in September). This exceedence indicates that PCE has migrated off-RTRVP property at a level exceeding its ACL and well above the MCL of 5 µg/L.
- Significant natural attenuation does not appear to be occurring in the upper groundwater plume. Geochemical results were not favorable for natural attenuation, and very limited PCE degradation product concentrations have been detected in groundwater.
- Testing results further characterized conditions associated with the storm sewer and the sanitary sewer cross-connect. Water samples collected from the KRBO (located adjacent to the RTRVP) in 1997, 1998, and 1999 by E&E and Hart Crowser (1999) documented the presence of PCE impact above WQSs and MCLs in storm water/groundwater discharging from the storm sewer outfall at the Kenai River. Samples from MH-1 (located adjacent to the RTRVP) collected in 1997 and 1998 were also impacted by PCE levels above WQSs. RI investigation results did not clearly identify the cross-connect as a conduit for contaminant migration.
- Interpretive maps of the top of the till surface and the water table contours indicate that the storm sewer system likely drains site groundwater north of the groundwater divide. The till surface was apparently excavated for the storm sewer system installation, and the backfill surrounding the storm sewer and the storm sewer itself provide a preferential flow pathway for groundwater.
- Between MW-21 and MW-22 in the upper plume area (approximately 115 lineal feet) approximately 2.4 gpm of water contaminated by PCE is migrating off-property. This estimate is based on hydraulic gradients measured in July through December 1999 and is not expected to have a significant seasonal variability.

- Approximate groundwater and PCE travel times from MW-16 to the storm sewer backfill ranged between 40 days and 200 days, respectively, estimated from measured hydraulic gradients and calculated hydraulic conductivities (slug test results). The approximate groundwater and PCE travel times from MW-25 to the storm sewer ranged between 22 days and 110 days, respectively. The dissolved PCE transport velocity is slower than groundwater velocity due to the adsorption of PCE onto organic carbon in the aquifer material.

8 SITE RISK SCREENING

Risk screening involves identifying chemicals of concern (COCs), evaluating the fate and transport of these chemicals at the site, developing conceptual site models (CSMs) to assess completed exposure pathways to receptors, and assessing site risks. Most of these elements were addressed by previous RTRVP investigations and evaluations. In particular, site risks were evaluated in the ACL development report (E&E, 1997c). This section of the RI/FS serves to reiterate the important risk-screening steps detailed in other reports, identify remaining data gaps, and present an updated CSM to summarize the human health and ecological risks posed by the site. Detailed human health and ecological risk assessments in accordance with CERCLA and current ADEC 18 AAC 75 regulations have not been performed for the RTRVP site.

8.1 COCS AND FATE & TRANSPORT

As discussed previously, COCs for the RTRVP site include PCE and its breakdown products TCE, cis- and trans-1,2-DCE, 1,1-DCE, and vinyl chloride.

A fate and transport analysis for site COCs is presented in Section 7 of this report. More detailed discussion of the impact of PCE on Kenai River surface water and sediments is presented here in Section 8.

8.2 CONCEPTUAL SITE MODEL

A CSM illustrates the conceptual understanding of the sources of contamination at the site, release and transport mechanisms, potential exposure pathways and routes, and receptors. The human health and ecological CSMs for the RTRVP site were originally created by E&E (1997c) for use in developing the ADEC-approved (ACLs) for site soil and groundwater (soil ACLs are presented in Table 3-3, and groundwater ACLs are presented in Table 3-2). These risk-based cleanup levels were calculated using site- and chemical-specific exposure and toxicity data, and equations obtained from USEPA's *Soil Screening Guidance* (USEPA, 1996a).

The E&E report addressed the risks to human health associated with most of the exposure pathways to contaminated soil and groundwater at the site; however, a recent review of the risk analysis identified several data gaps. These data gaps are summarized in the following bullets and are addressed in Section 8.3.

- The risks associated with the inhalation of contaminant vapors emanating from subsurface soils or groundwater and migrating through the former dry cleaner building subflooring into indoor spaces were not evaluated.
- Human health and ecological risks associated with contaminated surface water and sediment adjacent to the site (i.e., the Kenai River and its shoreline) were not fully addressed.
- Potential bioaccumulation and food-chain transfer of COCs were not evaluated.

Based on the most recent site data and the previously identified data gaps, the CSMs were modified and are presented here as a single figure (Figure 22). The modified CSM recognizes two distinct conceptual site zones at the RTRVP site: the Commercial/Residential Zone and the Aquatic Habitat Zone (see Figure 23 for the locations of these zones). In general, ecological risk is limited to the Aquatic Zone, because the development and use of the Commercial/Residential Zone preclude its use as significant ecological habitat. Human health risk is evaluated for both zones. The characteristics of the two zones are described in the following sections.

8.2.1 Commercial/Residential Zone

The Commercial/Residential Zone includes the RV Park, Laundromat (former dry cleaner), area residences, and the rest of the site down to the boardwalk (Figure 23). This zone is considered to have exposure scenarios different from the adjacent aquatic habitat for the following reasons:

- Access to the Commercial/Residential Zone via the road system provides for continued human activity that inhibits revegetation and wildlife activity; and
- The habitat structure and function, as well as the potential future uses of the Commercial/Residential Zone contrast greatly to adjacent aquatic habitat.

Potential human receptors within the Commercial/Residential Zone at the RTRVP site include residents, recreational and subsistence users accessing the Kenai River, and site maintenance workers (Figure 22). Off-property human receptors include year-round residents on adjacent properties. The resulting exposure scenarios are listed below:

- Residential Scenario (year-round residents at the RTRVP trailer park and properties adjacent to the RTRVP site);
- Industrial Exposure Scenario (short-term visits by maintenance workers at RTRVP); and
- Recreational & Subsistence Exposure Scenario (short-term visits by fishermen, campers, & other users of RTRVP).

These scenarios were evaluated for the following exposure pathways[1]:

- Inhalation of volatile chemicals inside the former dry cleaner building (volatilizing from groundwater and subsurface soil contamination);
- Incidental ingestion of groundwater-borne COCs during future site work at RTRVP (Industrial Exposure Scenario only);
- Dermal contact with groundwater-borne COCs during future site work at RTRVP (Industrial Exposure Scenario only); and
- Ingestion of and dermal contact with soil-borne COCs.

As indicated in Figure 22, the other potential exposure scenarios for the Commercial/Residential Zone were considered incomplete. Groundwater ingestion is not considered to be a completed exposure pathway, except for site workers who may incidentally ingest contaminated groundwater, for the following reasons:

  - Drinking water wells presently operational at the RTRVP property and at nearby properties are completed in the uncontaminated, confined aquifer.
  - Use of the site groundwater ACLs is based on the successful implementation and enforcement of site institutional controls prohibiting site work which may penetrate the aquitard confining the lower drinking-water aquifer (ADEC letter, dated August 27, 1997).

Visitors to the Commercial/Residential Zone may be exposed directly to contaminants in soil via incidental ingestion. Humans may also physically contact or inhale soil (i.e., fugitive dust) containing COCs; however, USEPA research indicates that soil ingestion

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[1] Surface water and sediment, as distinct media, are not present within the Commercial Zone.

risk values are adequately protective for dermal contact[2] and inhalation exposures to fugitive dusts of semivolatile organic compounds (USEPA, 1996a). Therefore, these pathways were eliminated.

8.2.2 Aquatic Habitat Zone

The Aquatic Zone includes the Kenai River and its shoreline for both the RTRVP site and adjacent off-property sites (Figure 23). Contamination from soil and groundwater in the Commercial/Residential Zone may impact surface water and sediment in Aquatic Zone via runoff, infiltration, and groundwater flow. Although windblown particles may transport COCs from the Commercial/Residential Zone, this pathway is considered negligible compared to surface and groundwater transport.

Significant ecological habitat is found adjacent to the RTRVP site along the Kenai River. The river and its shoreline are habitat for numerous mammals, birds, fish, and plant life (E&E, 1997c). Exposure scenarios for animals and plants in the Aquatic Habitat Zone are presented on Figure 22 and include:

- Uptake of COCs from surface water and sediment by fish and aquatic plants;
- Ingestion of and dermal contact with water-borne COCs by wildlife; and
- Ingestion of and dermal contact with sediment-borne COCs by wildlife.

Human use of the Kenai River is largely limited to fishing and other recreational activities. Consequently, the exposure scenario for humans is limited to the Recreational & Subsistence Use Scenario (short-term visits by fishermen, etc.).

This scenario was evaluated for the following exposure pathways:

- Ingestion of and dermal contact with surface water and
- Incidental ingestion of and dermal contact with sediment-borne COCs.

Because the Kenai River is an energetic environment, the vapor-phase fraction of any contaminant will quickly volatilize, leaving only dissolved constituents. For this reason, inhalation of airborne vapors volatilizing from contaminated surface water is not a complete exposure pathway. Inhalation of airborne vapors emanating from contaminated sediment is probably not a significant pathway because contaminants will tend to dissolve into water and only volatilize after significant dispersion within the water column.

Bioaccumulation and food chain transfer of COCs from plants and prey species living in aquatic habitats to species in higher trophic levels, including humans, are usually not considered significant if the COCs have log K_{OW} values >3.5 (USEPA, 1991). All of the RTRVP COCs have K_{OW} values of less than 2.7. This issue is discussed further in Section 8.3.5.

8.3 RISK-SCREENING DATA GAPS

8.3.1 Inhalation of Vapors in Indoor Air

The Johnson-Ettinger Model (Environmental Quality Management, Inc., 1991) for Subsurface Vapor Intrusion into Buildings was used to estimate the risk posed by indoor air exposure to groundwater and subsurface soil contaminated with PCE. This model is

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[2] Pentachlorophenol, which is not a COC for the RTRVP site, is the only chemical for which the USEPA Soil Screening Guidance directly considers dermal exposure (USEPA, 1996a).

a screening tool approved by the USEPA to estimate the transport of contaminant vapors emanating from subsurface soils or groundwater into indoor spaces located directly above or in close proximity to the source of contamination. The Johnson-Ettinger model is a one-dimensional analytical solution to convective and diffusive vapor transport into indoor spaces and provides an estimated attenuation coefficient that relates the vapor concentration in the indoor space to the vapor concentration at the source of contamination. Inputs to the model include chemical properties of the contaminant, saturated and unsaturated zone soil properties, and structural properties of the building. Model inputs specific to the RTRVP site are presented below.

- The Laundromat (former dry cleaner building) was used as the representative building for the model. Building dimensions were estimated from plan view maps of the site. Default values were used for hypothetical floor-wall seam cracks.
- Assuming that the representative building may be used for future residential purposes, the residential exposure scenario values presented in ADEC's *Guidance on Cleanup Standards, Equations, and Input Parameters* (1998) were used as model inputs.
- Depth to groundwater was assumed to be 6.5 feet below the building slab. Sand was used as the representative soil type to estimate vapor permeability. Within the model, sand has the highest value of hydraulic conductivity, resulting in conservatively large exposure concentrations. Default values were used for other soil properties such as bulk density and porosity.
- The concentrations of PCE used to calculate risk from vapor intrusion into indoor air were 13,000 µg/L for groundwater and 1.02 mg/kg for subsurface soil. The concentrations represent the 95 percent upper confidence level (UCL) of all groundwater PCE detections in the lower area and 150 soil sampling points. The development of the 95 percent UCLs for the RTRVP site is presented in Appendix M. The use of these values added to the conservatism of the calculated risk-based indoor air exposure concentrations.

The carcinogenic risks posed by residential exposure to PCE vapors entering the basement of the building from groundwater and subsurface soil were estimated at 4.6 X 10^{-5} and 5.2 X 10^{-6}, respectively. When risk screening indicates potential risk greater than 1 X 10^{-6}, additional evaluation of potential exposure is usually warranted. The basis for this finding is 40CFR300.430(e)(2)(I)(A)(2), cited here for convenience: "Acceptable exposure levels are generally concentration levels that represent an excess upper bound lifetime cancer risk to an individual of between 10^{-4} and 10^{-6} using information on the relationship between dose and response. The 10^{-6} risk level shall be used as the point of departure for determining remediation goals for alternatives when ARARs are not available or are not sufficiently protective because of the presence of multiple contaminants at a site or multiple pathways of exposure." The modeling spreadsheets are provided in Appendix N.

8.3.2 Risks Associated with Exposure to Surface Water & Sediments

As indicated in Figure 22, there are completed human health and ecological exposure pathways to Kenai River water column and sediments. Surface water and sediment quality, as they relate to risks attributed to the RTRVP site, are discussed in the following sections.

8.3.3 Surface Water Quality

The substantive ADEC requirements that relate to surface water quality are set forth in 18 AAC 70 Water Quality Standards. The standards set by this chapter apply to all surface-water bodies in the State of Alaska, including the Kenai River. They specify the degree of degradation that may not be exceeded in a surface-water body as a result of human actions. The water quality standards are largely set by the antidegradation policy in 18 AAC 70.015, the water quality criteria (WQC) in 18 AAC 70.020 (b), and the limits in 18 AAC 70.030. ADEC water-quality criteria for contaminants of concern at the River Terrace site are presented in Table 8-1.

Table 8-1: ADEC Water Quality Criteria for Site COCs

Contaminant of Concern	ADEC WQC (μg/L)
PCE	5
TCE	5
cis-1,2-DCE	70
trans-1,2-DCE	100
1,1-DCE	7
Vinyl Chloride	2

In 1997 and 1999, PCE, TCE, and cis-1,2-DCE were detected in the KRBO, which discharges directly into the Kenai River adjacent to the RTRVP site, at concentrations exceeding ADEC WQC (see Figure 23; E&E, 1997a and 1999b). In 1999, PCE, TCE, cis-1,2-DCE, and trans-1,2-DCE were detected in four of five samples [99RTRVP001SW (SW-1), 002SW (SW-1), 003SW (SW-2), and 004SW (SW-3)] collected from the Kenai River water column (see Figure 23). Of these samples, only PCE was detected at concentrations exceeding acceptable water-quality criteria (ADEC, 1998b). The surface water results were presented in Table 7-2, and the stormwater results were presented in Table 7-1. These data indicate that receptors in the Aquatic Habitat Zone may be deleteriously affected by River Terrace contaminants entering the Kenai River. The quality of pore water in RTRVP and adjacent property sediments is discussed in Section 8.3.4.2.

There is additional evidence that water quality in the lower Kenai River has been compromised. Reports prepared in 1992 and 1994 by the Alaska Department of Fish and Game compared the relative abundance of sensitive macroinvertebrates at several locations in the Kenai River (Litchfield & Kyle, 1992; Milner & Gabrielson, 1994). The studies found that the abundance of sensitive species (mayflies, stoneflies, and caddisflies) decreased significantly from 1990 to 1993 (the duration of the studies) in the vicinity of Kenai River Miles 18 to 20, which is near the Soldotna Bridge and the RTRVP site at River Mile 21. These species are intolerant of poor water quality and provide an indication of the overall health of the river. The reduction in numbers of these types of macroinvertebrates was determined statistically significant, but the cause of the reduction was not found (Litchfield & Kyle, 1992; Milner & Gabrielson, 1994). A 1999 report by Trout Unlimited also documented a decrease in the abundance of sensitive species (Randa, 1999).

8.3.4 Sediment Quality

Sediment quality benchmarks (SQBs) are necessary in addition to water quality criteria (WQC) because of the interaction between sediments and the water column. Chapman (Chapman 1989) provides five reasons for SQBs:

1. Various toxic contaminants found in only trace amounts in the water column accumulate in sediments to elevated levels.
2. Sediments serve as both a reservoir and a source of contaminants to the water column.
3. Sediments integrate contaminant concentrations over time, whereas water column contaminant concentrations are much more variable and dynamic.
4. Sediment contaminants in addition to water column contaminants affect benthic and other sediment-associated organisms.
5. Sediments are an integral part of the aquatic environment, providing spawning habitat, and feeding and rearing areas for many aquatic organisms.

To date, there are no regulatory sediment-quality criteria; however, ADEC 18 AAC 70.020 qualitatively addresses the effects of toxic organic substances such as PCE in sediments. With respect to these substances, the chapter states that “there may be no concentrations of toxic substances in water or shoreline or bottom sediments that, singly or in combination, cause, or reasonably can be expected to cause, toxic effects on aquatic life, except as authorized by this chapter.”

Jones et. al. (1997) developed freshwater SQBs for nonionic organics. The SQBs are based on the equilibrium partitioning approach. This approach calculates SQBs based on the water quality benchmarks (WQB), K_{oc} values for COCs, and the sediment TOC. Jones et. al. (1997) used the conventional chronic WQBs presented by Suter and Tsao (1996) to calculate the SQBs. The WQBs are the chronic National Ambient Water Quality Criteria (NAWQC), Tier II Secondary Chronic Values (SCVs), and the lowest chronic values for fish, daphnids, and nondaphnid invertebrates.

The chronic NAWQC are calculated by the EPA as the Final Acute Values (FAVs), which are the fifth percentile of the distribution of 48- to 96-hour LC50 values or equivalent EC50 values for each criterion chemical, divided by the geometric mean of quotients of at least three LC50 tests. If NAWQC were not available for a chemical, the Tier II method described in the EPA's *Proposed Water Quality Guidance for the Great Lakes System* was applied (EPA 1993a). The Tier II values are concentrations that would be expected to be higher than NAWQC in no more than 20% of cases.

Jones et. al. (1997) recommend comparing bulk sediment concentrations measured at a site to SQBs presented in their report, adjusted to the site-specific percent TOC. Two sets of SQBs are presented by Jones et. al. (1997): secondary chronic values and lowest chronic values for various species. The secondary chronic values are intended to be conservative predictors of effects, whereas concentrations that exceed lowest chronic value benchmarks indicate a risk of real effects.

8.3.4.1 River Terrace Site SQBs

Applicable secondary chronic SQBs for COCs detected during sampling by E&E in 1999 were adjusted for an average RTRVP sediment TOC of 0.36 percent (Jones et. al., 1997; E&E, 1999b). Jones et. al. also summarize other SQBs. USEPA’s Office of Solid Waste and Emergency Response (OSWER) has published Ecotox Thresholds (ETs) intended for screening contaminants at CERCLA sites. The preferred method for determining sediment ETs is to use USEPA-proposed sediment quality criteria (SQC) or ET SQB values. When SQCs are not available, ET SQBs are used. SQCs are not available for PCE and its degradation products; ET SQBs are available for PCE and TCE, but not DCE or vinyl chloride. The ETs are based on equilibrium partitioning;

therefore, they also are dependent upon TOC. SQBs and ET values for RTRVP COCs are summarized in Table 8-2.

Table 8-2: Sediment Quality Benchmarks and Ecotox Thresholds (μg/kg)

	PCE	TCE	Cis-1,2-DCE	Trans-1,2-DCE	Vinyl Chloride	1,1-DCE
SQBs[1]	410	220	400	400	NA	31
SQBs[2]	147	79	144	144	NA	11
ETs[3]	530	1600	NA	NA	NA	NA
ETs[4]	191	56	NA	NA	NA	NA

1 SQBs presented by Jones et. al., 1997; values normalized to 1% TOC
2 Jones et. al., 1997 SQBs normalized to average site-specific TOC of 0.36%
3 ET SQBs; values normalized to 1% TOC (USEPA, 1996b)
4 ET SQBs normalized to average site-specific TOC of 0.36%

PCE, TCE, and cis-1,2-DCE were detected in Kenai River sediments adjacent to the RTRVP site sampled in 1997 and 1999 (E&E, 1997a; E&E, 1999b). The sediment sampling results were presented in Table 7-3. Sediment sampling locations are presented on Figure 23. A comparison of the 1997 and 1999 sample results with the SQBs and ETs (adjusted for site-specific organic carbon), and the site-specific SQCs presented in Table 8-2 indicates that these criteria have been exceeded for PCE, TCE, and cis-1,2-DCE several times in Kenai River sediments. These exceedences are summarized below in Table 8-3. One sample collected at SD-5 on May 19, 1999 also contained 1,1-DCE at 330 μg/kg, a concentration exceeding the SQB and SQC (E&E, 1999b).

Table 8-3: Summary of RTRVP Site Sediment Samples Exceeding SQBs and ETs.

Location	Date Sampled	Cis-1,2-DCE (μg/kg)	Criteria Exceeded (μg/kg)	TCE (μg/kg)	Criteria Exceeded (μg/kg)	PCE (μg/kg)	Criteria Exceeded (μg/kg)
SD-5	5/19/99	460	A				
SD-6	5/7/97	450	A	110	A	400	A,B
SD-6	5/19/99	460	A			340	A,B
SD-6	5/22/99	180	A				
SD-7	5/19/99	340	A				
SD-7	5/22/99	180	A				
SD-8	5/7/97	670	A	170	A,B	510	A,B

A Jones et. al., 1997 SQBs normalized to average site-specific TOC of 0.36%
B ET SQBs normalized to average site-specific TOC of 0.36%

SQBs and ETs are helpful in determining whether contaminants warrant further assessment or are at a level that requires no further attention. If a chemical concentration or the reported detection limit exceeds a proposed lower benchmark, further analysis is needed to determine the hazards posed by that chemical. If, however, the chemical concentration falls below the lower benchmark value, the chemical may be eliminated from further study (Jones et. al., 1997). The sediment exceedences at the RTRVP clearly indicate that further assessment is warranted.

PCE and other organic contaminants tend to accumulate in sediments. The following subsections discuss this tendency and its possible impacts on Kenai River sediments near the RTRVP site.

8.3.4.2 Sorption in the Sediment Environment

A sediment environment can be characterized by three distinct matrices: overlying surface water, sediments, and pore water (water in between and surrounding sediment soil particles). The sediment can contain a variety of particle types from sand and silt to decaying plant material. Sediments generally have a fairly high organic content, which causes them to accumulate organic compounds.

High organic content soils act as natural carbon filters for organic compounds via a variety of physical and chemical interactions. The process through which these soils accumulate organic compounds has been reported by several researchers, but all the interactions involved are not completely understood. The overall process is described by Kadlec and Knight (1996): "Wetland soils have a high trapping efficiency for a variety of chemical constituents; they are retained within the hydrated soil matrix by forces ranging from chemical bonding to physical dissolution within the water of hydration. The combined phenomena are referred to as sorption" (Kadlec and Knight, 1996).

Kinetics favor adsorption over desorption, and organic contamination will tend to accumulate in organic carbon in the sediment. This property is the basis for activated carbon water treatment systems. Once all of the carbon adsorption sites are occupied, contamination will simply pass through the sediment. This phenomenon is termed breakthrough when applied to activated carbon treatment systems. Adsorbed contamination may slowly leach out of the sediment, degrade, or be taken up by organisms in contact with the sediment.

The overall effect of the sorption properties of sediments is to increase the concentration of contamination in pore water and sediment over time. Furthermore, measurement of contaminant concentrations in surface water is not a reliable method to assess water quality in sediment pore water. This phenomenon is explored in *Fundamentals of Aquatic Toxicology* (Rand, 1995):

> Conditions may vary considerably between the overlying water and the pore water as a result of the interaction between particulate matter and water. Because sediment is the site of chemical reactions such as sulfate reduction as well as mineral precipitation and dissolution, which can change ion ratios, the natural composition of chemicals in pore water may be quite different from that of the overlying water. Furthermore, the movement of particle-sorbed foreign chemicals to the sediment is an additional source of chemicals for pore water and is the main route for clearance of these chemicals from the water column. Consequently, concentrations of both natural and foreign chemicals in pore water may be many times higher than in the overlying water.

Furthermore, water quality standards are not reliable indicators of sediment toxicity because organisms that reside in sediments are exposed to different concentrations of contaminants from those in the water column. Chemicals in sediment may be present at higher concentrations and for longer time periods than chemicals dissolved in the surface water (Jones et. al., 1997).

8.3.4.3 Impact of PCE on Kenai River Sediments at the River Terrace Site

The maximum mass of PCE sorbed to organic carbon in Kenai River sediments adjacent to the RTRVP site can be estimated using the results of an analysis by Farrell, Hauck, and Jones (1999). They investigated TCE adsorption to water-saturated microporous adsorbents and found that the concentration of TCE correlated closely with the fraction of organic carbon content (Farrell, Hauk, and Jones, 1999). TCE has similar adsorption properties to PCE, and the two compounds would be expected to have similar organic carbon adsorption tendencies.

For example, if groundwater leaving the RTRVP site contains PCE at the level of the site ACL (840 µg/L) when it encounters Kenai River sediments, the sediments could theoretically retain up to 161 milligrams of PCE per kilogram of sediment. This concentration is calculated assuming that PCE adsorption approximately follows the Freundlich adsorption isotherm (Rand, 1995). The Freundlich isotherm for PCE in water predicts 46.1 milligrams of PCE would be adsorbed to 1 gram of organic carbon, given a PCE concentration of 840 µg/L in the water. The average TOC content of three samples collected from sediments in the Kenai River in 1999 was approximately 0.36 percent (E&E, 1999b). Based on the Freundlich isotherm estimates and the average TOC content, the Kenai River sediments can theoretically retain up to 161 mg/kg PCE, which greatly exceeds the SQB (adjusted for site TOC) of 147 µg/kg (see Table 8-2). Appendix O contains the Freundlich Isotherm calculation.

The highest measured PCE concentration in Kenai River sediments was detected in 1997 at 0.510 mg/kg, over two orders of magnitude below the theoretical holding capacity of the sediment (assuming 840 µg/L PCE concentration in water). Consequently, if groundwater is leaving the site at 840 µg/L, the concentration of PCE in river sediments will probably increase over time, unless the PCE concentration in groundwater is reduced significantly.

For comparison purposes, the Freundlich isotherm was also used to estimate the sorbed equilibrium concentration of PCE predicted for Kenai River sediments based on groundwater leaving the RTRVP site at the water quality standard of 5 µg/L (PCE). In this scenario, a value of 2.6 milligrams of PCE per gram of organic carbon or approximately 9 mg/kg PCE for the RTRVP sediments is the theoretical maximum concentration of PCE that could accumulate in Kenai River sediments. The implication of this calculation is that sediments exposed only to the water quality standard could also eventually exceed the sediment benchmark criteria for PCE, although at a much lower concentration than would be possible based on water at 840 µg/L. This theoretical finding is supported by researchers: “Mounting evidence exists of environmental degradation in areas where USEPA water quality criteria (WQC) are not exceeded, yet organisms in or near sediments are adversely affected (Chapman, 1989)” (Rand, 1995).

In the field, the theoretical maximum sorbed concentration is probably never reached. Many adsorption sites are isolated and unavailable and others are used up by other organics. In addition, the sorbed contamination may degrade over time. Nonetheless, the estimates presented above demonstrate that the Kenai River sediments have the capacity to accumulate contamination at concentrations that are orders of magnitude greater than the sediment detections in 1997 and 1999 and the sediment benchmarks.

The preceding analysis is equally applicable to TCE and cis-1,2-DCE, the other COCs detected in Kenai River sediments. Similar conclusions would be expected from this expanded analysis for the other COCs.

8.3.5 Bioaccumulation and Food-Chain Transfer

Bioaccumulation is the net accumulation of a chemical by an organism as a result of uptake from all routes of exposure (Suter, 1993). Bioaccumulation is of concern both for its possible effect on the organism and for the contamination of higher trophic levels, including humans, which may occur. Food chain transfer refers to the movement of chemicals through the food chain to higher trophic levels by herbivory and predation.

Risks from dietary exposure to chemicals with a log K_{OW} <3.5 is usually not considered significant (USEPA, 1991). The log K_{OW} values for the RTRVP chemicals of concern are presented in Table 8-4 (USEPA, 1996a). Since the log K_{OW} values of these chemicals are below 3.5, bioaccumulation and food chain transfer for these COCs are not considered significant.

Table 8-4: Log K_{ow} Values for Site COCs

Compound	Log K_{ow}
PCE	2.67
TCE	2.71
cis-1,2-DCE	1.86
trans-1,2-DCE	2.07
1,1-DCE	2.13
Vinyl Chloride	1.5

8.4 RISK SUMMARY

This preliminary risk evaluation indicates that the RTRVP site provides potential hazards to human and ecological receptors. The RTRVP CSM (Figure 22) illustrates the completed exposure pathways to contamination on the RTRVP site. Potential hazards identified in this preliminary risk evaluation are listed below:

- ACLs established for the site were derived from risk-based studies performed by E&E (E&E, 1997c); exceedences of these ACLs in Commercial Zone groundwater and soil indicate potential hazards to human and ecological receptors (Figure 23; E&E, 1997c).
- Vapor modeling, based on soil and groundwater PCE detections near the former dry cleaner building, indicates potential risk to building users. The carcinogenic risks posed by residential exposure to PCE vapors entering the basement of the building from groundwater and subsurface soil were estimated at 4.6×10^{-5} and 5.2×10^{-6}, respectively. When risk screening indicates potential risk greater than 1×10^{-6}, additional evaluation of potential exposure is usually warranted.
- COC detections in sediment in the Aquatic Zone (Section 8.3 of this report) indicate potentially deleterious ecological effects. Concentrations of PCE, TCE, and cis-1,2-DCE have been detected in sediments adjacent to the RTRVP site at levels exceeding SQBs, ETs, and SQCs. These exceedences indicate that further assessment is warranted at the RTRVP site.
- COC detections in surface water in the Aquatic Zone (Section 8.3 of this report) indicate potentially deleterious ecological effects. PCE has been detected in the KRBO, which discharges directly into the Kenai River, at levels exceeding its WQS. Furthermore, PCE has been detected in samples from the Kenai River water column, although at levels below its WQS.

- There is additional evidence that water quality in the lower Kenai River has been compromised. Reports prepared in 1992 and 1994 by the Alaska Department of Fish and Game found that the abundance of sensitive species of macroinvertebrates decreased significantly from 1990 to 1993 (the duration of the studies) in the Kenai River near the RTRVP site, but the cause of the decrease was not determined. These species are intolerant of poor water quality and provide an indication of the overall health of the river.

A detailed evaluation of the carcinogenic and noncarcinogenic effects of site COCs, including a cumulative risk analysis, on human and ecological receptors has not been performed for the RTRVP site. This preliminary risk evaluation indicates that further risk-based assessment may be needed at the RTRVP site.

PART 3 FEASIBILITY STUDY

9 REMEDIAL ACTION OBJECTIVES

Remedial action objectives (RAOs) are the goals that the remedial action alternatives proposed in the feasibility study are designed to achieve (EPA, 1988). RAOs include objectives that are applicable to all sites, in addition to objectives specific to the River Terrace site. Specific RAOs must ensure compliance with 18 AAC 75 and other applicable state and federal regulations hereafter referred to as Applicable or Relevant and Appropriate Requirements (ARARs), and protection of human health and the environment. Applicable regulations strictly apply to a site. Relevant and appropriate requirements may not strictly apply but need to be adhered to. In general, ARARs are the regulatory requirements the site must meet. Protection of human health and the environment is normally determined by comparing contamination levels to clean up levels specified in 18 AAC 75 or performing a site specific risk assessment.

9.1 GENERAL REMEDIAL ACTION OBJECTIVES

Overall goals for the remedial action are defined by both state and federal regulation. The main applicable state regulations are contained in the State of Alaska Oil and Hazardous Substances Pollution Control Regulations (18 AAC 75). Other state regulations which may impact development of remedial action objectives are State of Alaska Water Quality Criteria (18 AAC 70), State of Alaska Drinking Water Standards (18 AAC 80), and potentially Alaska Department of Fish and Game Statutes (AS Title 16) and regulation (e.g. 5 AAC 95).

The federal regulations that apply to the site are the Comprehensive Environmental Response, Compensation and Liability Act (CERCLA) (42 USC §§ 9601-9675) and the National Contingency Plan (NCP) which is a part of CERCLA, the Resource Conservation and Recovery Act (RCRA) (42 USC §§ 6901-6992k), and the Clean Water Act(CWA) (33 USC §§ 1251-1387). CERCLA is not directly applicable to the state's remedial investigation or cleanup of the site; however, CERCLA defines the necessary content of an RI/FS and is not inconsistent with state regulations. The overall goals listed below apply to CERCLA sites in general, are consistent with state law, and are appropriate for the River Terrace site.

- Protect human health by reducing the risk from the potential exposures, including cumulative risk posed by exposure to multiple contaminants in groundwater. Cleanup levels for individual contaminants may not be protective when multiple contaminants are present.
- Protect environmental receptors.
- Restore contaminated media for present and future land use where practicable and within a reasonable time frame given the particular circumstances of the site.
- Protect uncontaminated media by preventing releases from sources.
- Use permanent solutions and alternative treatment technologies to the maximum extent practicable.
- Meet state and federal ARARs.

Any hazardous waste generated at the site must be managed and disposed in accordance with RCRA and discharges to surface water must be in compliance with

CWA. In particular, contaminated groundwater is potentially F listed and/or characteristic hazardous waste. Hazardous waste groundwater can be treated by the generator on-site in a tank or container per 40 CFR 262.34. Treated listed waste must also meet other RCRA requirements prior to disposal back on the land, e.g. the facility will need to obtain a contained in decision from the EPA and will need to meet the land disposal restrictions for listed waste.

Specific RAOs derived from these goals are identified in Table 9-1. These RAOs were developed considering state requirements, site specific conditions, and the goals listed above.

Table 9-1. Specific Remedial Action Objectives

- Reduce the site's human health carcinogenic risk to less than 10^{-5} and non-carcinogenic risk to less than a hazard index of 1. This objective includes the requirement to reduce the site's cumulative risk to less than 10^{-5} and non-carcinogenic hazard index to 1 or less across all exposure pathways as stipulated in 18 AAC 75.325(g).
- Prevent contaminants above their respective Maximum Contaminant Levels (MCLs) from reaching potential drinking water sources.
- Maintain the integrity of wetlands areas.
- The remedial alternative must conform with the Kenai River Management Plan.
- Prevent contaminants above their respective Water Quality Criteria (WQC) and MCLs from reaching the Kenai River.
- Identify remedies that will have minimal impact on surface land use.
- Attain Alternate Cleanup Levels (ACLs) in all soil and groundwater as specified in an August 1997 letter to the site owner. Additional requirements, specified herein, apply to any off River Terrace property location where the ACLs will be used as the cleanup level.
- Attain 18 AAC 75 Method 2 soil cleanup levels and Table C groundwater cleanup levels off River Terrace property.

9.2 POTENTIAL CONTAMINANT-SPECIFIC ARARS

Potential chemical-specific ARARs are typically health-based or risk-based numerical values, or methodologies that result in numerical values when applied to a specific site. These values establish the acceptable amount or concentration of a chemical that may be found in, or discharged to, an environmental medium (soil, sediment, groundwater, surface water, or air).

The potential contaminant-specific ARARs for River Terrace include:

- Resource Conservation and Recovery Act (RCRA) (40 CFR 261 and 18 AAC 62; Listed and Characteristic Wastes),
- State of Alaska Soil and Groundwater Cleanup Levels (18 AAC 75 Articles 3 and 9, as amended through January 22, 1999),
- State of Alaska Water Quality Criteria (18 AAC 70, as amended through March 1, 1998),
- State of Alaska Drinking Water Standards (18 AAC 80, as amended through November 24, 1994 and public review draft of March 15, 1999),

- Clean Water Act (40 CFR 122), and
- National Toxics Rule (40 CFR 131.36)

As stated in Table 9-1, ACLs have been established for the River Terrace site; these ACLs are explained in Section 9.5 below.

9.3 POTENTIAL LOCATION-SPECIFIC ARARS

Potential location-specific ARARs are requirements that affect the management of hazardous constituents, or the units in which they are managed, due to the location of the unit(s). Potential location-specific ARARs might be triggered, for example, if groundwater remediation that required the construction of new surface wastewater treatment units was selected as a remedial action. Examples of sensitive locations for such units include wetlands, floodplains, historic areas, and wildlife refuges. Location specific ARARs will be determined in the Record of Decision (ROD) after the preferred alternative has been selected.

9.4 POTENTIAL ACTION-SPECIFIC ARARS

Potential action-specific ARARs are technology-based or activity-based requirements that may be triggered by the particular remedial action chosen for a site. Potential action-specific ARARs do not affect the selection of a remedial action but instead may pose restrictions on the methods by which a selected alternative may be achieved. Examples of potential action-specific ARARs include landfilling or transport of wastes off site subject to the Solid Waste Disposal Act and discharge of pollutants into surface waters subject to the Clean Water Act. Action specific ARARs will be determined in the ROD after the preferred alternative has been selected. All investigation derived waste and stockpiled soil must meet the conditions of the contained-in decision issued by the EPA[3].

9.5 REMEDIAL ACTION OBJECTIVES FOR RIVER TERRACE

RAOs are specific cleanup levels and related requirements to be met at the River Terrace site. In general, contaminated sites in Alaska are subject to soil and groundwater cleanup levels provided by the ADEC's Oil and Hazardous Substance Control Regulations (18 AAC 75 Articles 3 and 9). Groundwater cleanup levels provided in 18 AAC 75 generally reflect the EPA's drinking water MCLs, for those contaminants with established MCLs. Soil cleanup levels provided in 18 AAC 75 were established at levels considered protective of human health and the environment.

9.5.1 River Terrace Site Soil and Groundwater RAOs

For the River Terrace site soil and groundwater, ACLs were established for PCE, PCE degradation products, and petroleum hydrocarbons in a letter, dated August 27, 1997, from Lynn Tomich Kent of the ADEC to Mr. Gary Hinkle, the site owner. The letter stipulates that ACLs apply to all aquifers between surface grade and the lower drinking water aquifer, and the lower drinking water aquifer must meet drinking water standards on the RTRVP property. The August 27 letter also specifies that the ACLs were selected to be protective of growth and propagation of fish, shellfish, other aquatic life, and wildlife in freshwater as specified in 18 AAC 70.020(a) and (b).

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[3] Letter from Dave Bartus to R. Sundet and M. Schwenne, 2 March 2000.

The ACLs for soil and groundwater do not meet current ARARs, i.e. 18 AAC 75 (promulgated on January 22, 1999), but are still considered applicable because the ACLs were approved prior to promulgation of the current 18 AAC 75. The ACLs do not meet current ARARs because the ACLs are higher than the tabulated cleanup levels and a risk assessment or ACL development, in accordance with the regulation, has not been performed. The ACLs were established using a number of documents cited in the August 27 letter. In particular, risk assessment methodology was used in a document titled *Development of Alternative Cleanup Levels Report, River Terrace RV Park, Soldotna, Alaska* dated August 1997 (E&E 1997). This document does not meet the requirements for a CERCLA risk assessment or the current state regulations; however, it was considered adequate to establish ACLs in 1997. Current regulation provides for four methods to develop soil cleanup levels. General methodology of each method is listed below.

- Method one relies on a site score sheet and is only applicable for petroleum constituents.
- Method two relies on tabulated cleanup levels.
- Method three allows for calculation of site specific ACLs using standard formulas.
- Method four allows for development of risk based cleanup levels.

Both methods three and four require a public participation as specified in 18 AAC 75.325(j) before ACLs can be established.

As stipulated in the August 27, 1997 letter, all on RTRVP property soil and groundwater (above the lower drinking water aquifer) must meet the ACLs. Cleanup levels for contamination that has migrated off-property were not established in the August 27, 1997 letter. Consequently, the current 18 AAC 75 regulation applies off-property. Method two, tabulated cleanup levels, are applicable unless the responsible party opts to pursue an alternate cleanup level using methods three or four.

Water quality standards are applicable off-property because data indicates the site is hydraulically connected to the Kenai River and contamination has been found in the river and its sediments. Application of the ACLs off-property would require compliance with 18 AAC 75.345. Specifically, this regulation requires preparation of a risk assessment [18 AAC 75.345(b)(3)], demonstration that the surface water quality standards will be met [18 AAC 75.345(f)], monitoring [18 AAC 75 745(g), (h) and (i)], demonstration that the contamination will not affect a potential drinking water source [18 AAC 75.345 (b)(2)(A) & (B)], consent of each land owner affected [18 AAC 75.340(f)(2)], and seek public participation [18 AAC 75.325(j)].

If the responsible party decides to request application of the ACLs off-property, the practical feasibility of this action needs to be considered. Physical constraints of the site render application of the ACLs off-property in the upper area impractical. Specifically, contaminated water above the water quality standard discharges into the Kenai River at the storm sewer outfall. The bridge abutment and utilities in the vicinity are potential obstacles to the installation of an effective groundwater treatment system in this area. Consequently, treatment will need to occur before contamination reaches the storm sewer backfill, in effect requiring treatment to the water quality standard at the property boundary.

Currently, the primary off RTRVP property landowner affected is the Alaska Department of Transportation (ADOT). ADOT has verbally indicated they would not object to application of the ACLs in their right-of-way (Dan Breeden, personal communication with

Rich Sundet). Consequently, application of the ACLs off RTRVP property (on the ADOT property) is possible. Technical constraints, i.e. the bridge abutments and underground utilities, make application of the groundwater cleanup levels off property unlikely. Application of the soil alternate cleanup levels is more feasible if it can be demonstrated that the contamination does not threaten surface water or potential drinking water sources.

9.5.2 Kenai River RAOs

Site surface water (e.g., the Kenai River) must meet the Alaska Water Quality Criteria (18 AAC 70), which is the ARAR for surface water. The Kenai River RAOs are applicable to any soil or groundwater below the ordinary high water line (OHW) of the Kenai River, either on or off property or in the ADOT right-of-way. Water Quality Criteria become Water Quality Standards when applied to a site. The WQSs for PCE and its breakdown products, by application of 18 AAC 70.020(b) are the MCLs, if the MCL exists for a particular contaminant. The groundwater ACLs specified in the 27 August 97 letter are well above the WQSs applicable to the Kenai River. The basis for the Kenai River WQSs is delineated in the following discussion.

The Kenai River is currently protected for use classes (1)(A), (1)(B), and (1)(C), (18 AAC 70.050). Class (1)(A) includes the following fresh water uses:

(i) potential drinking water;

(ii) agriculture, irrigation, & stock watering;

(iii) aquaculture; and

(iv)industrial.

Class (1)(B) includes the following fresh water uses:

(i) water contact recreation; and

(ii) secondary recreation).

Class (1)(C) includes the growth and propogation of fish, shellfish, other aquatic life, and wildlife.

Three of the applicable water use categories for the Kenai River cite the MCLs as the appropriate water quality standard: (1)(A)(i) potential drinking water, (1)(A)(ii) where contact with a product destined for human consumption is present (e.g. fish), and (1)(B)(i) contact water recreation. Where MCLs are not available for a compound, the Gold Book (*EPA Quality Criteria for Water,* 1986) apply. Consequently, to comply with the surface water WQSs, groundwater seeping into surface water must meet the appropriate MCL, or Gold Book Value if an MCL is not available.

A mixing zone is not appropriate because the mixing zone regulations were developed for permitted point source discharges and the contamination at the RTRVP is not a point source discharge.

State regulation requires sentinel monitoring of groundwater hydrologically connected to surface water. 18 AAC 75.345(g) states, “If the groundwater point of compliance is established at or near a property boundary or if the groundwater is closely connected hydrologically to a surface waterbody, the department will, if the department determines that sentinel monitoring is necessary to ensure protection of human health, safety, or welfare, or the environment, require a responsible person to develop sentinel monitoring wells that monitor for any hazardous substances likely to migrate to the applicable point of compliance at concentrations that exceed the cleanup levels.” The EPA also recognizes general policies for assessing attainment of remedial action objectives. The general policy for establishing a point of compliance at surface water bodies is stated in the preamble to the NCP (Federal Register, Bol 55, No. 46, pp. 8713, March 8, 1990). “For surface waters, the selected levels should be attained at the point or points where the release enters the surface waters.” This policy clearly states that the point of entry, not the water body as a whole or part of it, is the compliance point.

If the Kenai River were reclassified such that it was no longer classified as (1)(A)(i) - Water Supply, drinking, culinary, and food processing the MCLs would still be the water quality standard because use classes (1)(A)(ii) - agriculture, irrigation, & stock watering and (1)(B)(i) – Contact Water Recreation would still apply.

Points of compliance must be established to accommodate the lack of a reliable means to monitor water quality at the exact groundwater/surface water interface. A point of compliance is a location hydraulically upgradient from the surface water body where standards applicable to the surface water body are applied to the groundwater. The August 27, 1997 letter established MW-5, MW-6 and MW-7 as the points of compliance at the River Terrace Site. Additional site data collected since the letter was issued indicate a larger groundwater plume is present than originally thought. Specifically, the plume extends under the bridge and along the Sterling Highway right-of-way. Additional points of compliance will be needed to adequately monitor the site for entry of contaminated groundwater into the Kenai River. A number of existing monitoring wells are currently installed that are suitable as points of compliance at the river terrace site. Currently installed monitoring wells suitable as sentry wells include MW-5, MW-6, MW-7, MW-8, MW-12, MW-13, MW-20 and MW-27. The actual points of compliance will be presented in a proposed plan and designated in the ROD. Additional monitoring wells may need to be installed to monitor the site efficiently. In particular, additional sentry wells will be needed at the DOT Right of Way boundary to ensure the Table C cleanup levels are met at the RTRVP property boundary.

9.5.3 Chemical-Specific RAO Summary for the River Terrace Site

Soil, groundwater, and surface water ACLs and ARAR levels are summarized in Table 8-2. As discussed above, the ARAR for surface water is 18 AAC 70 and 18 AAC 75.345(f) by reference to 18 AAC 70, and the ARAR for groundwater and soil cleanup levels is 18 AAC 75. Method two cleanup levels from 18 AAC 75, Table B1 and Table C are listed on Table 8-2 for comparison purposes. In addition, multiple contaminants are present so cumulative risk may need to be evaluated for ground water.

Some contaminant degradation is expected to occur as groundwater travels from the points of compliance to the river. Because of the proximity of the points of compliance, i.e. current sentry wells, to the river ordinary high water (OHW) line (the average distance from the sentry wells to the OHW line is approximately 15 feet), lateral dispersion is the primary mechanism for decreasing contaminant concentration. Using the Bioscreen model (Newell and McLeod, 1996) with site-specific organic carbon and groundwater characteristics, the expected degree of attenuation between the OHW line of the river and a point of compliance 15 feet upland was calculated. The calculated attenuation factor was three times. This factor is valid only if a single compliance point in the plume is at or above the modified cleanup level, e.g. if one sentry well is at 15 ug/L then all the remaining wells must be less than 15 ug/L. If the entire plume is at a concentration near the maximum allowed, then no effective dispersion will occur. Based on the model predictions, points of compliance concentrations not expected to result in groundwater entering the surface water in exceedence of the WQSs were calculated and are also presented in Table 9-2. Bioscreen model input parameters and modeling results are presented in Appendix N.

Table 9-2. Preliminary Remedial Action Objectives for River Terrace

							Remedial Action Goals for On River Terrace Property Water Table Aquifer		
Media	**Contaminants of Concern**	**Maximum Concentration**	**Maximum Concentration Location (Date)**	**Maximum Concentration Location (Date) 1999 sampling**	**Drinking Water MCLs/ 18 AAC 75 soil cleanup levels**	**Water Quality Standards 18 AAC 70**	**Point of Compliance Remediation Goal**	**Remediation Goal**	**Basis**
Groundwater (mg/L)	Benzene	0.00756	MW-4A (8/1/98)	0.0042 MW-9 (7/9/99)	0.005	0.005	NA	0.005	18 AAC 75
	PCE	5.5	MW-16 (9/2/99)	5.5 MW-16 (9/2/99)	0.005	0.005	NA	0.84	ACL[1]
	TCE	0.97	MW-6 (7/22/97)	0.86 MW-20 (7/8/99)	0.005	0.005	NA	21.9	ACL[1]
	cis-1,2-DCE	4.6	MW-20 (7/8/99)	4.6 MW-20 (7/8/99)	0.07	0.07	NA	11.6	ACL[1]
	trans-1,2-DCE	0.43	MW-4A (10/20/98)	0.033 MW-4A, MW-27 (10/99)	0.1	0.1	NA	11.6	ACL[1]
	1,1-DCE	0.004	MW-4A, MW-9 (12/28/98)	0.0024 MW-20 (10/27/99)	0.007	0.007	NA	0.007	ACL[1]
	Vinyl Chloride	0.0076	MW-20 (7/8/99)	0.0076 MW-20 (7/8/99)	0.002	0.002	NA	0.002	ACL[1]
							Remedial Action Goals for Off River Terrace Property Water Table Aquifer		
Groundwater (mg/L)	Benzene	0.003	MW-8 (10/21/98)	0.00105 MW-8 (4/15/99)	0.005	0.005	NA	0.005	18 AAC 75
	PCE	0.96	MW-8 (10/21/98)	0.92 MW-25 (9/3/99)	0.005	0.005	NA	0.005	18 AAC 75
	TCE	0.35	MW-8 (10/21/98)	0.21 MW-8 (7/10/99)	0.005	0.005	NA	0.005	18 AAC 75
	cis-1,2-DCE	1.9	MW-8 (10/21/98)	1.01 MW-8 (4/15/99)	0.07	0.07	NA	0.07	18 AAC 75
	trans-1,2-DCEne	0.021	MW-8 (10/21/98)	0.0084 MW-12 (10/26/99)	0.1	0.1	NA	0.1	18 AAC 75
	1,1-DCE	0.001	MW-8 (10/21/98)	ND	0.007	0.007	NA	0.007	18 AAC 75
	Vinyl Chloride	ND	---	ND	0.002	0.002	NA	0.002	18 AAC 75

Table 9-2: Preliminary Remedial Action Objectives for River Terrace (continued)

							Remedial Action Goals for Surface Water		
Media	**Contaminants of Concern**	**Maximum Concentration**	**Maximum Concentration Location (Date)**	**Maximum Concentration Location (Date) 1999 sampling**	**Drinking Water MCLs/ 18 AAC 75 soil cleanup levels**	**Water Quality Standards 18 AAC 70**	**Point of Compliance Remediation Goal**	**Remediation Goal**	**Basis**
Surface Water(mg/L)	Benzene	ND	---	ND	0.005	0.005	0.005	0.005	AWQC
	PCE at KRBO	0.023	SW-1 (5/97)	0.017 SW-1 (5/19/99)	0.005	0.005	0.005	0.005	AWQC
	PCE in River	0.0025	SW-3 (5/19/99)	0.0025 SW-3 (5/19/99)	0.005	0.005	0.015	0.005	AWQC
	TCE	0.0019	SW-3 (5/19/99)	0.0019 SW-3 (5/19/99)	0.005	0.005	0.015	0.005	AWQC
	cis-1,2-DCE	0.015	SW-3 (5/19/99)	0.014 SW-3 (5/19/99)	0.07	0.07	0.021	0.07	AWQC
	trans-1,2-DCE	0.000081	SW-3 (5/19/99)	0.000081 SW-3 (5/19/99)	0.1	0.1	0.3	0.1	AWQC
	1,1-DCE	ND	---	ND	0.007	0.007	0.021	0.007	AWQC
	Vinyl Chloride	ND	---	ND	0.002	0.002	0.002	0.002	AWQC

							Remedial Action Goals for Soil On River Terrace Property Post 1997 and 1998 Removal		
Media	**Contaminants of Concern**	**Maximum Concentration**	**Maximum Concentration Location (Date)**	**Maximum Concentration Location (Date) 1999 sampling**	**Drinking Water MCLs/ 18 AAC 75 soil cleanup levels**	**Water Quality Standards 18 AAC 70**	**Table B1 Cleanup Level (based on mig. to GW < 40” precip.)**	**Remediation Goal**	**Basis**
Soil **(mg/kg)**	PCE	20	SB-3A (6/29/98)	2.00 MW-19 (6/25/99)	NA	NA	0.03	11.5	ACL[1]
	TCE	0.88	SB-0/B1 (6/22/99)	0.88 SB-0/B1 (6/22/99)	NA	NA	0.027	300	ACL[1]
	cis-1,2-DCE	0.62	MW-20 (6/25/99)	0.62 MW-20 (6/25/99)	NA	NA	0.2	72.1	ACL[1]
	trans-1,2-DCE	0.007	MW-19 (6/25/99)	0.007 MW-19 (6/25/99)	NA	NA	0.4	87.3	ACL[1]
	1,1-DCE	ND	---	ND	NA	NA	0.03	7.1	ACL[1]
	Vinyl Chloride	ND	---	ND	NA	NA	0.009	2.1	ACL[1]
	Benzene	ND	---	ND	NA	NA	0.02	0.1	ACL[1]

Table 9-2: Preliminary Remedial Action Objectives for River Terrace (continued)

							Remedial Action Goals for Soil Off River Terrace Property		
Media	**Contaminants of Concern**	**Maximum Concentration**	**Maximum Concentration Location (Date)**	**Maximum Concentration Location (Date) 1999 sampling**	**Drinking Water MCLs/ 18 AAC 75 soil cleanup levels**	**Water Quality Standards 18 AAC 70**	**Table B1 Cleanup Level (based on mig. to GW < 40" precip.)**	**Remediation Goal**	**Basis**
Soil **(mg/kg)**	PCE	0.380	MW-12 (10/12/98)	0.190 MW-25 (9/1/99)	NA	NA	0.03	0.03	18 AAC 75
	TCE	0.009	MW-25 (9/1/99)	0.009 MW-25 (9/1/99)	NA	NA	0.027	0.027	18 AAC 75
	cis-1,2-DCE	0.006	MW-25 (9/1/99)	0.006 MW-25 (9/1/99)	NA	NA	0.2	0.2	18 AAC 75
	trans-1,2-DCE	ND	---	ND	NA	NA	0.4	0.4	18 AAC 75
	1,1-DCE	ND	---	ND	NA	NA	0.03	0.03	18 AAC 75
	Vinyl Chloride	ND	---	ND	NA	NA	0.009	0.009	18 AAC 75
	Benzene	ND	---	ND	NA	NA	0.02	0.02	18 AAC 75

[1] ACLs established in letter dated 8/27/97 from Lynn Tomich Kent (ADEC) to Mr. Gary Hinkle (site owner).

Definitions:

ADWS - Alaska Drinking Water Standards (18 AAC 80)
RG - Remediation goal
AWQC – Alaska Water Quality Criteria (18 AAC 70)
bgs – below ground surface
ND – not detected
Off-property wells include: MW-1A, MW-1C(deep), MW-2, MW-8, MW-12, MW-13, MW-14, MW-18, MW-22, MW-25, MW-27, MW-28, MW-30, MW-31, MW-34, SB-1(deep), and SB-1(shallow)
NA – MCLs and Water Quality Standards do not apply to soil

10 IDENTIFICATION AND SCREENING OF TECHNOLOGIES

In this step of the FS, potentially applicable remedial technologies are identified for the River Terrace site and then screened to reduce these to a reasonable number for development of remedial alternatives (USEPA, 1988a). Remedial technologies were identified by drawing upon a variety of sources, including the USEPA/United States Air Force's Remediation Technologies Screening Matrix Reference Guide (USEPA, 1993a). During the screening step, process option and entire technology types are selected for consideration based on technical implementability for the specific contaminants present at River Terrace. Those technologies that are not appropriate for the site conditions or would present significant technical obstacles to successful implementation at the site are screened out.

As discussed in Section 1.3.2 of this report, the primary contaminants of concern at the River Terrace site are PCE and its degradation products dissolved in groundwater. Soil contamination is also present at the site. However, since previous site remediation efforts have included excavation and treatment of the contaminated soil and with the exception of a few known “hot spots” that are above the ACLs or potentially contaminated locations that may be above the ACLs (e.g., under the former dry cleaner building), the remaining soil concentrations are below the site specific ACLs for soil. Site conditions and PCE DNAPL migration characteristics make it difficult and prohibitively expensive to identify all unknown locations of PCE contamination that may be above the ACLs for soil, with the exception of the area beneath the former dry cleaner building. Therefore, remedial technologies specifically designed for soil remediation were not considered. It is understood that treatment of the remaining soil contamination, or source of the groundwater contamination, may be beneficial to achievement of the overall RAOs. Therefore, at least one of the remedial action alternatives includes treatment of the source area contamination for both the upper and lower contaminant plume locations.

If the RAOs used in this feasibility study are revised, such as by a reclassification of use of the Kenai River, then the selected remedial alternative may need to be revised or a different remedial alternative selected to comply with the new RAOs.

Geological and hydrogeological site conditions combined with the physical, chemical, and degradation properties of PCE and its degradation products determine which remedial processes may be applicable for addressing remaining PCE contamination at the site. Section 4 of this report provides a detailed discussion on the physical and chemical characteristics of PCE and Sections 6 and 7 provide details on the geological and hydrogeological site conditions.

10.1 FEASIBILITY STUDY PROCEDURES

The procedures used in preparing this feasibility study follow the USEPA *Guidance for Conducting Remedial Investigations and Feasibility Studies Under CERCLA* (USEPA, 1988). Even though River Terrace is not a CERCLA site the application of this guidance is generally appropriate and is the standard followed throughout the industry. The costing of remedial alternatives was conducted in accordance with the USEPA *Remedy Cost Estimating Procedures Manual: A Guide to Developing and Documenting Remedial Alternative Cost Estimates During the Feasibility Study* (USEPA, 1998c).

10.2 INITIAL SCREENING OF TECHNOLOGIES

The identification and screening of remedial technologies is a two-step process that considers the environmental conditions at River Terrace and the RAOs.

Per USEPA guidance, an initial screening is performed to reduce the number of technology types based on technical implementability (USEPA, 1988a). The initial screening step focuses on general response actions, which are types of remedial actions that could potentially satisfy the RAOs. General response actions that are not applicable to the contamination types found at the site are screened out of further consideration. Next, potential remedial technologies and specific process options for each technology are identified that could be implemented for each potentially applicable remedial alternative.

The second screening step focuses on choosing several remedial technologies for detailed discussion and comparison. For this FS, seven remedial technologies were evaluated in detail. As required by the National Contingency Plan (NCP), the “no action” alternative is one of the remedial technologies evaluated [NCP 300.430(g)(6)]. In general, the remedial technologies considered include a relatively low cost/low technology, a moderate cost/moderate technology, and a relatively high cost/high technology alternative.

A list of the seven remedial technologies is provided in Section 10.4, and in Section 11, the seven remedial technologies are evaluated in detail for both the upper and lower contaminant plumes at the River Terrace site.

10.3 GENERAL RESPONSE ACTIONS

General response actions for River Terrace were identified for the upper unconfined aquifer located between the surface grade and the lower artesian aquifer, which is presently being used as a drinking water supply. Six general response actions were identified for water: (1) no action, (2) intrinsic remediation or natural attenuation, (3) institutional controls, (4) containment, (5) collection/*ex situ* treatment/discharge, and (6) *in situ* treatment. These response actions are discussed below.

No Action. The "no action" alternative is used for comparison of other response actions (USEPA, 1988a). Although a "no action" alternative may include some type of environmental monitoring, actions taken to reduce the potential for exposure (e.g., site fencing and deed restriction) are not included as a component of this alternative.

Intrinsic Remediation. Intrinsic remediation is a general response action that does not involve an active remedial response. Intrinsic remediation is not a "technology" but is the documentation that natural processes are reducing contaminant concentrations to acceptable levels. The processes involved include dilution, volatilization, biodegradation, adsorption, and chemical reactions with subsurface materials.

Intrinsic remediation is not the same as "no action," although it may be perceived as such. Consideration of this option requires modeling and evaluation of contaminant degradation rates to determine feasibility, and special approvals, such as institutional controls, may be required. Sampling and analysis must be conducted throughout the process to confirm that attenuation of contaminants is proceeding at rates consistent with meeting cleanup objectives.

Institutional Actions. Institutional actions limit human exposure to the groundwater and surface water. Two types of institutional actions may be appropriate: limited actions (e.g., fencing, warning signs, etc.) or administrative controls (e.g., deed restrictions on groundwater use, excavations, etc.). Institutional actions are often used in combination with other general response actions.

Containment. Containment uses barriers to prevent or minimize the spread of contaminated water. Containment methods include capping and hydraulic barriers. Containment would not reduce the toxicity or volume of contamination; however, it could serve to limit contaminant mobility to the area within the barriers by reducing the contaminant's ability to migrate beyond the physical barriers. Containment is used as part of a remedial alternative that limits migration rather than eliminating contaminant concentrations.

Collection/*Ex situ* Treatment/Discharge. This response action consists of collecting contaminated water, either by extraction of groundwater or surface water collection. Collected water can be treated using a variety of biological, physical, or chemical methods prior to on site or off site discharge.

***In Situ* Treatment**. This response action consists of treating contaminants in place. Contaminated water may be treated *in situ* using biological, physical, or chemical methods. The technologies involved will often treat the soil in contact with the contaminated water as well.

10.4 REMEDIAL TECHNOLOGIES FOR GROUNDWATER

Each of the remedial technologies that are potentially applicable for addressing contaminated water at River Terrace are described in detail in the following sections, organized by general response actions. The general response actions that were determined to be inappropriate for use at River Terrace are also presented, along with the reason why they were determined to be inappropriate. Only remedial technologies applicable for addressing chlorinated solvents in groundwater and that are protective of human health and the environment and meet ARARs are appropriate for more detailed evaluation.

10.4.1 Intrinsic Remediation

Intrinsic remediation relies on the natural ability of the groundwater to lower contaminant concentrations primarily through biological processes, but also through physical and chemical processes, until cleanup levels are achieved. Intrinsic remediation generally is a long-term response action that continues until cleanup levels have been reached throughout the area of attainment. Conditions that potentially favor the use of intrinsic remediation occur when contaminants in groundwater are expected to attenuate to risk-based or ARAR levels within the property boundaries or prior to reaching potential receptors. Additionally, the contaminants shall not currently pose or be expected to pose an unacceptable risk to human health or the environment.

10.4.2 Institutional Action

Institutional actions include administrative controls such as access restrictions and alternative water supplies. Access restrictions provide legal controls and limitations for current and proposed development or usage. A governmental institution is usually responsible for administering and enforcing these controls. Other types of institutional actions that are often termed limited actions (e.g., fencing or warning signs) are not appropriate for groundwater contamination.

Access Restrictions. Deed restrictions may be used to prevent the construction and use of wells in areas where groundwater is contaminated. Deed restrictions are specifically required for the RSRVP property by the ADEC (letter from Lynn T. Kent, dated August 27, 1997). This deed restriction is intended to prevent the use of all groundwater aquifers between the surface grade and the lower artesian aquifer as a drinking water supply at the RSRVP property for as long as the water levels exceed the drinking water MCLs. Examples of other deed restrictions may include fencing, capping, and deed restrictions that preclude excavating or building on the site.

Alternative Water Supply. Alternative water supplies are used to reduce exposure to contaminated water by supplying an alternative, safe water supply for the intended use. Alternative water supplies are typically supplied by connecting homes and businesses to nearby municipal water systems, supplying bottled water, providing new "clean" wells completed in alternative aquifers, or by construction of a water treatment system to remove contaminants.

Data collected by Hart Crowser and OASIS Environmental (see Appendix D) shows that the water supply aquifer at River Terrace has not been affected by the shallow groundwater contamination; therefore, an alternative water supply is not currently necessary at this site.

10.4.3 Containment

Containment of groundwater and surface water contamination is used to reduce or prevent risk by preventing transport of contaminants to human or environmental receptors. Because containment technologies cannot indefinitely "hold back" the contaminants, the containment mechanism will eventually be bypassed unless combined with another treatment technology. Containment technologies are primarily used to control where plumes migrate; therefore allowing protection of environmentally sensitive areas (i.e., natural wetlands) or uncontaminated water supply wells. They are also used for containment of contamination within property boundaries. Alternatively, containment technologies can be used to divert "clean" water around or away from contaminated zones.

In general, containment technologies are not applicable for use at River Terrace. Groundwater is the contaminated media of concern at this site. Therefore, any type of a barrier wall that would contain the groundwater would result in a “bathtub” effect that would eventually overflow and result in release of contaminants. Hydraulic barriers (e.g., pumping wells) could be used to contain the contaminated groundwater but would require pumping large quantities of contaminated groundwater that would then require *ex situ* treatment. The combination or hydraulic barriers and *ex situ* treatment is more correctly termed collection-and-treatment and is discussed under the next section on Collection.

For the reasons just described, containment alone is not considered a viable alternative for the River Terrace site. Therefore, detailed descriptions of groundwater containment technologies are not provided in this FS.

10.4.4 Collection

Collection of contaminated groundwater and surface water, followed by treatment and/or disposal, has historically been the most common method for remediation, e.g., pump-and-treat. Collection methods can be passive, relying on interceptor trenches and drains, or active, relying on pumps to extract the groundwater. Floating contaminants on the surface water can be collected using absorbent booms. Extraction technologies are used to recover contaminated groundwater. Active extraction depends on pumps to withdraw the groundwater from depths ranging from a few feet to several hundred feet. Passive extraction technologies can also be applied to shallow groundwater, where topographic conditions cause the water table to be expressed as springs or seeps.

Groundwater collection technologies by themselves are considered a form of groundwater containment, and are discussed in the section on containment. In general, groundwater collection technologies are combined with some form of *ex situ* treatment and discharge. This section discusses only the collection alternatives; potential treatment alternatives are discussed in the *Ex situ* Treatment sections.

When considering groundwater collection alternatives, where the groundwater is brought above grade for treatment or discharge, it is always important to consider the regulatory implications. Depending on the groundwater characteristics, such as contaminant concentration, the collected water may be regulated as a hazardous waste under RCRA and State of Alaska Hazardous Regulation requirements.

Numerous case studies have shown that groundwater extraction is not an effective means for remediating contaminated groundwater emanating from source areas containing residual soil contamination. The solubility of the residual phase and effect of sorption on soil matrices prevent the removal of contaminants within what is generally considered a reasonable time frame. Dense non-aqueous phase liquids (DNAPLs) and in particular PCE are slow to dissolve in groundwater (groundwater concentrations often represent only one percent of the compound's solubility) and thus, numerous pore volumes of "clean" water are required to pass through the DNAPL region to remove these contaminants. Thus, the main limitation of pump-and-treat technology is the long time that may be required to achieve an acceptable level of cleanup. Where residual phase components are present, pump-and-treat removal may need to be combined with other remedial alternatives (e.g., vacuum extraction) that better address residual phase contaminants; and/or hydraulic containment rather than cleanup may be the realistic remedial objective.

Another limitation of pump-and-treat or collection systems is due to the sorption of dissolved contaminants. For linear sorption, a distribution coefficient and retardation factor can be defined for defined for most chemicals. The retardation factor indicates the speed of a contaminant relative to the water velocity. For example, dissolved PCE was found to have a distribution coefficient of 0.2 mL/g in a porous medium with a bulk density of 1.65 g/cm3 and a porosity of 0.25. The resulting PCE velocity is approximately 40 percent of that for water flowing through the same porous media based on the retardation factor. This means that it would take 40 percent longer to reach a cleanup goal for PCE compared to the cleanup time for a nonsorbing compound.

For the reasons provided above groundwater collection (e.g. pump-and-treat) is not generally effective for contaminant removal and achievement of cleanup levels. However, groundwater collection is a suitable means for controlling contaminant migration and preventing it from migrating off property boundaries and from reaching sensitive areas or environments.

Subsurface Collection. Interceptor trenches, French drains, and extraction wells are the most common forms of subsurface collectors. These techniques were originally developed to de-water areas containing shallow water tables. An interceptor trench or French drain consists of a trench dug through the areas to be dewatered. The trench, typically 6 inches to a few feet wide, is filled with sewer rock or pea gravel, and may contain perforated drainpipe. The trench sides may be lined with geosynthetic cloth to prevent soil from filling the void spaces within the sewer rock or pea gravel. One or more collection points within the trench will contain a pump to remove the collected groundwater, allowing additional groundwater to enter the trench and flow to the collection point. Where topography allows, the drain may terminate above ground, allowing the water to gravity flow into a surface collector. Interceptor trenches and French drains are most applicable to shallow aquifers of limited thickness. The drain is usually installed down gradient of the leading edge of the contaminant plume to intercept the contamination.

Extraction wells containing a single pump are typically sized from 4 to 12 inches in diameter. Most groundwater extraction wells depend on submersible pumps to extract the water. However, where the water table is near the surface, aboveground pumps may be used. For most installations, including light non-aqueous phase liquids (LNAPLs), the well screen will extend from a few feet above the water table to some depth within the aquifer. For DNAPL type contaminants, the well screen may be located just above the first aquitard or natural barrier within DNAPL contaminant pools or at locations where the contamination is expected to occur. Extraction wells are most applicable to deeper aquifers and aquifers of greater saturated thickness.

Surface Collection. Surface water collection is used to intercept contaminated surface water, such as seeps and springs, as well as precipitation running off contaminated soil areas. Surface collectors can also be used to redirect clean surface water away from contaminated areas. Surface water collection technologies include earthen berms and ditches, and concrete trenches.

Surface water collection is not an appropriate technology for addressing dissolved-phase groundwater contamination at either the upper or lower contaminant plumes at River Terrace.

10.4.5 *In Situ* Treatment

In situ treatment of groundwater is a desirable approach to remediation because the processes avoid the additional steps of extraction and discharge. The *in situ* technology types presented here include biological, physical, chemical, and thermal treatments.

***In Situ* Biological Treatment**. *In situ* biological treatment of groundwater involves stimulating and enhancing biological growth in the contaminated zone to reduce the contaminant concentrations. Microorganisms that can use some or all of the contaminants as substrate will normally exist in a contaminated environment. Biological transformations can be either aerobic (use oxygen) or anaerobic. Stimulation of the microorganisms to increase their biological growth and degradation of contaminants is accomplished by providing the necessary environmental conditions and essential microbial nutrients. In some cases, the biological degradation is limited by low concentrations of inorganic nutrients (nitrogen and phosphorus) or microbial populations that are adapted to consume the particular contaminants present. Nutrient additions or seeding of the groundwater with appropriate microorganisms may be required to enhance biological activity.

An alternative method for *in situ* biological treatment involves the installation of a permeable reactive barrier or treatment wall, which consists of a porous substrate (e.g., gravel) placed into a trench dug through the saturated zone. The trench allows for better control of the subsurface environment. For instance, it may be designed to increase oxidation of the groundwater by allowing addition of air into the porous bed material or by adding an oxygen-releasing agent to the porous material. As groundwater passes through the trench, contaminants are degraded by the enhanced biological activity.

Aerobic: For aerobic treatment, the *in situ* treatment systems typically consist of oxygen enhancement by injection into the groundwater. The most common methods of introducing oxygen are forced air (air sparging) and addition of compound that releases oxygen. Two of the compounds used for oxygen release are hydrogen peroxide and a slow Oxygen Release Compound (ORC™) that is made by Regenesis. Alternatively, nitrate can be used to stimulate degradation in the absence of oxygen (anoxic); in this case nitrate serves as an alternative electron acceptor. A third type of *in situ* biological treatment involves stimulation of co-metabolic degradation of chlorinated hydrocarbons by injection of oxygen and methane (or some other suitable substrate).

It is generally believed that biodegradation of PCE is not viable under aerobic conditions. While co-metabolic degradation is possible, it is difficult to control and maintain the subsurface environment in a condition favorable for the biodegradation to occur. For these reasons aerobic biodegradation of PCE is considered technically impracticable. However, aerobic biodegradation of some of the degradation products of PCE, such as VC, do favor aerobic conditions and are considered a viable alternative, if required.

Anaerobic: *In situ* enhancement of aquifers to promote anaerobic degradation is a new technology. As discussed in Section 4, the only pathway for PCE biodegradation appears to be reductive dechlorination under anaerobic conditions. One method of promoting reductive dehalogenation involves the addition of lactic acid to the groundwater environment. The technology is similar to aerobic treatment in that the primary goal is to promote the environmental conditions necessary for biodegradation. The underlying assumption is that the necessary bacteria are already present in the environment.

Reductive dechlorination by anaerobic bioremediation is a multi-step process. First, the aquifer has to be driven to an anaerobic condition, if it is not already in that state. Obviously, this has to be achieved to support the growth and development of anaerobic microorganisms. To achieve this state, most of the oxygen and other electron acceptors such as nitrate and sulfate, must be consumed. Once anaerobic conditions are achieved, the lactic acid is biodegraded by anaerobic organisms (such as acetogens) to form pyruvic acid and then acetic acid. It is through this process that hydrogen is formed; two moles of hydrogen as H_2 are derived in the conversion of lactic acid to acetic acid. The resulting hydrogen can then be used by reductive dehalogenators that are capable of dechlorinating compounds such as PCE, TCE, and trichloroethane (TCA) as well as their daughter products. However, other competing microbial processes also consume hydrogen. The most common being methanogenesis – a methane generating reaction that involves the combination of carbon dioxide with hyrdrogen.

The reductive dechlorination of PCE may proceed all the way to ethene, however, each reaction becomes progressively more difficult to carry out; subsequently, the DCEs, particularly *cis*-DCE, and VC tend to accumulate in anaerobic environments.

The USEPA is currently evaluating this technology at one site. The test site procedures involve weekly injection of hundreds of gallons of sodium-lactate in wells surrounding a chlorinated solvent hot spot. Weekly injections are necessary to keep the aquifer anaerobic. Initial results indicate that the procedure is effectively remediating the “hot spot” (Personnel Communication, Lance Peterson, USEPA, Region 10). A potential concern with this technology is the ability to manage and control the size of the anaerobic zone created by the sodium lactate additions. If too much sodium-lactate is added to the lower contaminant plume at River Terrace, it could result in driving a portion of the Kenai River system anaerobic.

Regenesis Corporation produces a Hydrogen Release Compound (HRC™), which uses a proprietary substrate to slowly release lactic acid over a period of months. The time-release feature of the product alleviates the need for weekly injections of sodium-lactate. However, HRC™ still needs to be replenished on some time interval - the vendor claims that the product will last for several months between applications.

As discussed in Sections 5 and 7, chemical data collected at the River Terrace site indicate an anaerobic environment exists across much of the lower portion of the site and that reductive dechlorination is naturally occurring in this portion of the site. This is beneficial in that it reduces the amount of sodium-lactate required to drive the system anaerobic and thus, more of the sodium-lactate can be used to produce hydrogen to degrade the PCE.

The low permeability and heterogeneity of the till soils at the lower contaminant plume of the River Terrace site may inhibit successful application of this technology. Obtaining wide coverage throughout the contaminated water bearing zones will be difficult because there is no assurance that lactic acid could be preferentially introduced into the water bearing zones where contamination resides. Advective transport processes are negligible in low permeability soils where the dominant transport mechanism is diffusion. Under these conditions, the remediation process is slowed down considerably resulting in extended periods-of-time being required to complete the remediation process. Locations where residual DNAPLs are present will also extend the time frame for biodegradation to achieve acceptable cleanup levels.

Enhancing anaerobic treatment by sodium lactate or HRC addition is considered viable for both the upper and lower contaminant plumes at River Terrace, provided it can be effectively introduced into the areas of contamination and that no adverse impacts occur to the Kenai River (i.e., creating anaerobic conditions within the sediments or the river itself). It should be noted that this technology is still in the development stage and is not yet considered a full-scale technology. This will be discussed further in Section 11.

Co-metabolic: Co-metabolism is one form of secondary substrate transformation in which enzymes produced for primary substrate oxidation are capable of degrading the second substrate fortuitously, even though the secondary substrate does not afford sufficient energy to sustain the microbial population. The addition of methane or methanol supports methanotrophic activity, which has been demonstrated effective for the degradation of chlorinated solvents, such as VC and TCE, by co-metabolism. Toluene, propane, and butane have also been demonstrated to stimulate the co-metabolism of TCE. Co-metabolic technologies may be classified as long-term technologies that may take several years or decades to clean up a plume.

The U.S. Department of Energy (DOE) conducted an innovative technology study at their Savannah River Site to demonstrate the use of gaseous nutrient injection for *in situ* bioremediation of soil and groundwater contaminated with TCE and PCE. The study combined air sparging/gaseous nutrient injection with soil vapor extraction to produce a more efficient in situ remediation system. The gaseous nutrient injection included nitrogen, phosphorous, and 4 percent methane. Almost 17,000 lbs. of VOCs were removed or degraded over 384 days of operation. The vacuum extraction component was responsible for 12,100 lbs. of VOCs and the biological component degraded and mineralized an additional 4,838 lbs. of VOCs. The mass balance calculations indicate that bioremediation destroyed 40 percent more VOCs than if air sparging was used by itself.

This technology is still at the pilot scale of development and would require considerable treatability study efforts be conducted at the River Terrace site before it could be considered a fully viable alternative.

***In Situ* Physical Treatment**. *In situ* physical treatment processes in water typically rely on the differences in the physical characteristics between the water and dissolved contaminants. Volatility is the most common physical property used in groundwater treatment, where air bubbles are passed through the water in order to strip (remove) the more volatile organic contaminants. *In situ* air sparging has been used at numerous sites to remove volatile organics, including PCE and its degradation products, from groundwater and DNAPL contaminated zones. In the process, a number of air injection wells or horizontal headers are installed to inject air into the saturated zone. Contaminants are removed from the water by contact between the water and air bubbles. Increasing the air-to-water interface by designing the wells to inject small air bubbles is desirable. Contaminants stripped from the water will move into the vadose zone where they may be adsorbed by soils, emitted to the atmosphere, or collected by vapor extraction wells for discharge or further treatment. Air sparging effectiveness can be greatly reduced by site-specific soil conditions: limited contact area between the injected volume of air and surrounding water and unfavorable partitioning coefficients. Heating of the air prior to injection may improve removal efficiency. A potential disadvantage of air injection is that soluble metal ions may be converted to insoluble oxides that then precipitate within the sparging wells and void spaces in the porous media around the well. Air sparging can also be used to stimulate *in situ* aerobic biodegradation by increasing the oxygen content in the groundwater.

Air sparging is considered effective in soils having a hydraulic conductivity greater than 10^{-3} to 10^{-4} cm/s. A compound is considered strippable if its Henry's Law Constant is greater than 10^{-5} atm/m^3/mole. Henry's Law Constants for PCE and TCE are in the 10^{-2} atm/m^3/mole and thus have good strippability. Slug test data indicate hydraulic conductivity values in the range of 4.6×10^{-4} cm/sec in the till deposits (MW-4A), 1.9×10^{-2} in the alluvial deposits along Kenai River, and 1.0×10^{-2} (MW-24) in the near surface fill/alluvial deposits at the lower portion of the River Terrace site (Table 6-3). Slug test data indicate hydraulic conductivity values of 6.2×10^{-3} cm/sec (MW-16) to 1.6×10^{-1} cm/sec (MW-25) in the near surface water bearing deposits at the upper portion of the River Terrace site (Table 6-3). Therefore, air sparging is considered viable for treating the shallow unconfined aquifers in the alluvial deposits along the Kenai River, in the surficial sand and gravel deposits in the upper contaminant plume area, and in the fill deposits across various parts of the site. Air sparging is not considered viable for treatment within the silty-till deposits that underlay these surficial deposits.

***In Situ* Chemical Treatment**. Chemical treatment involves using chemical reactions to transform hazardous compounds into less toxic substances. Several chemical treatment processes are available but the most common are chemical oxidation and permeable reactive barriers.

Chemical oxidation can be accomplished by injecting chemicals such as potassium permanganate, ozone, or hydrogen peroxide. The oxidants tend to be nonspecific and may be expended in oxidizing organics other than the target compounds. The degree of oxidation will vary depending on the actual contaminants and various by-products formed. The unknown stability of the oxidants in the subsurface environment combined with the potential formation of undesirable by-products makes *in situ* oxidation difficult to control. Because of the large plume size and difficulty in controlling an *in situ* chemical oxidation process, this technology is eliminated from further consideration.

Permeable reactive barriers may include a variety of reactive media combinations, but zero-valent iron [Fe(0)] (more commonly referred to as metallic iron) is the most common reactive media used. Reactive iron barriers have the ability to reduce oxidized inorganic species and reductively dehalogenate chlorinated contaminants such as PCE and TCE. The reductive dehalogenation of these compounds occurs due to electron transfers from the iron to the halocarbon at the iron surface. This process results in the halogen ions (e.g., Cl^-) being replaced by hydrogen species, ultimately yielding ethene or ethane that can be mineralized via biodegradation (USEPA, 1998a).

Permeable reactive barriers may also be designed for plume control but with significant differences to the pump-and-treat technology that is commonly used for plume control. Pump-and-treat operations often require significant maintenance and energy expenditures. This effect is significant since the sources of DNAPL contamination, such as residual saturation, often cannot be easily located and may continue to generate a continuous plume of dissolved halocarbons for extended periods-of-time. However, permeable reactive barriers are *in situ* systems that operate in a passive manner, do not bring contaminants to the surface, and should operate for at least 5 years with minimal, if any, maintenance. After 5 years, the reactive media may become saturated and/or plugged with fine material and bacteria, and may require maintenance (removal of fouling materials). This is accomplished by either using augers to mix the reactive media or replacing the reactive media. Iron treatment walls have been installed at PCE contaminated sites that have continued to operate for more than 5 to 10 years without significant maintenance problems (USEPA, 1998a).

Permeable reactive barriers are a viable remedial alternative for use at both the upper and lower contaminant plumes at the River Terrace site and will be included in the detailed analysis.

***In Situ* Thermal Treatment**. Thermal destruction of groundwater contaminants *in situ* is typically not implementable because of the high cost of heating the groundwater. However, thermal processes can be used to increase the removal efficiency of volatile organics from the groundwater. A representative *in situ* thermal treatment process identified for this screening includes stripping of volatile organics with high temperature steam. Steam stripping was originally developed in the petroleum industry to enhance the extraction of crude oil as reservoir pressures were depleted, and for high viscosity oils. The process involves injecting steam into the saturated zone where it reduces the viscosity and increases the volatility of non-aqueous phase contaminants. The reduced viscosity allows contaminants to flow to groundwater extraction wells for removal and treatment. The volatilized contaminants are typically removed for subsequent treatment using soil vapor extraction wells.

Due to the relative ease of volatilizing and air stripping PCE from the soil and groundwater and because of the high costs of implementing and operating steam stripping, this technology was eliminated from further consideration. Soil vapor extraction and air sparging are considered more cost effective, while still maintaining most of the effectiveness added through thermal treatment enhancements.

10.4.6 *Ex situ* Treatment

Historically, a common approach for remediation of contaminated groundwater has been to extract the groundwater using an extraction well network, followed by *ex situ* treatment, i.e., pump-and-treat. The primary limitations of *ex situ* treatment are a result of limitations in contaminant removal through the groundwater extraction or collection process. The primary factors being low contaminant solubility, sorption and desorption processes (i.e., contaminant retardation), and heterogeneities in the subsurface environment that result in preferential flow patterns (i.e., diffusion limited contaminant removal). These processes were discussed previously in Section 10.3.4 on water collection. Another drawback to *ex situ* treatment is due to the potential for increased regulatory requirements. This may include Hazardous Waste considerations and permitting requirements for discharge of the treated water.

Most *ex situ* treatment*,* technologies were developed from water and wastewater treatment processes. The technologies use biological, physical, and chemical processes to concentrate, degrade, or chemically alter the contaminants, or transfer of the contaminants to a different carrying medium (such as water to air).

In general, *ex situ* groundwater treatment technologies are similar in nature to the *in situ* technologies already presented. For this reason, the technologies will not be repeated here. One of the primary differences for *ex situ* operations is that they allow for better control and engineering of the treatment process in comparison to *in situ* treatment.

10.4.7 Discharge

Water that has been treated may be discharged on or off the RTRVP property by a variety of methods depending on the treated water quality. Virtually all methods of discharge, even discharge to a publicly owned treatment works (POTW), will require obtaining some type of discharge permit (e.g., a state general permit, municipal permit, underground injection control permit, or an National Pollution Discharge Elimination System [NPDES] permit) and meeting substantive requirements prior to discharge. This includes the requirement to manage all generated wastewater in accordance with 40 CFR 262 regulations.

On Site Discharge. On site discharge to a local water body (e.g., Kenai River) or reinjection to the RTRVP property groundwater is typically a desirable option because the discharge returns the water to its natural position in the hydrologic cycle. This type of discharge helps reduce or prevent adverse environmental impacts associated with surface and groundwater interception or withdrawal, including impacts such as reducing stream flows or drying up natural wetlands. The majority of the groundwater flow at the River Terrace site naturally discharges into the Kenai River. Direct discharge of treated water to the river may be allowable depending on the treated water quality.

Injection wells can be used to return treated water to the subsurface. As an additional benefit, these wells may provide a hydraulic barrier along the downgradient edge of the contaminant plume to prevent further migration of the contaminants. The wells can also be used to flush contaminants from the soil by allowing reinjection upgradient of the plume, effectively increasing the groundwater flow through the contaminated area. For environmental applications, injection wells are typically completed in the vadose zone. Well sizes are typically larger than required for extraction or monitoring because sufficient soil area must be provided to allow the water to enter and percolate through the soil. Well sizes of twelve-inches in diameter or larger are common. The number and placement of wells will be dependent on the soil permeability and natural hydraulic gradient and on whether the wells will be used to provide hydraulic control of the contaminant plume. For highly permeable soil, a few large wells may be sufficient for discharge. In less permeable soils, a large number of wells may be required. For shallow groundwater, the injection wells may be replaced with trenches or infiltration basins.

Off Site Discharge. Off site discharge of treated water can be accomplished by transporting the water to a nearby stream or river via pipeline, transporting off site for injection into a deep well, or transporting to a POTW for disposal (USEPA, 1990b). Discharge of water into an off site river, such as the Kenai River, would require an NPDES permit. Deep well injection of treated water off site would require a suitable well that is permitted under the Underground Injection Control (UIC) permit. Modified oil exploration and production wells that are no longer in service have been utilized for discharge of wastewater at other sites. Extracted water that is a hazardous waste and can’t be discharged off site would require transport to an out-of-state RCRA facility for disposal at a substantial cost.

10.5 RESULTS OF TECHNOLOGY SCREENING

The technology types and process options identified for contaminated groundwater at the River Terrace site have been evaluated for technical implementability. Site conditions, such as contaminant types and site physical characteristics, were used to eliminate technology types and process options that cannot be technically implemented to meet RAOs. The rationale for eliminating or retaining technology types and process options was summarized in the previous sections by general response actions within each media type. The no action alternative was retained for all media to serve as a basis for comparison of technologies.

A second screening step was completed to choose eight remedial alternatives (including the "No Action" alternative) for detailed evaluation for each of the two-contaminant plumes. The development of these alternatives involved combining remedial technologies that were judged most applicable to the types of contamination at the River Terrace site. The eight remedial technologies that passed the screening steps are presented below in Table 10-1.

Table 10-1. River Terrace Remedial Alternatives

General Response Action	Remedial Technology
PCE and Degradation Products in Groundwater	
No Action	None
Intrinsic Remediation/Institutional Controls	Intrinsic Remediation, Groundwater Monitoring, and Access Restrictions
In situ Chemical Treatment	Permeable Reactive Barrier of zero-valent iron
In situ Physical Treatment	*In situ* Air Sparging
Containment with Treatment	Funnel and gate system with engineered in-well air stripping
Ex situ Treatment	Groundwater collection, tray air stripping, and discharge
In situ Biological Treatment	Reductive Anaerobic Biological *In situ* Treatment Technology (RABITT)
Contaminant Source Removal	Source Area Excavation

11 DETAILED ANALYSIS OF REMEDIAL ACTION ALTERNATIVES

In this section, detailed evaluations are performed for the seven remedial alternatives identified at the end of the previous section for each the lower and upper contaminant plume areas.

While not formally discussed in this detailed analysis of remedial action alternatives, it is important to realize that any remedial action performed at River Terrace will required a concentrated effort to coordinate with all of the parties involved or that may be impacted by the selected remedial alternative. For instance, the ADOT is planning an upgrade of the Sterling Highway that passes along the west property boundary. In addition, the proximity of the site along the Kenai River means that coordination and permitting with Fish and Game and other state or federal agencies will be required to ensure protection of the river and its habitat.

11.1 APPROACH USED FOR DETAILED ANALYSIS

The detailed analysis of alternatives consists of the following components:

- An assessment and a summary profile of each alternative against the evaluation criteria.
- A comparative analysis among the alternatives to assess the relative performance of each alternative using specific evaluation criteria.

A summary of these steps is provided in the following sections.

11.1.1 Alternative Development Process

Four multimedia alternatives, including the no action alternative, have been assembled for detailed analysis. These multimedia alternatives consist of combinations of media-specific alternatives that were developed and screened in Section 10. The alternatives were assembled using criteria specified by the state of Alaska and the National Contingency Plan (NCP).

- For source control actions, a range of alternatives that include treatment for reducing the toxicity, mobility, or volume of the contaminants were developed. The range of alternatives include 1) an alternative that removes or destroys contaminants to the maximum extent feasible, eliminating or minimizing the need for long-term management; 2) other alternatives that address the principal threats posed by the site but vary in the degree of treatment employed; and 3) an alternative that involves little or no treatment but provides protection of human health and the environment by preventing or controlling exposure to contaminants.
- For groundwater response actions, a limited number of remedial alternatives that attain site-specific remediation levels within different restoration times using one or more different technologies were developed.
- One or more innovative treatment technologies, if these methods offer the potential for comparable or superior performance or implementability, fewer or lesser adverse impacts than other available approaches, or lower costs for similar levels of performance compared to demonstrated treatment technologies.
- The no action alternative, which may be no further action if some removal or remedial action has already occurred at the site.

11.1.2 Evaluation Criteria

The state of Alaska has established five criteria for evaluation of remedial alternatives, and the NCP contains nine criteria for evaluation of remedial alternatives. The nine NCP criteria were chosen for use in this FS, because they are more rigorous than and inclusive of the five state criteria.

The nine NCP evaluation criteria used in the detailed analyses and brief definitions of each are presented in Table 11-1. The five state criteria are listed below, with the equivalent or most similar NCP criteria given in parentheses:

- Practicable (*implementability NCP criterion*),
- Protectiveness (*protective of human health and the environment NCP criterion*),
- Short- and long-term effectiveness (*combines the short- and long-term effectiveness NCP criteria*),
- Regulations (*compliance with ARARs NCP criterion*), and
- Public input (*community acceptance NCP criterion*).

The evaluation criteria used in the detailed analysis are divided into three categories: threshold criteria, balancing criteria, and modifying criteria. Threshold criteria are those conditions that must be met for the alternative to be viable, and they must be related directly to statutory findings that will be made in the ROD; these criteria must be met. Balancing criteria form the primary basis for comparing alternatives, these criteria relate the alternative to the site-specific conditions. Modifying criteria factor in agency and community concerns; an alternative could be effective and technically implementable, but not viable based on these considerations.

The detailed evaluations focus on the threshold and balancing criteria. Modifying criteria (agency and community acceptance) are not included in this analysis since they depend upon the results of agency and public review. Modifying criteria are considered in the proposed plan stage of the NCP process.

Table 11-1. Remedial Alternative Evaluation Criteria

Criteria Type	Evaluation Criteria	Definition
Threshold Criteria	Protective of human health and the environment	Protection of both human health and the environment is achieved through the elimination, reduction, or control of exposures to contaminated media. All migration pathways must be addressed.
	Compliance with ARARs	Attainment of applicable or relevant and appropriate requirements under federal environmental laws and state environmental or facility siting laws, or provide grounds for invoking applicable waivers.
Balancing Criteria	Long-term effectiveness and permanence	Protects human health and the environment after the remedial objectives have been met.
	Reduction in toxicity, mobility, or volume through treatment	The degree to which recycling or treatment reduces the toxicity, mobility, or volume of the contaminated media.
	Short-term effectiveness	Protects human health and the environment during construction and implementation. Degree of threat and the time period to achieve RAOs are also considered.
	Implementability	The ease or difficulty of implementing the alternative. Considers technical and administrative feasibility as well as the availability of services and materials.

Criteria Type	Evaluation Criteria	Definition
	Cost	Costs include design, construction, startup, and present-worth costs for long-term monitoring and maintenance. Accuracy to within -30% and +50% (USEPA, 1998c).
Modifying Criteria	State Acceptance	The state's position and key concerns related to the preferred alternatives.
(These assessments may not be completed until comments to the proposed plan are received.)	Community Acceptance	The community's apparent preferences for or concerns about alternatives.

11.1.3 Identification of Remedial Alternatives

Each remedial alternative identified in Table 10-1 was evaluated using the seven threshold and balancing NCP criteria. The individual criteria scores were summed to derive a total score for that alternative. The total scores were then compared among all of the alternatives in the Comparative Analysis.

As discussed in Section 11.1.1, the multimedia alternatives were developed to present a range of remedial options, from low-cost and low-effectiveness options (e.g., no action) to high-cost, high-effectiveness options (e.g., active remediation). The remedial alternatives were then applied to both of the two contaminant plume areas – those being the Lower Contaminant Plume between the former dry cleaner building and the Kenai River, and the Upper Contaminant Plume adjacent to the former dry cleaner building (Figures 18 and 19).

Remedial Alternatives for Lower Contaminant Plume: The remedial alternatives selected for detailed analysis at the Lower Contaminant Plume of the River Terrace site are listed below. The primary RAO of these technologies was to prevent contaminants from entering the Kenai River at concentrations above the Alaska WQS (18 AAC 70). One additional alternative (Alternative RT-G) includes treatment of the contaminant source area for the Lower Contaminant Plume with the goal of meeting the same RAO within a shorter restoration time.

Alternative RT-A: No action

Alternative RT-B: Intrinsic remediation

Alternative RT-C: Permeable Reactive Barrier

Alternative RT-D: *In situ* air sparging curtain

Alternative RT-E: Funnel and gate with in well air stripping

Alternative RT-F: Groundwater extraction wells with *Ex situ* air stripping

Alternative RT-G: Reductive Anaerobic Biological *In situ* Treatment Technology

Remedial Alternatives for Upper Contaminant Plume: The remedial alternatives selected for detailed analysis at the Upper Contaminant Plume of the River Terrace site are listed below. Three of the technologies (UT-C to UT-E) are based on some type of a barrier system and are designed to prevent contaminants from leaving the site property. The last two technologies (UT-F and UT-G) are based on treatment of the source contamination area with the ultimate goal of meeting site specific ACLs and preventing off property migration of contaminants above Alaska WQS or other potentially applicable regulatory limits that may apply to adjacent properties.

For this scenario, it was assumed that contaminants from the upper contaminant plume are entering the Sterling Highway right-of-way and are being transported to the Kenai River along the storm sewer system that underlies the highway. Site contamination may also be reaching the storm sewer system along the River Terrace sewer system lift station cross-connect that ties into MH-1 (Figure 18), although this contaminant migration pathway is believed to be only a minor contributor of contaminants observed at the storm sewer outfall as explained in Section 6.3.1.4.

Alternative UT-A: No action

Alternative RT-B: Intrinsic remediation

Alternative UT-C: Permeable Reactive Barrier

Alternative UT-D: *In situ* air sparging curtain

Alternative UT-E: Funnel and gate with in well air stripping

Alternative UT-F: *In situ* air sparging and soil vapor extraction

Alternative UT-G: Reductive Anaerobic Biological *In Situ* Treatment Technology

Because contamination entering the storm-drain piping is presently discharging directly to the Kenai River at a concentration above the Alaska WQS for PCE an interim action is being taken to treat this discharge before it enters the river. A water treatment system based on air stripping technology is to be installed inside the storm sewer piping just prior to discharging into the Kenai River to remove the PCE contamination. This system is expected to remain in place until one of the remedial alternatives is in place and operational or contaminant concentrations decline to below the Alaska WQS.

The ADOT is planning to upgrade the Sterling Highway where it passes along the western boundary of the River Terrace site property. One of the changes to be made during this upgrade is to abandon in place the lower portion of the existing storm sewer system between the Kenai River and Kobuk Street, and construct a new storm sewer system that no longer discharges into the Kenai River. At that time, it would be possible to grout a portion of the existing storm sewer piping and the backfill around the piping to prevent further migration of PCE contaminants along this pathway. However, it is unknown how these changes may affect the current hydrogeologic environment. One possibility is that the present groundwater flow paths may be altered such as to promote off-property migration of the upper contaminant plume in a northwesterly direction, thus allowing the contaminant plume to extend beyond the western side of the Sterling Highway toward and past monitoring well MW-34 (see Figure 18 for reference).

11.1.4 Approach for Comparative Analysis

A comparative analysis was performed to identify the advantages and disadvantages of each remedial alternative relative to the other alternatives. The relative performance of each alternative is evaluated with respect to each of the State/CERCLA evaluation criteria, using the numerical scoring system presented in Table 11-2. The scores have no independent value; they are only meaningful when compared among the different alternatives.

Table 11-2. Evaluation Criteria Rating System

Evaluation Criteria	*Condition*	*Value*
Protective of Human Health and the Environment	Is fully protective	Yes
	Is not protective	No
Compliance with ARARs	Complies with all ARARs	Yes
	Does not comply	No
Long-Term Effectiveness and Permanence	Effective and permanent	5
	Future release possible	3
	No removal or destruction	0
Reduction in Toxicity, Mobility, or Volume Through Treatment	Eliminates toxicity, mobility, volume	5
	Reduces toxicity, mobility, volume	3
	No reduction or treatment	0
Short-Term Effectiveness	Low risk and high protection	5
	Limited risk and limited protection	3
	High risk and low protection	0
Implementability	High technical, administrative, and logistic feasibility	5
	Limited technical, administrative, or logistic feasibility	3
	Technically unproven, permitting uncertain, or resources unavailable	0
Cost	Actual predicted present worth costs were normalized to a 0 to 5 scale, with the Highest Cost Alternative earning a 0, and the no action alternative earning a 5	0 to 5
State Acceptance[1]	To be determined	TBD
Community Acceptance[1]	To be determined	TBD

NOTE: TBD = To Be Determined.

[1] These criteria are typically evaluated following comment on the FS report and the Proposed Plan. They will be addressed in the ROD.

As shown, the rating for threshold criteria can be one of two possibilities: the criterion is either fully met or not met. Therefore, no numerical values are assigned to the threshold criteria.

For balancing criteria, the rating can range from zero to five: if the criterion factors are fully met a five is scored and if the criterion factors are not met a zero is scored. The numerical comparative analysis focuses on the balancing criteria. Determination of scoring values for each alternative is based on comparisons between the alternatives.

11.1.4.1 Balancing Criteria Scoring

An explanation of the balancing criteria scoring procedure is provided in this section.

Long-Term Effectiveness and Permanence: This criterion is used to distinguish between long-term, lasting technologies and technologies that are potentially reversible or can result in lasting contamination. For example, technologies relying on contaminant containment (e.g., slurry walls that could potentially fail, or binding contaminants in concrete that could eventually leach) would score lower than technologies relying on contaminant removal. All of the technologies evaluated in this FS, except the no action alternative, involve contaminant treatment or removal. The no action alternative earned a rating score of 0 for this criterion, intrinsic remediation earned a rating score of 1 for this criterion, and the active remedial technologies earned a rating score of 3 to 5 for this criterion.

Reduction in Toxicity, Mobility, or Volume Through Treatment: This criterion is used to distinguish between technologies that reduce or treat contaminant volume and technologies that do not actively treat contamination. For example, technologies relying on contaminant containment or treatment to prevent further migration would score lower than technologies relying on source-area contaminant treatment or removal.

Short-Term Effectiveness: This criterion is used to balance risks inherent in implementation with short-term effectiveness. The highest score, five, is earned by technologies offering low exposure risks and high protection. All of the active remedial technologies evaluated in this FS involve some implementation risk (e.g., risk to workers during system installation), which is balanced by their increased short-term effectiveness. All passive remedial technologies (e.g., no action) involve less short-term effectiveness, which is balanced by no implementation risk.

Implementability: This criterion is used to differentiate technologies that are easier to implement from technologies that are more difficult to implement. The no action and intrinsic remediation alternatives require no or minimal effort to implement; these alternatives generally earn a score of five for this criterion. The active remedial technologies evaluated in this FS require significant capital costs and design considerations, and some contained significant implementability concerns; these alternatives earned a score of four or less for this criterion.

Cost: This criterion is used to rate, on a relative scale, the different costs associated with each technology. The total present worth costs of each remedial alternative were estimated, and then the costs were normalized on a zero to five scale. The most expensive remedial alternative earns a score of zero, and the no action alternative (least expensive) earns a score of five.

11.1.4.2 Comparative Analysis Discussion

To aid in comparing alternatives, the total score and effectiveness-to-cost quotients for each alternative were calculated. The total score is the sum of the five balancing criteria scores. The effectiveness-to-cost quotient is the sum of the three effectiveness criteria divided by the total cost in millions ($1,000,000). The higher the effectiveness-to-cost quotient, the more cost-effective the alternative is. To assist in identifying preferred alternatives, effectiveness-to-cost quotients provide a qualitative measure of the ability of the alternative to provide remediation versus the cost required achieving the remedial goals.

The validity of the comparative analysis is limited by several assumptions. First, the analysis assumes that all contaminant transport pathways are of equal importance. Similarly, it assigns equal importance to each CERCLA criterion, since each is weighted the same. The analysis also does not quantify synergistic effects between combinations of groundwater, wetland, and soil pathways. Finally, the comparative analysis relies upon the five subjective scores of the balancing factors for each alternative.

Estimating the time required to achieve RAOs is difficult to predict. The controlling factor in this estimate will be the time required for the residual phase PCE to be dissolved/desorbed by the groundwater flowing through these contaminated areas. Due to the complexity of the flow system and unknown residual-phase concentrations, it is impossible to accurately predict a remediation time frame. This is further compounded by the possible presence of free phase DNAPLs at the RTRVP property as discussed in Section 7.7.

If free phase PCE is not present, our experience and results from other sites suggests that a time frame of approximately 10 to 20 years or longer will be needed to achieve RAOs without any source remediation or removal. An estimate of the remaining site contamination and time for restoration was determined and is presented in Appendix N and discussed in Section 7.6. For the purposes of this feasibility study, a single time frame of 15 years was assumed for all passive treatment alternatives (i.e., permeable reactive barrier). Remedial technologies that included treatment of the source contamination area were assumed to achieve RAOs in a 5-year time frame. Since a similar time frame was applied to the similar types of remedial alternatives (i.e., barrier wall vs. source treatment), the selected time frames will have minimal impact on the comparative analysis of the alternatives. Should restoration times take longer than estimated here, their impact on the total remedial costs is relatively minor, due to the present value of money used in the cost estimates, provided no major capital expenditures occur in the future.

When selecting an apparently best alternative, preference is generally given to the highest-scoring remedial alternative. Ultimately however, the regulatory agencies and the community must agree on which alternative, or combinations of alternatives, are the most desirable to achieve the RAOs and to protect the Kenai River based on effectiveness, implementability, and cost.

11.2 ANALYSIS OF ALTERNATIVES FOR LOWER CONTAMINANT PLUME

In this section, each of the seven remedial alternatives is evaluated in detail, using the numerical scores presented in Table 11-2. Conceptual designs and cost estimates for each of the remedial alternatives are provided in Appendix Q.

Groundwater and surface water monitoring will be required for successful implementation of any of the remedial alternatives, with the exception of no action, and will have similar costs. Therefore, the monitoring costs were calculated separately and added into all of the remedial alternatives. The proposed monitoring program includes quarterly monitoring for the first three years of operation, followed by semiannual monitoring for the next two years, and annual monitoring thereafter. The monitoring frequency may be modified by the ADEC in order to comply with desired remedial objectives.

Monitoring is an important decision tool in determining rate of progress and effectiveness of selected remedial alternatives. In some cases, modifications to the selected remedial alternative may be required in order for the alternative to comply with the remedial objectives within desired restoration time and monitoring will help to determine this.

Alternative RT-A: No Action

Alternative RT-A, the no action alternative, is used as a baseline reflecting current conditions without remediation. This alternative is used for comparison with each of the other alternatives. Although natural processes may result in reduction of contaminant concentrations to acceptable levels over time, this alternative does not include any long-term monitoring, modeling, or treatability studies to evaluate the effectiveness of these processes. This alternative is applicable to all contaminant types found in water, soil, and wetland environments.

CERCLA Criteria Scoring Results – Alternative RT-A

Criteria	Score
Protection of Human Health and the Environment	No
Compliance With ARARs	No
Long-Term Effectiveness and Permanence	0
Reduction in Toxicity, Mobility, and Volume Through Treatment	0
Short-Term Effectiveness	1
Implementability	5
Costs	5
Total Score	11

Protection of Human Health and the Environment. Alternative RT-A provides no control of exposure to the contaminated soil, groundwater, and surface water and no reduction in risk to human health and the environment posed by the site contamination. It also allows for continued migration of the contaminant plume and further degradation of the groundwater. (Rating = No)

Compliance With ARARs. Because no action is taken, Alternative RT-A would not comply with ARARs such as the cleanup level for PCE and its daughter products in soil and water. (Rating = No.)

Long-Term Effectiveness and Permanence. This alternative includes no controls for exposure and no long-term management measures. Under this alternative, all current and potential future risks would remain. (Rating = 0.)

Reduction in Toxicity, Mobility, and Volume Through Treatment. This alternative provides no reduction in toxicity, mobility, or volume of the contaminated soil or groundwater through treatment. (Rating = 0)

Short-Term Effectiveness. There would be no additional risks posed to the community, the workers, or the environment because of this alternative being implemented. However, release of contaminants from the subsurface environment to the Kenai River would continue for the foreseeable future. (Rating = 1)

Implementability. There are no implementability concerns posed by this remedy since no action would be taken. (Rating = 5)

Cost. The total present-worth cost of Alternative RT-A is estimated to be $0 since there would be no action. (Rating = 5)

Alternative RT-B: Intrinsic Remediation

Description. Intrinsic remediation would not involve active remedial technologies. Groundwater, soil, and surface water would be left in their current state, and natural processes would continue to reduce contaminant concentrations. Dilution, adsorption, volatilization, precipitation, complexation, and biological degradation of the contaminants occur in the groundwater and subsurface soils. Intrinsic remediation would allow these processes to continue to occur as they have in the past, without disturbances potentially caused by implementation of active remedial technologies.

Intrinsic remediation is not the same as "no action." Implementation of this alternative requires modeling and evaluation of contaminant attenuation. This alternative would also include a groundwater and surface-water monitoring program to confirm predicted results. The water samples would be collected periodically and analyzed for the contaminants of concern. The data generated would be used to monitor degradation and provide an early indication of possible impacts, allowing time for remedial response to mitigate the impact. Intrinsic remediation involves no excavation or handling of contaminated materials. Therefore, site workers are not at risk during implementation and there is no risk to the community from extraction and treatment of contaminated water.

The target contaminants for intrinsic remediation are usually nonhalogenated volatile and semivolatile organics and fuel hydrocarbons. Halogenated volatiles and semivolatiles can also be allowed to naturally attenuate, although the process may be less effective and may only be applicable to some compounds within these contaminant groups. The primary contaminants of concern at the River Terrace site are halogenated volatiles, which are more difficult to treat via intrinsic remediation.

Based on current and historical sampling results, groundwater is leaving the site at levels above the RAOs established for this site. Based on these sampling results, the intrinsic remediation alternative is not protective or compliant with ARARs at the site. As discussed previously, it is estimated that site contamination will continue to migrate off property at concentrations above the Alaska WQS for approximately another 15 years or more. A detailed cost evaluation for this alternative is presented in Appendix Q.

CERCLA Criteria Scoring Results – Alternative RT-B

Criteria	Score
Protection of Human Health and the Environment	No
Compliance With ARARs	No
Long-Term Effectiveness and Permanence	1
Reduction in Toxicity, Mobility, and Volume Through Treatment	1
Short-Term Effectiveness	2
Implementability	5
Costs	3.7
Total Score	12.7

Protection of Human Health and the Environment. By intrinsic remediation and institutional controls (already implemented), alternative RT-B would provide some reduction in risk to human health and the environment posed by site contamination. However, this alternative cannot guarantee that groundwater leaving the site is protective of the Kenai River surface water. (Rating = No)

Compliance with ARARs. Because active remediation is not included, Alternative RT-B would not meet ARARs (such as the cleanup level for PCE in groundwater) in the near term. Over time, intrinsic remediation is expected to reduce contaminant concentrations; however, groundwater currently leaving the site is not compliant with the ACLs established for the site. (Rating = No)

Long-Term Effectiveness and Permanence. Intrinsic remediation is effective in the long-term; however, there is continued risk of future releases and migration of contaminants as a result of this alternative. (Rating = 1)

Reduction in Toxicity, Mobility, and Volume Through Treatment. By intrinsic remediation and institutional controls, Alternative RT-B would provide some reduction in risk to human health and the environment posed by site contamination. The primary short-term risk to human health or the environment that is not addressed by this alternative is potential migration of dissolved-phase PCE and its degradation products to the Kenai River. (Rating = 1)

Short-Term Effectiveness. There would be no additional risks posed to the community or the environment because of this alternative being implemented. This alternative does not provide short-term effectiveness for ecological protection from contaminants in the Kenai River. (Rating = 2)

Implementability. This alternative has low implementability concerns; it requires that only a long-term monitoring plan be implemented. (Rating = 5)

Cost. The total present-worth cost of Alternative RT-B is estimated to range from \$314,000 to \$674,000. Costing details are provided in Appendix Q. On a normalized cost scale (0 to 5), the total present-worth cost of Alternative RT-B earned a rating of 3.7. (Rating = 3.7)

Alternative RT-C: Permeable Reactive Barrier

Description. A permeable reactive treatment wall is installed across the flow path of a contaminant plume, allowing the water portion of the plume to passively move through the wall (Figure 24). This type of barrier allows the passage of water while removing dissolved contaminants by physical, chemical, and/or biological processes. The mechanically simple barriers may contain such agents as zero-valent iron, chelators (ligands selected for their specificity for a given metal), sorbents, microbes, and others. Successful application of this technology requires sufficient characterization of the groundwater hydrology and contamination.

An iron treatment wall consists of iron granules or other iron-bearing minerals for the treatment of chlorinated contaminants such as tetrachloroethylene (PCE), trichloroethylene (TCE), dichloroethylene (DCE), and VC (VC). As the iron is oxidized, a chlorine atom is removed from the compound by one or more reductive dechlorination mechanisms, using electrons supplied by the oxidation of iron. The iron granules are dissolved by the process, but the metal disappears so slowly that the remediation barriers can be expected to remain effective for many years, possibly even decades. The effectiveness of the iron treatment varies depending on the contaminant properties. The reaction rates for TCE and PCE are more rapid than the reaction rates for DCE and VC (USEPA, 1998a). Typically, permeable reactive barriers are designed to provide adequate residence time for the degradation of the parent compound and all toxic intermediate products that are produced. It is estimated that an iron treatment wall will result in complete conversion of the site contaminants to non-toxic compounds, however, bench-scale studies will be required to predict system performance (verify degradation rates) and provide data for field design. Iron treatment wall vendors have also stated that downgradient impacts, such as iron staining, from the installation of an iron treatment wall are unlikely as this has not been observed at any of the sites where the technology has been applied (Personnel Communication, EnviroMetal Technologies).

In situ treatment walls have several advantages over other treatment methods. *In situ* technologies do not require exposing contamination to the surface for remediation. Operation and maintenance costs are minimal; no energy input is required, because the treatment occurs under the natural groundwater gradient. The only operation and maintenance costs associated with this technology would be periodic replacement or rejuvenation of the reaction medium, which may be necessary if the media becomes plugged or its reactive surface capacity is diminished. A permeable reactive barrier installed at the Borden Aquifer, Ontario, Canada, showed only minimal amounts of calcium carbonate precipitate in the wall after five years of operation and it was estimated that the wall should remain active for at least another five years (USEPA, 1999).

Vidic and Pohland (1996) present a summary on the status of treatment wall technology. Examples of pilot-, field-, and commercial-scale treatment walls for chlorinated organic compounds are provided in this summary. Most of the systems reviewed included slurry walls (or other impermeable barriers) on both sides of a permeable treatment wall. The treatment walls were charged with a range of sand/iron mixtures (from 100 percent granular iron to a minimum of 22 percent [by weight] iron filings mixed with 78 percent sand). Treatment wall widths varied from 0.6 meters to 3.2 meters and groundwater residence times within the wall varied between 2 days and 15 days. Influent TCE concentrations ranged from 0.05 mg/L to 250 mg/L; the walls consistently removed 90 percent or more of the influent TCE concentrations.

Full-Scale installations of permeable reactive barriers indicate removal efficiencies of 90 to 95 percent for PCE and TCE, respectively (USEPA, 1999). A former drycleaner site in Germany with initial maximum plume concentration of 20,000 μg/L for PCE had effluent concentrations for PCE of less than 100 μg/L after treatment with a granular iron permeable reactive barrier. A pilot-scale demonstration at the Borden Aquifer in Ontario, Canada, showed that a permeable reactive barrier reduced TCE concentrations by 90 percent and PCE concentration by 86 percent (USEPA, 1999). Initial site concentrations were 250,000 μg/L for TCE and 43,000 μg/L for PCE.

The River Terrace permeable reactive wall would extend across the lower portion of the contaminant plume parallel to the Kenai River as shown in Figure 24. The wall would be approximately 220-feet long by 20-feet in depth with an active treatment layer of approximately 6 feet. Additional details and assumption used for the permeable reactive wall are provided in Appendix Q along with the estimated costs. Because this system functions as only a treatment barrier the expected restoration time is the same as for intrinsic remediation, the difference being that the treatment system would prevent contaminants from migrating into the Kenai River during that period of time.

CERCLA Criteria Scoring Results – Alternative RT-C

Criteria	Score
Protection of Human Health and the Environment	Yes
Compliance With ARARs	Yes
Long-Term Effectiveness and Permanence	5
Reduction in Toxicity, Mobility, and Volume Through Treatment	3
Short-Term Effectiveness	3
Implementability	1
Costs	1.5
Total Score	13.5

Protection of Human Health and the Environment. Alternative RT-C is expected to provide protection of human health and the environment. Dissolved PCE contamination should continue to be remediated by the reactive treatment wall throughout the expected lifetime of this project. (Rating = Yes)

Compliance With ARARs. ARARs compliance for dissolved PCE and its daughter products is expected downgradient of the reactive treatment wall shortly after installation. Over time, intrinsic remediation is expected to reduce all site groundwater and residual-phase contaminant concentrations to below ARAR levels. (Rating = Yes)

Long-Term Effectiveness and Permanence. This alternative removes dissolved PCE from the groundwater, as it passes through the reactive wall, and instrinsic remediation addresses the residual soil contamination. Once installed, the permeable reactive barrier will continue to treat the groundwater even after the RAOs are achieved. Iron treatment walls have the potential for fouling and clogging through the precipitation of minerals in the groundwater, however, removal or mixing of the fouled material can often be done to rejuvenate the wall. This could range from as frequently as every five years in highly mineralized or oxygenated groundwater to a frequency of every 10 to 15 years in less mineralized waters (USEPA, 1998a). Reductions in groundwater and residual contaminant concentrations are considered permanent. (Rating = 5)

Reduction in Toxicity, Mobility, and Volume Through Treatment. This alternative provides a reduction in toxicity, mobility, and volume of contaminated groundwater passing through the treatment wall. No attempt is made to address source area contaminants. (Rating = 3)

Short-Term Effectiveness. There should be only minimal additional risks posed to the community or the environment because of this alternative being implemented. However, worker exposure may occur during the reactive treatment wall installation. The reactive iron treatment wall is not an effective short-term technology for addressing on site contamination; however, it treats groundwater leaving the site to ensure no downgradient migration of the dissolved contamination. (Rating = 3)

Implementability. This alternative has high implementability concerns, regarding planning and implementation of PCE treatment wall installation activities. Costs for installation of an iron permeable reactive barrier in the lower contaminant plume area, as with all subsurface construction in a contaminated area, may significantly fluctuate based on:

- The need for dewatering during excavation,
- The means and costs of contaminated groundwater and soil disposal,
- The structural stability of the soils and potential need for excavation support particularly below the water table,
- Underground utilities within the excavation area will need to be temporarily terminated and later reconnected or permanently relocated, and
- Health and safety concerns for construction personnel working in both a contaminated area and an excavation operation simultaneously.

Equipment, materials, and labor for this type of installation are generally available from larger construction firms. However, granular iron suitable for use will need to be purchased and shipped from the Midwest section of the United States. Excavation and placement of the granular iron may be problematic depending on sloughing of trench walls. It is assumed that sheet pile walls will be required during excavation and iron media placement activities. Implementation of institutional controls and a long-term monitoring plan have a minor contribution to implementability concerns. (Rating = 1)

Cost. The total present-worth cost of Alternative RT-C is estimated to range from \$848,000 to \$1,818,000. Costing details are provided in Appendix Q. On a normalized cost scale (0 to 5), the total present-worth cost of Alternative RT-C earned a rating of 1.5. (Rating = 1.5)

Alternative RT-D: Air Sparging Curtain

Description. This alternative would involve injecting air into the contaminated groundwater, creating an underground stripper that removes contaminants through volatilization. This process is designed to operate at high airflow rates in order to effect volatilization (as opposed to the lower airflow rates used to stimulate biodegradation). Air sparging is usually operated in tandem with a soil vapor extraction (SVE) systems that capture volatile contaminants stripped from the saturated zone. Air sparging is a full-scale remedial technology. Volatilized compounds that enter the vadose zone are often aerobically biodegraded or they are further volatilized into the atmosphere. Alternatively, these VOCs are captured and removed using SVE and the collected vapors are discharged to the atmosphere or treated. If necessary, activated carbon can be used to control emissions from an SVE system, although monitoring and air dispersion modeling are often sufficient to mitigate risk to human health.

Monitoring of the groundwater and SVE discharge would be required to document the effectiveness of this alternative and determine if the compliance objectives are being met.

The target contaminant groups for air sparging are halogenated and nonhalogenated VOC and fuels. Air sparging technology is generally applicable to volatile compounds such as PCE, TCE, DCE, and VC.

The River Terrace air sparging system would contain a sparging curtain across the downgradient edge of the groundwater plume (Figure 25). This sparge curtain would consist of 40 air-sparging wells connected to blowers housed in a connex or small building. Additional details and estimated costs are presented in Appendix Q. Because this system functions as only a treatment barrier the expected restoration time is the same as for intrinsic remediation, the difference being that the treatment system would prevent contaminants from migrating into the Kenai River during that period of time.

CERCLA Criteria Scoring Results – Alternative RT-D

Criteria	Score
Protection of Human Health and the Environment	Yes
Compliance With ARARs	Yes
Long-Term Effectiveness and Permanence	3
Reduction in Toxicity, Mobility, and Volume Through Treatment	2
Short-Term Effectiveness	3
Implementability	4
Costs	1.8
Total Score	13.8

Protection of Human Health and the Environment. Within the 15-year remediation timeframe, Alternative RT-D is expected to provide protection of human health and the environment. (Rating = Yes)

Compliance With ARARs. With the remediation of dissolved PCE downgradient of the sparge curtain, Alternative RT-D would meet some ARARs in the near term. Intrinsic remediation is expected to adequately address the remaining groundwater and residual phase contaminants within the 15-year timeframe. (Rating = Yes)

Long-Term Effectiveness and Permanence. This alternative removes dissolved PCE from the groundwater, as it passes through the air sparge curtain, and instrinsic remediation addresses the residual-phase contamination. The hydraulic conductivity values at the site are near the lower limit considered acceptable for air sparging and may limit its effectiveness. Reductions in groundwater and residual-phase contaminant concentrations are considered permanent. (Rating = 3)

Reduction in Toxicity, Mobility, and Volume Through Treatment. This alternative provides a reduction in toxicity, mobility, and volume of contaminated groundwater passing through the air sparge curtain. Air sparging is often not completely effective in the removal of dissolved groundwater contaminants due to air channelization and heterogeneties within the subsurface. No attempt is made to address source area contaminants. (Rating = 2)

Short-Term Effectiveness. There should be only minimal additional risks posed to the community or the environment because of this alternative being implemented. However, worker exposure may occur during air sparging well installation. This is an active remedial technology resulting in an aggressive removal of dissolved PCE. (Rating = 3)

Implementability. This alternative has only minor implementability concerns, primarily regarding planning and implementation of the air sparging wells. Implementation of institutional controls and a long-term monitoring plan have a minor contribution to implementability concerns. (Rating = 4)

Cost. The total present-worth cost of Alternative RT-D is estimated to range from $764,000 to $1,637,000. Costing details are provided in Appendix Q. On a normalized cost scale (0 to 5), the total present-worth cost of Alternative RT-D earned a rating of 1.8. (Rating = 1.8)

Alternative RT-E: Funnel and Gate with In Well Air Stripping

Description. This alternative includes collecting and treating the contaminated water without removing it from the shallow ground water zone. A funnel wall, consisting of an impermeable barrier between the hard packed till-layer and the ground surface will trap and direct the contaminated groundwater plume to a permeable gate area for treatment. The funnel will most likely be a vertically buried impermeable liner with a gravel-packed drainage trench to enhance groundwater collection and channeling.

An in well air stripping system is then used to remove the volatile contaminants. The collected groundwater is directed to two diffused air bubble stripping wells that will have baffled chambers. Within each chamber, air is injected into the water by a fine bubble diffuser to enhance volatilization. Air strippers provide one of the most aggressive and controllable methods of treating contaminated water, and they are particularly effective at volatilizing the types of chemical contaminants found at this site.

It was assumed that off gases from the air stripping operations could be released to the atmosphere without treatment. If off gas concentrations are higher than anticipated additional costs for off gas treatment will be required.

Monitoring of the groundwater and air stripper discharge would be required to document the effectiveness and regulatory compliance of this alternative.

A conceptual design of the treatment system is shown on Figure 26. As before the barrier wall would extend across the downgradient edge of the groundwater plume and direct the groundwater to the in well air stripping system. Additional details and estimated costs are presented in Appendix Q. Because this system functions as only a treatment barrier the expected restoration time is the same as for intrinsic remediation, the difference being that the treatment system would prevent contaminants from migrating into the Kenai River during that period of time.

CERCLA Criteria Scoring Results – Alternative RT-E

Criteria	Score
Protection of Human Health and the Environment	Yes
Compliance With ARARs	Yes
Long-Term Effectiveness and Permanence	5
Reduction in Toxicity, Mobility, and Volume Through Treatment	3
Short-Term Effectiveness	3
Implementability	1
Costs	1.1
Total Score	13.1

Protection of Human Health and the Environment. Within the 15-year remediation timeframe, Alternative RT-E is expected to provide protection of human health and the environment. (Rating = Yes)

Compliance With ARARs. With the capture and remediation of dissolved PCE collected by the funnel and gate system, Alternative RT-E would meet ARARs in monitoring wells downgradient of the impermeable wall. Intrinsic remediation is expected to adequately address the remaining groundwater and residual phase contaminants within the 15-year timeframe. (Rating = Yes)

Long-Term Effectiveness and Permanence. This system is intended to intercept the flow of contaminants into the Kenai River and aggressively treats the contaminated shallow groundwater that is collected before discharge back to the groundwater system. In comparison to the *in situ* remedial action alternatives, this alternative has a greater degree of flexibility in the treatment of the collected groundwater and thus a higher degree of certainty in achieving the desired RAOs. However, there is a small potential for escapement of groundwater underneath, around, or through the joints of an impermeable wall. Reductions in groundwater contaminant concentrations are considered permanent. (Rating = 5)

Reduction in Toxicity, Mobility, and Volume Through Treatment. This alternative provides a reduction in toxicity, mobility, and volume of contaminated groundwater collected by the funnel and gate system. No attempt is made to address source area contaminants. (Rating = 3)

Short-Term Effectiveness. There should be minimal if any additional risks posed to the community or the environment because of this alternative being implemented. The volatilization of contaminants may results in some increased human exposure. Additionally, worker exposure may occur during installation of the impermeable wall and in well air stripper. (Rating = 3)

Implementability. This alternative has moderate to high implementability concerns, primarily regarding planning and implementation of the impermeable barrier wall. Site soil conditions are an important factor in the difficulty with which the impermeable wall can be constructed. Costs for installation of an impermeable barrier, as with all subsurface construction in a contaminated area, may vary significantly based on:

- The need for dewatering during excavation,
- The means and costs of contaminated groundwater and soil disposal,
- The structural stability of the soils and potential need for excavation support particularly below the water table,

- Underground utilities within the excavation area will need to be temporarily terminated and later reconnected or permanently relocated, and
- Health and safety concerns for construction personnel working in both a contaminated area and an excavation operation simultaneously.

Planning and implementation of the in well air stripper is relatively straightforward and adds only a minor contribution to implementability concerns. (Rating = 1)

Cost. The total present-worth cost of Alternative RT-E is estimated to range from $934,000 to $2,002,000. Costing details are provided in Appendix Q. On a normalized cost scale (0 to 5), the total present-worth cost of Alternative RT-E earned a rating of 1.1. (Rating = 1.1)

Alternative RT-F: Extraction Wells with Air Stripping

Description. This alternative uses groundwater extraction wells to capture and direct shallow-groundwater flow to an above ground treatment system. The collected water will be pumped to the surface for treatment with air stripping equipment. Once treated, the water will be returned to a drainage gallery in the alluvial deposits along the Kenai River. For the purpose of this feasibility study, it was assumed that any groundwater removed from the ground for on site treatment would be allowed under the hazardous waste regulations. If not allowed this alternative would effectively be eliminated from consideration.

Air strippers work by introducing air into contaminated water to maximize the air-water interface and volatilize contaminants. Three general types of air strippers are: packed tower, low-profile tray, and diffused bubble air strippers.

In the packed tower air-stripping system, water is pumped to the top of a tower and allowed to trickle over the packing material inside the air stripper. As the water flows downward over the packing, it spreads more thinly, creating a greater surface area. These thin films of water are met by a counter-flow of air blown in from the bottom of the tower. Packed towers are typically tall units that must be stationary for operation. This is the oldest form of air stripping and is still widely used.

Low-profile tray air strippers represent a large percentage of the type of air strippers used at newer remediation sites. The most common type of low-profile air stripper is the tray-type unit in which a shallow layer of water is allowed to flow along one or more trays. Air is blown through hundreds of holes in the bottom of the trays to generate a froth of bubbles that significantly enhance contaminant volatilization. Manufacturers often claim 99 percent removal rates from tray air strippers. Additionally, low-profile systems are much smaller than the packed tower type and are more resistant to media failure due to clogging (iron fouling). They are often configured on a mobile platform with all necessary ancillary devices to provide a complete portable water treatment solution.

Diffused air strippers are typically a series of tanks, or a single tank with a series of baffles. Air is introduced from the bottom by fine bubble diffusers to enhance volatilization. They are often more economical, since diffused air bubble type strippers may be built for a site-specific application using locally procured components. Such systems are probably less efficient than the prefabricated, packed tower or low profile type systems.

Monitoring of the groundwater and air stripper air and water discharges would be required to document the effectiveness of this alternative. Activated carbon can be used to control emissions from an air stripping system, although monitoring and dispersion modeling is often sufficient to assess risk to human health.

The River Terrace groundwater extraction system would contain a series of 15 or more extraction wells located across the downgradient edge of the groundwater plume (Figure 27). These extraction wells would pump groundwater to the air stripper building for treatment prior to being discharged back the Kenai River alluvial deposits. Additional details and estimated costs are presented in Appendix Q. Because this system functions as only a treatment barrier the expected restoration time is the same as for intrinsic remediation, the difference being that the treatment system would prevent contaminants from migrating into the Kenai River during that period of time.

CERCLA Criteria Scoring Results – Alternative RT-F

Criteria	Score
Protection of Human Health and the Environment	Yes
Compliance With ARARs	Yes
Long-Term Effectiveness and Permanence	4
Reduction in Toxicity, Mobility, and Volume Through Treatment	2
Short-Term Effectiveness	2
Implementability	3
Costs	0.76
Total Score	11.76

Protection of Human Health and the Environment. Within the 15-year remediation timeframe, Alternative RT-F is expected to provide protection of human health and the environment. (Rating = Yes)

Compliance With ARARs. With the capture and remediation of dissolved PCE collected by the extraction wells, Alternative RT-F would meet ARARs in monitoring wells downgradient of their location. Intrinsic remediation is expected to adequately address the remaining groundwater and residual phase contaminants within the 15-year timeframe. (Rating = Yes)

Long-Term Effectiveness and Permanence. This system is intended to intercept the flow of contaminants into the Kenai River. However, some escapage of groundwater is expected to occur between the extraction wells. Air strippers provide one of the most aggressive and controllable methods of treating contaminated water, and they are particularly effective at volatilizing the types of chemical contaminants found at this site. Reductions in groundwater contaminant concentrations are considered permanent. (Rating = 4)

Reduction in Toxicity, Mobility, and Volume Through Treatment. This alternative provides a reduction in toxicity, mobility, and volume of contaminated groundwater collected by the extraction wells. No attempt is made to address source area contaminants resulting in longer restoration times. (Rating = 2)

Short-Term Effectiveness. There should be minimal if any additional risks posed to the community or the environment because of this alternative being implemented. The volatilization of contaminants may results in some increased human exposure. Additionally, worker exposure may occur during groundwater extraction well installation. The use of extraction wells will not be as effective as an impermeable barrier in preventing the flow of contaminated groundwater towards the Kenai River. (Rating = 2)

Implementability. Groundwater extraction well installation is commonly performed and has only minor implementability concerns. The materials necessary for the air stripper system are available from a vendor in Oregon and can readily be shipped to Kenai. Additional materials and labor can be obtained locally for drilling, trenching, and system installation. However, this alternative, unlike the other alternatives, may contain significant regulatory issues and costs associated with the above ground treatment and discharge of treated wastewater -- re-injection to shallow groundwater or, especially, into a storm water or sanitary sewer system. If above ground treatment of the groundwater is not allowed under RCRA this alternative would essentially be eliminated from consideration. Planning and implementation of institutional controls and a long-term monitoring plan have a minor added contribution to implementability concerns. (Rating = 3)

Cost. The total present-worth cost of Alternative RT-F is estimated to range from $793,000 to $1,699,000. Costing details are provided in Appendix Q. On a normalized cost scale (0 to 5), the total present-worth cost of Alternative RT-F earned a rating of 1.7; it is the third most expensive alternative evaluated. (Rating = 1.7)

Alternative RT-G: In situ Biological Treatment

Description. This alternative would involve injecting sodium lactate or HRC into the contaminated groundwater, creating a suitable anaerobic environment that removes contaminants through biological activity. Once anaerobic conditions are achieved the lactic acid is converted to hydrogen through biodegradation, this hydrogen can then be used by reductive dehalogenators that are capable of dechlorinating compounds such as PCE, TCE, and TCA as well as their daughter products. However, other competing microbial processes may also consume the hydrogen, the most common being methanogenesis.

Although anaerobic conditions favor PCE and TCE degradation, some daughter products like DCE and VC can be degraded faster under aerobic conditions. Given that VC accumulation is of particular concern due to its high toxicity, optimal results for chlorinated compound remediation may also require the addition of oxygen to the groundwater at a point downgradient of the lactate injection to promote aerobic biodegradation of the VC.

Because the sodium lactate and HRC are consumed during the dechlorination process, these materials must be replenished for the chlorinated compound remediation to continue. Sodium lactate may require weekly applications whereas HRC is expected to last for several months due to its time-release feature. For this reason, it is assumed that HRC would be preferred method of application.

An HRC field demonstration conducted at a dry cleaning site in Wisconsin showed that the PCE plume mass declined by over 70 percent and the DCE plume mass increased by over 3,500 percent 253 days after adding 240 pounds of HRC. Increases in the VC concentration were also observed although specific concentrations were not reported (regenesis.com/hrctb311.htm). Other case studies reported by Regenesis showed similar results with declines in the PCE/TCE plume mass and increases in the DCE and VC plume masses.

The method of HRC application for the Lower Contaminant Plume at River Terrace consists of an HRC barrier across the downgradient edge of the groundwater plume (Figure 28). This HRC barrer would be constructed using two rows of 40 4-inch injection wells to allow for frequent reapplication of the HRC. Due to the potential for increased VC concentrations and the potential for driving the groundwater and Kenai River sediments anaerobic, re-oxygenation of the groundwater was assumed necessary. A series of 50 ORC injection wells between the HRC barrier and the Kenai River is proposed for re-oxygenation of the groundwater.

The remaining contaminated plume/soil area would be treated using 100 HRC injection points during the first year, with 25 new HRC injection points being installed annually for reapplication at any remaining contaminated hot spots. Appendix Q includes additional details and the estimated cost for implementing this alternative. Because this alternative includes injection of HRC into the source contamination area, it is expected that the restoration time will be shorter than that for intrinsic remediation. A restoration time of five years was assumed for the feasibility study.

Monitoring of the groundwater chemistry and contaminant concentrations would be required to document the effectiveness of this alternative and determine if the compliance objectives are being met. This monitoring will also be used to determine if potential VC accumulations require treatment prior to their entering the Kenai River. For this feasibility study, it was assumed that treatment of VC would not be necessary.

CERCLA Criteria Scoring Results – Alternative RT-G

Criteria	Score
Protection of Human Health and the Environment	Yes
Compliance With ARARs	Yes
Long-Term Effectiveness and Permanence	4
Reduction in Toxicity, Mobility, and Volume Through Treatment	4
Short-Term Effectiveness	4
Implementability	2
Costs	0
Total Score	14

Protection of Human Health and the Environment. Within the 5-year remediation timeframe, Alternative RT-G is expected to provide protection of human health and the environment. (Rating = Yes)

Compliance With ARARs. With the remediation of dissolved PCE downgradient of the HRC injection barrier, Alternative RT-G would meet the ARARs in the near term. However, additional treatment may be required if unacceptable increases in the DCE or VC concentrations occur. In situ biological remediation of the source contaminated area is expected to adequately address the remaining groundwater and residual phase contaminants within the 5-year timeframe. (Rating = Yes)

Long-Term Effectiveness and Permanence. This alternative removes dissolved PCE from the groundwater via biodegradation processes, as it passes through the HRC injection barrier. The HRC injection point grid addresses the source area PCE contamination. Frequent applications of the HRC are required for continued effectiveness. Reductions in groundwater and residual-phase contaminant concentrations are considered permanent. However, there is a possibility for breakdown products, such as DCE and VC, to occur. (Rating = 4)

Reduction in Toxicity, Mobility, and Volume Through Treatment. This alternative provides a reduction in toxicity, mobility, and volume of contaminated groundwater passing through the HRC injection curtain. An HRC injection grid is used to address source area contaminants. Biologically mediated reactions are sometimes difficult to control and add a degree of uncertainty to the treatment method. (Rating = 4)

Short-Term Effectiveness. There should be no additional risks posed to the community or the environment because of this alternative being implemented. However, worker exposure may occur during HRC injection well installation. This is an active remedial technology resulting in an aggressive removal of dissolved PCE. (Rating = 4)

Implementability. This alternative has moderate implementability concerns, primarily regarding planning and implementation of the HRC injection wells. Numerous HRC injection points are required to ensure complete coverage of the contaminated area and re-oxygenation of the groundwater with ORC is required. Both the HRC and ORC injection wells need to be replenished on a frequent basis (at least twice a year) for the duration of the remedial treatment activities. Biological treatment alternatives also require specific environmental site conditions and microorganisms for them to be effective. Implementation of a long-term monitoring plan has a minor contribution to implementability concerns. (Rating = 2)

Cost. The total present-worth cost of Alternative RT-G is estimated to range from $1,195,000 to $2,561,000. Costing details are provided in Appendix Q. On a normalized cost scale (0 to 5), the total present-worth cost of Alternative RT-G earned a rating of zero; it is the most expensive alternative evaluated. (Rating = 0)

11.3 ANALYSIS OF ALTERNATIVES FOR UPPER CONTAMINANT PLUME

The following is an analysis of the seven remedial alternatives considered for the Upper Contaminant Plume region of the River Terrace site.

Alternative UT-A: No Action

Alternative UT-A, the no action alternative, is used as a baseline reflecting current conditions without remediation. This alternative is used for comparison with each of the other alternatives. Although natural processes may result in reduction of contaminant concentrations to acceptable levels over time, this alternative does not include any long-term monitoring, modeling, or treatability studies to evaluate the effectiveness of these processes. This alternative is applicable to all contaminant types found in water, soil, and wetland environments.

CERCLA Criteria Scoring Results – Alternative UT-A

Criteria	Score
Protection of Human Health and the Environment	No
Compliance With ARARs	No
Long-Term Effectiveness and Permanence	0
Reduction in Toxicity, Mobility, and Volume Through Treatment	0
Short-Term Effectiveness	1
Implementability	5
Costs	5
Total Score	11

Protection of Human Health and the Environment. Alternative UT-A provides no control of exposure to the contaminated soil, groundwater, and surface water and no reduction in risk to human health and the environment posed by the site contamination. It also allows for continued migration of the contaminant plume and further degradation of the groundwater. (Rating = No)

Compliance With ARARs. Because no action is taken, Alternative UT-A would not comply with ARARs such as the cleanup level for PCE and its daughter products in soil and water. (Rating = No.)

Long-Term Effectiveness and Permanence. This alternative includes no controls for exposure and no long-term management measures. Under this alternative, all current and potential future risks would remain. (Rating = 0.)

Reduction in Toxicity, Mobility, and Volume through Treatment. This alternative provides no reduction in toxicity, mobility, or volume of the contaminated soil or groundwater through treatment. (Rating = 0)

Short-Term Effectiveness. There would be no additional risks posed to the community, the workers, or the environment because of this alternative being implemented. However, release of contaminants from the subsurface environment to the Kenai River would continue for the foreseeable future. (Rating = 1)

Implementability. There are no implementability concerns posed by this remedy since no action would be taken. (Rating = 5)

Cost. The total present-worth cost of Alternative UT-A is estimated to be $0 since there would be no action. (Rating = 5)

Alternative UT-B: Intrinsic Remediation

Description. Intrinsic remediation would not involve active remedial technologies. Groundwater, soil, and surface water would be left in their current state, and natural processes would continue to reduce contaminant concentrations. Dilution, adsorption, volatilization, precipitation, complexation, and biological degradation of the contaminants occur in the groundwater and subsurface soils. Intrinsic remediation would allow these processes to continue to occur as they have in the past, without disturbances potentially caused by implementation of active remedial technologies.

Intrinsic remediation is not the same as "no action." Implementation of this alternative requires modeling and evaluation of contaminant attenuation. This alternative would also include a groundwater and surface-water monitoring program to confirm predicted results. The water samples would be collected periodically and analyzed for the contaminants of concern. The data generated would be used to monitor degradation and provide an early indication of possible impacts, allowing time for remedial response to mitigate the impact. Intrinsic remediation involves no excavation or handling of contaminated materials. Therefore, site workers are not at risk during implementation and there is no risk to the community from extraction and treatment of contaminated water.

The target contaminants for intrinsic remediation are usually nonhalogenated volatile and semivolatile organics and fuel hydrocarbons. Halogenated volatiles and semivolatiles can also be allowed to naturally attenuate, although the process may be less effective and may only be applicable to some compounds within these contaminant groups. The primary contaminants of concern at the River Terrace site are halogenated volatiles, which are more difficult to treat via intrinsic remediation.

Based on current and historical sampling results (including sampling results from the outfall of the Sterling Highway storm sewer) PCE contaminated water is leaving the site at levels above the Alaska WQS. Based on these sampling results, the intrinsic remediation alternative is demonstrated to not be protective or compliant with ARARs at the site. As discussed previously it is estimated that site contamination will continue to migrate off property at concentrations above the Alaska WQS for approximately another 15 years or more. A detailed cost evaluation for this alternative is presented in Appendix Q.

CERCLA Criteria Scoring Results – Alternative UT-B

Criteria	Score
Protection of Human Health and the Environment	No
Compliance With ARARs	No
Long-Term Effectiveness and Permanence	1
Reduction in Toxicity, Mobility, and Volume Through Treatment	1
Short-Term Effectiveness	2
Implementability	5
Costs	3.9
Total Score	12.9

Protection of Human Health and the Environment. By intrinsic remediation and institutional controls (already implemented), alternative UT-B would provide some reduction in risk to human health and the environment posed by site contamination. However, this alternative cannot guarantee that groundwater leaving the site is protective of the Kenai River surface water. (Rating = No)

Compliance with ARARs. Because active remediation is not included, this alternative would not meet ARARs (such as the cleanup level for PCE in the Kenai River water) in the near term. Over time, intrinsic remediation is expected to reduce contaminant concentrations; however, groundwater currently leaving the site is not compliant with the established Alaska WQS. (Rating = No)

Long-Term Effectiveness and Permanence. Intrinsic remediation is effective in the long-term; however, there is continued risk of contaminant migration because of this alternative. (Rating = 1)

Reduction in Toxicity, Mobility, and Volume through Treatment. By intrinsic remediation and institutional controls, Alternative UT-B would provide some reduction in risk to human health and the environment posed by site contamination. The primary short-term risk to human health or the environment that is not addressed by this alternative is potential migration of dissolved-phase PCE and its degradation products to the Kenai River. Potential exposure to vapor accumulation in underground utilities is also a risk. No mitigation of the volatilized PCE vapors observed adjacent to the former dry cleaner building is provided by the alternative. (Rating = 1)

Short-Term Effectiveness. There would be no additional risks posed to the community or the environment because of this alternative being implemented. This alternative does not provide short-term effectiveness for ecological protection from contaminants in the Kenai River. (Rating = 2)

Implementability. This alternative has low implementability concerns; it requires only that a long-term monitoring plan be implemented. (Rating = 5)

Cost. The total present-worth cost of Alternative UT-B is estimated to range from $314,000 to $674,000. Costing details are provided in Appendix Q. On a normalized cost scale (0 to 5), the total present-worth cost of Alternative UT-B earned a rating of 3.9. (Rating = 3.9)

Alternative UT-C: Permeable Reactive Barrier

Description. A permeable reactive treatment wall is installed across the flow path of a contaminant plume, allowing the water portion of the plume to passively move through the wall. This type of barrier allows the passage of water while removing dissolved contaminants by physical, chemical, and/or biological processes. The mechanically simple barriers may contain such agents as zero-valent iron, chelators (ligands selected for their specificity for a given metal), sorbents, microbes, and others. Successful application of this technology requires sufficient characterization of the groundwater hydrology and contamination.

An iron treatment wall consists of iron granules or other iron-bearing minerals for the treatment of chlorinated contaminants such as PCE, TCE, DCE, and VC. As the iron is oxidized, a chlorine atom is removed from the compound by one or more reductive dechlorination mechanisms, using electrons supplied by the oxidation of iron. The iron granules are dissolved by the process, but the metal disappears so slowly that the remediation barriers can be expected to remain effective for many years, possibly even decades. The effectiveness of the iron treatment varies depending on the contaminant properties. The reaction rates for TCE and PCE are more rapid than the reaction rates for DCE and VC (USEPA, 1998a). Typically, permeable reactive barriers are designed to provide adequate residence time for the degradation of the parent compound and all toxic intermediate products that are produced. It is estimated that an iron treatment wall will result in complete conversion of the site contaminants to non-toxic compounds, however, bench-scale studies will be required to predict system performance (verify degradation rates) and provide data for field design. Iron treatment wall vendors have also stated that downgradient impacts, such as iron staining, from the installation of an iron treatment wall are unlikely as this has not been observed at any of the sites where the technology has been applied (Personnel Communication, EnviroMetal Technologies).

In situ treatment walls have several advantages over other treatment methods. *In situ* technologies do not require exposing contamination to the surface for remediation. Operation and maintenance costs are minimal; no energy input is required, because the treatment occurs under the natural groundwater gradient. The only operation and maintenance (O&M) costs associated with this technology would be periodic replacement or rejuvenation of the reaction medium, which may be necessary if the media becomes plugged or its reactive surface capacity is diminished. A permeable reactive barrier installed at the Borden Aquifer, Ontario, Canada, showed only minimal amounts of calcium carbonate precipitate in the wall after five years of operation and it was estimated that the wall should remain active for at least another five years (USEPA, 1999).

Vidic and Pohland (1996) present a summary on the status of treatment wall technology. Examples of pilot-, field-, and commercial-scale treatment walls for chlorinated organic compounds are provided in this summary. Most of the systems reviewed included slurry walls (or other impermeable barriers) on both sides of a permeable treatment wall. The treatment walls were charged with a range of sand/iron mixtures (from 100 percent granular iron to a minimum of 22 percent [by weight] iron filings mixed with 78 percent sand). Treatment wall widths varied from 0.6 meters to 3.2 meters and groundwater residence times within the wall varied between 2 days and 15 days. Influent TCE concentrations ranged from 0.05 mg/L to 250 mg/L; the walls consistently removed 90 percent or more of the influent TCE concentrations.

Full-Scale installations of permeable reactive barriers indicate removal efficiencies of 90 to 95 percent for PCE and TCE, respectively (USEPA, 1999). A former drycleaner site in Germany with initial maximum plume concentration of 20 mg/L for PCE had effluent concentrations for PCE of less than 100 μg/L after treatment with a granular iron permeable reactive barrier. A pilot-scale demonstration at the Borden Aquifer in Ontario, Canada, showed that a permeable reactive barrier reduced TCE concentrations by 90 percent and PCE concentration by 86 percent (USEPA, 1999). Initial site concentrations were 250,000 μg/L for TCE and 43,000 μg/L for PCE.

The permeable reactive wall for the River Terrace Upper Contaminant Plume would extend along the western property boundary parallel to the Sterling Highway and on across a portion of the northern portion of the site as shown in Figure 29. The wall would be approximately 240-feet long by 24-feet in depth with an active treatment layer of approximately 7 feet. Additional details and assumption used for the permeable reactive wall are provided in Appendix Q along with the estimated costs. Because this system functions as only a treatment barrier the expected restoration time is the same as for intrinsic remediation, the difference being that the treatment system would prevent contaminants from migrating into the Sterling Highway storm sewer system during that period of time.

CERCLA Criteria Scoring Results – Alternative UT-C

Criteria	Score
Protection of Human Health and the Environment	Yes
Compliance With ARARs	Yes
Long-Term Effectiveness and Permanence	5
Reduction in Toxicity, Mobility, and Volume Through Treatment	3
Short-Term Effectiveness	3
Implementability	2
Costs	3.2
Total Score	16.2

Protection of Human Health and the Environment. Alternative UT-C is expected to provide protection of human health and the environment. Dissolved PCE contamination should continue to be remediated by the reactive treatment wall throughout the lifetime of this project. (Rating = Yes)

Compliance With ARARs. ARARs compliance for dissolved PCE and its daughter products is expected downgradient of the reactive treatment wall shortly after installation. Over time, intrinsic remediation is expected to reduce all site groundwater and residual-phase contaminant concentrations to below ARAR levels. (Rating = Yes)

Long-Term Effectiveness and Permanence. This alternative removes dissolved PCE from the groundwater, as it passes through the reactive wall, and instrinsic remediation addresses the residual soil contamination. Iron treatment walls have the potential for fouling and clogging through the precipitation of minerals in the groundwater, however, removal or mixing of the fouled material can often be done to rejuvenate the wall. This could range from as frequently as every five years in highly mineralized or oxygenated groundwater to a frequency of every 10 to 15 years in less mineralized waters (USEPA, 1998a). Reductions in groundwater and residual contaminant concentrations are considered permanent. (Rating = 5)

Reduction in Toxicity, Mobility, and Volume Through Treatment. This alternative provides a reduction in toxicity, mobility, and volume of contaminated groundwater passing through the treatment wall. No attempt is made to address source area contaminants. No mitigation of the volatilized PCE vapors observed adjacent to the former dry cleaner building is provided by this alternative. (Rating = 3)

Short-Term Effectiveness. There should be only minimal additional risks posed to the community or the environment because of this alternative being implemented. However, worker exposure may occur during the reactive treatment wall installation. The reactive treatment wall for PCE is not an effective short-term technology for addressing on site contamination; however, it treats groundwater leaving the site to ensure no downgradient migration of the dissolved contamination. (Rating = 3)

Implementability. This alternative has high implementability concerns, regarding planning and implementation of PCE treatment wall installation activities. Soil conditions at the Upper Contaminant Plume consist of cobbles and gravel making excavation and sheet pile driving rather difficult. Costs for installation of an iron permeable reactive barrier, as with all subsurface construction in a contaminated area, may vary significantly based on:

- The need for dewatering during excavation,
- The means and costs of contaminated groundwater and soil disposal,
- The structural stability of the soils and potential need for excavation support particularly below the water table,
- Underground utilities within the excavation area will need to be temporarily terminated and later reconnected or permanently relocated, and
- Health and safety concerns for construction personnel working in both a contaminated area and an excavation operation simultaneously.

Equipment, materials, and labor for this type of installation are generally available from larger construction firms. However, granular iron suitable for use will need to be purchased and shipped from the Midwest section of the United States. Excavation and placement of the granular iron may be problematic depending on sloughing of trench walls. Implementation of institutional controls and a long-term monitoring plan have a minor contribution to implementability concerns. (Rating = 2)

Cost. The total present-worth cost of Alternative UT-C is estimated to range from $522,000 to $1,118,000. Costing details are provided in Appendix Q. On a normalized cost scale (0 to 5), the total present-worth cost of Alternative UT-C earned a rating of 3.2. (Rating = 3.2)

Alternative UT-D: Air Sparging Grid

Description. This alternative would involve injecting air into the contaminated groundwater, creating an underground stripper that removes contaminants through volatilization. This process is designed to operate at high airflow rates in order to effect volatilization (as opposed to the lower airflow rates used to stimulate biodegradation). Air sparging is a full-scale remedial technology. Volatilized compounds that enter the vadose zone are often aerobically biodegraded or they are further volatilized into the atmosphere. Alternatively, these VOCs are captured and removed using SVE and the collected vapors are discharged to the atmosphere or treated. If necessary, activated carbon can be used to control emissions from an SVE system, although monitoring and air dispersion modeling are often sufficient to mitigate risk to human health.

The target contaminant groups for air sparging are halogenated and nonhalogenated VOC and fuels. Air sparging technology is generally applicable to volatile compounds such as PCE, TCE, DCE, and VC.

Groundwater pump-and-treat combined with air sparging was used to cleanup PCE and TCE contamination at the Gold Coast Superfund Site in Miami, Florida (USEPA, 1998b). Initial PCE and TCE groundwater concentrations at the site were 100 mg/L and 48 mg/L, respectively. Remediation consisted of five extraction wells operating at a combined flow of 44 gpm or a total of 80 million gallons of water removal over a 4-year period. Groundwater cleanup was achieved in four years after excavation and air-sparging of the DNAPL source areas. A total of 1,961 pounds of TCE and PCE were removed from the site at a total cost of approximately $700,000 or $360 per pound of contaminant removed.

The River Terrace Upper Contaminant Plume air-sparging grid system would consist of a grid of sparging wells across the entire region of the groundwater plume (Figure 30). This sparging grid would consist of 65 air-sparging wells and 10 VES wells connected to blowers housed in a connex or small building. Six passive VES wells are included in the floor of the former dry cleaner building to enhance removal of PCE contaminants in the soils under the building. Appendix Q includes additional details and the estimated cost for implementing this alternative. Because this alternative includes sparging and vapor extraction of the source contamination area, it is expected that the restoration time will be shorter than that for intrinsic remediation. A restoration time of ten years was assumed for the feasibility study.

Monitoring of the groundwater chemistry and contaminant concentrations would be required to document the effectiveness of this alternative and determine if the compliance objectives are being met.

CERCLA Criteria Scoring Results – Alternative UT-D

Criteria	Score
Protection of Human Health and the Environment	Yes
Compliance With ARARs	Yes
Long-Term Effectiveness and Permanence	4
Reduction in Toxicity, Mobility, and Volume Through Treatment	4
Short-Term Effectiveness	3
Implementability	4
Costs	1.2
Total Score	16.2

Protection of Human Health and the Environment. Within the 10-year remediation timeframe, Alternative UT-D is expected to provide protection of human health and the environment. (Rating = Yes)

Compliance With ARARs. With aggressive remediation of dissolved and residual phase PCE and other contaminants, Alternative UT-D is expected to meet ARARs within the 10-year remediation timeframe. However, the quantity of contamination that exists underneath the building is unknown and the actual remediation timeframe could be longer. (Rating = Yes)

Long-Term Effectiveness and Permanence. This alternative removes dissolved and residual phase PCE from the groundwater and soil, as the injected air passes through the groundwater into the vadose zone. Sparging wells are placed in the contaminant source area to speed up the remediation process. The hydraulic conductivity values at the site are considered acceptable for air sparging and should not limit its effectiveness, however, a pilot study is recommended to test the effectiveness of air sparging. Reductions in groundwater and residual-phase contaminant concentrations are considered permanent. (Rating = 4)

Reduction in Toxicity, Mobility, and Volume Through Treatment. This alternative provides a reduction in toxicity, mobility, and volume of contaminated groundwater and soil. Air sparging is often not completely effective in the removal of dissolved groundwater contaminants due to air channelization and heterogeneties within the subsurface. Sparging wells are placed in a grid pattern across the entire contaminant area in order to address source area contaminants. Mitigation of the volatilized PCE vapors observed adjacent to the former dry cleaner building and mitigation of potential exposure from vapor accumulation in underground utilities along the Sterling Highway are also provided by this alternative. (Rating = 4)

Short-Term Effectiveness. There should be minor additional risks posed to the community or the environment because of the volatilization of contaminants. Worker exposure may occur during air sparging well installation. This is an active remedial technology resulting in an aggressive and rapid removal of dissolved PCE. (Rating = 3)

Implementability. This alternative has minor implementability concerns, primarily regarding planning and installation of the air sparging wells. Planning and implementation of institutional controls and a long-term monitoring plan have a minor contribution to implementability concerns. (Rating = 4)

Cost. The total present-worth cost of Alternative UT-D is estimated to range from $1,087,000 to $2,330,000. Costing details are provided in Appendix Q. On a normalized cost scale (0 to 5), the total present-worth cost of Alternative UT-D earned a rating of 1.2; it is the second most expensive alternative evaluated. (Rating = 1.2)

Alternative UT-E: Air Sparging Curtain

Description. This alternative would involve injecting air into the contaminated groundwater, creating an underground stripper that removes contaminants through volatilization. This process is designed to operate at high airflow rates in order to effect volatilization (as opposed to the lower airflow rates used to stimulate biodegradation). Air sparging is usually operated in tandem with an SVE system that captures volatile contaminants stripped from the saturated zone. Air sparging is a full-scale technology. Volatilized compounds that enter the vadose zone are often aerobically biodegraded or they are further volatilized into the atmosphere. Alternatively, these VOCs are captured and removed using SVE and the collected vapors are discharged to the atmosphere or treated. If necessary, activated carbon can be used to control emissions from an SVE system, although monitoring and air dispersion modeling are often sufficient to mitigate risk to human health.

The target contaminant groups for air sparging are halogenated and nonhalogenated VOC and fuels. Air sparging technology is generally applicable to volatile compounds such as PCE, TCE, DCE, and VC.

Monitoring of the groundwater and SVE discharge would be required to document the effectiveness of this alternative and determine if the compliance objectives are being met.

The River Terrace Upper Contaminant Plume air sparging barrier system would contain a sparging curtain across the downgradient edge of the groundwater plume (Figure 31). This sparge curtain would consist of 35 air-sparging wells and 5 VES wells connected to blowers housed in a connex or small building. Blower silencers and sound deadening insulation of the building may be required to reduce noise levels. Additional details and estimated costs are presented in Appendix Q. Because this system functions as only a treatment barrier the expected restoration time is the same as for intrinsic remediation, the difference being that the treatment system would prevent contaminants from migrating into the Sterling Highway storm sewer system during that period of time.

CERCLA Criteria Scoring Results – Alternative UT-E

Criteria	Score
Protection of Human Health and the Environment	Yes
Compliance With ARARs	Yes
Long-Term Effectiveness and Permanence	3
Reduction in Toxicity, Mobility, and Volume Through Treatment	2
Short-Term Effectiveness	3
Implementability	4
Costs	2.5
Total Score	14.5

Protection of Human Health and the Environment. Within the 15-year remediation timeframe, Alternative UT-E is expected to provide protection of human health and the environment. (Rating = Yes)

Compliance With ARARs. With the remediation of dissolved PCE downgradient of the sparge curtain, Alternative UT-E would meet some ARARs in the near term. Intrinsic remediation is expected to adequately address the remaining groundwater and residual phase contaminants within the 15-year timeframe. (Rating = Yes)

Long-Term Effectiveness and Permanence. This alternative removes dissolved PCE from the groundwater, as it passes through the air sparge curtain, and instrinsic remediation addresses the remaining dissolved phase and residual-phase contamination. The hydraulic conductivity values at the site are considered acceptable for air sparging and should not limit its effectiveness. Reductions in groundwater and residual-phase contaminant concentrations are considered permanent. (Rating = 3)

Reduction in Toxicity, Mobility, and Volume Through Treatment. This alternative provides a reduction in toxicity, mobility, and volume of contaminated groundwater passing through the air sparge curtain. Air sparging is often not completely effective in the removal of dissolved groundwater contaminants due to air channelization and heterogeneties within the subsurface. No attempt is made to address source area contaminants. No mitigation of the volatilized PCE vapors observed adjacent to the former dry cleaner building is provided by this alternative, however, potential exposure to vapor accumulation in underground utilities along the Sterling Highway are mitigated. (Rating = 2)

Short-Term Effectiveness. There should be minor additional risks posed to the community or the environment because of this alternative being implemented. However, worker exposure may occur during well installation. This is an active remedial technology resulting in an aggressive removal of dissolved PCE. (Rating = 3)

Implementability. This alternative has minor implementability concerns, primarily regarding planning and implementation of the air sparging wells. Drilling of wells is possible even through the gravel and cobbles present at the site. Planning and implementation of institutional controls and a long-term monitoring plan have a minor added contribution to implementability concerns. (Rating = 4)

Cost. The total present-worth cost of Alternative UT-E is estimated to range from $730,000 to $1,565,000. Costing details are provided in Appendix Q. On a normalized cost scale (0 to 5), the total present-worth cost of Alternative UT-E earned a rating of 2.5. (Rating = 2.5)

Alternative UT-F: Source Area Excavation

Excavation Description. This alternative would involve excavating upper plume contaminated soils adjacent to and underneath the former dry cleaner building and *ex situ* treatment of contaminated soils in treatment cells. Excavation and *ex situ* treatment of contaminated soil is a proven remedial technology.

This alternative would include the design, construction, and operation of a soil treatment cell. Soils in the treatment cell will be remediated by SVE. Blowers will aerate the soil, causing the VOCs (e.g., PCE and its degradation products) to volatilize. Vapors may be released to the atmosphere or alternatively they can be treated with activated carbon. The soil treatment cell process is a full-scale process that has been used at numerous sites to remediate contaminated soil. This technology was used to successfully remediate contaminated soil removed during the October 1997 and June 1998 excavations at the RTRVP site. Soil monitoring would need to be performed as part of this alternative, except for performance monitoring to document the removal of contaminated soil from the excavation and performance monitoring of the soil treatment process.

The target contaminant groups for SVE are halogenated and nonhalogenated volatile organic compounds, and fuel hydrocarbons. SVE is a full-scale technology that has been applied at numerous sites for *in situ* and *ex situ* treatment of volatile contaminants.

As an *ex situ* remedy, the excavation associated with this alternative poses a potential health and safety risk to site workers through skin contact and air emissions. Personal protective equipment, at a level commensurate with the contaminants involved would be required during excavation operations.

Application of the Technology at the RTRVP site: The excavation will encompass an area of approximately 9,000 square feet adjacent to the former dry cleaner building, with an average depth of 35 feet (Figure 32). Based on soil sample results, the 12 to 15 feet of soil above the water table is uncontaminated, and will be used as backfill in the excavation. The contaminated material from below the water table will be placed into four SVE soil remediation cells located near the previous RTRVP soil remediation cells.

The advantage of this technology includes the direct removal of contaminated soils from the source area resulting in almost immediate site cleanup. Disadvantages include higher costs, and the possibility of missing a portion of the source area contamination that may be below the existing facility or that may have penetrated deeper into the till layer.

Appendix Q includes additional details and the estimated cost for implementing this alternative. Because this alternative includes removal of contaminated soils, it is expected that the restoration time will be shorter than that for intrinsic remediation. A restoration time of five years was assumed for the feasibility study.

Implementation of this alternative will also involve groundwater and surface water monitoring and institutional controls. It is estimated that this monitoring will be required for a period of 5 years, but the actual monitoring period may vary depending on how soon the remedial action objectives are met.

CERCLA Criteria Scoring Results – Alternative UT-F

Criteria	Score
Protection of Human Health and the Environment	Yes
Compliance With ARARs	Yes
Long-Term Effectiveness and Permanence	5
Reduction in Toxicity, Mobility, and Volume Through Treatment	4
Short-Term Effectiveness	3
Implementability	1
Costs	0.0
Total Score	13.0

Protection of Human Health and the Environment. Alternative UT-F is expected to provide protection of human health and the environment. (Rating = Yes)

Compliance With ARARs. With the removal of PCE-contaminated soils and intrinsic remediation of any dissolved PCE downgradient of the former dry cleaner building, Alternative UT-F would meet the ARARs in the near term. (Rating = Yes)

Long-Term Effectiveness and Permanence. This alternative removes PCE from the soil by excavation and treats groundwater affected by any remaining PCE by intrinsic remediation. Excavated soil will be treated ex situ in a soil treatment cell. Reductions in groundwater and residual-phase contaminant concentrations are considered permanent. (Rating = 5)

Reduction in Toxicity, Mobility, and Volume Through Treatment. This alternative provides a reduction in toxicity, mobility, and volume of contaminated groundwater and soil. Excavation and *ex situ* treatment are used to address the source area contaminants. However, residual contamination that likely exists in the subsurface till materials may be missed resulting in incomplete source removal. The effects of any remaining PCE will be addressed through intrinsic remediation. (Rating = 4)

Short-Term Effectiveness. There will be additional risks posed to the community or the environment because of this alternative being implemented. Fencing will be required to minimize risks posed to the community from contaminated soil and groundwater that is brought to the surface. Worker exposure may occur during excavation activities. This is an active remedial technology resulting in an aggressive removal of contaminants. (Rating = 3)

Implementability. This alternative has relatively high implementability concerns. Implementing the soil removal will adversely impact the former dry cleaner building since it will need to be moved or destroyed. Construction of the soil treatment cells will adversely impact the use of the RTRVP parking units since many of them will be covered by the soil cells. Significant conflicts with buried underground utilities are also anticipated in the excavation area. Costs for excavation as with all subsurface construction in a contaminated area, may significantly fluctuate based on:

- The need for dewatering during excavation,
- The means and costs of contaminated groundwater and soil disposal,
- The structural stability of the soils and potential need for excavation support particularly below the water table,
- Underground utilities within the excavation area will need to be temporarily terminated and later reconnected or permanently relocated, and

- Health and safety concerns for construction personnel working in both a contaminated area and an excavation operation simultaneously.

Equipment, materials, and labor for this type of installation are generally available from larger construction firms. Implementation of a long-term monitoring plan has a minor added contribution to implementability concerns. (Rating = 1)

Cost. The total present-worth cost of Alternative UT-F is estimated to range from $1,433,000 to $3,071,000. Costing details are provided in Appendix Q. On a normalized cost scale (0 to 5), the total present-worth cost of Alternative U-F earned a rating of 0.0; it is the most expensive alternative evaluated. (Rating = 0.0)

Alternative UT-G: In Situ Biological Treatment

Description. This alternative would involve injecting sodium lactate or HRC into the contaminated groundwater, creating a suitable anaerobic environment that removes contaminants through biological activity. Once anaerobic conditions are achieved the lactic acid is converted to hydrogen through biodegradation, this hydrogen can then be used by reductive dehalogenators that are capable of dechlorinating compounds such as PCE, TCE, and TCA as well as their daughter products. However, other competing microbial processes may also consume the hydrogen, the most common being methanogenesis.

Although anaerobic conditions favor PCE and TCE degradation, some daughter products like DCE and VC can be degraded faster under aerobic conditions. Given that VC accumulation is of particular concern due to its high toxicity, optimal results for chlorinated compound remediation may also require the addition of oxygen to the groundwater at a point downgradient of the lactate injection to promote aerobic biodegradation of this VC.

Because the sodium lactate and HRC are consumed during the dechlorination process, these materials must be replenished for the chlorinated compound remediation to continue. Sodium lactate may require weekly applications whereas HRC is expected to last for several months due to its time-release feature. For this reason, it is assumed that HRC would be preferred method of application.

An HRC field demonstration conducted at a dry cleaning site in Wisconsin showed that the PCE plume mass declined by over 70 percent and the DCE plume mass increased by over 3,500 percent 253 days after adding 240 pounds of HRC. Increases in the VC concentration were also observed although specific concentrations were not reported (regenesis.com/hrctb311.htm). Other case studies reported by Regenesis showed similar results with declines in the PCE/TCE plume mass and increases in the DCE and VC plume masses.

The method of HRC application for the Upper Contaminant Plume at River Terrace consists of an HRC injection grid. This HRC grid would be constructed using 100 HRC injection points during the first year, with 25 new HRC injection points being installed annually for reapplication at the remaining contaminated hot spots (Figure 33). Additionally injections of liquid HRC or sodium lactate to the soils underneath the building will be performed to promote biodegradation of the PCE contamination found under the building. Appendix Q includes additional details and the estimated cost for implementing this alternative. Because this alternative includes injection of HRC into the source contamination area, it is expected that the restoration time will be shorter than that for intrinsic remediation. A restoration time of ten years was assumed for the feasibility study.

Monitoring of the groundwater chemistry and contaminant concentrations would be required to document the effectiveness of this alternative and determine if the compliance objectives are being met. This monitoring will also be used to determine if potential VC accumulations require treatment prior to their entering the Kenai River. For this feasibility study, it was assumed that treatment of VC would not be necessary.

CERCLA Criteria Scoring Results – Alternative UT-G

Criteria	Score
Protection of Human Health and the Environment	Yes
Compliance With ARARs	Yes
Long-Term Effectiveness and Permanence	4
Reduction in Toxicity, Mobility, and Volume Through Treatment	4
Short-Term Effectiveness	4
Implementability	2
Costs	2.9
Total Score	16.9

Protection of Human Health and the Environment. Within the 10-year remediation timeframe, Alternative UT-G is expected to provide protection of human health and the environment. (Rating = Yes)

Compliance With ARARs. With the remediation of dissolved PCE downgradient of the HRC injection area, Alternative RT-G would meet the ARARs in the near term. However, additional treatment may be required if unacceptable increases in the DCE or VC concentrations occur. In situ biological remediation of the source area is expected to adequately address the remaining groundwater and residual phase contaminants within the 10-year timeframe. However, the quantity of contamination that exists underneath the building is unknown and the actual remediation timeframe may be longer. (Rating = Yes)

Long-Term Effectiveness and Permanence. This alternative removes dissolved PCE from the groundwater and soil, through *in situ* biological treatment as the HRC is dissolved and carried with the groundwater. Frequent applications of the HRC are required for continued effectiveness. Reductions in groundwater and residual-phase contaminant concentrations are considered permanent. However, there is a possibility for breakdown products, such as DCE and VC, to occur. (Rating = 4)

Reduction in Toxicity, Mobility, and Volume Through Treatment. This alternative provides a reduction in toxicity, mobility, and volume of contaminated groundwater and soil. An HRC injection grid is used to address the source area contaminants. In situ bioremediation also offers at least partial mitigation of the volatilized PCE vapors observed adjacent to the former dry cleaner building and partial mitigation of potential exposure from vapor accumulation in underground utilities along the Sterling Highway. (Rating = 4)

Short-Term Effectiveness. There should be no additional risks posed to the community or the environment because of this alternative being implemented. However, worker exposure may occur during HRC injection well installation. This is an active remedial technology resulting in a fairly aggressive removal of contaminants. (Rating = 4)

Implementability. This alternative has moderate implementability concerns, primarily regarding planning and implementation of the HRC injection wells. Numerous HRC injection points are required to ensure complete coverage of the contaminated area and frequent applications of the HRC are required for continued effectiveness. The area over the contaminant injection points will need to remain free of obstacles, such as buildings, to allow for reapplication of the HRC. Biological treatment alternatives also require specific environmental site conditions and microorganisms for them to be effective. Implementation of a long-term monitoring plan has a minor contribution to implementability concerns. (Rating = 2)

Cost. The total present-worth cost of Alternative UT-G is estimated to range from $602,000 to $1,290,000. Costing details are provided in Appendix Q. On a normalized cost scale (0 to 5), the total present-worth cost of Alternative UT-G earned a rating of 2.9; it is the second least expensive alternative evaluated with the exception of intrinsic remediation and the no action alternatives. (Rating = 2.9)

11.4 COMPARISON OF ALTERNATIVES

A comparative analysis of the alternatives developed for River Terrace is provided in Table 11-3. The relative numerical values for each of the criteria are shown; the estimated cost is also expressed in thousands of dollars for comparison. The numerical values were developed in Section 9.2, which provides a discussion of the alternatives and how each meets the criteria.

Comparison of the alternatives for River Terrace is facilitated by evaluation of total scores and the effectiveness-to-cost quotients (Table 11-3**Error! Reference source not found.**). The total score is the sum of the seven criteria scores. The effectiveness-to-cost quotient is the sum of the five effectiveness criteria divided by the total cost in millions of dollars. The higher the cost quotient, the more cost-effective the alternative. This quotient provides a qualitative comparison of the ability of the alternative to provide remediation versus the cost required achieving the remedial goals. Effectiveness-to-cost quotients were not calculated for the "no action" alternative since no costs were developed for this alternative and the effectiveness-to-cost quotient scores would be meaningless.

To aid the reader in comparison of the remedial alternatives Table 11-3 includes a column that lists the theoretical effectiveness of the treatment technology. Where possible this value is based on numbers reported for field case studies reported in the literature. However, since the effectiveness of any *in situ* treatment alternative is dependent on the site conditions (i.e., hydraulic conductivity, sorption characteristics, etc.) these numbers often reflect “ideal” or favorable site conditions and the expected treatment effectiveness may be somewhat lower.

11.4.1 Limitations of Comparative Analysis

The comparative analysis is limited by several assumptions. First, it assumes that all contaminant types are of equal importance. Similarly, the comparative analysis assigns equal importance to each criterion rather than trying to rank them by some method.

The best overall remedial approach for River Terrace may not necessarily include the highest scoring remedial alternative. The numerical scoring reflects only the CERCLA balancing criteria scores; the threshold criteria (protection of human health and the environment and compliance with ARARs) must be met in order for an alternative to be viable. Ultimately, the regulatory agencies and the community must agree on an alternative or set of alternatives, based on effectiveness, implementability, acceptability, and cost.

11.4.2 Limitations of Feasibility Study

The overall achievement of the remedial objectives of the feasibility study is limited based on the site characterization data available, uncertainty in the data, and uncertainty in the implementation of the remedial alternative. The remedial designs and cost estimates included in the feasibility study are intended for comparison of the various alternatives. They should not be construed to represent actual remedial designs or true remedial design and construction cost estimates.

There are a large number of uncertainties involved with preparing a feasibility study. For instance, estimating the time required to achieve RAOs is often uncertain. Site specific parameters and conditions, such as the mass or volume of contamination, are often only partially known and thus contain other uncertainties that affect the overall remediation timeframe. Remedial technologies are often dependant on knowing these site parameters and conditions and the effectiveness of the remedial technology may not be known until site specific pilot studies have been completed.

Continued monitoring of the site conditions and contaminant concentrations may be used to help address some of these uncertainties. For instance, periodic review of the groundwater monitoring information will need to be conducted to determine if the selected remedial alternative(s) are achieving the desired remedial objectives. If monitoring data indicates that remedial objectives are not being met or will not be achieved within the desired timeframe then additional corrective measures or modifications may be required.

For the purposes of the feasibility study it was assumed that none of the excavated soils, removed groundwater, or investigation-derived wastes (IDW) would be classified as a RCRA hazardous waste. The EPA has provided a Contained-in Determination for investigation-derived wastes that have been or will be generated during remedial investigation work associated with the River Terrace site. This contained-in determination applies only to those wastes that comply with the Contained-in levels stated in the USEPA letter for the River Terrace site and that are to be disposed of on the RTRVP property. To be exempt from the RCRA Subtitle C regulations, soils must also be shown not to exhibit any hazardous characteristic under 40 CFR Part 261 Subpart C.

If the assumption that excavated soils or removed groundwater are not classified as RCRA hazardous waste turns out to be incorrect, it could result in a major increase in the cost estimate for some alternatives depending on the amount of waste generated but especially with pump-and-treat alternatives.

11.5 SUMMARY

11.5.1 Lower Contaminant Plume

The feasibility study component of this report for the Lower Contaminant Plume evaluated seven alternative remediation technologies potentially appropriate for the site. Two of the remedial alternatives were determined to not be protective of human health and the environment and not to comply with ARARs, these Alternatives are RT-A (No Action) and RT-B (Intrinsic Remediation). The five remaining technologies were determined viable and cost estimates (total present cost) were developed for each of these alternatives. Estimated costs include 15 years of operation and maintenance and 15 years of monitoring for the four barrier type alternatives (RT-C through RT-F) and the Intrinsic Remediation Alternative (RT-B). Estimated costs include 5 years of operation and maintenance and 5 years of monitoring for the one alternative (RT-G) that includes treatment of the contaminant source area. The alternatives and the estimated cost to implement it are listed below:

• RT-B	Intrinsic Remediation (for comparison)	\$314K to \$674K
• RT-C	Permeable reactive barrier	\$848K to \$1,818K
• RT-D	*In situ* air sparging curtain	\$764K to \$1,637K
• RT-E	Funnel and gate system	\$934K to \$2,002K
• RT-F	Extraction wells with air stripping	\$793K to \$1,699K
• RT-G	*In situ* biological treatment	\$1,195K to \$2,561K

The funnel and gate system Alternative RT-E also includes three options for groundwater treatment using an in well air stripping system, a permeable reactive iron barrier, or an *ex situ* air stripping treatment system. The estimated costs for each of these options is included in Appendix Q.

Any of the alternatives selected will require institutional controls to prohibit installation of a well as a drinking water source or other intrusive activities that would not be appropriate during site remediation.

11.5.2 Upper Contaminant Plume

The feasibility study component of this report for the Upper Contaminant Plume evaluated seven alternative remediation technologies potentially appropriate for the site. Two of the remedial alternatives were determined to not be protective of human health and the environment and not to comply with ARARs, these Alternatives are UT-A (No Action) and UT-B (Intrinsic Remediation). The five remaining technologies were determined viable and cost estimates (total present cost) were developed for each of these alternatives. Estimated costs include 15 years of operation and maintenance and 15 years of monitoring for the two barrier type alternatives (UT-C and UT-E) and the Intrinsic Remediation Alternative (UT-B). Estimated costs include 10 years of operation and maintenance and 10 years of monitoring for the two alternatives (UT-D and UT-G) that included treatment of the contaminant source area. Estimated costs include 5 years of operation and maintenance and 5 years of monitoring for the one alternative (UT-F) that included removal of the contaminant source area by excavation. The alternatives and the estimated cost to implement it are listed below:

• UT-B	Intrinsic Remediation (for comparison)	\$314K to \$674K
• UT-C	Permeable reactive barrier	\$522K to \$1,118K
• UT-D	*In situ* air sparging grid	\$1,087K to \$2,330K
• UT-E	*In situ* air sparging curtain	\$730K to \$1,565K
• UT-F	Source Area Excavation	\$1,433K to \$3,071K
• UT-G	*In situ* biological treatment	\$602K to \$1,290K

Any of the alternatives selected will require institutional controls to prohibit installation of a well as a drinking water source or other intrusive activities that would not be appropriate during site remediation.

Table 11-3 Comparative Analysis of Remedial Alternatives

(insert Table 11-3)

11.6 DETAILED ANALYSIS OF REMEDIAL ACTION ALTERNATIVES FOR ACLS

At the request of the property owner, a detailed evaluation of alternatives for reaching ACLs is included in Appendix S.

Also, a detailed evaluation of alternatives for excavating all remaining contaminated soil at the RTRVP site is included in Appendix S2.

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Appendix A

Photograph Log

River Terrace Site Remedial Investigation

Appendix B

Site Surveying Data

River Terrace Site Remedial Investigation

Appendix C

Historical Soil Analytical Data

River Terrace Site Remedial Investigation

Appendix D

Historical Groundwater Analytical Data

River Terrace Site Remedial Investigation

Appendix E

Monitoring Well and Soil Boring Logs

River Terrace Remedial Investigation
1999-2000 RI and Previous Investigations

Appendix F

Monitoring Well Development Sheets and Groundwater Sample Data Sheets

River Terrace Remedial Investigation
July, November, December 1999 and January 2000

Monitoring Well Development Sheets

July 1999

Groundwater Sample Data Sheets

September 1999

Groundwater Sample Data Sheets

October 1999

Groundwater Sample Data Sheets

November 1999

Groundwater Sample Data Sheets

December 1999

Groundwater Sample Data Sheets

January 2000

Groundwater Sample Data Sheets

Appendix G

Slug Test Data
River Terrace Site Remedial Investigation
July 1999-November 1999

Appendix H

Cation/Anion Analysis Graphs

River Terrace Site Remedial Investigation
October 1999-December 1999

Appendix I

Quality Assurance Review

River Terrace Remedial Investigation
July, November, December 1999 and January 2000

Appendix J

PCE Trend Charts

River Terrace Remedial Investigation Monitoring Wells July 1997 to October 1999

Appendix K

Migration to Groundwater Calculations

River Terrace Site Remedial Investigation

Appendix L

SESOIL Modeling to Support “Contained-In” Determination

River Terrace Site Remedial Investigation

Appendix M

Spill Volume Calculations

River Terrace Site Remedial Investigation

Appendix N

Johnson-Ettinger Air Vapor Model Runs
Bioscreen Summary

River Terrace Remedial Investigation
Former Dry Cleaner Building

Appendix O

Freundlich Isotherms

River Terrace Site Remedial Investigation

Appendix P

ADEC ACL Determination Letter

River Terrace Remedial Investigation

Appendix Q

Remedial Alternatives Cost Analysis

River Terrace Site Feasibility Study