REMEDIAL INVESTIGATION REPORT

LOCKWOOD SOLVENT GROUNDWATER PLUME SITE

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ACRONYMS AND ABBREVIATIONS

°C	Degrees Centigrade
°F	Degrees Fahrenheit
μg/kg	Micrograms per kilogram
μg/L	Micrograms per liter
μg/m^3	Micrograms per cubic meter
ATSDR	Agency for Toxic Substances and Disease Registry
bgs	Below ground surface
Beall	Beall Trailers of Montana, Inc.
BTEX	Benzene, toluene, ethylbenzene, total xylenes
CalEPA	California Environmental Protection Agency
CECRA	Montana Comprehensive Environmental Cleanup and Responsibility Act
CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act
cfs	Cubic feet per second
cis-1,2-DCE	cis-1,2-dichloroethylene
cm^2	Square centimeter
COPC	Contaminant of potential concern
CFR	Code of Federal Regulations
DCA	Dichloroethane
DCE	Dichloroethene
DEQ	Montana Department of Environmental Quality
DEQ/RD	DEQ/Remediation Division
DNRC	Montana Department of Natural Resources and Conservation
DOE	U.S. Department of Energy
EPA	U.S. Environmental Protection Agency
ERA	Ecological Risk Assessment
FS	Feasibility study
ft/day	Feet per day
ft/ft	Feet per foot
ft/year	Feet per year
gpd/ft	Gallons per day per foot
gpm	Gallons per minute
gpm/ft	Gallons per minute per foot
GPS	Global positioning system
HEAST	Health Effects Assessment Summary Tables
HHRA	Human Health Risk Assessment
HI	Hazard index
HQ	Hazard quotient

ACRONYMS AND ABBREVIATIONS
(Continued)

IRIS	Integrated Risk Information System
IW	Industrial well
LOAEL	Lowest-observed-adverse-effects level
LSGPS	Lockwood Solvent Groundwater Plume Site
LUST	Leaking underground storage tank
LWDS	Lockwood Water and Sewer District
MCL	Maximum contaminant level
mg/kg	Milligram per kilogram
mL	Milliliter
MW	Monitoring well
NAPL	Nonaqueous phase liquid
nM	nanoMole
NOAEL	No-observed-adverse-effects level
NPL	National Priorities List
PAH	Polynuclear aromatic hydrocarbon
PCE	Tetrachloroethene
PID	Photoionization detector
Pioneer	Pioneer Technical Services, Inc.
ppm	Parts per million
PRG	Preliminary remediation goal
PVC	Polyvinyl chloride
QA	Quality assurance
QAPP	Quality assurance project plan
QC	Quality control
RAGS	Risk Assessment Guide for Superfund
RI	Remedial investigation
RI/FS	Remedial investigation/feasibility study
RIWP	Remedial investigation work plan
RW	Residential well
SOP	Standard operating procedure
SSL	Soil Screening Level
SVOC	Semivolatile organic compound
SWL	Static water level
TCA	Trichloroethane
TCE	Trichloroethene
TtEMI	Tetra Tech EM Inc.
trans-1,2-DCE	trans-1,2-dichloroethene

ACRONYMS AND ABBREVIATIONS
(Continued)

UCL95	95th percentile upper confident limit on the mean
USGS	U.S. Geological Survey
USFWS	U.S. Fish and Wildlife Service
VC	Vinyl chloride
VOA	Volatile organic analysis
VOC	Volatile organic compound
WQB-7	Water Quality Bureau Circular 7

EXECUTIVE SUMMARY

This report presents the findings of the remedial investigation (RI) conducted at the Lockwood Solvent Groundwater Plume Site (LSGPS) in Billings, Montana. The RI was conducted by Tetra Tech EM Inc. (TtEMI) under the direction of the Montana Department of Environmental Quality Remediation Division (DEQ/RD) pursuant to the RI work plan (RIWP) (TtEMI 2002a). This investigation was performed in accordance with the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) and the Montana Comprehensive Environmental Cleanup and Responsibility Act (CECRA). The RI has been conducted under a Cooperative Agreement with the U.S. Environmental Protection Agency (EPA). The objectives of the RI are as follows:

- Characterize the nature and extent of contamination at the site
- Describe the geology, hydrogeology, and physical features at the site
- Identify potential contaminant migration pathways and receptors
- Determine the fate and transport of contaminants at the site
- Provide data to support a baseline human health risk assessment and ecological risk assessment for use in identifying areas and contaminants of concern
- Present data for use in developing the feasibility study, including development and screening of remedial alternatives.

SITE BACKGROUND

The LSGPS is a 580-acre site on the outskirts of Billings, in Yellowstone County, Montana, that has been found to have soils and groundwater contaminated with chlorinated solvents. The primary contaminants of potential concern (COPC) are the volatile organic compounds (VOC) tetrachloroethene (PCE), trichloroethene (TCE), dichloroethene (DCE) (including both cis- and trans- isomers), and vinyl chloride (VC). On December 1, 2000, the LSGPS was officially placed on the EPA National Priorities List.

Current land use within the LSGPS is characterized as residential, commercial, and "light" industrial. Examples of commercial and light industrial businesses in the area include trucking, vehicle repair, truck tank manufacturing, chemical repackaging, machine shops, and auto salvage. At this time, the primary source of domestic use water in the LSGPS is from the Lockwood Water and Sewer District (LWSD) Public Water Supply. However, limited full-use domestic, other domestic (such as irrigation), commercial, and nondomestic use water is known to come from the shallow alluvial aquifer via several individual wells.

Based on the geography of the LSGPS and DEQ/RD's desire to present site figures on readily reproducible size paper, the LSGPS has been broken into three geographic areas. These three areas are as follows:

- Area A: The northern part of the LSGPS, generally including the area west of Klenck Lane; north of Taylor Place and the Coulson irrigation ditch; and south and east of the Yellowstone River
- Area B: The southeastern part of the LSGPS, generally including the area north of U.S. Highway 87 East and a line south of the Beall Trailers of Montana, Inc. (Beall) facility, west of a line extending south from the east end of Gordon Drive, east of the Interstate Highway 90 Lockwood interchange, and south of Taylor Place
- Area C: The southwestern part of the LSGPS, including the area west of the Interstate Highway 90 Lockwood interchange and Steffes Road, south of the Coulson irrigation ditch, east of the Yellowstone River, and north of the lower Lockwood irrigation ditch.

PREVIOUS INVESTIGATIONS

In October 1986, LWSD personnel discovered the presence of benzene and chlorinated solvents in LWSD water supply wells. That discovery led to the initiation of a number of investigations of underground storage tanks and a petroleum pipeline in the vicinity of the LWSD facility. In June 1998, the DEQ/RD Site Response Section performed an integrated assessment of the LSGPS through its multi-site cooperative agreement with the EPA. This assessment and subsequent investigations by DEQ/RD, EPA, and others indicated that chlorinated solvents at the LSGPS have impacted groundwater, surface water, soil, soil vapor, and indoor air.

The results of previous investigations identified two potential source areas where elevated concentrations of COPCs are found in soil and associated groundwater. These areas included the Brenntag West Inc. (Brenntag) and Beall properties. Previous investigations, although incomplete, did not identify contaminated soil at other areas of the site. However, the investigations identified contaminant concentrations in groundwater at the site that exceeded state human health standards and federal maximum contaminant levels (MCL), and which pose a potential threat to human health and the environment. The MCL is the maximum permissible level of a contaminant in water, which is delivered to any user of a public water system.

REMEDIAL INVESTIGATION

In May 2002, DEQ/RD and their contractor, TtEMI, initiated remedial investigation and data collection activities at the LSGPS as described in the RIWP. Surface and subsurface soil investigation and sampling, and monitoring well construction and development began in June 2002 and continued through July 2002. Aquifer testing was conducted in August 2002. The first round of groundwater sampling for VOCs was conducted in July and August 2002. Surface water and sediment sampling was conducted in May 2002 for the Coulson irrigation ditch. Additional surface water and sediment sampling of the AJ Gravel pond and a wetlands was conducted in August 2002. A second round of groundwater sampling for VOCs was conducted in October 2002. Groundwater sampling for remediation by natural attenuation parameters was also conducted in October 2002.

NATURE AND EXTENT OF CONTAMINATION

Two primary sources of groundwater contamination have been identified at the LSGPS. These sources have been identified as the Brenntag source area and the Beall source area. Soil samples with COPC concentrations above site-specific soil screening levels (SSL) were reported in both areas. In the Brenntag source area, concentrations of COPCs above SSLs were reported in surbsurface soil samples; the lateral and vertical limits of source material have not been fully defined. However, based on preliminary data, the volume of source material is tentatively estimated at 35,500 cubic yards.

In the Beall source area, concentrations of COPCs above SSLs were reported in both surface and subsurface soil samples; however the limits of the identified source material at the Beall source area have not been fully defined. Additional source material is likely present beneath the oil-water separator and associated steam clean bay drainage pipe. Due to the limited width of potentially contaminated soil beneath these features, the volume of this source area has not been estimated. Additional subsurface investigation of this area may further delineate this source but it may be difficult to locate contaminated soil without removal and inspection of the inactive oil-water separator and piping.

The highest concentrations of VOCs in groundwater at the LSGPS have been reported within and downgradient of the Brenntag and Beall source areas. COPC concentrations in groundwater above MCLs are found within and downgradient of both source areas. In addition, low level concentrations of VOCs in groundwater have been detected throughout the site. The extent of contamination has been delineated on the southern and eastern boundaries by sample locations where no COPCs were detected.

Groundwater contamination is delineated (bounded) on the western and northern edges by the Yellowstone River; no samples were collected on the west or north side of the Yellowstone River.

Surface water samples collected from the AJ Gravel pond contained concentrations of COPCs above MCLs. COPCs were detected in Coulson irrigation ditch at concentrations that were below MCLs. COPCs in the AJ Gravel pond and Coulson irrigation ditch are considered the result of contaminated groundwater discharge into these surface waters. Contaminated groundwater likely discharges into the irrigation ditch only during non-irrigation periods. Surface water in Coulson irrigation ditch flows off-site to the northeast. No sediment samples contained COPC concentrations above SSLs.

FATE AND TRANSPORT OF CONTAMINATION

The Brenntag source area is characterized by historic releases of what are believed to be pure phase PCE and possibly TCE, as well as petroleum products and other unidentified organic compounds. Residual non-aqueous phase liquid (NAPL) sources may be present below the water table in the source area as suggested by COPC concentrations in the groundwater, but none were detected during RI sampling activities. The VOC plume in the alluvial aquifer extends from the Brenntag source area to the Yellowstone River. A portion of the plume discharges into the AJ Gravel pond. The bedrock aquifer downgradient from the Brenntag source area does not appear to be impacted by VOCs.

Geochemical evidence suggests that groundwater within and downgradient of the Brenntag source area is undergoing active anaerobic biodegradation. Statistical evaluation of monitoring well data within the Brenntag source area indicates that there is no upward or downward trend in the concentration of TCE and cis-1,2-DCE over time. Decreasing trends of PCE and TCE, coupled with increasing concentrations of the daughter products cis-1,2-DCE and VC, were observed in downgradient plume monitoring wells. These statistics support the geochemical data that suggest biodegradation is active. Groundwater modeling results suggest that the overall length and width of the COPC plume within and downgradient of the Brenntag source area have stabilized.

The Beall source area is characterized by historic releases of waste water likely containing TCE used in tank trailer cleaning operations. No evidence of NAPL release was detected in soil boring samples or indicated by maximum concentration of COPCs in groundwater samples. The VOC plume in the alluvial aquifer extends from the Beall source area toward the Yellowstone River. A portion of the plume discharges into the Coulson irrigation ditch during the non-irrigation season. The bedrock aquifer downgradient from the Beall source area has been impacted by low levels of VOCs.

Aerobic biodegradation may be occurring within the Beall source area and downgradient to Lockwood Road; however the geochemical evidence of degradation is limited. Statistical analysis of groundwater data suggests there is no upward or downward trend in the concentration of TCE or cis-1,2-DCE; the source of contamination appears to be stable. Groundwater modeling results support a conclusion that the plume downgradient of the Beall source area is slowly increasing in size.

A groundwater-surface water mixing model was developed to assess the impact of groundwater contamination on surface water quality in the Yellowstone River. Model results suggest that the incremental impact of COPCs in groundwater on surface water quality in the Yellowstone River would not be measurable at the lowest mean monthly flow in the river for the period of record.

BASELINE HUMAN HEALTH RISK ASSESSMENT

Baseline human health and ecological risk assessments were conducted under the RI following EPA guidelines. Media samples were collected at appropriate locations across LSGPS as outlined in the RIWP for use in characterizing baseline human health and ecological risks.

For the human health risk assessment, both cancer risks and noncancer hazard indices were calculated for an array of current and future residential and industrial exposure scenarios at the LSGPS. Both the reasonable maximum exposure and a central tendency exposure scenario were considered for each receptor selected for quantitative assessment. Because of its considerable size, the LSGPS was divided into exposure subareas for the evaluations, resulting in discrete human health risk assessments in 9 areas: Area A (source and nonsource areas), Area B (source and nonsource areas), Area C, the AJ Gravel pond, the Coulson irrigation ditch, the wetland area on Cerise Road, and the Yellowstone River. The latter three areas had no COPCs retained for quantitative human health risk assessment; therefore, risks and hazards were calculated for the 6 remaining areas.

Cancer risks associated with exposure to COPCs classified as carcinogens were characterized as an estimate of the probability (excess risk) that an individual will develop cancer over a 70-year lifetime as a direct result of exposure to potential carcinogens. For example, a cancer risk of 1×10^{-6} indicates that an individual has a 1-in-1 million probability of developing cancer during a lifetime as a result of the assumed exposure conditions.

The National Oil and Hazardous Substances Pollution Contingency Plan (NCP) indicates that for known or suspected carcinogens, acceptable additional cancer risk falls within a range between one person in ten thousand (1 x 10^{-4}) and one person in a million (1 x 10^{-6}), known as the risk management range for this risk assessment. Risks below 1 x 10^{-6} are considered to be insignificant. Risks above 1 x 10^{-4} may indicate the need for further evaluation or remediation.

The human health risk assessment concluded that the following scenarios are within the risk management range or considered to be insignificant risks under the NCP:

- Resident adults and children in each of the subareas, assuming only indoor air inhalation exposures as a result of subsurface vapor intrusion.
- Resident adults who use contaminated well water to wash cars or irrigate their lawn in each of the subareas.
- Resident adolescents who recreate with contaminated well water in kiddie pools or sprinklers in each of the subareas.
- Recreators who fish from or wade/dip their arms in the AJ Gravel pond.
- Utility/construction workers in any of the subareas.
- Industrial workers in Area A nonsource, Area B source, Area B nonsource, and Area C subareas who use the public water supply or are supplied an alternate source of drinking water.
- Resident adults and children in Area A nonsource. Area B nonsource, and Area C subareas who use groundwater as a potable water source for whole-house use and/or drinking water source.
- Industrial workers in Area A nonsource. Area B nonsource, and Area C subareas who use groundwater as a potable water source for interior use and/or drinking water source.

Those scenarios and receptors that had cancer risks that may indicate the need for further evaluation or remediation are as follows:

- Industrial workers in Area A source subarea that use groundwater as a potable water source for interior use and/or drinking water source but may spend 4 hours of each workday in contact with Area A source subarea groundwater.

- Resident adults and children in Area A source and Area B source subareas that use contaminated groundwater for whole-house use, including bathing, drinking, and washing.

- Industrial workers in Area A source and Area B source subareas that use contaminated groundwater for unrestricted workplace use, including drinking and washing.

Important uncertainties in these reasonable maximum exposure conclusions were summarized in the baseline human health risk assessment.

ECOLOGICAL RISK ASSESSMENT

The baseline ecological risk assessment included a detailed screening of all detected chemicals in each medium sampled at the LSGPS specifically for ecological effects. The most conservative available ecological screening benchmarks were employed along with updated toxicity information in an attempt to close data gaps identified in the screen. The LSGPS ecological risk assessment found:

- All surface water concentrations at the LSGPS (including the maximum detected concentrations present in the AJ Gravel pond as well as modeled concentrations in the Yellowstone River) were below conservative direct contact and food-chain-protective aquatic benchmarks.

- All sediment concentrations at the LSGPS were below sediment benchmarks.

- All soil concentrations at the LSGPS were below soil benchmarks.

- A conservative desktop food model was employed to evaluate top-level avian carnivores, such as the bald eagle, because of that species' special status and possible home range overlap with the LSGPS. Based on an assumption that 100 percent of the eagle's diet was contaminated fish from the AJ Gravel pond (contaminated with the maximum detected concentrations of VOCs in any surface water body at the LSGPS), and that 100 percent of the eagle's daily water intake was contaminated at the maximum VOC concentration, no unacceptable risks to bald eagles were concluded.

The results of the ecological risk assessment for the LSGPS indicate the site does not pose an unacceptable risk to ecological receptors. Based on these findings, no action is required to address ecological risk at the LSGPS.

CONCLUSIONS AND RECOMMENDATIONS

The RI identified two sources of groundwater contamination within the site: the Brenntag source area and the Beall source area; however, the full extent of soil contamination at these source areas has not been defined. The groundwater within and downgradient of these sources contains chemicals at concentrations greater than both federal and state regulatory standards and poses unacceptable risks to human health. Soil contamination at the site does not pose unacceptable risks to humans. Contamination does not pose an unacceptable risk to ecological receptors at the site or within the Yellowstone River.

Some data gaps have been identified and additional work is recommended at the site to address these data gaps. The following additional work is recommended at the Brenntag source area and the portion of the plume downgradient of the Brenntag source area:

- Analysis of groundwater samples collected from wells MW101, MW102, and MW104 for other organic compounds such as petroleum hydrocarbons, ethanol and methanol to identify and quantify the carbon mass in this area in order to evaluate attenuation and reductive dechlorination processes.
- Installation of soil borings and collection of subsurface soil samples in the vicinity of wells PT001 through PT004 and the tank farm area to delineate the extent of source material and to evaluate the extent of potential NAPL sources in this area.
- The bottom of the AJ Gravel pond needs to be profiled in order to calculate the mass of COPCs entering the pond and estimate the contaminant mass loss from the groundwater to surface water.

The following additional work is recommended at the Beall source area and portion of the plume downgradient of the Beall source area:

- Investigation by excavation of the oil water separator and steam clean bay drain lines to delineate the location of soil contamination beneath these features.
- Installation of a new upgradient monitoring well should be installed further east of MW202 in order to monitor background water quality conditions at the facility.
- Installation of up to five additional monitoring wells within the bedrock aquifer to further delineated vertical and horizontal extent of VOC contamination.
- Installation of two to three additional alluvial aquifer monitoring wells in the residential area northwest of the Beall Trailer facility and northeast of MW210 to delineate the higher concentrations of VOCs in the portion of the plume downgradient of the Beall source area and downgradient of U.S. Highway 87 East.
- Installation of monitoring wells to further define the boundary of the plume in the vicinity of wells MW207 and MW209. The current plume boundary in this area is tentatively defined by residential wells located east of MW207 and MW209.

1.0 INTRODUCTION

This remedial investigation (RI) report presents the geologic, hydrogeologic, and chemical data gathered during the investigation field efforts conducted at Lockwood Solvent Groundwater Plume Site (LSGPS) from July 2002 through October 2002. Tetra Tech EM Inc. (TtEMI) received Task Order No. 10 from the Montana Department of Environmental Quality, Remediation Division (DEQ/RD), under DEQ Contract No. 402014. The purpose of this task order is to prepare an RI report for the LSGPS. This RI report includes the required elements as delineated by Task No. 2 in Task Order No. 10.

The LSGPS is a 580-acre site on the outskirts of Billings, Montana, that has been found to have chlorinated solvent contamination in soils and groundwater. The primary contaminants of potential concern (COPC) are the volatile organic compounds (VOC), tetrachloroethene (PCE), trichloroethene (TCE), dichloroethene (DCE) (including both cis- and trans- isomers), vinyl chloride (VC), and carbon tetrachloride. On December 1, 2000, the LSGPS was officially placed on the U.S. Environmental Protection Agency's (EPA) National Priorities List. Investigation and data collection activities were conducted as described in the remedial investigation work plan (RIWP) (TtEMI 2002a), final work plan for the LSGPS groundwater monitoring program (Pioneer 2001a), and the final groundwater and surface water monitoring work plan (TtEMI 2002c).

1.1 REPORT ORGANIZATION

This RI is organized with text divided into sections, followed by tables and figures, which are followed by appendices. The contents of Sections 1.0 through 8.0 are briefly described below.

Section 1.0, Introduction—describes the report organization, report purpose and objectives, the site and site history, and summarizes previous and ongoing investigations.

Section 2.0, Investigation Procedures and Field Methods—describes the investigative procedures, field methods and activities, quality assurance (QA) and quality control (QC) methods and samples, handling of investigation-derived waste, opportunistic sampling, and deviations from the RIWP.

Section 3.0, Physical Characteristics and Environmental Setting—describes the LSGPS climate, geology, surface water hydrology, hydrogeology, population data, and land use.

Section 4.0, Nature and Extent of Contamination—summarizes the current understanding of the nature and extent of contamination based on field observations, field parameters, and validated analytical data results from soil, surface water, sediment, and groundwater samples. In addition, this section identifies contaminants of potential concern, sources of contamination, and data gaps.

Section 5.0, Contaminant Fate and Transport—summarizes the properties of contaminants of potential concern, fate and transport processes, and evaluates hydrogeologic information and fate and transport modeling.

Section 6.0, Human Health Risk Assessment—summarizes the human health exposure assessment, toxicity assessment, risk characterization, uncertainty analysis, and human health risk-based remedial action objectives.

Section 7.0, Ecological Risk Assessment—summarizes the ecological exposure assessment, toxicity assessment, risk characterization, uncertainty analysis, and ecological risk-based remedial action objectives.

Section 8.0, Summary and Conclusions—presents RI report conclusions and recommendations and describes the feasibility study (FS) process.

The appendices follow the text and are as follows:

- Appendix A – Field Logs
- Appendix B – Photographic Logs
- Appendix C – Proposed and Actual Samples Collected
- Appendix D – Monitoring Well and Borehole Logs
- Appendix E – Monitoring Well Construction Data Tables
- Appendix F – GPS and Other Location Information
- Appendix G – Laboratory Analytical Results
- Appendix H – Data Validation Reports and QA/QC Summary
- Appendix I – Climate Data
- Appendix J – Aquifer Test Graphical Solutions
- Appendix K – Monitoring Well Water Level Hydrographs
- Appendix L – Contaminant Fate and Transport Modeling
- Appendix M – Risk Assessment Methodology and Results
- Appendix N –Ecological Risk Assessment Tables
- Appendix O – Toxicity Profiles and Benchmarks for COPCs

1.2 PURPOSE AND OBJECTIVES

The purpose of the RI is to collect data necessary to adequately characterize the site for the purpose of developing and evaluating effective remedial alternatives that address human health and environmental risks at the site. Activities developed and conducted under the LSGPS RI included project scoping, data collection, data evaluation and interpretation, and risk assessment. Activities for the LSGPS RI were performed in accordance with the *Final Remedial Investigation Work Plan, Sampling and Analysis Plan, and Health and Safety Plan, Lockwood Solvent Groundwater Plume Site* (Tetra Tech 2002a). The primary objectives of the RI for the LSGPS include the following:

- Delineating the nature and extent of contamination at the LSGPS, including identifying and characterizing contaminant sources
- Collecting data necessary to assess the baseline risks to human health and the environment posed by contamination at the LSGPS
- Collecting site-specific data necessary to develop and evaluate viable remedial alternatives

The primary objective of this document is to describe the results of the RI at the LSGPS. This RI report has been prepared in accordance with EPA's Guidance for Conducting Remedial Investigations and Feasibility Studies under the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) Interim Final (EPA 1988a). In addition, this RI report summarizes historic site activities, response efforts, and other information pertinent to characterizing the LSGPS.

1.3 SITE DESCRIPTION

The LSGPS is located on the outskirts of Billings, Montana, in the Northeast ¼ of the Northeast ¼ of Section 34, the North ¼ of Section 35, a portion of the Southeast ¼ of Section 27, and most of Section 26, Township 1 North, Range 26 East of the Montana Principal Meridian, in Yellowstone County, Montana. Figure 1-1 presents the general LSGPS location and the LSGPS vicinity map is presented on Figure 1-2. The site encompasses approximately 580 acres. The boundaries of the site extend from Rosebud Lane on the south and Klenck Lane and Maier Road on the east to the Yellowstone River on the north and west. The LSGPS consists of residential and light industrial commercial facilities. At this time, the primary source of domestic use water in the LSGPS is from the Lockwood Water and Sewer District (LWSD) Public Water Supply. However, other domestic (such as irrigation), commercial, and nondomestic use water is known to come from the shallow aquifer via several individual wells.

The area is zoned controlled industrial (minor industry) according to the Yellowstone City/County Planning Department. The current land use within the LSGPS includes residential, commercial, and "light" industrial. Examples of commercial and light industrial businesses in the area include trucking, vehicle repair, truck tank manufacturing, chemical repackaging, machine shops, and auto salvage.

Based on the geography of the LSGPS and DEQ/RD's desire to present site figures on readily reproducible size paper, the LSGPS has been broken into three geographic areas. These three areas (Figure 1-2) are as follows:

- Area A: The northern part of the LSGPS, which includes the area west of Klenck Lane; north of the railroad tracks, Taylor Place, and Coulson irrigation ditch; and south and east of the Yellowstone River
- Area B: The southeastern part of the LSGPS, generally including the area north of U.S. Highway 87 East and a line south of the Beall Trailers of Montana, Inc. (Beall) facility, west of a line extending south from the east end of Gordon Drive, east of the Interstate Highway 90 Lockwood interchange, and south of Taylor Place
- Area C: The southwestern part of the LSGPS, including the area west of the Interstate Highway 90 Lockwood interchange and Steffes Road, south of Coulson irrigation ditch, east of the Yellowstone River, and north of the lower Lockwood irrigation ditch

1.4 SITE HISTORY

This section provides a chronology of the history of the LSGPS.

The LWSD, formerly known as the Lockwood Water Users Association, is located at 736 Cerise Road and provides public water to a portion of the LSGPS. Prior to 1991, the LWSD treatment plant obtained its water from four supply wells located near the Yellowstone River and within the current boundaries of the LSGPS. In 1986, a CENEX petroleum pipeline ruptured near the LWSD Treatment plant wells. Subsequent investigations of the pipeline rupture and related leaking underground storage tanks (LUST) revealed elevated levels of benzene, toluene, ethylbenzene, and xylenes in the LWSD treatment plant wells. TCE was also detected in the LWSD Treatment plant wells at a concentration of 2.0 micrograms per liter (µg/L). Although this is below EPA's maximum contaminant level (MCL) of 5.0 µg/L, DEQ/RD ranked the LSGPS as a high-priority site under the Montana Comprehensive Environmental Cleanup and Responsibility Act (CECRA). This step was taken because a known release of TCE to the groundwater at levels below water quality standards had occurred. The MCL is the maximum permissible level that a contaminant in water can be delivered to any user of a public water system; MCLs are found in 40 Code of Federal Regulations (CFR) 141 and 142, and EPA 2002f.

From 1991 to the present, the LWSD Treatment plant diverted surface water from the Yellowstone River for its public water supply system, and it currently maintains three production wells for potential future use.

In June 1998, the DEQ/RD's Site Response Section performed an integrated assessment of the LSGPS through its multisite cooperative agreement with EPA. The investigation focused on collecting samples upgradient of the LWSD Treatment plant wells and the petroleum release sites, and included 23 soil samples and 30 groundwater samples from Geoprobe holes or existing wells. As a precautionary measure, DEQ/RD personnel also collected 14 groundwater samples from residential wells in the northern portion (Lomond Lane area) of the currently identified LSGPS. Results from this study and subsequent sampling tentatively identified Beall as a potential source for TCE and its chemical breakdown products in the groundwater. In addition, nine residential wells, mainly in the Lomond Lane area, exceeded MCLs for PCE, TCE, cis-1,2-DCE, VC, and/or carbon tetrachloride.

The DEQ/RD immediately provided bottled water to residents whose wells contained groundwater contaminants exceeding or approaching MCLs. The DEQ/RD tentatively identified 21 additional residential wells and 10 commercial wells that were affected by VOCs; however, the well water from those locations was not used for human consumption. In September 1998, the DEQ/RD initiated a second investigation using CECRA authorities and funding to identify the source of VOC contamination in the Lomond Lane area.

At the request of the DEQ/RD (May 1999), the Agency for Toxic Substances and Disease Registry (ATSDR) completed a health consultation and determined there was a short-term exposure risk for people showering or bathing in the contaminated groundwater (ATSDR 1999). The ATSDR recommended that residents be provided with alternative whole-house water within 1 year.

In response to the ATSDR's findings, the DEQ/RD requested that EPA provide whole-house alternative water, control the sources of groundwater contamination, perform indoor air quality investigations and mitigation (if necessary), continue groundwater monitoring, and proceed with preparation of a hazard ranking system package for the LSGPS to evaluate whether NPL listing was warranted. EPA's Emergency Removal Program responded to the DEQ/RD's request for assistance at the LSGPS under the management of Peter Stevenson, EPA's On-Scene Coordinator, while David Williams, EPA's Region 8 hazard ranking system NPL Coordinator, prepared the hazard ranking system package.

A PCE source investigation in the Lomond Lane area began in September 1999, and available data suggested a potential PCE source on the property of Brenntag West Inc. (formerly known as hci Dyce Chemical Inc.) (Brenntag). Under EPA direction, Brenntag personnel continued the source investigation on their property; however, they were unable to identify a source of the PCE contamination.

Between September 1999 and April 2000, the indoor air from seven homes in the Lomond Lane area was tested for the presence of chlorinated solvent vapors. EPA initially sampled the indoor air at two residences located above the highest concentrations of VOCs in the groundwater. Both of these houses used cisterns for whole-house water; therefore, any concentrations detected in the indoor air were assumed to originate from groundwater vapor migration. The September sampling did not indicate a concern with air concentrations in living spaces; however, one residence did have contaminant levels at the screening values in the crawl space below the house.

EPA returned to the same two homes in January 2000 and sampled again to evaluate wintertime indoor air concentrations. Based on EPA's screening level of 3.1 micrograms per cubic meter ($\mu g/m^3$) for PCE in ambient air, results indicated a concern with vapor contaminant concentrations in living spaces. PCE concentrations detected in the air in living spaces at the two homes ranged from 30 to 130 $\mu g/m^3$. EPA performed vapor mitigation in these homes, and post-mitigation sampling indicated a reduction of PCE vapors to concentrations below screening levels. Based on these results, EPA sampled indoor air at five additional homes in the Lomond Lane Area. Three residences had PCE concentrations below the 3.1 $\mu g/m^3$ screening level (ranging from a method detection limit of 0.03 $\mu g/m^3$ to 0.9 $\mu g/m^3$), and two residences had PCE concentrations above 3.1 $\mu g/m^3$ (ranging from 3.5 to 9.1 $\mu g/m^3$). EPA provided information regarding mitigation techniques.

Groundwater monitoring occurred in the fall of 1999, during the initial investigations performed by EPA, and again in June 2000. DEQ/RD personnel performed groundwater monitoring of the LSGPS in November 2000. During the summer of 2000, EPA's Emergency Removal Program extended the LWSD public water supply line to the Lomond Lane Area, and 14 residences were connected to the public water supply by August 2000.

Beall requested that the DEQ/RD allow it to initiate actions under Montana's CECRA. The DEQ/RD and EPA replied that they planned to conduct a remedial investigation/feasibility study (RI/FS) for the LSGPS using EPA funding, with DEQ/RD acting as the lead agency. As a result, the agencies did not have the resources to oversee the efforts of all individual property owners within the LSGPS, and would not review or provide approval for the work Beall proposed to perform. The agencies stated that Beall was free to proceed with implementing the sampling proposed in its work plan without additional approval or comments from the agencies, and that any data Beall generates that meets QA requirements would be reviewed for screening and planning uses in the RI/FS. Beall completed field work for Phase A of its interim action work plan in October 2000 (Pentacore Resources, LLC 2001). Phase A was designed to identify the COPCs at Beall's property. Phase B was designed to characterize the extent and magnitude of the impacts of COPCs identified in Phase A, but has not been implemented to date.

The LSGPS was proposed for NPL listing in May 2000. The public comment period ended July 11, 2000, and final listing occurred on December 1, 2000.

DEQ/RD began an indoor-air monitoring program at residences in the Lomond Lane area in April 2001, in accordance with an indoor air monitoring plan (DEQ 2001a). Indoor air monitoring was conducted by DEQ/RD personnel in April, May, and July 2001, and by Pioneer Technical Services Inc. (Pioneer) personnel in October 2001 and February 2002. Sampling under this program was completed in February 2002, and TtEMI compiled results in a comprehensive indoor air sampling and analytical results report (TtEMI 2002d).

DEQ/RD began a monthly groundwater and surface water monitoring and quarterly sampling program in July 2001 following the final work plan for the LSGPS groundwater monitoring program (Pioneer 2001a). Under this program, groundwater wells were monitored monthly for static water levels, and surface water locations were monitored for flows. Groundwater samples were collected at monitoring, residential, and industrial wells in July 2001 (Pioneer 2002a), October 2001 (Pioneer 2002b), and January 2002 (TtEMI 2002b). Quarterly monitoring reports were generated after each quarterly sampling event. The reports described quarterly sampling and monthly monitoring activities, and summarized the data collected during each quarter. A summary of the quarterly monitoring groundwater data is presented in Table 2 of the RIWP (TtEMI 2002a) and additional monitoring data from July 2002 and October 2002 is presented in this RI. Monitoring under this program continued through September 2002, with additional quarterly sampling conducted in April and July 2002. During the summer of 2002, TtEMI conducted field work as outlined in the RIWP (TtEMI 2002a). In addition to the collection of groundwater, surface water, and soil samples, 62 monitoring wells were installed as part of these activities. The groundwater and surface water monitoring program was extended through September 2003 under the Final Groundwater/Surface Water Monitoring Plan (TtEMI 2002c). Site monitoring and sampling were conducted by TtEMI in October 2002 under this work plan.

1.5 SUMMARY OF PREVIOUS AND ONGOING INVESTIGATIONS

The LWSD began construction of a water treatment plant in 1986. In October 1986, LWSD discovered the presence of benzene and chlorinated solvents in the groundwater supply. That discovery led to the initiation of a number of investigations, including the following:

- 1986—CENEX Pipe Leak Investigation (Northern Engineering and Testing)
- 1990—EZ Shoppe Investigation (Western States Tank & Testing)
- 1991—CENEX and Conoco Pipeline Investigation (LSE, Inc.)
- 1991—Phase 1 LUST Trust Program Investigation (Chen Northern, Inc.)
- 1991—EZ Shoppe Investigation (Matney-Frantz)
- 1992—Ken's Interstate Conoco Investigation (Braun Intertec)
- 1992-1993—LUST Trust Program Investigation (Tetra Tech, Inc.)
- 1993—Montana Department of Health and Environmental Services/Water Quality Bureau Field Investigation (MDHES/WQB)
- 1994—EZ Shoppe Corrective Measures (Resource Technologies, Inc.)
- 1995—Phase IV LUST Trust Investigation (MSE/HKM)
- 1998—Billings Lockwood Pumping Test Investigation and Groundwater Monitoring (MSE/HKM)
- 1998-1999—Integrated Site Assessment (Pioneer/DEQ)
- 1998—Lockwood-Lomond Lane Sampling Investigation (Pioneer/DEQ)
- 1999—ATSDR Health Consultation (ATSDR)
- 1999—Volatile Organic Compound (VOC) Groundwater Plume Delineation and Potential Source Area Assessment (Lockheed Martin Technology Services Group)
- 2000—Site Investigation, Dyce Chemical Facility, (Maxim Technologies, Inc)
- 2001—Beall Interim Action Phase A, (Pentacore Resources, LLC)
- 2001—Preliminary Residential Air Sampling, (DEQ 2001a)
- 2001-2002—Groundwater Monitoring, (Pioneer 2002a, 2002b)
- 2002—Final Quarterly Monitoring Report January 2002 (TtEMI 2002c)
- 2002—RI report for the LSGPS (TtEMI, in progress)

A detailed description of the individual investigations and findings prior to the 2002 RI sampling event (TtEMI 2002a) can be found in the data summary report for the LSGPS (Pioneer 2001b), and is not repeated here.

2.0 INVESTIGATION PROCEDURES, FIELD METHODS, AND DEVIATIONS FROM THE WORK PLAN

This section describes the procedures used to collect data during LSGPS RI field activities, including utility clearance, sampling methods and equipment, water level and flow measurements, surveying, analytical program and data validation, and management of investigation-derived waste. This section also describes any deviations from the investigation procedures and field methods outlined in the RIWP (TtEMI 2002a), as well as the rationale for the deviations. Deviations from the RIWP were implemented only after they had been reviewed and approved by the DEQ/RD.

Sampling requirements for this project called for collecting and analyzing samples from groundwater, surface water, sediments, and soil. Field activities followed the procedures and methods detailed in the RIWP (TtEMI 2002a), except where noted, and were performed in accordance with TtEMI's health and safety plan for the LSGPS. Copies of field logs are included in Appendix A and a photographic log is included in Appendix B. A summary of proposed and actual samples collected under the RI is included as Appendix C. Figures 2-1 through 2-5 show the proposed and actual sampling locations.

2.1 UNDERGROUND UTILITY LOCATION AND CLEARANCE

Preliminary utility locations were evaluated by reviewing several information sources, including the following:

- Morrison Maierle (underground water line maps)
- Qwest (underground phone lines)
- AT&T Broadband (underground cable lines)
- Yellowstone Valley Electric (underground electrical lines)
- Montana Dakota Utilities (underground gas line maps)

The LWSD and Lockwood Irrigation District were consulted when conducting sampling and drilling activities near their utilities. All information was reviewed to identify proposed sample locations that might intersect or otherwise interfere with known utility corridors. In addition, TtEMI contacted the

Utility Locate Service before starting intrusive or subsurface field activities. TtEMI also contracted with One Call Locators in Billings to locate all underground utilities within the property fence at the Beall Trailers facility. TtEMI consulted with Brenntag when conducting drilling and sampling activities at their property.

The utility locator was responsible for clearing all proposed sample locations that required subsurface investigation. TtEMI was responsible for confirming proposed locations had been cleared before subsurface activities began. For alternate or opportunistic sample locations, TtEMI coordinated with DEQ/RD to evaluate and finalize the sample locations.

2.2 SOIL SAMPLING PROCEDURES AND DEVIATIONS FROM THE WORK PLAN

The following sections describe the sampling methods and equipment used for collecting surface and subsurface soil samples, field screening for nonaqueous phase liquids (NAPL) in soil using the Flute, Inc. ribbon sampler, and installing soil vapor implants. Any deviations from the work plan are included in the sampling methodology sections.

2.2.1 Surface Soil Sampling

Sampling of surface soils followed the procedures provided in standard operating procedure (SOP) 005 (Appendix B in the RIWP, TtEMI 2002a). Sample locations are shown on Figures 2-1 through 2-5. Grab surface-soil samples were collected from a discrete location and depth between 0 and 2 feet below ground surface (bgs) using a hand-operated core sampler with plastic sleeves. If a layer of vegetation was present at a surface soil sampling location, the organic material was removed before coring. The stainless steel hand-operated core sampler was driven into the surface soil (see Photograph No. 1, Appendix B). The core was then removed and the soil sample was cut from the plastic sleeve and placed into a glass sample container.

Deviations from the RIWP. Four surface soil samples were not collected because of conditions encountered in the field. Surface soil sample SS103 at the Brenntag facility was not collected because of hand auger refusal in an asphalt area. Surface soil samples SS200, SS201, and SS202 at the Beall Trailers facility were not collected because it was decided that drilling through the concrete floor was not practical at that time.

2.2.2 Subsurface Soil Sampling

Subsurface soil samples were collected by using hydraulic push and hollow-stem auger sampling methods. Hydraulic direct-push methods were also used to drill boreholes for Flute ribbon sampler installation (Section 2.2.3) and temporary borehole installation for groundwater sampling (Section 2.3.4). Borehole logs are included in Appendix D. Proposed and actual sampling locations are shown on Figures 2-1 through 2-5. A description of the hydraulic direct-push system and hollow-stem auger is provided in the following text.

Hydraulic Direct-Push Borehole Soil Sampling

The hydraulic direct-push technology consists of a hydraulic drive-point system mounted on the rear of a truck. The operation of the sampler followed general practices listed in standard SOP 005 (Appendix B of the RIWP). The direct-push rig used a hydraulic hammer probe to drive a 1.5-inch-diameter core barrel sampler. The method used an open tube or closed piston sampler, depending on the stability of the soils encountered.

To obtain a continuous lithologic sample using both methods, the barrel sampler was fitted with a clear polymer liner. A new liner was installed into the sampler before each drive. During open-tube sampling, the lined sampler was fitted with a drive shoe and driven to the desired depth. The barrel sampler was retracted, the sample liner was removed from the sample barrel, and the soil sample was visually logged. If the open borehole began to cave or slough into the borehole, a closed piston sampler was employed. Using this method, the sample barrel was equipped with a piston and a drive point that locked into the cutting shoe. This step allowed the sealed barrel sampler to pass through the sloughed material and to be opened at the desired depth. The sample barrel was then driven to the starting depth of the drive with the piston assembly locked. The piston assembly was then released from the cutting shoe allowing it to move to the top of the sampler as the liner was filled during the drive. The barrel sampler was then driven an additional 2 feet. The sampler was then removed from the hole, and the liner was removed and the soil core was logged.

A soil sample was collected by cutting a 4-inch long section of the soil core and placing the sample into the sample container and put in a cooler with ice. A portion of the sample was also placed into a Ziploc bag to measure headspace soil gas using a photoionization detector (PID).

Hydraulic direct-push borings were abandoned by sealing the borehole with granular bentonite to the surface. Grouting material was emplaced from the bottom to the top of the boring with a tremie pipe. Asphalt patch material was used to repair the top of the borehole at sampling locations where asphalt was present.

Deviations from the RIWP. Two soil samples (at locations SB100 and SB101) were to be collected at the LSGPS from soil borings using hydraulic direct-push technology (Figure 2-2). However, no soil samples were collected at borings SB100 and SB101 using the direct-push method because field observations PID screening, odor and visual observations) indicated no obvious contamination was present in the borehole soil.

Hollow-Stem Auger Borehole Soil Sampling

A majority of the soil borings for the LSGPS were installed using a hollow-stem auger drill rig. Soil borings were either continuously cored or cored at selected intervals.

Subsurface soil samples were collected with a split-spoon sampler using the procedures described in SOP 005 (Appendix B of the RIWP). Soil samples were collected by taking a 4-inch long section of the soil core selected for sample analysis and immediately placing the sample into a glass sample container. A portion of the sample was also placed into a Ziploc bag to measure headspace soil gas using a PID.

At the Brenntag facility, a minimum of three soil samples were collected for laboratory analysis from each soil boring based on observed soil boring lithology and results from the ribbon sampler. For risk assessment purposes, at least one of the samples collected from boreholes at the Brenntag facility was collected from the 2- to 10- foot depth interval.

At the Beall facility, four soil samples were collected for analysis from each soil boring based on observed lithology, and they include one soil sample collected from the base of two gravel layers present in lithology at the facility, another soil sample collected from soil above the water table, and one sample collected from the alluvial material directly on top of the bedrock. Additionally, up to two additional samples were collected from each soil boring based on PID field screening results. For risk assessment purposes, at least one of the samples collected from boreholes at the Beall facility was collected from the 2- to 10- foot depth interval.

For boreholes completed in areas A, B, and C outside of the Brenntag and Beall facilities, a maximum of two subsurface soil samples were collected per borehole based on field observations and PID measurements.

Deviations from the RIWP. On several occasions, heaving sands, large boulders, or poor sample recovery prevented a subsurface soil sample from being collected at a specific location in a borehole. At the Brenntag facility, not all three soil samples were collected in each borehole based on soil lithology and lack of contamination found on the ribbon sampler. At the Beall facility, not all four subsurface soil samples were collected in each borehole because distinct gravel layers were not always present or the PID reading at a target layer was zero. In some instances, a subsurface soil sample was collected immediately above an observed clay layer if an obvious gravel layer was not observed in lithology at the Beall facility.

Two boreholes (SB209 and SB210) were not drilled at the Beall Trailers facility because it was determined that additional data from the boreholes were not needed because of the limited contamination found in other soil samples collected from boreholes in the vicinity.

A proposed subsurface soil sample from the 10-foot depth was not collected from the borehole at monitoring well MW112 based on PID screening readings of zero parts per million (ppm) and no obvious staining or odors.

No soil samples were collected from monitoring wells MW120, MW121, MW125, and MW307 (see Figures 2-1, 2-5, 4-1, and 4-7) for the following reasons:

- The borehole at well location MW120 was drilled twice on July 10 and 11, 2002; the holes caved in because of heaving sands and large gravels. A soil sample could not be collected using the split-spoon sampler in these boreholes. The well at MW120 was redrilled a third time on July 16, 2002, and the monitoring well installed without using the split-spoon sampler. Therefore, no soil sample was collected from well location MW120.
- No soil samples were collected from monitoring wells MW121 and MW125 because PID measurements were 0.0 ppm for all headspace samples monitored in each borehole.
- One subsurface soil sample was collected from the 9- to 11-foot bgs interval at the boring located at MW307; however, the sample was inadvertently omitted from shipment to the laboratory. No other soil samples were collected from this boring.

The RIWP stated that soil sampling for VOCs was to employ the use of the EasyDraw syringe and PowerStop handle system (TtEMI 2002a). The system required three steps including (1) loading of the sampling device by inserting the syringe on the appropriate slot on the handle, (2) collecting the sample by pushing the syringe into the soil being sampled, and (3) ejecting the sample into a laboratory-supplied vial by removing the handle device and pushing on the syringe plunger. However, the average grain size of the subsurface soil material was too large and prevented an adequate sample volume from being recovered with the Easy Draw sampling system. As a result, the use of the EasyDraw and PowerStop handle system was not used. All soil samples were collected by placing soil directly into glass sample containers.

As part of the collection of subsurface soil samples at the LSGPS, 12 subsurface soil samples were proposed to be collected from six monitoring well locations (MW110, MW117, MW202, MW204, MW301, and MW308) and analyzed for physical parameters including bulk density, porosity, moisture content, and grain size. Two soil samples were to be collected from each of the six well locations. One subsurface soil sample was to be collected from the saturated zone, and one was to be collected from the unsaturated zone. The samples were to be collected by inserting Shelby tubes or brass sleeves into the split-spoon sampler of the hollow-stem auger drilling apparatus and removing and capping the ends of the tubes or sleeves with plastic caps.

Of the six proposed locations, physical parameter analyses were obtained from only two locations (MW202 and SB203) in Area B. Sampling location MW204 was replaced by location SB203. The drilling crew attempted to collect soil samples for physical parameters in Areas A and C; however, the samples could not be collected because of poor sample recovery (at MW117); gravel and rocks becoming jammed in or crushing the sampling tubes (MW301 and MW308); and in one instance (MW110), the sample container was damaged.

Six subsurface soil samples were collected using hollow-stem auger drilling methods and submitted for total organic carbon analysis, as proposed in the RIWP. Two samples were collected from each investigation area (Areas A, B, and C) to characterize contaminant migration. The six proposed locations (MW110, MW117, MW202, MW204, MW301 and MW308) for total organic carbon analysis were relocated to six alternate locations (MW118, MW127, MW210, MW300, MW305, and MW311) based on field conditions and an evaluation of preliminary sampling data.

2.2.3 Ribbon Sampler Installation and Field Screening

Ribbon samplers were installed using Flute, Inc. technology in six selected hydraulic direct-push borings from the surface through the vadose and saturated zones to bedrock (29.1 to 32.8 feet bgs) to evaluate the presence of NAPL (see Figure 2-2 and Table 2-1). Field screening with the ribbon samplers was conducted based on the locations of suspected sources of contamination at the Brenntag facility. These locations included areas of suspected spills of pure-phase solvents, areas downgradient of the tank farm, and at areas where a previous site inspection performed by Maxim showed elevated levels of VOCs in groundwater (Maxim 2000). Using the data collected by Maxim, a ribbon sampler was placed at a borehole location where VOC concentrations in groundwater exceeded 1 percent solubility of a pure-phase compound. This concentration suggested the possible presence of NAPL in the saturated zone.

The ribbon sampler system consisted of an impermeable flexible liner and an exterior covering on the liner that reacts with NAPL to form a bright dye stain on a white background (Photograph No. 2, Appendix B). The liner and cover system were emplaced via a hydraulic direct-push borehole (Photograph No. 3, Appendix B). The pressurized liner pushed the reactive cover tightly against the borehole wall. Ribbon samplers were deployed for 1.75 to 2.5 hours; the manufacturer requires at least 1 hour deployment for dyes to react with NAPL compounds. The reactive cover was removed by inverting and peeling the liner from the hole. In this manner, the cover did not come in contact with the borehole wall anywhere else as it was removed. The cover was then examined for the presence and extent of layers and globules of NAPL in the borehole (Photograph No. 4, Appendix B).

After sampling, ribbon sampler borings were sealed with granular bentonite to the surface. Grouting material was emplaced from the bottom to the top of the boring with a tremie pipe. Asphalt patching material was used to repair the ground surface at sampling locations where asphalt was present.

Black-colored staining was observed at three of the ribbon sampler locations: RS101, RS102, and RS104 (see Table 2-1 and Photograph Nos. 5, 6, and 7 in Appendix B). The black staining was found to indicate contact with reduced sediments and anaerobic conditions in the subsurface soil and not from the presence of NAPL.

Deviations from the RIWP. The original ribbon sampler installation at location RS104 failed and had to be abandoned because the sampler could not be retrieved due to soil sloughing in the borehole. The failed ribbon sampler was recovered and disposed of properly. Location RS104 was redrilled 5 feet from the previous location, and a new ribbon sampler was successfully installed and retrieved.

2.2.4 Soil Vapor Sampling and Field Screening

As proposed in the RIWP, soil vapor implants were installed at select monitoring well locations to provide the flexibility to sample soil vapor downgradient of the Beall Trailers facility. The implants consisted of an anchor attached to a double woven, stainless steel wire screen, from which tubing was attached to the outside of a monitoring well polyvinyl chloride (PVC) casing with plastic zip ties, and the casing was lowered into an open borehole. Two implants were installed at each selected monitoring well at 10 feet and 20 feet bgs above the water table. Sandpack was placed around the implants approximately 2 feet above and 2 feet below each implant. Granular bentonite was placed between the sandpacks, and was used to fill the annular space above and below the sandpacks. The remainder of the monitoring well PVC casing and screen were installed and constructed as described in Section 2.3.1

As stated in the RIWP, only the installation of soil vapor implants was proposed at the site; no soil vapor sampling was planned under the RI and no soil vapor samples were collected. Any future soil vapor sampling was to be evaluated and performed under a separate task order, as approved by the DEQ/RD.

Deviations from the RIWP. The RIWP proposed that soil vapor implants be installed at five monitoring well locations (MW211, MW212, MW213, MW214, and MW215). However, the only two soil vapor implant locations (MW213 and MW214) were selected because these wells were closest to residential dwellings that may be sampled for VOCs in indoor air.

2.2.5 Opportunistic Soil Sampling

Opportunistic samples, or contingency samples, are samples not specifically proposed in the RIWP, but are generally identified in the planning stages to allow for samples to be collected based on special or unexpected field conditions encountered during the RI field sampling effort. These samples could include samples collected during one event, or several samples collected over time, or could include collecting a sample for supplementary laboratory analysis. Opportunistic samples were only collected as directed by the DEQ/RD.

Opportunistic soil samples were collected in selected soil boreholes based on the results of the ribbon sampler, headspace PID field screening data, and visual observations. Generally, one to three opportunistic soil samples were tentatively assigned to each proposed soil borehole in the RIWP; however, not all of the opportunistic soil samples were collected. Opportunistic soil samples are listed in Appendix C.

2.3 MONITORING WELL INSTALLATION, GROUNDWATER SAMPLING, AND DEVIATIONS FROM THE WORK PLAN

This section describes well installation, development, slug testing, and sampling procedures used at the LSGPS. Deviations from the work plan are included in each subsection. Monitoring well logs are presented in Appendix D, and monitoring well construction data tables are included in Appendix E.

2.3.1 Monitoring Well Installation

Monitoring wells were installed in the boreholes drilled by hollow-stem auger and air rotary drilling methods. As proposed in the RIWP, monitoring well installations followed SOP 020 (Appendix B of the RIWP). New monitoring wells were constructed in accordance with State of Montana monitoring well standards. Material specifications and completion depths were recorded during well construction and documented on monitoring well borehole logs (Appendix D) and in monitoring well construction data tables (Appendix E).

Generally, monitoring wells were constructed using 2-inch, inside diameter, schedule 40 PVC well casing with flush-threaded joints. The well was screened using schedule 40 PVC screen with 0.02-inch machined slots. A filter pack consisting of 10/20 mesh silica was emplaced in the annular spacing. The wells were screened from the alluvium or bedrock interface to approximately 2 feet above the water table to allow for seasonal groundwater fluctuations based on site-wide historical water levels. Screening the wells across the water table also allowed for light NAPL measurement, if present. Therefore, the riser and screen length were determined in the field based on total boring depth and depth to the water table. In general, monitoring wells were constructed with a 20- to 25-foot screen and were screened through gravel and across the bedrock-gravel interface.

The monitoring well screen and riser material was assembled and lowered by hand through the augers to the desired depth. The 10/20 mesh silica sand was poured or tremied through the augers to form the filter pack extending at least 2 feet above the screen. Photograph No. 8 (Appendix B) shows a driller setting the PVC casing while the augers are removed from the borehole. As the filter pack was poured, the auger was slowly raised out of the borehole. Bentonite pellets were poured through the augers and were hydrated with potable water to form an annular seal at least 2 feet thick above the filter pack. A grout mixture of 95 percent cement and 5 percent bentonite was poured or tremied through the augers to create a bentonite seal to within 2 feet bgs. Concrete was then poured in the space above the grout to form a surface seal. The protective well casing was set in the concrete.

The concrete surface seal was sloped to allow for drainage of surface water away from the well. Wells were completed at the surface with either a flush-mount or stickup completion. Most wells were completed with a stickup, with the casing rising above the ground surface approximately 3 feet. Flush-mount surface completions were installed in areas of heavy vehicular traffic or at the property owners' request. The volume of material placed in the borehole to form the filter pack, annular seal, and surface seal was calculated and recorded on the monitoring well completion log or borehole log (Appendix D). Flush-mount and stickup completions are listed in Appendix E.

Two bedrock monitoring wells (MW128 and MW219) were constructed following SOP 020 (Appendix B in the RIWP). These two monitoring wells were installed with a casing advance air rotary drill rig and constructed as a dual-cased well with protective surface 8-inch casing set 5 feet into the top of bedrock and grouted to prevent cross-contamination between the gravel aquifer and the bedrock. The monitoring wells were installed by drilling out the surface casing grout and setting a 2-inch diameter PVC casing and screen. The remainder of the construction and completion procedures for these two monitoring wells is the same as described in the previous text.

Deviations from the RIWP. Seventeen proposed monitoring well locations were relocated because of conditions encountered in the field; a list of the wells and the rationale for re-locating the wells are presented in Table 2-2. Only wells that were relocated more than approximately 50 feet from their proposed location were included in the table. Figures 2-1 through 2-5 show proposed and actual well locations. A list of proposed and actual samples collected during the RI is provided in Appendix C.

Minor changes to the well installation procedures (for example, using a different brand of filter pack material or using a smaller screen slot size) are recorded on the borelogs in Appendix D and logbooks in Appendix A.

2.3.2 Monitoring Well Development and Purging

Monitoring well development and purging activities were performed in accordance with SOP 021 (Appendix B in the RIWP). The monitoring well was allowed to stabilize for a minimum of 24 hours after completion to allow for adequate curing of the grout. Before well development, the depth to water and total depth of the well were measured using a water-level indicator. These measurements were used to determine the well casing volume and minimum purge volume. Before purging, the well was surged by manually raising and lowering a surge block through the water column for a minimum of 10 minutes.

After the well was surged, a portable pump was used to evacuate three casing volumes of water from the well. A new piece of disposable tubing was attached to the pump and lowered to variable depths below the water table to evacuate the water from the well and introduce groundwater into the well from the aquifer. Water quality parameters including pH, specific conductance, and temperature were measured during the removal of each casing volume of water to provide baseline information. Water quality meters were calibrated each day and checked periodically. Water quality parameters were recorded on well development forms (see Appendix E).

If the well dried up before the specified amount of purge water had been withdrawn, the well was allowed to recharge. After the well had recharged, additional water was removed, and the well was purged until a total of three well casings had been removed or the well went dry a second time. After the well was purged dry twice, the well was considered to be developed and a groundwater sample could then be collected (see Appendix E).

Well development and purge water generated was temporarily contained in a 500-gallon tank and subsequently treated and disposed of as investigation-derived waste. See Section 2.8 for additional information on procedures for IDW handling and disposal.

Deviations from the RIWP. An air compressor that used fresh air was also used to develop some of the wells at the LSGPS during RI field activities. This system consisted of attaching a high-pressure air hose to the bottom of a 1-inch diameter PVC pipe and lowering the apparatus to the bottom of a monitoring well. The air compressor was turned on and high-pressure air forced water and sediment out of the well through the 1-inch PVC pipe, which was elbowed and directed into a 55-gallon drum (see Photograph No. 9, Appendix B). The high-pressure air system was able to remove several feet of sediment and sand from the bottom of a monitoring well more efficiently than the proposed methods.

2.3.3 Slug Testing

As proposed in the RIWP, slug removal, or rising head, aquifer tests were conducted at select monitoring wells to estimate hydraulic conductivity in the aquifer underlying the site. A slug removal test consisted of the instantaneous removal of a known solid volume (slug) from the standing column of water in a well, decreasing the hydraulic head in the monitoring well while measuring the rate of water level recovery. The slugs were 10 feet long and consisted of a sealed, 1.5-inch outside diameter, PVC pipe filled with sand.

A few hours before each slug test was conducted, the water level was measured with a water-level indicator to determine the static water level in the monitoring well. The slug was then placed in the monitoring well and allowed to sit for a period of time that allowed for the water level in the monitoring well to return to a static level (usually at least 30 minutes). After the water had returned to a static level, the slug test was performed by rapidly removing the slug from the monitoring well and then measuring the water level as soon as possible, and afterwards, every few seconds until the water level was approximately 90 percent of pre-test levels. The results of the slug tests are presented in Section 3.4.2 and Appendix J.

Deviation from the RIWP. Slug tests were proposed to be performed at nine monitoring wells (MW102, MW110, MW117, MW128, MW202, MW204, MW219, MW301, and MW308). However, monitoring wells MW102, MW202, MW204, and MW219 were replaced with MW107, MW203, MW213, and MW215. These locations were selected based on a review of lithologic information obtained during drilling and were believed to provide more representative hydrogeologic data. Proposed and actual locations of slug tests are also presented in Appendix C.

2.3.4 Groundwater Sampling

Groundwater samples were collected using low-flow sampling methods as described in the RIWP (SOP 015, Appendix B) in the RIWP. Existing monitoring wells were sampled during quarterly sampling and monitoring activities in July 2002. All newly installed monitoring wells were sampled in August 2002. All existing and newly installed monitoring wells were sampled again in October 2002 following the Final Groundwater/Surface Water Monitoring Plan (TtEMI 2002c). In addition to VOC analysis, 26 groundwater samples were analyzed for natural attenuation parameters including total nitrogen, nitrate and nitrite, dissolved hydrogen, ammonia, total iron, total arsenic, sulfate, chloride, alkalinity, total organic carbon, and dissolved gases (methane, ethane, and ethene). Groundwater samples analyzed for natural attenuation parameters were collected from select new monitoring wells in October 2002. The results of the October 2002 sampling event for natural attenuation parameters are summarized in Section 5.2. A second round of groundwater sample collection for natural attenuation parameters occurred in April 2003. The results of the April 2003 sampling event will be summarized under a separate task order.

Groundwater samples were collected from monitoring wells following well development and purging, as follows:

- Monitoring wells with static water levels less than 30 feet bgs were purged and sampled using a 12-volt peristaltic pump (see Photograph No. 10 in Appendix B). Disposable downhole Teflon sample tubing was installed in each well sampled with a peristaltic pump. Water samples for laboratory analyses were collected by disconnecting the downhole tubing from the flow-through cell and reconnecting to the flexible silicon tubing on the peristaltic pump. The pump discharge was reduced to no greater than 0.5 liters per minute. VOC samples were collected by allowing the pump discharge to flow gently down the inside of laboratory-supplied 40-milliliter (mL) volatile organic analysis (VOA) vials preserved with hydrochloric acid. The sample was chilled to maintain a temperature below 4 degrees Centigrade (^{o}C).
- Monitoring wells with static water levels greater than 30 feet bgs were purged with a positive displacement or submersible pump that was decontaminated after each use. The pump was only used for purging until field parameters stabilized. After stabilization, the pump was shut off and slowly removed from the well. Immediately following removal of the pump, a bottom-filling Teflon bailer was lowered into the well and allowed to sink slowly into the water until filled. The bailer was then raised to the surface. Samples were collected by slowly filling a hydrochloric acid-preserved 40-mL VOA vial from the bottom-filling device attached to the bailer. The sample was chilled to maintain a temperature below 4^{o}C.
- Dissolved hydrogen samples were collected from 26 wells that were sampled for natural attenuation parameters. After all sample containers were filled for the other chemical analyses, the inlet tube of the Microseeps Bubble Strip Sampler cell assembly (Photograph No. 11, Appendix B) was attached to the pump, and an outlet tube was inserted into a 5-gallon bucket. The pump was turned on and the cell assembly was checked for leaks according to the manufacturer’s specifications (Microseeps 2003). After any leaks were sealed, the flow rate (in mL per minute) was measured using a 1-liter soda pop bottle placed in the 5-gallon bucket. The flow rate value was entered into a calculation table provided by the manufacturer to determine the appropriate equilibrium time needed to collect a dissolve hydrogen sample. Twenty milliliters of ambient air was injected with a syringe into the cell assembly. The groundwater was allowed to pump through the cell for the appropriate time, allowing dissolved hydrogen to collect in the 20 mL air space in the cell. When the equilibrium time was reached, the pump was turned off and one mL of gas was withdrawn and purged from the syringe. Using the empty syringe, 15 mL of sample gas was removed from the cell and injected into a sample vial sealed with a septum. The sample was chilled to maintain a temperature below 4^{o}C.
- Most of the residential and industrial wells at the LSGPS were sampled from associated spigots outside the building or from the taps inside the building because the top of the wells were constructed so that they would not allow sampling and measurement equipment to be lowered inside the well. Residential wells RW001, RW007, and RW027 were sampled from a spigot on

the side of the home; residential well RW002 was sampled from the kitchen tap; residential wells RW005 and RW006 were sampled from the well head; and industrial wells IW002 and IW003 were sampled from the well head.

During sample collection for natural attenuation parameters, field screening test kits were used to measure additional groundwater conditions. Field screening test kits included ferrous iron, total iron, hydrogen sulfide, manganese, and dissolved oxygen. Field test kit procedures were performed in accordance with the manufacturer's directions that were provided with each kit. The results of the field test kits are summarized in Table 5-2. Additional information on groundwater sampling can be found in Table 2-3.

As proposed in the RIWP, six groundwater grab samples were collected from three boreholes (SB100, SB101, and SB102) drilled at east edge of the Brenntag facility (Figure 2-2). Two groundwater samples were collected from each borehole; one from the top of the water table and one from immediately above bedrock. A disposable 1-inch, inside diameter PVC screen was placed in the open borehole to collect the groundwater sample. A peristaltic pump equipped with disposable tubing was used to purge and sample the groundwater using low-flow methods. No field parameters were collected for groundwater samples collected from open boreholes because of high turbidity and sediments in the water. The groundwater samples were collected in 40-mL VOA vials and immediately placed in a cooler with ice.

Deviations from the RIWP. Groundwater samples at four monitoring well locations were not collected as proposed in the RIWP including existing wells MW034, MW039, MW082, and MW086, all of which are located in Area C (Figure 2-5). These samples were inadvertently omitted from being sent to the laboratory for VOC analysis (see Appendix C, Area C) during the August 2002 RI sampling event. It was later determined that data collected from nearby wells were sufficient for the needs of the RI program and that data from these four wells were not needed at the time. These wells may be included in the next groundwater sampling event scheduled for April 2003.

Several of the proposed locations chosen for natural attenuation sample analysis were revised and relocated based on the preliminary analytical results received from RI groundwater sampling in August 2002. A list of proposed and actual sample locations for natural attenuation parameter analysis is provided in Appendix C.

2.3.5 Opportunistic Groundwater Sampling

Four opportunistic or contingency groundwater samples were collected from monitoring wells installed at the Beall Trailer facility to determine the placement of and need for other proposed monitoring wells. The approximate boundary of groundwater contamination was evaluated by stepping out from monitoring wells already installed. The opportunistic groundwater samples were collected from monitoring wells MW200, MW203, MW205, and MW206 and were analyzed for VOCs using quick-turnaround analyses. The samples were collected prior to well development and were intended for field screening purposes only. The analytical results for these samples were qualitative and were not validated. Table 2-4 summarizes the COPCs detected in these samples. Based on the results of the sample analysis (each sample showed a detection of at least one COPC, and several detections were at levels above screening criteria), it was determined that all wells at and near the Beall Trailers facility be installed as proposed in the RIWP.

Deviations from the RIWP. No deviations were noted regarding the collection of opportunistic groundwater samples.

2.4 SURFACE WATER AND SEDIMENT SAMPLING AND DEVIATIONS FROM THE WORK PLAN

This section describes the field methods used for collecting surface water and sediment samples, collecting water level and flow measurements, opportunistic sampling, and any deviations from the RIWP.

2.4.1 Surface Water and Sediment Sampling

Surface water and sediment samples were collected during quarterly sampling and monitoring activities conducted in April and May, 2002. Surface water and sediment samples were collocated and analyzed for VOCs. Water quality parameters were measured at each sampling location for pH, specific conductance, temperature, dissolved oxygen, oxidation reduction potential, and turbidity. Water quality parameters and additional details are shown in Appendix A.

Collocated surface water and sediment samples were collected first from the most downstream location and last from the farthest upstream sample location so that any sediment disturbances from samples collected upstream did not affect downstream samples. The following sequence of sampling activities occurred at each location:

- A surface water sample was collected
- Water quality parameters were measured
- A sediment sample was collected
- Depth of surface water and water level was measured

Surface water samples were collected as grab samples by directly immersing sample containers under the water surface (see Photograph No. 12, Appendix B) as described in TtEMI SOP 009 in the *Final Groundwater/Surface Water Monitoring Work Plan* (Tetra Tech 2002c).

Sediment samples were collected using a disposable scoop and placing the sediment directly in a glass sampling container, which was put in a cooler with ice. Sediment samples were collected from the top 6 inches of the sediment layer.

Proposed and actual surface water and sediment locations are presented on Figures 2-1 and 2-5 and are listed in Appendix C.

2.4.2 Opportunistic Surface Water and Sediment Sampling

Opportunistic surface water and sediment samples were collected in the Coulson irrigation ditch, a wetland area south of Cerise Road, and the AJ Gravel pit pond in April 2002 (Figures 2-1 and 2-5). These samples were collected during the April 2002 quarterly sampling activity (Tetra Tech 2002c) as a low-flow sampling event using the methods described in Section 2.4.1. The analytical results of these opportunistic samples were intended to be used for the subsequent RI, which began after the flow in Coulson irrigation ditch was scheduled to be increased to high-flow conditions.

Deviations from the RIWP. The RIWP proposed that opportunistic surface water samples be collected during high-flow and low-flow conditions in the Coulson irrigation ditch. Low-flow sample data were collected in April 2002. High-flow samples were proposed to be collected in July 2002; however, the pump that supplied water from the Yellowstone River to the ditch was turned off unexpectedly and no high-flow samples were collected.

2.5 WATER-LEVEL AND FLOW MEASUREMENTS

This section describes water-level and flow measurements collected according to the procedures outlined in the RIWP (TtEMI 2002a) and monitoring work plan (TtEMI 2002c), and discusses deviations from the work plans.

Monthly water levels were measured in monitoring wells and water levels and flow rates were measured at surface water locations at the end of each month from January through October 2002. Water levels and flow rates were measured on a quarterly basis starting in January 2003. Water levels in monitoring wells were also measured during the quarterly monitoring and RI sampling events.

2.5.1 Monitoring Well Water-Level Measurements

Static water levels were measured in monitoring wells according to procedures outlined in SOP 014 (Appendix B in the RIWP). A water-level probe attached to a measuring tape and reel was lowered into a well until the alarm sounded. A measurement was taken by reading the depth from the graduated tape, at a mark located on the north side of the top of the well casing. This value was recorded in a field logbook (see Appendix A). Monitoring well water-level measurements are summarized in Table 2-5.

Deviations from the RIWP. Water-level measurements at 12 monitoring well locations were inadvertently not collected as proposed in the RIWP. These monitoring wells included the following:

MW032
MW033
MW034
MW035
MW039
MW045
MW079
MW080
MW081
MW082
MW083
MW084

It was later determined that data collected from nearby wells were sufficient to meet the RI program objectives. Water levels from these wells will be measured during the next field sampling event scheduled for January 2003.

2.5.2 Surface Water Level and Depth Measurements

Surface water level and depth measurements were taken in Coulson and Lower Lockwood irrigation ditches, AJ Gravel pond (see Photograph No. 13, Appendix B), and a wetland area south of Cerise Road (see Photograph No. 14, Appendix B) using a graduated survey rod and tape measure lowered in the center channel of an open ditch or culvert. Rebar stakes were installed as temporary survey points at selected sampling locations. Surface water level and depth measurements are presented in Table 2-6.

Deviations from the RIWP. Water-level and flow measurements were not taken at surface water location SW011 because of its proximity to surface water monitoring location SW004. As shown on Figure 2-1, SW004 and SW011 are located on either side of Lomond Lane, near the intersection with Island Park Road (see also Section 2.5.3).

2.5.3 Surface Water Flow Measurements

Surface water flow in Coulson and Lower Lockwood irrigation ditches were measured using a Marsh-McBirney water current meter as described in SOP 095 (Appendix B of the RIWP) (see Photograph No. 15, Appendix B). On the occasions when the water in the ditches was not flowing and only standing water was present, no flow measurement was taken and the observation was recorded in the field logbook. A summary of surface water flow measurements at the LSGPS is presented in Table 2-7.

Deviations from the RIWP. The water level and flow rate were not measured at surface water station SW011 during the RI sampling effort. Because this location is across the street from surface water station SW004 (at the intersection of Lomond Lane and Island Park Road), it was determined that data from SW011 were not necessary.

2.5.4 Opportunistic Water Level and Flow Measurements

Opportunistic water level and flow measurements were collected in April 2002 at surface water stations. These monitoring activities are described in further detail in Section 2.4.2.

2.6 GLOBAL POSITIONING SYSTEM LOCATING

The locations of proposed and existing monitoring wells, and all new and existing surface water stations were determined using Global Positioning System (GPS) methods to an accuracy of plus or minus 0.1 foot horizontally and plus or minus 0.01 foot vertically. Horizontal coordinates were reported in accordance with the existing state planar coordinate system. Vertical coordinates were reported as feet above mean sea level. At wells, horizontal and vertical measurements were recorded for the top of well casing and for ground surface immediately adjacent to the well. At surface water stations, horizontal and vertical measurements were recorded for the top of rebar stakes and top of culverts. The results of the GPS locating are found in Appendix F. Photograph No. 16 (Appendix B) shows a typical GPS equipment setup.

Deviations from the RIWP. The vertical position of 10 monitoring wells (MW011, MW017, MW100, MW119, MW121, MW126, MW127, MW210, MW213, and MW308) and three surface water sample locations (SW008, SW010, and SW012) were not able to be taken (no GPS reading) because of obstructions, such as buildings and trees, or the quality of the GPS reading was questionable. As a result, these sample locations were measured in December 2002 from known reference points using a Trimble rotating laser to an accuracy of 0.01 foot. The results of the Trimble survey are summarized in Appendix F.

2.7 ANALYTICAL PROGRAM AND DATA VALIDATION

Samples obtained for laboratory analysis were collected and analyzed according to the procedures outlined in the RIWP (TtEMI 2002a). Laboratory analytical data were evaluated for quality following data validation procedures established by EPA including “National Functional Guidelines for Organic Data Review” (EPA 1999b) and EPA “National Functional Guidelines for Inorganic Data Review” (EPA 1994a). Data that were collected during quarterly groundwater sampling events were validated by a validation subcontractor (Portage Environmental) and data collected for the RI sampling program were validated by a TtEMI chemist. The quality of the data was assessed through the use of field QC samples, which were collected at regularly scheduled intervals as described in the RIWP (TtEMI 2002a). Appendix G contains all laboratory analytical results reports. Appendix H summarizes the results of the data validation and discusses field QC samples.

Deviations from the RIWP. No deviations from the RIWP were noted for the analytical program and data validation procedures.

2.8 MANAGEMENT OF INVESTIGATION-DERIVED WASTE

The management of investigation-derived waste for the RI and quarterly groundwater sampling efforts was conducted in accordance with the RIWP and quarterly groundwater and surface water monitoring work plan. A substantial quantity of investigation-derived waste was generated during the investigations.

Solid matrix investigation-derived waste included the remainder of homogenized soil extracted by hydraulic direct-push samplers and drill cuttings. Liquid matrix investigation-derived waste included wastewater from decontamination procedures, monitoring well development and purging, and preparation of equipment rinsate samples. Soil cuttings and other solid matrix investigation-derived waste that were not obviously contaminated (that is, there was no obvious staining, odors, or product present; and PID readings showed no measurable VOCs in soil) were containerized in drums and subsequently transferred to larger roll-off bins. Heavily contaminated solid matrix investigation-derived waste was segregated and stored in 55-gallon drums. Drums were clearly labeled for inventorying, sampling, and eventual disposal.

Composite investigation-derived waste soil samples were obtained from roll-off bins and 55-gallon drums and sent to the analytical laboratory for analysis to determine proper disposal. The investigation-derived waste soil samples were analyzed for VOCs, semivolatile organic compounds (SVOC), polynuclear aromatic hydrocarbons (PAH), chlorinated and nonchlorinated pesticides, chlorinated herbicides, pH, ignitability, reactivity, and metals. Based on the sampling results, none of the investigation-derived waste soil was characterized as a hazardous waste. The soil was properly disposed of at a local landfill operated by Browning-Ferris Industries. The last shipment of investigation-derived waste soil to the landfill occurred on January 29, 2003. No investigation-derived waste soil remains on site at this time.

investigation-derived waste wastewater was temporarily stored in open stock tanks, treated by pumping the water through a granular activated carbon filter unit, then transferring the water to an on-site 500-gallon tank. One composite investigation-derived waste water sample per 500 gallons of treated water was sent to the laboratory and analyzed for VOCs, SVOCs, PAHs, chlorinated and nonchlorinated pesticides, chlorinated herbicides, pH, flash point, and metals. None of the liquid matrix investigation-derived waste was characterized as a hazardous waste and all COPCs were reported as below the detection limit. After the investigation-derived waste water sample was analyzed and the results reviewed, the liquid matrix investigation-derived waste was disposed of by DEQ/TtEMI in a settling pond at the LWSD. The last volume of investigation-derived waste was disposed of in the settling pond on December 2, 2002. The granular activated carbon filter system will remain at the site for use during future sampling.

3.0 PHYSICAL CHARACTERISTICS AND ENVIRONMENTAL SETTING

Physical characteristics of the LSPGS pertinent to the RI include the climate, geology, surface water hydrology, hydrogeology, and land use. The following sections describe these characteristics in detail.

3.1 CLIMATE

The first weather records taken by the U.S. Army in the Billings area were in 1883, 2 years before the city was incorporated. Records were kept at several locations downtown from that time to the establishment of a Weather Bureau Station at the Billings Airport on June 13, 1934. The airport station served as a State Forecast Center from 1941 to 1953. Current operational National Weather Service facilities and stations in Billings include the Automated Surface Observation System and Doppler Next Generation Radar station at the Billings Airport. Additional weather observation stations are located at the Billings Water Treatment plant and the Billings NWS office. Climate data are provided in Appendix I.

Billings, Montana, is located at an elevation of 3,100 to 3,600 feet above mean sea level. Billings is situated in the transition area between the Great Plains and the Northern Rocky Mountains and has a climate that takes on some of the characteristics of both regions. Its climate may be classified as semiarid; however, with abundant irrigation water available from the Yellowstone River and the favorable distribution of the precipitation, it is possible to raise a variety of crops in the area.

About a third of the annual precipitation falls during May and June, with June being the wettest month. Billings receives an average 14.6 inches of precipitation per year. In the 51-year period of record, the maximum amount of precipitation in 1 year was 26.81 inches in 1978; the minimum amount of precipitation in 1 year was 8.41 inches in 1979. The period of least precipitation is from November through February. These four months normally produce less than 20 percent of the annual precipitation. The heaviest snows occur during the spring and fall months when the temperature and moisture conditions are most favorable. Heavy snows of 6 inches or more also occur during November and December. The occurrence of thawing periods normally prevents the snow from accumulating to great depths on the ground. Thunderstorms are most frequent during the summer months. These storms are frequently accompanied by strong, gusty winds and occasional hail. Destructive hailstorms, however, are rather infrequent.

Winter is usually cold and generally affords several mild periods of a week to several weeks in length. The winter cold periods are ushered in by moderately strong north to northeast winds and snow. True blizzard conditions are not observed very often in Billings; however, in the surrounding rural areas, blizzard conditions may develop several times during the winter. Cold weather warms with the onset of moderate to strong southwest winds. This wind is sometimes a foehn condition (chinook), but is more often a drainage wind moving down the Yellowstone Valley, which transports warmer air of Pacific origin to the area. Occasionally an open winter occurs when cold Arctic outbreaks pass far to the east and temperatures stay above zero degrees. Spring brings a period of frequent and rapid fluctuations in the weather. It is usually cloudy and cool with frequent periods of rain and/or snow. As the season progresses, snows become less frequent until late May and June when rain is the rule. The last freezing temperatures in spring usually occur before mid-May although they have occurred as the end of June. Billings has an average annual temperature of 47.1 degrees Fahrenheit (°F). The maximum monthly average temperature of 77.85 °F was reported in July 1960, and the minimum monthly average temperature of 5.05 °F was reported in January 1950.

The summer season is characterized by warm days with abundant sunshine and low humidity. The nights are cool because of the altitude and the cool air drainage into the valley from the higher terrain. Seldom is there a protracted rainy spell during this season. Frequent thunderstorms bring threatening afternoon cloudiness but usually only small amounts of rain.

The first freezing temperatures of the fall season occur in late September, but they have been noted as early as late August. Over the years, the fall months have been about evenly distributed between cold, wet ones, and mild, dry, pleasant ones. The change to severe winter weather usually arrives after the middle of November. There have been years when the more severe type of winter weather has been delayed until late in December.

3.2 GEOLOGY

The LSGPS is located in central Montana, which is predominantly underlain by Cretaceous shale and sandstone deposited in an inland sea environment. The cliffs surrounding the Billings area and along the Yellowstone River are composed of Eagle Sandstone. These units, along with younger sedimentary deposits associated with the Yellowstone River, comprise the principal geologic units at the LSGPS. The following sections present the geomorphology, regional geology, and site geology of the LSGPS.

3.2.1 Geomorphology

The LSGPS is situated within the active fluvial valley of the Yellowstone River immediately downriver and east of Billings, Montana. The site is situated on the southeast side of the Yellowstone River Valley north of Burlington Northern—Santa Fe Railroad and U.S. Highway 87 bridge crossings. The Yellowstone River in the vicinity of Billings exhibits typical braided channel environment characterized by high bed load sediment transport. The valley is defined by a relatively flat floodplain, low relief alluvial terraces, colluvial fans, and eroded bedrock hills and escarpments that define the valley margins. In the vicinity of LSGPS, the present river course is on the northwest side of the valley. As a result, little or no floodplain is developed on the northwest side of the river from the LSGPS. The LSGPS in located almost entirely within the floodplain and terrace area of the valley. Portions of the south side of the site are within areas of eroded bedrock hills with minor outcrop exposures. Elevation relief varies from approximately 3,160 feet above mean sea level at the uppermost terrace area to 3,090 feet above mean sea level at the riverbank.

As many as two alluvial terraces may be present within the site. The uppermost and oldest alluvial terrace boundary appears to have been located at or near the Interstate Highway 90 highway corridor. A second lower terrace boundary may be located along the approximate 3,100-foot elevation contour, southeast and parallel to the course of the Coulson irrigation ditch (Figure 3-1). The presence of alluvial terraces represents the dissected remnant of the former Yellowstone River floodplain. Terraces within the LSGPS have been obscured by the extensive construction of highways, railroads, and buildings and overprinting of colluvial fan deposits originating from bedrock upland areas to the southeast. The ephemeral drainage terminates into a chute channel along the Yellowstone River east of Corcoran pond.

The Yellowstone riverbank area in the LSGPS is composed of abandoned chute channels and gravel bars extending from the Lockwood water treatment plant area downriver to the Sandy-Lomond Lane area, where a cut bank along the river is developed. A second island and chute channel complex is developed farther downriver from Sandy-Lomond Lane area. These chute channels were observed to contain water during elevated water table events resulting from river stage increases in early July 2002. However, local residents indicated that these channels contain flow in some years during high-river stage levels in spring runoff periods.

3.2.2 Regional Geology

Alluvial and colluvial deposits within the greater Yellowstone floodplain consist of variable thickness of clay, silts, sand, and gravel. Alluvial deposits contain gravel of variable lithology derived from sources in the Yellowstone River drainage area while colluvial deposits are from locally derived bedrock sources, such as the Eagle Sandstone. Thickness of recent sedimentary deposits within the site varies from less than 10 feet near the bedrock contact to greater than 65 feet thick southeast of Interstate Highway 90 in the vicinity of Beall Trailers.

The majority of bedrock encountered in RI borings was identified as Upper Cretaceous Eagle Sandstone Formation. The Eagle Sandstone is resistant to erosion, as evident by the cliffs in and around Billings, Montana and in the vicinity of the LSPGS. Shale was also observed in some RI borings and likely represents thin layers of shale within the Eagle Sandstone. Outcrops of Eagle Sandstone occur near the LSGPS and exposures are visible where Interstate Highway 90 crosses the Yellowstone River. Eagle Sandstone is also exposed immediately south of the Town Pump Conoco station and the 2nd Shift Casino in the southwest portion of the U.S. 87 and Interstate Highway 90 interchange. From borehole data collected during the RI, the erosional contact between the alluvium and bedrock in the south portion of the LSGPS appears to be steeply dipping. Borehole data from previous investigations in the vicinity of the Lockwood Water Treatment plant and east toward the Interstate Highway 90 and U.S. 87 interchange indicated a less steeply dipping contact and possibly a narrow erosional bench in the bedrock surface.

Lopez (2000) mapped five geologic units within the LSGPS. These are, from youngest to oldest, Quaternary alluvium, Quaternary colluvium, Quaternary alluvial terrace gravels, Upper Cretaceous Claggett shale, and the Upper Cretaceous Eagle Sandstone. These units are generally described as follows:

- Quaternary alluvium (Holocene): gravel, sand, silt, and clay along active channels of rivers, creeks, and tributaries. Coarse, well-rounded gravel restricted mainly to the active channel of the Yellowstone River drainage. Most sediment in tributary drainages is sand, silt, and clay derived from local Cretaceous bedrock.

- Quaternary colluvium (Holocene and Pleistocene): locally derived slope-wash deposits mainly of sand, silt, and clay. Typically thin veneer-concealing bedrock, but locally as thick as 30 feet and commonly grades into Quaternary alluvium.

- Quaternary alluvial terrace gravels (Holocene and Pleistocene): composed of mostly cobbles and pebbles with minor amounts of sand and silt. Clasts are mainly granitic igneous rocks, granitic gneiss, schist, and quartzite, with much less limestone and sandstone 20 to 40 feet thick.

- Claggett Shale (Upper Cretaceous): brownish-gray fissile shale with minor interbeds of light brownish-gray, very argillaceous sandstone. Light brownish-gray to light-brown, calcareous concretions common, usually fossiliferous. The upper contact is gradational and conformable and is placed at the change to ledge-forming sandstones of the Judith River Formation.

- Eagle Sandstone (Upper Cretaceous): light brownish-gray to very pale-orange, very fine to fine-grained, cross bedded sandstone, burrowed to bioturbated in part. Locally contains calcareous, light-brown sandstone concretions up to 15 feet in diameter.

Regional geologic mapping by Lopez (2000) indicates a regional north-south trending anticline in the vicinity of Billings. This structural fold has limited influence on the course of the Yellowstone; however, the river course through the east side of Billings has down cut a narrow channel through a thick section of resistant Eagle Sandstone. Clagett Shale does not appear to be present in the LSGPS.

Intensive fracturing of the Eagle Sandstone is not readily apparent in outcrops near the LSGPS. Regional structural analysis by Lopez (2000) indicates the presence of northeast-southwest trending faults in the vicinity of LSGPS. No indication of significant vertical displacement of bedrock was observed in RI borings at the LSGPS. The elevation of the bedrock surface is likely controlled by erosional processes during river down cutting.

3.2.3 Site Geology

The LSGPS is situated on Quaternary alluvium and alluvial terrace deposits. Upper Cretaceous Eagle Sandstone bedrock is exposed and present near surface in the south portion of the site; however, most bedrock exposures are covered by colluvium deposits or have been buried by construction fill. The general contact between alluvium and bedrock follows the trace of the Lower Lockwood irrigation ditch from the Yellowstone River to the Beall property. This contact also marks the approximate extent of the shallow alluvial aquifer at the site (Figure 3-1). South of the Lower Lockwood irrigation ditch, bedrock was encountered at very shallow depths while north of the ditch alluvium in the upper alluvial terrace area had an average thickness of 65 feet. Alluvium thickness in the lower terrace area averaged 40 feet.

Three geologic cross-sections have been constructed across the LSGPS site to show relationships between surface topography, shallow aquifer, alluvium thickness, groundwater flow, and bedrock. Sections were constructed from boring log information collected during RI drilling activities. RI borehole logs are included in Appendix D. Figure 3-2 shows the map view location of each of the geologic cross-sections and Figures 3-3 through 3-5 are sections A-A', B-B', and C-C' respectively. A bedrock structure map is included as Figure 3-6.

3.3 SURFACE WATER HYDROLOGY

Major surface water features at the LSGPS include the Yellowstone River, manmade features including the Coulson and Lockwood (upper and lower) irrigation ditches, mixed wetlands with ponds near the Yellowstone River, and the ponds associated with gravel pit mining operations, AJ Gravel Pit and Corcoran ponds. These features are described in detail in the following sections.

3.3.1 Yellowstone River

The Yellowstone River is the main surface water feature in the LSGPS and the centerline of the channel marks the western boundary of the site. A U.S. Geological Survey (USGS) gauging station is located within the LSGPS and has been used since 1904. The Yellowstone River at the point of the gauging station (river mile 360.3) drains approximately 11,795 square miles. The Yellowstone River has an average flow of approximately 7,000 cubic feet per second (cfs) and a base flow of 3,700 cfs. The lowest mean monthly flow for the 76-year period of record is 2,491 cfs (USGS 2003).

3.3.2 Coulson Irrigation Ditch

Previous investigation studies (Tetra Tech, Inc. 1993; Lockheed Martin 1999) have reported the Coulson irrigation ditch surface water elevations consistent with static water level (SWL) elevations in nearby groundwater wells during periods of no irrigation flow. Because the Coulson irrigation ditch flows generally perpendicular to groundwater flow, it has the potential of receiving contaminated groundwater seepage and transporting it downstream.

The Coulson irrigation ditch originates at a diversion structure on the Yellowstone River south (upriver) from the LWSD Treatment plant. The Montana Department of Natural Resources and Conservation (DNRC) water right filing for the Coulson irrigation ditch is September 3, 1895 for 26.26 cfs from the Yellowstone River. The ditch supplies irrigation water for the Coulson Water Users Association downriver from the LSGPS. The installation dates of the present Coulson irrigation ditch and diversion structure is not known; however, they likely predate the majority of road and building development in the area. The ditch flows by gravity directly from the river diversion to the northeast, through the Auto Auction property, and then continues northeast, passing north of the Brenntag property. The ditch exits the LSGPS beneath Klenck Road and into an open field area. Stage elevation of the ditch indicates that the ditch is at or near local shallow groundwater elevations.

Irrigation flow in 2002 was observed in Coulson irrigation ditch during the May, June, and July monitoring events. Flow in the ditch was measured at several monitoring stations beginning in 2001. The maximum recorded flow in 2002 was 54.3 cfs at station SW005, while three other ditch monitoring stations recorded flows between 18.4 and 23.3 cfs for the same event.

During RI activities, water was observed in the ditch during periods of no diversion from the river and likely represents a combination of groundwater influx and minor stormwater drainage. Stormwater drainage into Coulson irrigation ditch was observed at one location within the LSGPS near surface water station SW302. At this location there is a tributary ditch entering the Coulson irrigation ditch from the southeast and appears to drain the railroad right-of-way area south of Cerise Road. Wetland vegetation (cattails) was observed in some portions of the railroad right of way area.

3.3.3 Upper and Lower Lockwood Irrigation Ditches

The Upper and Lower Lockwood irrigation ditches lie well above the water table and may provide some seasonal recharge to shallow groundwater depending on leakage rates and flow within the ditch. Sections of both Upper and Lower Lockwood irrigation ditches are piped in subsurface culverts where they cross under Interstate Highway 90, U.S. 87, and the Beall facility. The Upper and Lower Lockwood irrigation ditches originate at a pump intake located at the Coulson diversion structure on the Yellowstone River south (upriver) from the LWSD Treatment plant. The Montana DNRC water right filing for the Lockwood Irrigation District is May 31, 1914, for 35,500 gallons per minute (gpm) from the Yellowstone River. The installation dates of the current Lockwood irrigation ditches (upper and lower) are not known. The lower ditch is pumped approximately 40 feet higher than the river to a point where it discharges on the Eagle Equipment property. From there it flows eastward, under Interstate Highway 90, and eventually through the Beall property. The upper ditch is pumped approximately 100 feet higher than the Yellowstone River, to the south of Interstate Highway 90 where it discharges and flows generally eastward, along the hillside to the south of the LSGPS. Both ditches flow through moderately permeable sediments 40 to 50 feet above the water table and are likely losing water at variable rates along the length of the channels.

Water level and flow measurement data have been collected at five locations in the Lower Lockwood irrigation ditch since August 2001. Flow was observed in the ditch in 2002 during the May, June, July, and August monitoring events. Minimum and maximum recorded flows ranged form 1.8 to 4.03 cfs.

3.3.4 Ponds

The two ponds, AJ Gravel pond and the Corcoran pond, are located south of the Yellowstone River, near the north end of the LSGPS. The ponds are the result of aggregate mining activities and, hence, the water elevations in the ponds are a reflection of water table elevations. Corcoran Pond was mined in the 1960s and no recent mining operations have taken place. Depths or configurations of the bottom of the ponds are not available.

During active mining operations at AJ Gravel, water was pumped from the AJ Gravel pond to the Yellowstone River or Corcoran pond. In 1998, the DEQ Open Cut Mining Program prohibited discharge to the Yellowstone River as a result of detection of VOCs in the pond water. To continue operation, seepage water was transferred via pumping to a series of ponds within the property boundary to dewater the work area(s). During 2000, when the public water line was installed along Lomond Lane, the AJ Gravel pond was pumped for approximately 2 months in order to dewater the waterline trenches. This type of operation continued through December 2000; however, no dewatering has taken place since 2000.

3.3.5 Wetland Area

A permanent wetland area with small open ponds is located in the west portion of the LSGPS east of the Yellowstone River. The wetlands extend from east of Cerise Road near monitoring well MW305 northeast toward the Sandy-Lomond Lane area (Figure 3-1). The morphology of the wetlands indicated they were formed in a former chute channel originating from the Yellowstone River and some portions of the wetlands have year round open water and likely receive groundwater flux. The former chute channel was likely cutoff from the active channel of the river by natural processes or by filling activities that may have occurred in the area downstream of the LWSD treatment plant.

3.4 HYDROGEOLOGY

Hydrogeologic data collected during the RI was used to (1) define the major hydrostratigraphic units at the LSGPS, (2) characterize the hydraulic characteristics of the units, and (3) identify the predominant groundwater flow directions and velocities at the site. This information has also been used to develop a site conceptual model for groundwater contaminant transport at the LSGPS (see Section 5.4).

3.4.1 Vadose Zone

The vadose zone consists of the unsaturated interval present from the ground surface to the top of the capillary fringe near the water table. The zone thickness varies considerably at the LSGPS because of the increasing thickness of alluvial terrace deposits and colluvium away from the Yellowstone River toward the edge of the river valley. Thickness ranges from greater than 45 feet at the Beall facility to less than 10 feet near the Yellowstone River. Significant thickness of low permeability (clay) units were not observed in the vadose zone in the LSGPS. Fine-silty clay, silt, and fine sand units were identified in the vadose zone throughout the site. These units originated as overbank deposits from the Yellowstone River with some colluvium fan deposits in the southeast portion of the site. Minor amounts of gravel were observed in some borings and are likely from colluvium deposition on the southeast side of the LSGPS. Ponding or perching of groundwater was not generally observed at the LSGPS; however, the majority of the site is developed and storm water is generally directed to drainage ditches thus limiting infiltration.

3.4.2 Alluvial Aquifer

The alluvial aquifer hydrostratigraphic unit at the LSGPS consists of the saturated alluvial sand and gravel unit that overlies the Eagle Sandstone bedrock. The aquifer thickness is variable, but generally ranges from 20 to 30 feet. The materials are derived from bed load channel deposition of the Yellowstone River and are composed of fine to coarse silty sands with gravels and cobbles. The sand and gravel interval is generally saturated and the top of the water table coincides with the top of the sand and gravel interval in the west and northwest portions of the LSGPS. Fine sediments that typically make up the vadose zone also comprise the upper saturated portion of the alluvial aquifer in the southeast half of the LSGPS (upper terrace area). Permeability characteristics of the alluvial aquifer are variable, but wells generally exhibit little or no drawdown during pumping. Aquifer tests and aquifer parameter estimates from previous investigations are summarized in Table 3-1.

During the RI activities, TtEMI conducted nine slug tests on selected RI monitoring wells. Eight of the tested wells were completed in the alluvial aquifer and one well was completed in bedrock. Field data were analyzed using AQTESOLV Version 3.0 and graphical solutions are presented in Appendix J. Table 3-2 summarizes the slug test data for the wells. Hydraulic conductivities in the eight alluvial aquifer wells that were tested ranged from 0.17 to 0.63 feet per day (ft/day). The single bedrock well slug test in well MW128 had a calculated hydraulic conductivity of 0.88 ft/day. These test data for the alluvial aquifer are representative of silty sand lithology while the bedrock well is representative of consolidated sandstone. Slug test data are generally considered to be an order of magnitude lower than pump test aquifer data due to the limited stress imparted to the aquifer in slug testing. Considering this, slug test data from the alluvial aquifer were below the range reported from previous pumping tests (130 to 676 ft/day) and also hydraulic conductivities estimated from specific capacity data (10 to 600 ft/day) (see Table 3-1).

3.4.3 Bedrock Aquifer

The bedrock hydrostratigraphic unit at the LSGPS is composed of the Eagle Sandstone and associated interbedded shale units. The top of the bedrock unit was penetrated in nearly all RI borings; however, samples could not always be recovered to identify lithology. Two monitoring wells (MW219 and MW128) were completed in the bedrock to evaluate contaminant transport downgradient of two suspected source areas. Bedrock lithology is predominantly composed of Eagle Sandstone although black shale was observed in some borings. Groundwater within the bedrock aquifer is likely contained within a system of interconnected fractures and joints and possibly some preserved primary and secondary porosity. A structure map on the top of the bedrock surface indicates a generally northwest slope to the Yellowstone River (Figure 3-6). Bedrock slope is to the north in the vicinity of the alluvial aquifer boundary near the Interstate Highway 90 and U.S. 87 interchange and at Beall Trailer. An erosional area with a corresponding increase in alluvial aquifer thickness was observed in the vicinity of well MW115 and in wells MW114 and MW120 in the area northwest of Sandy Lane and Bone Ave. This low was likely the result of river down cutting. Based on a review of contaminant distribution data, the bedrock low does not appear to have any significant effect on contaminant transport.

3.4.4 Groundwater Flow Direction

Water-level maps (Figures 3-7 and 3-8) prepared for the alluvial aquifer for the August 28, 2002 and October 28, 2002 measurement events and indicate a general northwest flow to the Yellowstone River. Water-level elevations for the two events ranged from a maximum of 3112.8 feet in well MW201 at Beall Facility to a minimum of 3068.0 feet in well MW124 adjacent to the Yellowstone River. The overall site gradient was approximately 0.009 feet per foot (ft/ft) for both events. Local groundwater flow variations were noted in the vicinity of Beall Trailers where a component of north groundwater flow was observed.

In addition, a more westerly component of flow is noted in the vicinity of Cen-Dak Leasing and Planet Lockwood. Some water table mounding may be present in the vicinity of well MW302 because of discharge from the Planet Lockwood Septic drain field.

3.4.5 Groundwater Recharge and Discharge

Recharge areas are those where a component of the groundwater flow is downward and discharge areas are those where flow is upward. Recharge of groundwater to the alluvial aquifer at LSGPS is likely occurring though two processes, (1) infiltration from surface features, such as ditches, and rainfall on unpaved surfaces, and (2) groundwater flow from the bedrock at the upgradient alluvial aquifer boundary. Although not quantified during the RI, net infiltration from the surface within the LSGPS is likely limited by the low net precipitation in the area, controlled storm water runoff, and by paved cover and buildings. Recharge from the bedrock aquifer is likely occurring along most the alluvial aquifer boundary with saturated bedrock. Groundwater recharge for the LSGPS is occurring in the bedrock hills to the south and southeast of the LSGPS. Groundwater from this area flows downgradient toward the Yellowstone River and enters the alluvial aquifer at the contact (alluvial aquifer boundary) with the bedrock aquifer.

3.4.6 Hydraulic Relationship between the Alluvial and Bedrock Aquifers

Geologic boundaries such as the contact between the alluvial and bedrock aquifer at LSGPS can significantly alter groundwater flow patterns depending on contrasts in hydraulic properties between the two aquifers. Limited slug test data from the RI indicate that hydraulic conductivity of the alluvial and bedrock aquifers are within the same order magnitude. However, seepage velocities based on groundwater quality data do not support this conclusion. It is likely that the slug tests underestimate hydraulic conductivities in the alluvial aquifer.

Two monitoring well pairs were installed during the RI in the Eagle Sandstone bedrock to monitor water quality as well as to evaluate the hydraulic gradient between the alluvial aquifer and the bedrock aquifer. Water levels from both alluvial/bedrock well pairs indicate a variable vertical gradient at the two well pairs during three groundwater level measurement events (Table 3-3).

Monitoring well pair MW007/MW128 was measured in August and September 2002. Data indicated a slight (downward) gradient, suggesting groundwater flow or discharge from the alluvial aquifer to the bedrock aquifer. This observation is not supported by groundwater quality data since no contamination has been detected in the two groundwater samples collected from the bedrock monitoring well (MW128). Essentially no vertical gradient was observed in the MW007/MW128 well pair during the October 2002 measurement event.

Monitoring well pair MW205/MW219 was measured in August 2002 and initially showed a positive (downward) gradient, indicating groundwater flow or discharge from the alluvial aquifer to the bedrock aquifer. Measurements made in September and October 2002 indicate a negative (upward) gradient from the bedrock aquifer to the alluvial aquifer. Bedrock monitoring well (MW219) had low level detections of VOCs in two groundwater samples collected in August and October 2002. These data generally support an overall downward gradient from the VOC-contaminated alluvial aquifer that is contributing to detectable VOC contamination in the upper portion of the bedrock aquifer.

3.4.7 Hydraulic Relationship between Groundwater and Surface Water

Groundwater and surface water interaction takes places where groundwater discharges or “daylights” to surface water bodies such as a pond, spring, or river. Discharge from surface water to groundwater can also result from a rise in river stage, such as during seasonal (spring) runoff periods, resulting in surface water recharging into shallow groundwater. Hydraulic barriers may develop in some quiescent surface water settings that may inhibit groundwater flux to and from a surface water body. An example of barrier would be fine-grained bottom sediments deposited in a pond or ditch bottom. Areas of groundwater and surface water interaction at the LSGPS include the Coulson irrigation ditch, the Yellowstone River, AJ Gravel pond, and Corcoran pond.

Groundwater influx or seepage into the Coulson irrigation ditch likely occurs during periods of no flow where the bottom of the ditch intercepts the water table. Bottom elevations measured in Coulson irrigation ditch varied from 3090.21 feet above mean sea level near the LWSD treatment plant to 3089.22 feet above mean sea level where the ditch crosses Klenck Lane at the western boundary of the LSGPS. Comparison of these data to RI monitoring well water levels in October and November 2002 indicate that the majority of Coulson irrigation ditch within the LSGPS is below the water table and likely receives groundwater. Water has been observed in several portions of the Coulson irrigation ditch within the LSGPS throughout periods when irrigation water was not flowing. Results from water sampling in Coulson irrigation ditch conducted during previous investigations and the RI reported VOCs at 7 of 8 sample locations during nonirrigation flow periods, indicating that groundwater is entering the ditch (see also Section 4.5.2). During periods of ditch flow (May, June, and July), surface water is recharging groundwater and likely creates a water table mound near the Coulson irrigation ditch. This water table mound would rapidly dissipate after flow ceases in the Coulson irrigation ditch. Storm water and snow melt may also enter the ditch during periods of no irrigation flow, but likely contributes little to Coulson irrigation ditch flows.

Groundwater in the alluvial aquifer flows toward the Yellowstone River. Although the depth of the river channel adjacent to the LSGPS is not known, it is likely that the entire groundwater flux from the alluvial aquifer enters the river. Bedrock aquifer groundwater flux to the river may also occur depending on vertical movement of groundwater beneath the river and erosional depth of the river channel. Some recharge of surface water from the Yellowstone River to the alluvial aquifer likely occurs during spring runoff when the river stage is high. Water levels in depressions adjacent to the Yellowstone River and AJ Gravel pond were observed to rise during July 2002 RI activities. The propagation of this water table rise may extend some distance away from the river, but the effects on groundwater flow may be reduced gradient and contaminant dilution. There are likely no hydraulic barriers between the alluvial and bedrock aquifers and the Yellowstone River.

AJ Gravel pond was created by mining aggregate contained within the Quaternary alluvium. The deepest depths of mining below the water table are approximately 9 to 10 feet in the north end of the pond and 10 to 16 feet in the south end of the pond (Womack and Associates, Inc. 2003). From adjacent monitoring well water levels, the elevation of the pond surface coincides with the elevation of the water table. From this information, the pond receives groundwater flux from the upgradient (south) side and discharges to the downgradient (north) side (see also Section 4.5.1). Inspection of the pond indicates that it receives little or no storm water runoff and only periodic rainfall and precipitation. The surface of the pond also freezes to an unknown depth during winter months. Elevated levels of VOCs have been detected in pond water samples and in adjacent upgradient monitoring well MW005 and downgradient monitoring wells MW126 and MW127. These data suggest that the pond is acting as an open conduit for alluvial aquifer groundwater flow and little or no hydraulic barrier is present in the bottom of the pond. VOC contamination is also bypassing the pond in the lower portion of the alluvial aquifer. Similar conditions are likely present in the Corcoran pond; however, based on monitoring well data and flow direction is it is unlikely that VOCs are entering Corcoran pond.

3.5 POPULATION AND LAND USE

The LSGPS is located outside the city limits of Billings, Montana and south of the Yellowstone River, and consists of chlorinated solvent contamination in soils and groundwater. The area is zoned controlled industrial (minor industry) according to the Yellowstone City/County Planning Department. The current land use within the LSGPS includes residential, commercial, and "light" industrial operations. There are 81 commercial and light industrial businesses within the boundaries of the LSGPS including trucking companies, a chemical distribution plant, an auto auction facility, auto repair shops, construction companies, a water treatment plant, open agricultural land, a lumber yard, and three active gas stations. An estimated 75 residential single family residences, two trailer parks, and one apartment complex are currently located within the boundaries of the LSPGS.

At the time of the initial investigations, the primary source of drinking, domestic, commercial, and non-domestic water for the residences and businesses in the LSGPS came from the shallow aquifer from private wells. During the environmental investigations, however, residences with private wells that were found to contain contaminant concentrations above MCL/WQB-7 criteria were placed on the public water supply system. DEQ has confirmed that all businesses within the LSGPS site boundaries are either connected to the public water supply or provide an alternate drinking water supply for their employees.

4.0 NATURE AND EXTENT OF CONTAMINATION

This section describes the results of RI sampling activities conducted at the LSGPS and the nature and extent of contamination found at the site. Where applicable, data collected outside of the RI, including historic data, are referenced and discussed. Media that were sampled during the RI include surface soil, subsurface soil, surface water, sediment, and groundwater. Data summary tables for VOC analyses are provided in Appendix G and results for PCE, TCE, cis-1,2-DCE, trans-1,2-DCE, and VC are presented on Figures 4-1 through 4-9.

On the figures for Section 4.0, analytical results for soil, sediment, and water samples were screened against all of the criteria listed below to identify media and locations where contaminant concentrations may be of concern. These exceedances are highlighted in shaded boxes on the figures. However, the discussion of soil and sediment results in the text for Section 4.0 only include comparisons to site specific SSLs.

For sediment and soils, screening criteria include:

- EPA Region 9 industrial and residential preliminary remediation goals (PRG) (EPA 2002f)
- EPA Region 9 SSLs based on protection of groundwater (EPA 2002f)

- Site-specific SSLs (calculations included in Appendix G) developed based on EPA guidance (EPA 1996c)

For surface water and groundwater, screening criteria include:

- Federal MCLs (EPA 2002f)
- Montana water quality standards (WQB-7 criteria) (DEQ 2001b)

As stated in the RIWP, two of the primary objectives of this RI are to (1) investigate contaminant sources areas and (2) define the nature and extent of VOC groundwater and soil contamination in the LSGPS investigation area (TtEMI 2002a). From the data collected during previous investigations, groundwater VOC contamination within the LSGPS was known to be widespread. The sampling investigation for nature and extent included using sampling transects that were located across the site to evaluate the extent of contamination from suspected sources and to locate other, previously unidentified potential sources of contamination. The following sections describe the contaminants of potential concern, sources and extent of soil and groundwater contamination, and data gaps identified during the RI.

4.1 CONTAMINANTS OF POTENTIAL CONCERN

COPCs identified at the LSGPS in previous investigations include the following VOCs: PCE, TCE, cis-1,2-DCE, trans-1,2-DCE, VC, and carbon tetrachloride. Additional VOCs detected in soil and/or groundwater samples throughout the site during the RI and previous investigations include benzene, toluene, ethylbenzene and xylenes.

Typically, benzene, toluene, ethylbenzene and xylenes, when detected in the same sample, are referred to as BTEX and are a suite of contaminants generally associated with petroleum fuels. CERCLA includes a petroleum exclusion that was meant to separate petroleum spills from CERCLA response actions. The term “hazardous substance” as defined in CERCLA section 101(14) excludes petroleum, including crude oil or any fraction thereof, unless specifically listed or designated under CERCLA. The EPA interprets CERCLA section 101(14) to exclude crude oil and fractions of crude oil including the hazardous substances, such as BTEX compounds, that are found in those petroleum substances, from the definition of a hazardous substance. However, if an individual BTEX compound is identified, and is not directly related to a petroleum release (that is, instead there was storage or use of pure product), that compound may be listed as a COPC if it meets the established screening criteria.

Toluene and xylenes have been or are currently stored as pure phase products at the Brenntag facility; however, there are no records or indications of previous storage of ethylbenzene or benzene. Previous groundwater data collected by Maxim (2000) (Table 4-1) indicated 21,000 µg/L of toluene in a groundwater sample from Dyce-B-M-11-W. This concentration was well above the expected aqueous concentration from the dissolution of toluene as present in petroleum products. Therefore, historic toluene contamination at the Brenntag facility was not considered the result of a petroleum release. Other compounds including benzene, ethylbenzene, and xylenes were also detected in the sample at much lower concentrations and may have been impurities in the released toluene.

The BTEX concentrations in groundwater samples from monitoring wells completed during the RI are presented in Appendix G and summarized in Table 4-2. Current RI data indicate that BTEX concentrations are considerably lower than historic BTEX results and are indicative of a release of a petroleum product. Therefore, toluene and other BTEX compounds are not considered COPCs in this RI. Remediation of BTEX contamination resulting from the release of petroleum products will not be addressed by this RI. However, petroleum releases at the LSGPS that have been addressed in the past will continue to be addressed by the State of Montana underground storage tank corrective action programs.

4.2 AREA A INVESTIGATION

This section presents the results of RI sampling activities in Area A, including data from previous investigations.

4.2.1 Previous Investigations

Previous sampling in Area A consisted of sampling from Geoprobe borings by EPA (Lockheed Martin 1999) and DEQ/RD (Pioneer 1999) and installation and sampling of monitoring wells by EPA. Brenntag contracted Maxim Technologies to conduct a site investigation in August 2000. Maxim completed 12 Geoprobe boring nests within the Brenntag property boundary and collected multilevel groundwater samples from each boring nest. Groundwater samples from several borings indicated concentrations of PCE, cis-1,2-DCE, and toluene at aqueous concentrations greater than 1 percent pure phase solubility for those compounds. Only two soil samples from Borings I and M were collected and submitted for laboratory analysis. Cis-1,2-DCE was detected in the sample from Boring M at 12,000 micrograms per kilogram (µg/kg) and in the sample from Boring I of 2,900 µg/kg. VC was detected in a soil sample from

Boring I at a concentration of 1,900 µg/kg. No other VOCs were detected above the laboratory detection limit in either sample.

The VOC groundwater plume, later confirmed to emanate from the Brenntag facility, was detected in June 1998 during the DEQ/RD's Site Response Section Integrated Assessment of the LSGPS (Pioneer 1999). The investigation primarily focused upgradient (east) of the LWSD water supply wells (PWS001, PWS002, and PWS003) located adjacent to the Yellowstone River. As a precautionary measure, DEQ/RD personnel collected groundwater samples from selected residential wells in the Lomond Lane area, which resulted in the discovery of elevated VOC concentrations above MCL/WQB-7 criteria and also of those previously detected in the area of the LWSD treatment plant and adjacent areas. A total of nine groundwater samples collected from residential wells in the area exceeded MCL/WQB-7 criteria for PCE, TCE, cis-1,2-DCE, VC, and/or carbon tetrachloride. Groundwater samples were also collected from three Geoprobe boreholes completed north and west of the Brenntag facility. In 1999, the DEQ/RD initiated another Geoprobe groundwater investigation to identify the source(s) of VOC groundwater contamination in the area around Lomond Lane. Subsequent investigations by URS Corporation and Lockheed Martin for EPA identified a potential source area for the VOC contamination upgradient of the Lomond Lane residential area. EPA's contractor installed and sampled 11 monitoring wells in 1999 (monitoring wells MW001 through MW011).

Beginning in December 2001, Secor International Inc. (SECOR) conducted an investigation in the northwest portion of the Brenntag property (SECOR 2002). The project objective was to evaluate the area for installation of an air sparge and soil vapor extraction system to attempt to mitigate migration of VOCs from the Brenntag property. SECOR completed four soil borings which were all converted to monitoring wells (PT001, PT002, PT003, and PT004) (Figure 4-1). Two soil samples were collected from each boring and sampling results for PCE, TCE, cis-1,2-DCE, trans-1,2-DCE, and VC are provided in Table 4-3 and on Figure 4-1. Soil samples from borings PT001 and PT002 contained elevated concentrations of VOCs (primarily PCE) in the vadose zone which exceeded site specific SSLs and may indicate the presence of residual NAPL source in this portion of the Brenntag property. Concentrations of PCE in vadose soils exceeded site specific SSLs (0.172 milligrams per kilogram [mg/kg] for PCE) in six of the eight soil samples and with reported concentrations exceeding site specific SSLs ranging from 1.1 to 2,040 mg/kg. Other VOCs detected in the samples include 1,1,1,2-tetrachloroethane (TCA), 1,1,2-TCA, TCE, DCE, benzene, toluene, ethylbenzene, and xylenes (Appendix G). TCA compounds are likely present as impurities in the parent PCE which was apparently released. Groundwater samples were collected by SECOR from the monitoring wells in January 2002. Sample analysis indicated groundwater

VOC concentrations exceeding MCL/WQB-7 criteria in all four monitoring wells. PCE was reported in a groundwater sample from monitoring well PT002 at a concentration of 18,000 µg/L. Unvalidated data results were provided to the DEQ and subsequently TtEMI in October 2002 but were not available during RIWP development. This data is considered useable for characterization purposes, but not for risk assessment unless validated.

Subsequent groundwater sampling of the well network installed by EPA (monitoring wells MW001 through MW011) was conducted by DEQ through 2002 and is expected to continue in 2003. Mann-Kendall statistical tests were performed on selected wells from this network as well as additional select monitoring wells to evaluate concentration trends with time and are presented and discussed in Section 5.4.3.1. Historic monitoring data is summarized in previous data summary and monitoring reports (Pioneer 2001b, Pioneer 2002a, Pioneer 2002b, and TtEMI 2002b).

4.2.2 Remedial Investigation Activities

RI activities in Area A included field screening with Ribbon Samplers, subsurface and surface soil sampling, monitoring well installation and groundwater sampling.

4.2.2.1 Field Screening with Ribbon Samplers

During RI activities in June 2002, Tetra Tech installed six ribbon samplers (RS100 through RS105) (Figure 2-2) as a field screening tool to evaluate the presence or absence of NAPL layers in soil down to bedrock, and to assist in the determining where subsurface soil samples were to be collected from collocated soil borings (which were subsequently used to construct monitoring wells). Ribbon samplers were installed within the Brenntag property at locations adjacent to aboveground chemical storage tanks, near a former surface impoundment, and adjacent to former Geoprobe boring nests where elevated groundwater VOC concentrations were detected in the Maxim (2000) Geoprobe investigation.

Ribbon samplers did not detect the presence of NAPL in the saturated or vadose zones of subsurface soil down to bedrock. Black staining and elevated PID readings were observed in soils in the collocated borings for monitoring wells MW100 (6 to 9 feet bgs), MW102 (4 to 18.5 feet bgs), MW104 (4 to 27 feet bgs), and MW105 (24 to 28 feet bgs). Soils in the borehole for monitoring well MW104 also exhibited petroleum odors. Depth to water in the monitoring wells (MW100 to MW105) varied from approximately 6.6 to 10.5 feet bgs, indicating that soil staining and elevated PID readings persisted well below the estimated range of 1 to 3 feet (vertical) of water table fluctuation. Anaerobic degradation of VOC compounds was indicated by the observed black staining in the soils, which was a visual indication of reduced iron and manganese in the aquifer matrix. Although no NAPL was detected on the Ribbon Samplers to indicate where subsurface soil samples should be collected, the depth of subsurface soil samples from collocated soil borings were chosen based on the depths of observed soil staining, soil odors, highest PID readings, and soil lithology.

4.2.2.2 Subsurface Soil Sampling

Vadose zone and saturated zone soil sampling was conducted throughout Area A along Transects C, D, E, and F to evaluate the nature and extent of contamination (Figure 4-3). In general, sampling was performed in conjunction with monitoring well installation. Vadose zone thickness varied across Area A from about five to 15 feet and consisted of unconsolidated clayey silt with occasional fine-grained sand. Subsurface soil sampling results for PCE, TCE, cis-1,2-DCE, trans-1,2-DCE, and VC are provided on Figure 4-1.

Elevated headspace PID readings were detected in the vadose zone samples taken from the Brenntag facility, downgradient of the Brenntag facility, and at other locations in Area A (see PID readings recorded on borelogs in Appendix D). In addition, slight odors and black staining was observed in vadose zone samples collected from MW102 and MW104.

PCE was detected in seven of the 44 subsurface soil samples collected in Area A with concentrations ranging from 0.05J to 0.86 mg/kg. Concentrations in six of these samples were above the site specific SSL of 0.172 mg/kg. All Area A subsurface soil samples with detected PCE concentrations were located within or downgradient of the Brenntag facility.

TCE was detected in one subsurface soil sample (MW116, 19 to 21 ft bgs) at 0.053J mg/kg below the site specific SSL of 0.173 mg/kg. This subsurface soil sample was collected from a boring located downgradient of the Brenntag facility. TCE was not detected in any other subsurface soil sample collected in Area A.

Cis-1,2-DCE was detected in Area A in six of 44 subsurface soil samples, all of which were located at the Brenntag facility. Concentrations ranged from 0.051J to 0.56 mg/kg and all detections were below the site specific SSL of 1.071 mg/kg. Cis-1,2-DCE was not detected in subsurface soil samples collected outside of the Brenntag property in Area A.

Trans-1,2-DCE and VC were not detected in subsurface soil samples in Area A above laboratory detection limits.

4.2.2.3 Surface Soil Sampling

Surface soil samples were collected at the Brenntag facility during the RI to evaluate the presence of surface soil contamination, potential sources of contamination, and worker exposures to COPCs. Ten surface soil samples (SS100 through SS102 and SS104 through SS110) were collected on a grid pattern from the depth interval of 0 to 2 feet bgs (Figure 4-1). COPCs were not detected in any of the surface soil samples above the laboratory detection limit.

4.2.2.4 Groundwater Sampling

During RI field activities, TtEMI installed 29 monitoring wells in Area A (Figures 2-1 and 2-2). Groundwater sampling was conducted in all monitoring wells at LSPGS during two RI sampling events in July/August and October 2002. Groundwater sampling results for PCE, TCE, cis-1,2-DCE, trans-1,2-DCE, and VC are summarized on Figures 4-2A and 4-2B. Complete analytical results are provided in Appendix G. Detected PCE, TCE, and cis-1,2-DCE concentrations for monitoring wells along transects C through F are plotted on Figures 4-3C through 4-3F.

The monitoring well locations were selected to investigate the potential for additional sources of VOCs, if any, and to further define the VOC plume in the alluvial aquifer. Monitoring well locations were also selected to provide transects of monitoring points perpendicular to the current groundwater flow direction (Figure 4-3). The monitoring well transects also incorporated the existing well network installed by EPA's contractor in 1999. All newly installed monitoring wells in Area A that were completed in the alluvial aquifer were fully screened across the alluvial aquifer to bedrock.

Groundwater Sampling at the Brenntag Facility

Six of the 29 new monitoring wells (MW100 through MW105) were installed and collocated with six ribbon sampler locations at the Brenntag facility (Figures 2-2). Concentrations in monitoring wells installed by TtEMI at the Brenntag facility did not contain VOC concentrations greater than 1 percent pure phase solubility (a concentration which suggests the presence of free product), indicating that free product was not encountered in the groundwater samples collected during the RI. Groundwater VOC concentrations did exceed MCL/WQB-7 criteria for one or more VOCs in four of the six monitoring wells (MW101, MW102, MW104, and MW105). Groundwater samples collected from monitoring wells MW102 and MW103 contained detectable PCE, cis-1,2-DCE, and VC below MCL/WQB-7 criteria. Groundwater VOC concentrations in wells MW101, MW102, and MW104 were primarily cis-1,2-DCE with much lower concentrations of PCE, TCE, and VC. PCE concentrations abruptly increased in existing monitoring well MW011. The PCE concentration reported in the groundwater sample collected by SECOR in monitoring well PT002 was 18,000 µg/L in December 2001 (SECOR 2002). This concentration of PCE is the maximum detected within the LSGPS and exceeds the 1 percent solubility of pure phase PCE (1,500 µg/L).

Three Geoprobe soil borings (SB100, SB101, and SB102) were completed to bedrock east and south of the main tank farm at the Brenntag facility to further define the upgradient limit of the groundwater VOC plume (Figure 4-2A). Only groundwater samples (no soil) were collected from the borings. Two groundwater samples were collected from each boring, one at the top of the water table and one immediately above bedrock. The results of the groundwater samples are presented on Figure 4-2A. Only the highest concentration of the two groundwater sampling results for each borehole are shown on Figure 4-2A. All results for COPCs were nondetect except for a detection of PCE at 0.22J µg/L in a groundwater sample collected from boring SB101. VOC analytical results for all six groundwater samples are presented in Appendix G.

Groundwater Sampling Downgradient of the Brenntag Facility

Ten of the 29 RI monitoring wells were installed northwest and downgradient of the Brenntag facility and include monitoring wells MW116, MW117, MW121, MW122, MW123, MW124, MW125, MW126, MW127, and bedrock well MW128. Monitoring well MW128 was completed in the bedrock aquifer adjacent to monitoring well MW007 to evaluate water quality in the bedrock and to evaluate the vertical gradient between the two aquifers.

The highest concentrations of PCE downgradient from the Brenntag facility were detected in existing and new monitoring wells MW007, MW008, MW009, MW122, and MW011 and likely represent the best estimate of plume core concentrations. Maximum daughter degradation products (TCE, cis-1,2-DCE, and VC) concentrations presented a similar distribution along the plume centerline as PCE.

Groundwater Sampling in the Remainder of Area A

The remaining 13 monitoring wells were installed at other locations along transects in Area A and included monitoring wells MW106 through MW115 and MW118 through MW120. PCE was detected in four wells at concentrations ranging from 0.13J to 0.62 µg/L, all of which were less than MCL/WQB-7 criteria. TCE was detected in ten of the wells at concentrations ranging from 0.43J to 11 µg/L. TCE was detected above MCL/WQB-7 criteria (5.0 µg/L) in three groundwater samples collected from two monitoring well locations: (1) MW109 at concentrations of 10 and 11 µg/L, at the property of Ankrum Trucking and (2) MW110 at a concentration of 8.6 µg/L, located on the property of Keller Transport Inc. Cis-1,2-DCE was detected in all of the wells except monitoring well MW115, located on Bone Avenue. None of the cis-1,2-DCE detections exceeded MCL/WQB-7 criteria. Trans-1,2-DCE and VC were not detected above laboratory detection limits in any of the 13 monitoring wells.

4.3 AREA B INVESTIGATION

This section presents the results of RI sampling activities in Area B, including data from previous investigations.

4.3.1 Previous Investigations

Previous investigations at Area B included sampling soil and groundwater in five Geoprobe borings installed by Pioneer for the DEQ in 1998 (Pioneer 1999) and sampling six hollow-stem auger borings and one monitoring well installed by Pentacore Resources, LLC. (2001). Pioneer performed laboratory analysis on eight soil samples and five groundwater samples. TCE was detected in seven of the soil samples ranging from 0.0128 to 0.543 mg/kg, several of which exceeded the SSL of 0.173 mg/kg. TCE and cis-1,2 DCE were also detected in groundwater samples ranging from 246 to 3,200 µg/L for TCE and 24 to 11,000 µg/L for cis-1,2 DCE, which exceeded MCL/WQB-7 criteria levels. The majority of these samples were collected in the vicinity of the drainfield connected to the oil-water separators that received washwater discharge originating from the steam clean bay.

In 2000, Pentacore completed six hollow-stem auger soil borings within the Beall property boundary, collected 22 soil samples for laboratory analysis, and completed and sampled one monitoring well. Soil samples were analyzed for VOCs, among other compounds. TCE and cis-1,2-DCE were detected in 15 of the 22 samples ranging in concentration from 0.041 to 0.94 mg/kg for cis-1,2-DCE and 0.041 to 1.3 mg/kg for TCE and did not exceed the site specific SSL for either compound. Distribution of cis-1,2-

DCE and TCE in subsurface soil samples generally showed increasing concentrations near the water table (approximately 40 feet bgs) and horizontally in the boring nearest the oil-water separator.

Groundwater sampling consisted of collecting one sample from the single monitoring well MW012 installed by Pentacore in the drainfield area. The groundwater sample was analyzed for VOCs, among other compounds. For VOCs, only cis-1,2-DCE (227 µg/L) and TCE (48 µg/L) exceeded MCL/WQB-7 criteria (70.0 µg/L cis-1,2-DCE and 5.0 µg/L TCE). A summary of historical analytical data for groundwater COPCs for monitoring well MW012 is presented in Table 4-4. A Mann-Kendall statistical test was performed on groundwater data collected from monitoring well MW012 and is presented and discussed in Section 5.4.

4.3.2 Remedial Investigation Activities

RI activities in Area B included subsurface and surface soil sampling, monitoring well installation and groundwater sampling.

4.3.2.1 Subsurface Soil Sampling

During the RI, 67 subsurface soil samples were collected from 32 borings completed in Area B (Figure 2-3). Nine soil borings (SB200 through SB208) were completed at the Beall facility to determine if contamination remained in the vadose zone and if NAPL was present in the subsurface near the steam clean bay (Figure 2-4). Seven additional borings were completed as monitoring wells (MW200 through MW206) adjacent to the steam cleaning bay, the drainfield, the oil-water separator system, and upgradient of the facility. Twenty four subsurface soils samples were collected from 16 monitoring well locations downgradient of the Beall facility.

Subsurface soil samples were generally collected at 5-foot intervals through the vadose zone and into the saturated zone from the surface to 66 feet bgs. Subsurface soil samples were sent to the laboratory for VOC analysis. Soil sampling results for PCE, TCE, cis-1,2-DCE, trans-1,2-DCE, and VC are presented on Figures 4-4 and 4-5. Complete analytical results can be found in Appendix G.

VOCs were detected in 17 of the 67 subsurface soil samples. Detections of VOCs in nine of the 17 soil samples were from samples collected at depths greater than 40 feet bgs, and 14 of the samples were collected from depths greater than 25 feet bgs. Maximum soil VOC concentrations were detected in borings adjacent to the steam cleaning bay at the Beall facility (SB200 and SB201).

PCE was detected in two samples collected from locations (MW200 and MW201) near the steam cleaning bay at the Beall facility; the concentrations were 0.069J and 0.11J mg/kg and did not exceed SSL of 0.172 mg/kg. The PCE detections were found in samples collected from the 26.5- to 32.0-foot interval. PCE was not detected in subsurface soil samples collected from other locations in Area B.

TCE was detected in 16 subsurface soil samples collected at the Beall facility at concentrations ranging from 0.06J to 1.9J mg/kg. Eleven of the TCE detections exceeded the site specific SSL of 0.173 mg/kg. Most (12) of the TCE detections were found in samples collected between 28.0 and 56.0 feet bgs, with several (4) TCE detections found in samples collected between 4.0 and 20.0 feet bgs. Concentration ranges for TCE were within the same concentration range as those previously collected within the drainfield area (Pentacore Resources, LLC 2001). TCE was not detected in subsurface soil samples collected from other locations in Area B.

Cis-1,2-DCE was detected in 10 subsurface soil samples at the Beall facility at concentrations ranging from 0.058J to 0.62 mg/kg. Cis-1,2-DCE was not detected in any subsurface soil sample above the site specific SSL of 1.071 mg/kg. Concentration ranges for cis-1,2-DCE were within the same concentration range as those previously collected within the drainfield area (Pentacore Resources, LLC 2001). Cis-1,2-DCE was not detected in subsurface soil samples collected from other locations in Area B.

Trans-1,2-DCE and VC were not detected above laboratory detection limits in any subsurface soil sample collected in Area B.

4.3.2.2 Surface Soil Sampling

Surface soil samples (0 to 2 feet bgs) were collected at the Beall facility during the RI to evaluate the presence of surface soil contamination, potential sources of contamination, and worker exposure to COPCs. Fourteen surface soil samples (SS203 through SS216) were collected on a grid pattern (Figure 4-5). All of the surface soil samples were analyzed for VOCs. PCE, TCE, and cis-1,2-DCE were detected in ten of the 14 surface soil samples.

Three surface soil samples contained TCE above the site specific SSL of 0.173 mg/kg, ranging in concentration from 0.31 to 2.1 mg/kg. PCE and cis-1,2-DCE detections did not exceed the site specific SSLs of 0.172 mg/kg and 1.071 mg/kg. Trans-1,2-DCE and VC were not detected above laboratory detection limits in any surface soil sample collected in Area B.

4.3.2.3 Groundwater Sampling

During RI field activities, TtEMI installed 23 monitoring wells in Area B (Figures 2-3 and 2-4). Groundwater sampling was conducted in all monitoring wells at LSPGS during two RI sampling events in July/August and October 2002. Groundwater sampling results for PCE, TCE, cis-1,2-DCE, trans-1,2-DCE, and VC are summarized on Figures 4-6A and 4-6B. Complete analytical results are provided in Appendix G.

The monitoring well locations were selected to investigate the potential for additional sources of VOCs, if any, and to further define the VOC plume in the alluvial aquifer. Monitoring well locations were also selected to provide transects of monitoring points perpendicular to the current groundwater flow direction (Figure 4-3). All newly installed monitoring wells in Area B that were completed in the alluvial aquifer were fully screened across the alluvial aquifer to bedrock. Detected PCE, TCE, and cis-1,2-DCE concentrations for monitoring wells along transects A and B are plotted on Figures 4-3A and 4-3B.

Groundwater Sampling at the Beall Facility

Six monitoring wells MW200, MW201, MW203, MW204, MW205, and MW206 were completed in the alluvial aquifer (screen interval approximately 45 to 65 feet bgs); monitoring well MW219 was completed in the bedrock aquifer (screen interval approximately 95 to 105 feet bgs); and monitoring well MW202 was completed as a background well in the alluvial aquifer (screen interval approximately 33 to 73 feet bgs).

All groundwater samples collected from monitoring wells installed at the Beall facility contained one or more detectable VOCs (PCE, TCE, cis-1,2-DCE, trans-1,2-DCE, and VC) during the July/August and October 2002 RI sampling events (Appendix G). Groundwater concentrations of one or more VOCs exceeded MCL/WQB-7 criteria in seven existing and new monitoring wells (MW012, MW200, MW201, MW203, MW204, MW205, and MW206). The maximum TCE concentrations at the Beall facility were detected in monitoring well MW201 at a concentration of 1,850J µg/L and MW200 at a concentration of 1,870 µg/L, with decreasing concentrations downgradient to the west. Cis-1,2-DCE concentrations follow a similar distribution in the downgradient direction and generally demonstrate increasing concentrations relative to TCE (decreasing TCE to DCE ratio). Only one monitoring well (MW201) contained detectable VC at 0.7 µg/L during one sampling event in August 2002.

VOCs were also detected in the bedrock monitoring well (MW219) during both the July/August and October 2002 sampling events. In the sample collected during the July/August event, cis-1,2-DCE was detected at 0.28J µg/L and no other COPCs were detected; however, the sample collected during the October sampling event contained 1.8 µg/L of cis-1,2-DCE and 12 µg/L of TCE. Vertical gradient calculations (presented in Table 3-3) indicate a variable gradient with downward flow from the alluvial to the bedrock aquifer during the August measurement and upward flow during the September and October measurements. The extent of TCE and DCE contamination in the bedrock aquifer downgradient of the Beall facility is not known.

Monitoring wells at the Beall facility contained detectable TCE in one or both RI sampling events; however, well MW202 (the presumed background well) also had detectable cis-1,2-DCE at 0.22J µg/L during the October 2002 sampling event; but the cis-1,2-DCE detection did not exceed the MCL/WQB-7 criteria of 70.0 µg/L. Cis-1,2-DCE was detected in all seven monitoring wells during one or more monitoring events. Cis-1,2-DCE was also detected in monitoring wells MW200, MW201, and MW204 at concentrations above the MCL/WQB-7 criteria.

Groundwater Sampling Downgradient of the Beall Facility

During RI field activities beginning in June 2002, TtEMI installed 15 monitoring wells in Area B downgradient from the Beall facility. Groundwater sampling results for PCE, TCE, cis-1,2-DCE and VC are presented in Figures 4-6A and 4-6B.

The lateral extent of COPCs in groundwater currently covers an area from the Beall facility to the Yellowstone River. The majority of plume mass is currently observed east of Interstate Highway 90. Of the six monitoring wells located immediately downgradient from the Beall facility between U.S. Highway 87 and Interstate Highway 90, five of the wells (see Figures 4-6A and 4-6B) contained detectable TCE above the MCL/WQB-7 criteria of 5.0 µg/L and one well (MW210) also contained cis-1,2-DCE above the MCL/WQB-7 criteria of 70.0 µg/L during one or more RI sampling events. During the July/August sampling event, monitoring well MW210 contained the highest concentrations of TCE (30J µg/L) and cis-1,2-DCE (35 µg/L) downgradient of the Beall facility. During the October sampling event, monitoring well MW212 contained the highest concentrations of TCE (35 µg/L) and cis-1,2-DCE (52 µg/L) downgradient of the Beall facility.

Of the 15 monitoring wells installed downgradient of the Beall facility, nine of the wells were installed west of Interstate Highway 90. Low-level PCE, TCE, and cis-1,2-DCE groundwater contamination persists in the alluvial aquifer downgradient of the Beall facility, west of Interstate Highway 90. PCE was detected in six of 11 existing and new monitoring wells sampled during the RI in this area, however, concentrations were all below the MCL/WQB-7 criteria of 5.0 µg/L. TCE concentrations that exceeded the MCL/WQB-7 criteria of 5.0 µg/L were reported in six monitoring wells (MW013, MW023, MW209, MW216, MW217, and MW300) during one or more RI sampling events, with the highest concentration in MW217 and MW300 at 15 µg/L. Cis-1,2-DCE was not detected above MCL/WQB-7 criteria in this area. Monitoring well MW302 contained detectable VC at 1.1J µg/L during the July/August 2002 sampling event. VC was not detected in the well during the October 2002 sampling event. Monitoring well MW302 is located within an active septic drain field.

4.4 AREA C INVESTIGATION

This section presents the results of RI sampling activities in the Area C including data from previous investigations.

4.4.1 Previous Investigations

During previous investigations, contaminants detected in groundwater in Area C include VOCs (PCE; TCE; cis-1, 2 DCE; 1,1-DCE; 1,1,1-TCA; 1,1,2-TCA) and BTEX compounds (TtEMI 2002a). Several sources for the BTEX compounds have been identified and mitigated through the Montana LUST program; however, no source for the VOCs in this area has been positively identified. The highest and most persistent VOCs in groundwater in this area are TCE and cis-1,2-DCE. Previous investigations of VOCs in the vicinity of the LWSD have reported low levels of TCE (up to 8 µg/L) and cis-1,2-DCE (up to 4.1 µg/L) in surface water including the Coulson irrigation ditch.

4.4.2 Remedial Investigation Activities

RI activities in Area C included subsurface soil and groundwater sampling.

4.4.2.1 Subsurface Soil Sampling

Opportunistic subsurface soil samples were collected in Area C during the RI to identify potential sources of contamination and to evaluate worker exposures to COPCs. Sixteen subsurface samples were collected from nine hollow-stem auger borings during monitoring well installation (Figure 4-7) and were analyzed for VOCs. Samples were collected from the vadose zone (4 to 6 ft bgs), near the water table (9 to 12 ft bgs), and from the alluvial aquifer (to a depth of 30 feet). VOCs were not detected in any of the soil samples above the laboratory detection limit.

4.4.2.2 Groundwater Sampling

Low-level PCE, TCE, and cis-1,2-DCE groundwater contamination exists in the alluvial aquifer in Area C, downgradient of the Beall facility and west to the Yellowstone River. PCE was detected in four of 18 monitoring wells sampled during the July/August 2002 sampling event; however, concentrations were all below the MCL/WQB-7 criteria of 5.0 µg/L (Figures 4-8A). PCE was not detected above the laboratory detection limit during the October 2002 sampling event (Figure 4-8B). TCE was detected in 16 of 18 monitoring wells during one or more RI sampling events. Only one monitoring well (MW308) contained TCE (5.1 µg/L) above the MCL/WQB-7 criteria of 5.0 µg/L. Cis-1,2-DCE was detected in all wells sampled during the two RI sampling events with the exception of RW027; however, all reported concentrations were below the MCL/WQB-7 criteria of 70.0 µg/L. Trans-1,2-DCE and VC were not detected above laboratory detection limits in any well in Area C during RI sampling.

4.5 SURFACE WATER AND SEDIMENT INVESTIGATION

Surface water and sediment samples were collected from three surface water features located within the LSGPS including the AJ Gravel pond, the wetland area on Cerise Road, and the Coulson irrigation ditch. A description of each of these surface water features is provided in Section 3.3. The following sections describe the surface water and sediment results for the RI.

4.5.1 AJ Gravel Pond Sampling

Four monitoring stations (SW100, SW101, SW102, and SW104) were established at the AJ Gravel pond during the RI (Figure 2-1). Surface water and collocated sediment samples were collected from stations SW100, SW101, and SW102 and were analyzed for VOCs. No samples were collected from SW104; this station was established for recording water levels only. Sampling results for PCE, TCE, cis-1,2-DCE, trans-1,2-DCE, and VC are provided on Figure 4-9.

PCE exceeded MCL/WQB-7 criteria of 5.0 µg/L in samples from all three stations at concentrations ranging from 9.8 to 27 µg/L. TCE was detected below MCL/WQB-7 criteria of 5.0 µg/L in samples from all three stations at concentrations ranging from 2.8J to 4.1J µg/L. Cis-1,2-DCE was detected below MCL/WQB-7 criteria of 70.0 µg/L in samples from all three stations at concentrations ranging from 9.4 to 14 µg/L. Trans-1,2-DCE was not detected above the laboratory detection limit of 0.5 µg/L in any surface water sample collected at the pond. VC was detected above WQB-7 criterion of 0.2 µg/L in samples from all three stations at concentrations ranging from 0.81 to 2.2 µg/L. (The MCL criterion for VC is of 2.0 µg/L.) COPCs were not detected above laboratory detection limits in any sediment sample collected in the pond.

Water-level information from SW104 indicated that the elevation of the water in AJ Gravel pond is equivalent to the elevations of groundwater in adjacent monitoring wells MW126, MW127, and MW005. Additionally, COPC concentrations in the surface water samples collected at AJ Gravel pond are comparable to concentrations detected in groundwater from nearby upgradient (MW005) and downgradient (MW126 and MW127) monitoring wells. See Section 4.6.3 for a discussion on the influence of COPCs found in surface water at the AJ Gravel pond resulting from contaminated groundwater recharge. A discussion of the hydraulic relationship between groundwater and surface water at the AJ Gravel pond is provided in Section 3.4.7.

4.5.2 Wetland Area Sampling

In May 2002, one surface water and one collocated sediment sample were collected from station SW103 located in the wetland area by Cerise Road, and were analyzed for VOCs. Sampling results are shown on Figure 4-9. The surface water sample contained TCE at a concentration of 0.23 µg/L, which was below the MCL/WQB-7 criteria. This TCE concentration is comparable to the TCE concentrations collected in adjacent monitoring well MW305 at a concentration of 2.6 µg/L (Figure 4-8A) and monitoring well MW111 at a concentration of 2.5J µg/L (Figure 4-2A). No VOCs were detected above laboratory detection limits in the sediment sample. A discussion of the hydraulic relationship between groundwater and surface water at the wetland area, a former chute channel originating from the Yellowstone River, is provided in Section 3.4.7.

4.5.3 Coulson Irrigation Ditch Sampling

In May 2002, surface water and collocated sediment samples were collected from eight monitoring stations located in Coulson irrigation ditch during minimal flow conditions, prior to irrigation season flows. Sampling results are shown on Figure 4-9. VOCs including PCE, TCE, and cis-1,2-DCE, were detected in seven of the eight surface water samples, all at concentrations below MCL/WQB-7 criteria.

No VOCs were detected in a surface water sample collected at station SW303, the most upstream sample collected from the ditch. The highest concentrations of PCE, TCE, and cis-1,2-DCE were detected in surface water sample SW300, the next downstream monitoring station in the ditch from station SW303. PCE was reported at 0.13 µg/L; TCE was reported at 2.6 µg/L; and cis-1,2-DCE was reported at 0.98 µg/L. One additional detection (0.4 µg/L) of cis-1,2-DCE occurred in a sample collected at station SW301. TCE was detected in all surface water samples downstream of station SW300. Concentrations of TCE generally decreased downstream of SW300, with the lowest concentration (0.54 µg/L) reported at the most downstream station (SW005). It is likely that COPC concentrations in surface water in the Coulson irrigation ditch were being impacted by concentrations of COPCs found in groundwater discharging into the ditch. See Section 4.6.3 for a discussion on the influence of COPCs in surface water resulting from contaminated groundwater recharge. A discussion of the hydraulic relationship between groundwater and surface water at the Coulson irrigation ditch is provided in Section 3.4.7.

Eight sediment samples were collocated with the surface water samples in the Coulson irrigation ditch. Sediment sampling results are shown on Figure 4-9. No COPCs were detected above laboratory detection limits in any of the sediment samples.

4.6 SUMMARY OF NATURE AND EXTENT OF CONTAMINATION

This section presents a summary of the nature and extent of contamination at the LSGPS including a summary of contaminant sources and resultant groundwater contamination where COPC concentrations are greater than regulatory levels and may require remedial action.

4.6.1 Sources of Contamination

The RIWP (TtEMI 2002a) proposed that an additional 35 monitoring wells be installed across the site to, in part, identify potential sources of contamination. Monitoring wells installed during the RI (along with existing monitoring wells) are distributed across the entire site along six transects (Transects A through F), located at distances of approximately 700, 1,800, 2,600, 3,400, 4,000 and 4,500 feet downgradient of the Beall Trailers area (Figure 4-3). Wells installed along each of these transects were located at a spacing of approximately 350 feet to allow detection of potential groundwater contaminant plumes. In addition, wells were located to the maximum extent possible directly down gradient of potential sources. A summary of soil and groundwater sampling from transect wells is provided in Table 4-5.

Two primary sources of groundwater contamination have been identified at the LSGPS based on surface water, soil, soil gas, and groundwater sampling results from previous investigations and this RI. These sources have been identified as the Brenntag source area and the Beall source area. No other source areas have been identified based on the results of both groundwater and soil sampling (Table 4-5). A summary of surface and subsurface samples where chemical concentrations are greater than site specific SSLs is provided in Table 4-6. Surface and subsurface soil with COPC concentrations above site specific SSLs are considered potential sources for groundwater contamination. No surface or subsurface samples from locations outside the Brenntag and Beall source areas contained COPC at concentrations above SSLs.

In Area A, concentrations of COPCs above site-specific SSLs were reported in subsurface soil samples taken from PT001, PT002, PT003, MW105, MW116, MW117, and MW122. All of these sample locations are either within or downgradient of the Brenntag source area and all are considered associated with this source area. The highest concentrations of COPCs were found in PT001, PT002, and PT003. The limits of this identified source material at the Brenntag source area have not been fully defined and additional work is being proposed to address this data gap (see Section 8.0). The extent of the highest concentrations of source material can be approximated by the area encompassing wells PT001, PT002, PT003, MW100, and MW105 (approximately 1 acre). If the average depth of source material within this area is assumed to be 22 ft bgs (based on PCE concentrations in PT001 above SSLs at 20 to 22 ft bgs), the preliminary estimate of the volume of source material within the Brenntag source area is 35,500 cubic yards. No other sources have been identified in Area A.

In Area B, concentrations of COPCs above site-specific SSLs were reported in subsurface soil samples taken from MW200, MW201, SB200, SB201, SB202, and SB206, and surface soil samples SS205, SS208, and SS213. All of these sample locations are within the Beall source area and all are considered associated with this source area. The presence of COPCs at concentrations above site-specific SSLs in surface soil samples may suggest recent releases of VOCs at these locations. Concentrations of COPCs above SSLs reported in downgradient wells (for example MW204 and MW206) are considered associated with an upgradient contaminant release. However, the limits of this identified source material at the Beall source area have not been fully defined and additional work is being proposed to address this data gap (see Section 8.0). Additional source material is likely found beneath the oil-water separator and associated steam clean bay drainage pipe, based on groundwater COPC concentrations in this area. Maximum detected VOC concentrations of 1,870 µg/L TCE and 1,380 µg/L DCE in groundwater samples from the Beall facility do not suggest that NAPL was released or is present in the subsurface. Some degradation of TCE to DCE likely occurred before the compounds reached the groundwater either in the oil-water separator system or in the vadose zone during infiltration. Detected PCE is likely the result of PCE as an impurity in the parent TCE product based on relative concentration ratios. Because of the limited width of potentially contaminated soil beneath these features, the volume of this source area is difficult to estimate. Additional subsurface investigation of this area may further delineate this source but it may be difficult to locate contaminated soil without removal and inspection of the oil-water separator and piping. No other sources have been identified in Area B.

In Area C, no concentrations of COPCs above SSLs were reported in subsurface or surface soil samples. No sources have been identified in Area C.

4.6.2 Groundwater Impacts

The highest concentrations of VOC groundwater contamination at the LSGPS have been reported within and downgradient of the Brenntag and Beall source areas. In addition, lower level concentrations of VOC groundwater contamination has been reported throughout the site. Monitoring wells where samples have contained chemical concentrations greater than MCL/WQB-7 criteria are shown in Table 4-7. Concentrations in Table 4-7 are the maximums reported during the July/August and October 2002 sampling events. These well locations are shown on Figure 4-10. Site-wide COPC concentrations in groundwater for the July/August and October 2002 sampling events are present in Figures 4-10 through 4-18. PCE concentrations are shown on Figures 4-11 and 4-12. TCE concentrations are shown on Figures 4-13 and 4-14. Cis-1,2 DCE concentrations are shown on Figures 4-15 and 4-16. VC concentrations are shown on Figures 4-17 and 4-18.

4.6.3 Surface Water and Sediment Impacts

Surface water samples with COPC concentrations greater than MCL/WQB-7 criteria are shown in Table 4-8. No COPCs were reported in sediment above site specific SSLs. Concentrations of COPCs in surface water above regulatory criteria were reported in samples from stations SW100, SW101, and SW102, all located within the AJ Gravel Pond. COPCs in AJ Gravel pond are attributed to contaminated groundwater adjacent to the pond that discharges into the pond.

COPCs detected in Coulson irrigation ditch surface water are below MCL/WQB-7 criteria and are considered the result of contaminated groundwater recharge. Coulson irrigation ditch likely receives groundwater discharge when water levels in the ditch are below the groundwater table. Contaminated groundwater that discharges to the ditch appears to occur at and below station SW300 during nonirrigation flows. When irrigation flows are present in the ditch, water levels are elevated above those in adjacent groundwater and little or no groundwater to surface water discharge is expected.

4.6.4 Site Boundaries

The LSGPS site boundaries were originally designated in the RIWP (TtEMI 2002a) to encompass areas of known or suspected contamination. Based on the results of the RI, groundwater contamination has generally been delineated on the southern and eastern boundaries by samples with no COPCs above laboratory detection limits. The delineation on the eastern side of the site includes the use of two residential wells. The southern edge is defined by the alluvial aquifer/bedrock boundary which generally follows the Lower Lockwood irrigation ditch. The groundwater plume is delineated on the western and northern edges by the Yellowstone River. No sampling was conducted on the northern or western sides of the river during the RI because it is assumed that the alluvial aquifer discharges to the river.

The site boundaries (see Figure 1-2) have been adjusted slightly based upon the results of RI and may be revised in the future if determined necessary by DEQ/RD and EPA. The changes in the site boundaries presented in the RIWP include the following:

- The southern boundary has been adjusted to run from the Yellowstone River east along the Lower Lockwood irrigation ditch, U.S. Highway 87 East, and along a line south of the Beall Trailers facility.

- The southeastern boundary has been adjusted to run south on a line extending from the east end of Gordon Drive to U.S. Highway 87 East and along a line east of the Beall Trailers facility.

5.0 CONTAMINANT FATE AND TRANSPORT EVALUATION

This section provides an evaluation of the fate and transport of dissolved VOCs at the LSGPS. Site physical characteristics, contaminant characteristics, and an analysis of the extent of the contamination are combined in the evaluation of contaminant fate and transport. The contaminant fate and transport processes at the LSGPS are addressed in the following sections:

- Section 5.1 provides a description of the uses and physical, chemical, and toxicological properties of the COPCs
- Section 5.2 describes the fate and transport processes of the COPCs, particularly an evaluation of natural attenuation processes on the contaminant plumes originating from source areas
- Section 5.3 includes an assessment of the hydrogeologic data and provides general physical properties of the aquifer, groundwater flow velocities and directions, and contaminant flow velocities in groundwater
- Section 5.4 describes the approach, assumptions, and results of the contaminant fate and transport modeling performed for the groundwater plume at the LSGPS, and
- Section 5.5 summarizes the evaluation and results of contaminant fate and transport processes at the LSGPS

5.1 PROPERTIES OF CONTAMINANTS OF POTENTIAL CONCERN

The COPCs as identified in the RI are: PCE, TCE, cis- and trans-1,2-DCE, and VC. These COPCs are included in a larger group of chemicals called VOCs. The following subsections discuss the physical and chemical characteristics along with the toxicological effects of each of the COPCs. In addition, a general description of the typical breakdown products and parent compounds is provided.

5.1.1 Historical Uses of Contaminants of Potential Concern

This section describes the uses of PCE, TCE, cis-1,2-DCE, trans-1,2-DCE, and VC as they pertain to general industrial and manufacturing processes; historical uses of, or the presence of these chemicals in groundwater at the LSGPS are also discussed.

Tetrachloroethene

PCE is used as a dry cleaning and textile-processing solvent, for vapor degreasing and metal cleaning operations. It is also used in the production of chlorofluorocarbons; in aerosol formulations; as a carrier for rubber coatings, solvent soaps, printing inks, adhesives, sealants, polishes, lubricants, and silicones; and as a solvent in various consumer products such as typewriter correction fluid and shoe polishes (EPA 1994b). It is also used as a chemical intermediate. Chemical species that are produced in one step of an industrial process and consumed in subsequent steps are called intermediates. In textile processing, it is used as a scouring solvent that removes oils from fabrics after knitting and weaving operations, and as a carrier solvent for sizing and desizing and for fabric finishes and water repellents. PCE can be used to clean and dry contaminated metal parts and other fabricated materials. It is also present in many paint thinners, brake cleaners, and antifreeze.

At the LSGPS, PCE has been detected in monitoring wells at concentrations greater than 1,000 µg/L in Area A. Low-level concentrations (less than 10 µg/L) of PCE have been detected in Area B and Area C. The higher concentrations of PCE in groundwater in Area A suggest a historic release of this VOC may have occurred during previous storage and handling of products. Records indicate PCE has been stored and handled at the Brenntag facility. In Area B, PCE may have been a minor component (impurity) of the solvents used in a vapor degreasing operation at the Beall facility.

Trichloroethene

TCE is used primarily as a solvent to remove grease from metal parts but is also an ingredient in adhesives, paint removers, typewriter correction fluid, and spot removers (ATSDR 1997b). It is used to break down fats, waxes, resins, oils, rubber, paints, and varnishes (Merck Index 1996) and is also an excellent extraction solvent for greases and tars. TCE is used in the textile processing industry to scour cotton, wool, and other fabrics. The textile industry also uses TCE as a solvent in waterless dying and finishing operations. As a general solvent or as a component of solvent blends, TCE is used with adhesives, lubricants, paints, varnishes, paint strippers, pesticides, and cold metal cleaners. TCE is also used in the production of PVC. Other chemical intermediate uses of TCE include production of pharmaceuticals, polychlorinated aliphatic compounds, flame retardant chemicals, and insecticides (ATSDR 1997b). Various consumer products found to contain TCE include typewriter correction fluids, paint removers and strippers, adhesives, spot removers, and rug-cleaning fluids. Prior to 1977, TCE was used in veterinary medicine as a general and obstetrical anesthetic, a grain fumigant, surgical disinfectant, a pet food additive, an extractant of spice oleoresins in food, and for production of decaffeinated coffee (ATSDR 1997b). TCE also results as a degradation product of PCE.

At the LSGPS, TCE in Area A may be present as a result of PCE degradation, and may also have been released during the storage and handling of product. Records indicate that TCE has been stored and handled at the Brenntag facility. In Area B, concentrations of TCE have been detected in groundwater monitoring wells at concentrations greater than 1,000 µg/L suggesting this compound may have been used in a degreasing operation, or released during the storage or handling of product. Records indicate that TCE was stored and used in a degreasing operation at the Beall facility.

Dichloroethene

DCE is used to produce solvents and in chemical mixtures. It is a chemical intermediate in the synthesis of chlorinated solvents and compounds such as VC, and used in mixtures of paints, furniture stripper, metal degreaser, soaps, and as a gasoline additive. It was also an ingredient in WD-40, a common penetrating oil. It has also been used as a low-temperature extraction solvent for organic materials such as decaffeinated coffee, dyes, perfumes, lacquers, and thermoplastics (ATSDR 1996). It is a solvent to break down fats, phenol, and camphor (Merck Index 1996). There is evidence that other chlorinated hydrocarbons introduced into subsurface media can undergo biodegradation to cis-1,2-DCE or trans-1,2-DCE. The parent pollutants include such widely used solvents as TCE or PCE (ATSDR 1996).

At the LSGPS, cis-1,2-DCE may be present as a result of PCE and TCE degradation, and may have been released during storage and handling of product.

Vinyl Chloride

VC is a manufactured substance that is used to make PVC. PVC is used to make a variety of plastic products including pipes, wire and cable coatings, and furniture upholstery. VC also results from the breakdown of other substances, such as PCE, TCE, and cis-1,2-DCE. The presence of VC at the LSGPS is most likely related to degradation of PCE, TCE, or cis-1,2-DCE.

5.1.2 General Physical and Chemical Properties

This section describes the general physical and chemical properties of PCE, TCE, cis-1,2-DCE, and VC. Physical and chemical properties are summarized in Table 5-1.

Tetrachloroethene

PCE is a colorless, nonflammable liquid that evaporates readily, dissolves only slightly in water, and has a sharp, sweet odor. PCE can be released to the environment via soil, water, and air. In air, PCE breaks down to other compounds over several weeks (ATSDR 1997a). Because PCE is a liquid that does not bind well to soil, PCE released to soil can move through the soil and enter groundwater (EPA 1994b). Generally in the atmosphere PCE reacts with hydroxyl radicals and breaks down into phosgene, chloroacetylchlorides, formic acid, carbon monoxide, carbon tetrachloride, and hydrochloric acid (ATSDR 1997a). If released to the soil, volatilization will slowly occur with rapid leaching into the soil and groundwater. Under methanogenic conditions, PCE will breakdown into TCE and ultimately end up as VC. PCE can remain in the soil for a long period of time. Under anaerobic conditions in groundwater PCE undergoes reductive dehalogenation to TCE. In surface water PCE undergoes volatilization fairly rapidly and will break down into the same compounds as in the atmosphere (ATSDR 1997a).

Trichloroethene

TCE is a colorless, nonflammable liquid that evaporates quickly, dissolves only slightly in water, has a somewhat sweet odor, and a sweet burning taste. TCE can be released to the environment via soil, water, and air. TCE binds to soil particles and can remain in soil for a long time (ATSDR 1997b). Generally in the atmosphere, TCE reacts with hydroxyl radicals and breaks down into phosgene, dichloroacetylchloride, and formylchloride in a short period of time.

If released to the soil, volatilization will rapidly occur with some leaching. Volatilized TCE will eventually break down into the compounds previously identified. TCE binds to soil particles and can remain in the soil for a long period of time.

Under anaerobic conditions, TCE undergoes reductive dechlorination and will break down into cis- and trans-1,2-DCE. In surface water, TCE undergoes volatilization fairly rapidly and will break down into the same compounds as in the atmosphere. TCE may bind to particles in water and settle to bottom sediment (ATSDR 1997b).

Dichloroethene

DCE is a colorless, highly flammable liquid that evaporates rapidly, dissolves readily in water, and has a sharp, harsh odor. There are two forms of DCE, cis-1,2-DCE and trans-1,2-DCE, the difference being the relative location of chlorine atoms on the molecule. The 1,2-DCE isomer that has the two chlorine groups next to each other is referred to as the cis-isomer; the isomer with the two chlorine groups opposite each other is called the trans-isomer. Sometimes, both forms are reported together as 1,2-DCE, and sometimes the individual isomers (cis- and trans-) are reported separately (ATSDR 1996). Generally in the atmosphere, DCE reacts with hydroxyl radicals and undergoes photooxidation. It then breaks down into formyl chloride, monochloroacetyl chloride, dichloroacetaldehyde, which further reacts and forms phosgene, formyl chloride, formaldehyde, carbon monoxide, carbon dioxide, and hydrochloric acid in a short period of time. If released in the soil, some DCE will rapidly volatilize. A small amount will leach into the soil and subsequently to the groundwater. In the groundwater under anaerobic conditions cis-1,2-DCE and trans-1,2-DCE will degrade to VC. In surface water, DCE undergoes volatilization rapidly (ATSDR 1996).

Vinyl Chloride

VC is typically a colorless, flammable gas that evaporates easily, dissolves sparingly in water, and has a mild, sweet odor. VC can be in liquid form under high pressure and low temperatures. Liquid VC evaporates easily into the air. If it is near the surface of soil or water, it can also evaporate. VC breaks down in air within a few days and VC formed from the breakdown of other chemicals can enter groundwater (ATSDR 1997c). Generally in the atmosphere, VC rapidly reacts with hydroxyl radicals and breaks down into hydrochloric acid, formaldehyde, formyl chloride, acetylene, chloroacetal dehyde, and chloroacetyl chloranl epoxide. If released to the soil, it will undergo rapid volatilization and may easily leach into the soil because of its solubility in water and most organic solvents. In the groundwater, VC breaks down into ethane under aerobic conditions. In surface water, VC volatilizes and undergoes photodegradation.

5.1.3 General Toxicological Information

This section provides the general exposure routes and toxicological and carcinogenic effects to humans from PCE, TCE, DCE, and VC. Detailed toxicity information is provided in Appendix O.

Tetrachloroethene

PCE enters the body when breathed in with air or when consumed with contaminated food or water. It is less likely to be absorbed through skin contact. Breathing PCE for short periods of time can adversely affect the central nervous system. Effects range from dizziness, fatigue, headaches, and sweatiness to loss of coordination and unconsciousness. Contact with PCE liquid or vapor irritates the skin, the eyes, the nose, and the throat. These effects are not likely to occur at concentrations of PCE that are normally found in the environment (EPA 1994b).

Breathing PCE over longer periods of time can cause liver and kidney damage in humans. Repeated exposure to large amounts of PCE in air can cause memory loss and confusion. Laboratory studies show that PCE causes kidney and liver damage and cancer in animals exposed repeatedly by inhalation and by mouth. Repeat exposure to large amounts of PCE in air may likewise cause cancer in humans (EPA 1994b). It is generally agreed by the International Agency for Research on Cancer and EPA that PCE may cause cancer. EPA's weight-of-evidence classification is on the C-B2 continuum, where Class C indicates PCE is a possible human carcinogen, and B2 indicates PCE is a probable human carcinogen. At the present time, the agency has not adopted a final position on the weight-of-evidence classification. Additional specifics about the human and ecological effects of PCE can be found in Sections 6.4.4 and 7.5.1, respectively.

Trichloroethene

TCE can enter the body through inhalation of contaminated dust and vapors or through ingestion of contaminated soil and water. It can also enter the body through the skin. Short-term inhalation exposure to TCE can cause dizziness, headaches, lung irritation, poor coordination, sleepiness, facial numbness, and difficulty concentrating. Inhalation of large amounts of TCE may cause impaired heart function, unconsciousness, and, possibly, death. Ingestion of small amounts of TCE for long periods may cause liver and kidney damage, impaired immune system function, and impaired fetal development in pregnant women. Ingestion of large amounts of TCE can adversely affect the central nervous system, and may cause nausea, liver damage, impaired heart function, unconsciousness, and death. Skin contact with TCE for short periods may cause skin rashes (ATSDR 1997b). Some studies with mice and rats have suggested that high levels of TCE may cause liver or lung cancer. Recent studies by EPA have noted that the International Agency for Research on Cancer and the National Toxicology Program find TCE to be "reasonably anticipated to be a human carcinogen," which places TCE (like PCE) in the B1/B2 cancer class. Additional specifics about the human and ecological effects of TCE can be found in Sections 6.4.4 and 7.5.1, respectively.

Dichloroethene

Exposure to DCE can result from breathing contaminated air or by ingesting contaminated food or water. Exposure can also result from skin contact with the chemical. Inhalation of high levels of DCE can cause nausea, drowsiness, and sleepiness. Inhalation of very high levels can cause death. Long-term human health effects after exposure to low concentrations of DCE are not known. Lower doses of DCE caused effects on the blood such as decreased numbers of red blood cells, and also effects on the liver. EPA has determined that cis-1,2-DCE is not classifiable as to its human carcinogenicity, placing it in EPA Class D. Likewise, no EPA cancer classification is available for trans-1,2-DCE, also placing it in EPA Class D. Additional specifics about the human and ecological effects of DCE can be found in Sections 6.4.4 and 7.5.1, respectively.

Vinyl Chloride

VC can enter the body through inhalation of contaminated air, ingestion of contaminated food and water, or absorption through skin and eye contact. Inhalation of high levels of VC can cause dizziness and sleepiness. Breathing very high levels can cause unconsciousness and death. Long-term inhalation exposure can cause changes in liver structure. The effects of ingestion of high levels of VC are unknown. Skin contact with the chemical will cause numbness, redness, and blisters. Long-term dermal exposure may cause nerve damage and immune reactions. Results from several studies have suggested that inhalation or ingestion of drinking water containing low levels of VC may increase the risk of getting cancer. Workers who have had chronic inhalation exposure have showed increased risk of getting cancer in the liver. Chronic inhalation exposure may cause brain cancer, lung cancer, and some cancers of the blood. EPA has determined that VC is a known, or Class A, human carcinogen. Additional specifics about the human and ecological effects of VC can be found in Sections 6.4.4 and 7.5.1, respectively.

5.2 CONTAMINANT FATE AND TRANSPORT PROCESSES

General fate and transport processes that may attenuate concentrations of contaminants include dispersion, dilution, sorption, volatilization, abiotic degradation, and biodegradation. These processes are discussed in the following sections with emphasis on evaluation of the fate and transport of VOCs at the LSGPS; particularly the evaluation of natural attenuation processes and behavior of the portions of VOC plumes in Area A and Area B.

5.2.1 Nondestructive Attenuation Mechanisms

Nondestructive attenuation mechanisms result in decreasing contaminant concentrations, but no destruction of contaminant mass. These processes include sorption of contaminants to the aquifer matrix, dispersion or spreading of the contaminant mass in the aquifer during transport, dilution of the contaminant mass by recharge from infiltration or surface water, and volatilization of contaminants from the saturated zone to the vadose zone.

Sorption immobilizes contaminant mass by processes that fix or sorb compounds to the aquifer matrix. Organic carbon and clay minerals with large surface area to volume ratios generally act as sites of adsorption. Sorption causes slowing or retardation of the contaminant mass relative to the bulk groundwater velocity. Sorption is quantified using the coefficient of retardation or retardation factor. Calculation of site specific retardation factors for COPCs at LSGPS is presented in Section 5.3.5.

Dispersion describes the longitudinal and transverse spreading of the contaminant plume. Dispersion is caused by porosity scale velocity variations in the aquifer matrix during advective transport of contaminants. Dispersion results in the three dimensional mixing of the contaminants, but does not effect the total mass present in the plume. In relatively fast moving groundwater systems such as at LSGPS, the dominant dispersion process is mechanical mixing along tortuous flow paths in the aquifer matrix. Measurements of dispersivity are extremely difficult, and estimates are often based on plume length (Wiedemeir and others 1998). For this investigation, it was assumed that the longitudinal dispersivity ranges from about 10 feet in Area B to about 20 feet in Area A.

Dilution of contaminant mass may occur along the flow path of plumes through infiltration of recharge from precipitation, or recharge from surface water bodies resulting in dilution of contaminant concentrations along a flow path. Significant dilution of the contaminant mass is not likely occurring at the LSGPS due to overall low net infiltration rates. Surface water from Coulson irrigation ditch may locally recharge groundwater during flow irrigation periods.

Volatilization of contaminant mass may occur across the water table resulting in loss of contaminant mass to the vapor phase in the vadose zone. Soil gas measurements taken above VOC plumes indicate that this process is active at some sites depending on hydrogeologic characteristics. Volatilization of VOCs from groundwater and through the vadose zone can result in intrusion of VOCs from soil gas to subsurface structures and indoor air.

5.2.2 Destructive Attenuation Mechanisms

Destructive attenuation mechanisms of organic contaminants result in the removal of contaminant mass by biological and abiotic processes. Biological processes proceed by two broad mechanisms: (1) the utilization of the contaminant as a primary growth substrate by microorganisms using oxygen (aerobic) or minerals such as nitrate, ferric iron, sulfate, and carbon (anaerobic) as electron acceptors, or (2) where contaminants are fortuitously degraded by cometabolism with another substrate without producing energy for the microorganism. Abiotic processes include hydrolysis where halogens are replaced by hydroxyl (OH) groups. Abiotic processes are not considered to be as important except in particular cases such as carbon tetrachloride and chlorinated ethanes such as TCA (Wiedemeir and others 1998). The following sections provide a detailed evaluation of these parameters at the LSGPS.

5.2.3 Chlorinated Solvent Biodegradation

Chlorinated solvents in groundwater have been demonstrated to biodegrade by several mechanisms including reductive dechlorination of PCE, TCE, DCE, and VC, direct aerobic and anaerobic oxidation of DCE and VC, ferric iron reduction of VC, and cometabolism of TCE. One or more of these processes may be active within a particular plume. Reductive dechlorination, where the chlorinated compound is an electron acceptor, is the only biological process known to be able to completely degrade chlorinated compounds in groundwater.

Conditions necessary for reductive dechlorination include the following:

- Halospiring bacteria, particularly strains of *Dehalococcoides sp.*
- Sufficient electron donor for carbon and hydrogen
- Sulfate reducing or methanogenic conditions
- Dissolved hydrogen concentrations greater than one nanoMole (nM)

These conditions were evaluated at the LSGPS to determine if lines of evidence for natural attenuation exist and to provide data for use in evaluating it as a remedial alternative. Geochemical evidence supporting the presence of reductive dechlorination is presented in Section 5.2.5.

5.2.4 Behavior of Chlorinated Solvent Plumes

Wiedemeier and others (1998) classified chlorinated solvent plume behavior into three general categories based on the chlorinated solvent contaminant mass, the amount of natural or man-made organic carbon present in the aquifer, and the distribution, concentration and utilization of natural electron acceptors. Recent research by Major and others (2000) indicated that the absence of strains of *Dehalococcoides*-like (Dhc) microorganisms will stall reductive dechlorination of PCE and TCE at DCE. Alternatively, Nyer and others (2003) indicated that multispecies cultures are responsible for complete dechlorination at most sites and that biogeochemical management of native species is the most effective means of achieving remedial goals.

<u>Type I – Systems that are Anaerobic due to Man-made Carbon</u>

Type I plumes use man-made carbon sources, such as petroleum hydrocarbons, as the primary substrate or energy source. Man-made carbon sources would consist of petroleum compounds, ethanol, or other released organic compounds that are in contact with the source of chlorinated compounds. If insufficient man-made carbon is available, then with time the electron donor supply will be exhausted before complete degradation of the source has occurred. In addition, competing electron acceptors such as dissolved oxygen, sulfate, nitrate, and ferric iron may be competing for available donors. Type I behavior generally results in rapid and extensive degradation of more chlorinated compounds such as PCE and TCE (Wiedemeier and others 1999). Portions of the plume in Area A appear to be exhibiting Type I behavior where man-made carbon from released petroleum and possibly other compounds is the electron donor supply. Type I behavior is also exhibited in monitoring well MW302 which is screened under a septic drain field.

Type II – Systems that are Anaerobic due to Naturally Occurring Carbon

Type II plumes are characterized by high natural concentrations of native organic carbon in the aquifer matrix. Microbial utilization of the native carbon sources drive reductive dechlorination. Type II plumes typically degrade slower than Type I behavior, but may be sufficient to meet remedial time frames. Evaluation of electron acceptor supply and utilization becomes more important with Type II behavior plumes. Competition for electron acceptors may become critical and limit dechlorination (Wiedemeier and others 1999). Type II behavior is not likely occurring at LSGPS due to the relatively high concentrations of competing electron acceptors such as nitrate (> 1.0 mg/L) and the low concentrations of native organic carbon in the sandy gravel portions that make up the bulk of the alluvial aquifer.

Type III – Aerobic Systems due to No Fermentation Substrates

Type III behavior is characterized by low concentrations of native or man-made carbon and dissolved oxygen concentrations greater than 1.0 mg/L. Reductive dechlorination of PCE or TCE will not occur under aerobic conditions. Type III behavior may be occurring within the Beall source area, and downgradient to Lockwood Road where dissolved oxygen concentrations exceed 1.0 mg/L. Elevated groundwater dissolved oxygen values (>1.0 to 4.5 mg/L) also parallel the southern aquifer boundary indicating possible groundwater recharge from the bedrock uplands area.

Mixed Behavior

A single chlorinated solvent plume can exhibit one or more behavior types depending on local conditions and changes in electron donor and acceptor concentrations. The most beneficial situation involves reductive dechlorination of PCE, TCE, and DCE in the source area (Type I) then VC is oxidized downgradient either by Ferric iron reduction or aerobically (Type III).

5.2.5 Geochemical Characterization and Patterns

Geochemical patterns are assessed to establish the footprint of the plume (Wilson 2000). Geochemical patterns can aid in identifying the degradation mechanisms operating within the plume. Parameters expected within reductive dechlorination processes are identified in the *Technical Protocol for Evaluating Natural Attenuation of Chlorinated Solvents in Groundwater* (Weidemeier and others 1998). Groundwater contaminated with PCE or TCE undergoing reductive dechlorination typically reflects the following changes within the most contaminated zones immediately downgradient and within the plume source area:

- Ethene and ethane are present as degradation end products
- Methane is usually present
- Ferrous iron is usually present
- Dissolved oxygen and nitrate are absent
- Dissolved hydrogen concentrations are greater than one nM

The following sections present and discuss geochemical characterization data (Table 5-2) which were collected from 26 RI monitoring wells during the October 2002 sampling event.

5.2.5.1 Ethene and Ethane

Ethene and ethane are end products of reductive dechlorination. Ethene and ethane were detected in 8 of the 14 samples collected from monitoring wells in Area A (Table 5-2). Ethene concentrations ranged from 0.5 to 229 µg/L (8 wells) and ethane concentrations ranged from 0.8 to 2.2 µg/L (2 wells). The maximum concentration of ethene was detected in the Brenntag source area in monitoring well MW011.

Ethene was detected in monitoring well MW201 in the Beall source area at a concentration of 0.3 µg/L. Ethane was detected in the sample at 0.8 µg/L. Ethane was also detected in three monitoring wells, MW106, MW113, and MW119, located downgradient from the Beall source area. Ethane concentrations were low, and ranged from 0.3 to 0.4 µg/L. The laboratory detection limit was 0.2 µg/L. The presence of ethene and ethane coinciding with other COPCs indicates that reductive dechlorination is active and is generating these endpoint degradation products; however limited in the Beall source area. A plume centerline plot of ethene + ethane versus distance from the source area is presented in Figure 5-1. A map showing the distribution of ethane + ethane is presented in Figure 5-2.

5.2.5.2 Methane

Methanogenesis occurs only when fermentation also occurs. Methanogenesis generally occurs after dissolved oxygen, nitrate, and sulfate have been depleted. The presence of methane in groundwater indicates that hydrogen is being generated and reductive dechlorination processes are also likely to occur. Methane can also be generated from degradation of native organic carbon and petroleum hydrocarbons. Hydrogen concentrations are presented and discussed in Section 5.2.5.6.

Methane was detected in 13 of the 14 groundwater samples collected from monitoring wells in Area A. Methane concentrations in groundwater samples ranged from 0.9 to 919 µg/L; the maximum concentration was detected in monitoring well MW011 (Brenntag source area). Methane was not detected in the upgradient monitoring well MW001. A plume centerline plot of methane versus distance from the Brenntag source area is presented in Figure 5-1. A map showing the distribution of methane is presented in Figure 5-3. Maximum methane concentrations correspond with the high concentrations of COPCs in the Brenntag source area and the portion of the plume downgradient on the Brenntag source area in Area A.

Methane was detected in 4 of the 7 groundwater samples collected from monitoring wells in Area B; concentrations ranged from 0.6 to 2.2 µg/L. The maximum methane concentration was detected in monitoring well MW201 (Beall source area). The laboratory detection limit for methane was 0.2 µg/L. Methane was detected in 5 of the 7 downgradient monitoring wells with concentrations ranging from 0.5 to 4.2 µg/L. Overall, the low concentrations of methane indicate that very little methanogenesis is occurring in the portion of the plume in Area B.

5.2.5.3 Ferrous Iron

The presence of ferrous iron indicates the utilization of biologically available ferric iron as an electron acceptor during anaerobic biodegradation. Ferrous iron concentrations can be used as an indicator of the degradation of fuel hydrocarbons, vinyl chloride, or possibly native organic matter. Ferrous iron was detected in 4 of the 14 groundwater samples collected from monitoring wells in Area A; concentrations ranged from 0.7 to 0.8 mg/L. Ferrous iron was detected in three monitoring wells near the Yellowstone River (MW123, MW125, and MW126) as well as MW101 located near the Brenntag source area.

Ferrous iron was not detected in the Beall source area or in the portion of the plume immediately downgradient. Three monitoring wells northwest of the Interstate Highway 90 corridor (MW119, MW308, and MW302), contained detectable ferrous iron ranging in concentration from 0.2 to 3.2 mg/L. The maximum value was detected in monitoring well MW302, which is located within an active septic drain field. The lack of ferrous iron in the source areas indicates that ferric iron (Fe III) reduction is not an active degradation pathway.

5.2.5.4 Dissolved Oxygen

Dissolved oxygen is the most thermodynamically favorable electron acceptor used by microorganisms to degrade organic carbon. Dissolved oxygen is a principal control on redox (oxidation-reduction) processes in groundwater and is controlled by recharge, microbial respiration, biodegradation of organic material, and reactive mineral surfaces. Anaerobic bacteria cannot function at dissolved oxygen concentrations above 0.5 to 1.0 mg/L; therefore, reductive dechlorination cannot occur above these levels. After depletion of available dissolved oxygen by aerobic respiration, anaerobic microbes will utilize other electron acceptors in the following order:

Dissolved Oxygen ? Nitrate ? Ferric (III) Iron ? Sulfate ? Carbon Dioxide

Reductive dechlorination is most effective in the oxidation-reduction potential range of sulfate reduction and methanogenesis, but dechlorination of PCE and TCE can occur with the oxidation-reduction potential range of nitrate and ferric iron reduction (less than 50 millivolts).

Dissolved oxygen concentrations in the alluvial aquifer at the LSGPS vary with concentrations greater than 1.0 mg/L in the upgradient portions of the alluvial aquifer in Area B and along the south side of the site at the alluvial aquifer/bedrock boundary (Figure 5-4). Dissolved oxygen concentrations north of the boundary and northwest of the railroad corridor are generally less than 1.0 mg/L. Wells within the portion of the plume in Area A (Table 5-2) had an average dissolved oxygen concentration of 0.33 mg/L during the October 2002 sampling event. These concentrations indicate that reductive dechlorination may be active in this portion of the plume. A plume centerline plot of dissolved oxygen versus distance from the source area is presented in Figure 5-5. In contrast, wells within the portion of the plume in Area B had an average dissolved oxygen concentration of 3.47 mg/L. These concentrations indicate that reductive dechlorination may not be active in this portion of the plume. The portion of the plume downgradient of the Beall source area, with the exception of an area near the south alluvial aquifer boundary, exhibits anaerobic dissolved oxygen conditions of less than 1.0 mg/L (Figure 5-4); therefore, reductive dechlorination may be active. Four monitoring wells in the downgradient area of LSGPS exhibited dissolved oxygen concentrations greater than 1.0 mg/L; however, these may be due to localized recharge effect from surface water bodies.

5.2.5.5 Nitrate

Nitrogen exists in groundwater as nitrate (NO_3^-), nitrite (NO_2^-), and ammonia (NH_4^+). Nitrate functions as an electron receptor in denitrification or nitrate reduction after dissolved oxygen depletion occurs. For reductive dechlorination to occur, nitrate concentrations in groundwater near source areas must be less than 1.0 mg/L.

Nitrate concentrations (assuming negligible nitrite) in Area A ranged from nondetect to 2.86 mg/L. Nitrate was completely depleted (below detection) in monitoring well MW011; upgradient monitoring well MW001 exhibited a nitrate concentration of 3.04 mg/L. Nitrate concentrations within the downgradient portion of the plume in Area A ranged from nondetect (MW007) to 0.96 mg/L. Monitoring wells MW002 and MW004 contained nitrate at concentrations of 3.03 and 1.96 mg/L, respectively. Overall, there is a marked depletion of nitrate in the Brenntag source area indicating that denitrification is an active degradation pathway. A plume centerline plot of nitrate versus distance from the Brenntag source area is presented in Figure 5-5. A map showing the distribution of nitrate in Area A is presented in Figure 5-6.

Nitrate concentrations in Beall source area monitoring wells varied from 1.22 to 9.24 mg/L. Nitrate concentrations in downgradient wells ranged from 0.19 to 6.23 mg/L. Nitrate concentrations below 1.0 mg/L were detected in two monitoring wells (MW115 and MW119) located near the Yellowstone River. Nitrate concentrations in the portion of the plume in Area B are greater than 1.0 mg/L, indicating that reductive dechlorination is not active.

5.2.5.6 Dissolved Hydrogen

Molecular hydrogen (H_2) is the most important electron donor for reductive dechlorination. Dissolved hydrogen is continuously produced in anaerobic groundwater systems by fermentative microorganisms that decompose natural and man-made carbon. The carbon serves as both the oxidized and reduced species. This hydrogen is consumed by other microorganisms that use nitrate, ferric iron, sulfate, or carbon dioxide (methanogens) as terminal electron acceptors. The more oxidized the system, the more efficiently the hydrogen is used. Hydrogen accumulates under methanogenic conditions and is consumed under denitrifying conditions.

Hydrogen concentrations can be used to identify the active terminal electron acceptor process at a given site (Table 5-3). Reductive dechlorination typically requires sulfate reducing conditions (hydrogen concentration >1 nM).

Hydrogen concentrations were measured in 26 groundwater samples during the October 2002 sampling event. Samples were collected using the bubble strip method described in Section 2.3.4. Samples were analyzed by Microseeps, Inc. in Pittsburg, Pennsylvania. Hydrogen concentrations in samples ranged from 1.0 to 2.4 nM with a site-wide average of 1.67 nM (Table 5-2). Groundwater temperature at the site averages approximately 11.8ºC. At lower temperatures, reductive dechlorination may proceed at lower hydrogen concentrations (> 0.5 nM). Hydrogen concentrations at LSGPS indicate that sulfate reduction is the likely the terminal electron acceptor process in areas of active biodegradation.

5.2.5.7 Sulfate

Sulfate concentrations were measured by laboratory analysis in 26 samples during the October 2002 sampling event. Sulfate concentrations ranged from nondetect to more than 1,100 mg/L with an average concentration of 671 mg/L. High sulfate concentrations in groundwater are common in central Montana and result from the weathering of coal-bearing and pyritic Cretaceous rocks such as the bedrock in the Billings area. Sulfate concentrations greater than 20 mg/L may inhibit reductive dechlorination; however, in many plumes with high sulfate concentrations, reductive dechlorination still occurs (Wiedemeier and others 1998). One monitoring well in the LSGPS (MW308) exhibited complete depletion of sulfate, possibly due to biodegradation activity. Some depletion of sulfate in monitoring well MW011 in the Brenntag source area was observed indicating that sulfate reduction is an active degradation pathway. A plume centerline plot of sulfate versus distance from the Brenntag source area is presented in Figure 5-7. No discernable patterns of sulfate utilization in the vicinity of the Beall source area were observed.

5.2.5.8 Chloride

During biodegradation of COPCs dissolved in groundwater, chlorine in released into the aquifer and results in the accumulation of biogenic chloride. The result is chloride concentrations elevated in the plume relative to background. Chloride behaves conservatively in most groundwater systems and does not enter into reactions or readily adsorb to mineral surfaces.

Chloride concentrations were measured in 26 samples during the October 2002 sampling event. Site wide chloride concentrations ranged from 9 to 235 mg/L. The monitoring well upgradient from the Brenntag source area (MW001) exhibited a chloride concentration of 20 mg/L. Chloride concentrations were higher (235 mg/L) in monitoring well MW011; this well also exhibited the highest concentration of COPCs. This 10-fold increase in chloride concentration indicates the generation of biogenic chlorine in portions of the plume. A plume centerline plot of chloride versus distance from the Brenntag source area is presented in Figure 5-7. A map showing the distribution of chloride in Area A is presented in Figure 5-8.

The monitoring well upgradient from the Beall source area (MW202) exhibited chloride at a concentration of 9 mg/L. Chloride concentrations were in higher monitoring wells MW201 (44 mg/L) and MW204 (56 mg/L). This increase may also indicate previous generation of biogenic chlorine; however, reductive dechlorination processes do not appear to be currently active.

5.2.6 EPA Protocol Screening Evaluation

A screening process presented in the EPA Protocol for Preliminary Screening (Wiedemeier and others 1998)(EPA protocol) was performed at the two source area locations within the LSGPS to evaluate if reductive dechlorination is plausible. Groundwater analytical data from the Brenntag and Beall source areas were applied against protocol weighting criteria. There is no evidence for reductive dechlorination in other areas of the LSGPS outside of the Brenntag and Beall source areas.

5.2.6.1 Brenntag Source Area

Data from the Brenntag source area for the October 2002 sampling event were evaluated against EPA protocol for reductive dechlorination and are presented in Table 5-4. Total screening points for the Brenntag source area was 23, which, according to the EPA protocol, indicates that there is strong evidence for reductive dechlorination in this portion of the plume. In addition, this conclusion is supported by results from Mann-Kendall statistical testing and contaminant fate and transport modeling (see Section 5.4).

5.2.6.2 Beall Source Area

Data from the Beall source area for the October 2002 sampling event were evaluated against the EPA protocol for reductive dechlorination, and are presented in Table 5-5. Total screening points for the Beall source area was 9, which according to the EPA protocol indicates that there is limited evidence for reductive dechlorination in this portion of the plume. In addition, this conclusion is supported by results from Mann-Kendall statistical testing and contaminant fate and transport modeling (see Section 5.4).

5.3 EVALUATION OF HYDROGEOLOGIC DATA

Hydrogeologic data were evaluated to determine groundwater flow gradients and directions, to assess water transmitting properties of the aquifer, to estimate the velocity of groundwater flow, and to estimate the transport of COPCs in groundwater. The evaluations of these data are further described in the sections below.

5.3.1 Analysis of Groundwater Elevations

Sixty-two monitoring wells were installed at the LSGPS as part of the RI field effort. Groundwater elevation data from all monitoring wells were evaluated to establish spatial variations in hydraulic heads, and to interpret groundwater flow directions and gradients. Water-level maps (Figures 3-7 and 3-8) were prepared for the alluvial aquifer for the August 28/29 and October 28, 2002, measurement events. These maps indicate a general northwest flow of groundwater to the Yellowstone River. Water level elevations for the two events varied from a maximum of 3112.8 feet in monitoring well MW201 at Beall, to a minimum of 3068.0 feet in monitoring well MW124 adjacent to the Yellowstone River. The overall site gradient was 0.009 ft/ft for both events.

Local groundwater flow variations were noted in the vicinity of the Beall source area where a northerly groundwater flow component was observed. This variation may be due to the proximity of some monitoring wells to the alluvial aquifer/bedrock boundary. In addition, a westerly component of flow is noted in the vicinity of Cen-Dak Leasing and Planet Lockwood. Some water table mounding may be present in the vicinity of monitoring well MW302 because of discharge from the Planet Lockwood septic drain field. August 2002 groundwater elevations were used to estimate local hydraulic gradients in Area A and Area B. Based on these data, the hydraulic gradient in the portion of the plume downgradient of the Brenntag source area in Area A is estimated to be 0.0064; the hydraulic gradient in the portion of the plume downgradient of the Beall source area in Area B is estimated to be 0.0052. These gradients were incorporated into the groundwater models for the LSGPS Area A and Area B (Section 5.4).

5.3.2 Determination of Hydraulic Conductivity

Values of hydraulic conductivity for the alluvial and bedrock aquifers were initially estimated by performing a series of slug tests on eight alluvial monitoring wells and one bedrock monitoring well. A summary of RI slug test results is presented in Table 3-2. Values of hydraulic conductivity for the alluvial aquifer were subsequently refined based on calibration of contaminant transport models for Area A and Area B, and by incorporating results of pump testing conducted during previous investigations. Due to the coarse-grained nature of the sediments at the LSGPS, slug testing likely underestimates the actual hydraulic conductivity at this site. For the purpose of comparison, historic pump tests conducted at the LSGPS resulted in values of hydraulic conductivity more than 1,000 times larger than the values obtained by slug testing during the RI.

The initial hydraulic conductivity of the alluvial aquifer in Area A based on slug testing of monitoring wells MW110 and MW117 was 0.295 ft/day. The initial hydraulic conductivity value is considered to be very low and was subsequently refined during the process of model calibration; the final calibrated value for hydraulic conductivity for Area A is 70 ft/day. A value of 70 ft/day is more representative of coarse-grained sediments found in Area A (Todd 1981). This value is also more consistent with observed VOC migration in Area A.

The initial hydraulic conductivity of the alluvial aquifer in Area B based on slug testing of monitoring wells MW203 and MW213 was 0.275 ft/day. The initial hydraulic conductivity value was subsequently refined during the process of model calibration; the final calibrated value for hydraulic conductivity for Area B is 1.63 ft/day. A value of 1.63 ft/day is more representative of medium coarse-grained sediments found in Area B (Todd 1980). This value is also more consistent with observed VOC migration in Area B.

5.3.3 Summary of Aquifer Geotechnical Parameters

Geotechnical engineering parameter values for soils from borings at selected monitoring well locations were evaluated to provide estimates for model input parameters for the contaminant fate and transport model. Important engineering parameters include soil bulk density, porosity, and total organic carbon content (fraction of organic carbon) of aquifer soils. These parameters are required to estimate the retardation coefficient for each COPC in Area A and Area B. Sampling details are provided below.

Geotechnical engineering parameter values are based on analyses of soil samples collected by TtEMI with Shelby tubes from borings at locations SB203, MW110, MW118, MW127, MW202, MW203, MW210, MW300, and MW305 and geotechnical samples collected by SECOR from soil boring location PT004. These results were used to develop site-specific estimates for the retardation coefficient. Shelby tube retrieval from the sandy gravel portions of the alluvial aquifer proved problematic due to the large size (greater than 50 cm diameter) of some cobbles, therefore, test data may be biased toward finer portions of the alluvial aquifer where samples could be retrieved. Geotechnical parameters are summarized in Table 5-6.

5.3.4 Calculation of Groundwater Flow Velocity

The groundwater flow velocity, sometimes referred to as the seepage velocity, is an important component of contaminant transport investigations. It describes the advective process (bulk movement of water) of groundwater flow. The calculation used to determine the seepage velocity can be derived from a variation of Darcy’s Law (Fetter 1980):

$$v = (k \times i) / n_e$$

where:

v = seepage velocity, or pore water velocity (ft/day)
k = hydraulic conductivity (ft/day)
i = hydraulic gradient (ft/ft)
n_e = effective porosity (unitless)

Site-specific estimates for the seepage velocity were developed for Area A and Area B. The estimates for seepage velocity are based on values for engineering parameters discussed in Section 5.3.3, and hydraulic conductivity values described in Section 5.3.2.

For the portion of the plume downgradient of the Brenntag source area in Area A, the seepage velocity is estimated to be about 1.5 ft/day or about 558 feet per year (ft/year). This estimate is based on an average hydraulic conductivity of 70 ft/day, a hydraulic gradient of 0.0066 ft/ft, and an effective porosity of 0.27. For the portion of the plume downgradient of the Beall source area in Area B, the seepage velocity is estimated to be about 0.0356 ft/day or about 13.0 ft/year. This estimate is based on an average hydraulic conductivity of 1.63 ft/day, a hydraulic gradient of 0.0059 ft/ft, and an effective porosity of 0.27. Hydraulic gradients were calculated for flowpaths along the plume centerlines using July 2002 data. All parameter values are within the range of commonly accepted values for similar geologic settings and conditions, and are consistent with observed VOC migration in Area A and Area B.

5.3.5 Calculation of Contaminant Flow Velocities

It is not unusual for contaminant flow velocities for some COPCs to be less than the groundwater seepage velocity due to adsorptive processes within the aquifer. In these cases, the retardation factor describes the relative speed of contaminants in groundwater, and is defined to be the velocity of groundwater divided by the velocity of contaminants. The retardation factor can also be calculated using the following formula:

$$R = 1 + ([Koc][foc][Bd])/n$$

where:

R = Retardation factor
K_{oc} = Organic carbon partition coefficient
f_{oc} = Fraction of organic carbon in the aquifer matrix
Bd = Bulk density of the aquifer matrix
n = Porosity

The K_{oc} is chemical-specific, and each COPC is characterized by a specific value for K_{oc}. The f_{oc} is equal to the total organic carbon content of the aquifer divided by 100. The f_{oc}, bulk density, and porosity of the aquifer are based on values for engineering parameters discussed in Section 5.3.3. For this investigation, retardation factors for COPCs in groundwater were estimated for both Area A and Area B using the above formula. Values for the retardation factor used in contaminant transport modeling described in Section 5.4 are summarized below. Retardation factors for Area A and Area B are based on the limited field data that were available at the time groundwater modeling was conducted. Due to limited data, current estimates for both Area A and Area B are identical.

Retardation Factors for COPCs in Groundwater				
Area	**PCE**	**TCE**	**Cis-1,2-DCE**	**Vinyl chloride**
A	4.18	1.97	1.93	1.22
B	4.18	1.97	1.93	1.22

These retardation factors suggest that, TCE would transport approximately twice as far as PCE during the same time frame. Evidence of the relative rate of transport for PCE and TCE is observed in Area C by visual comparison of the maximum extent of migration for these two COPCs (Figures 4-11 through 4-14).

5.4 FATE AND TRANSPORT MODELING

Groundwater contaminant fate and transport modeling was conducted for the LSGPS to evaluate the dimensions of the groundwater plume over time, and to assess impacts to the Yellowstone River. Section 5.4.1 of this report identifies the objectives of modeling. Section 5.4.2 provides the rationale for selection of model codes. Section 5.4.3 discusses the approach to modeling. Section 5.4.4 presents the site conceptual fate and transport model. Section 5.4.5 provides a summary of model input parameters and assumptions. Section 5.4.6 presents the results of groundwater contaminant fate and transport modeling. Section 5.4.7 discusses important model limitations. Model input and output screens and model runs, as well as other supporting model documentation are provided in Appendix L-1 through Appendix L-6. Selected model equations are provided in Appendix L-7.

5.4.1 Objectives of Modeling

The objectives of modeling groundwater flow and contaminant fate and transport at the LSGPS were (1) to evaluate plume dimensions over time and to determine if the portions of the plume downgradient of the Brenntag and Beall source areas in Area A and Area B are currently increasing in size (length and width), stable, or decreasing in size, and (2) to estimate the impact of contaminated groundwater at the LSGPS on surface water quality in the Yellowstone River. To meet these objectives, a variety of computational tools were used including (1) statistical tools to analyze trends in groundwater quality data over time, (2) commonly used and available groundwater flow and contaminant transport model codes to simulate the migration and biodegradation of COPCs, and (3) a spreadsheet style hydrogeological groundwater-surface water mixing model (Appendix L-7). Modeling was conducted for four COPCs including PCE, TCE, cis-1,2-DCE, and VC. The rationale for selection of specific modeling software and computer codes is discussed below.

5.4.2 Rationale for Selection of Model Codes

Selected model software included the State of Wisconsin Mann-Kendall statistical software, BIOCHLOR version 2.2, and Microsoft Excel 6.0. Mann-Kendall statistical testing software (Form 4400-215) developed by the State of Wisconsin, Department of Natural Resources (WDNR), Remediation and Redevelopment Program (WDNR 2001) was used to evaluate water quality trends with time at selected monitoring well locations. The EPA-approved BIOCHLOR version 2.2 (Aziz and Others 2002) was used to simulate the migration and biodegradation of portions of the plume in Area A and Area B. Lastly, a groundwater-surface water mixing model was developed in a spreadsheet format using Microsoft Excel version 6.0 to evaluate the impact of contaminated groundwater at the LSGPS on surface water quality in the Yellowstone River.

Mann-Kendall Statistical Test

The Mann-Kendall test (WDNR 2001) is a nonparametric statistical test that can be used to evaluate trends in concentration over time in monitoring wells at the LSGPS: trends can be increasing, stable, or decreasing. Stable is defined to mean neither increasing nor decreasing. The test involves computing a Mann-Kendall statistic (S) that is the difference between the number of pair-wise slopes that are positive and the number that are negative. If S is a large, positive value, there is evidence of an increasing trend in the data. If S is a large, negative value, there is evidence of a decreasing trend in the data. The Mann-Kendall test must show decreasing COPC concentrations at an appropriate confidence level to demonstrate that monitored natural attenuation (or active remediation) is actively occurring. Confidence levels of 80 and 90 percent are routinely used by state regulatory agencies (WDNR 2001) in monitored natural attenuation studies. For the analyses prepared for the LSGPS, test statistics were calculated and presented for both the 80 and 90 percent confidence levels. These statistics can be used to support conclusions regarding trends in plume shape and size over time.

BIOCHLOR Version 2.2

The BIOCHLOR model code uses an analytical transport module with sequential first-order decay for simulating in situ biotransformation of VOCs. BIOCHLOR is an EPA-approved screening model that is routinely used by a variety of regulatory agencies and is supported by the International Groundwater Modeling Center. The model code BIOCHLOR version 2.2 was selected because it is a fully functional analytical solute transport model with sequential first order decay for simulating in situ biodegradation (EPA 2002a; Aziz 2002). BIOCHLOR is primarily designed for simulating the sequential reductive dechlorination of chlorinated ethenes, which is modeled as a sequential first order decay process.

Microsoft Excel 6.0 Mixing Model

A groundwater-surface water mixing model was constructed in Microsoft Excel 6.0 to evaluate the potential impact of contaminated groundwater on surface water quality in the Yellowstone River. Model calculations are based on a modified Darcy groundwater flux analysis, as presented by Todd (1981) (see Appendix L-7). The mixing model was used to estimate the incremental increase in the concentration of COPCs in surface water in the Yellowstone River at low flow using July and August 2002 groundwater quality conditions from seven sentinel monitoring wells. Microsoft Excel 6.0 was considered the most appropriate analytical tool to conduct a site-specific hydrogeologic mass balance because this software easily lends itself to the accounting nature of this particular hydrogeologic assessment.

5.4.3 Modeling Approach

The approach to groundwater modeling for the LSGPS was developed specifically to determine if the dimensions of COPC plumes are currently increasing in size (length or width of plume), stable, or decreasing in size. In addition, the impact of contaminated groundwater on surface water quality in the Yellowstone River was evaluated. Mann-Kendall statistical tests for trends and the analytical fate and transport model BIOCHLOR were used to support conclusions about changes in plume dimensions over time. A groundwater-surface water mixing model was used to perform an impact analysis of potential changes to surface water quality in the Yellowstone River. A detailed approach for each model is provided in the following sections.

5.4.3.1 Mann-Kendall Statistical Test for Trends

Time series data for 13 total wells, including 8 monitoring wells (MW002, MW005, MW007, MW008, MW009, MW010, MW011, and MW012) and five residential wells (RW001, RW002, RW005, RW006, and RW007), were evaluated for trends using computerized Mann-Kendall statistical test software (WDNR 2001). Two monitoring wells (MW011 and MW012), were selected because they are representative of groundwater quality at locations near suspected source areas; two monitoring wells (MW002 and MW010) are located along the flanks of the Brenntag source area in Area A; two monitoring wells (MW007 and MW009) are located within the core portions of the plume downgradient of the Brenntag source area in Area A; and two monitoring wells (MW005 and MW008) are located near the leading edge of the portion of the plume downgradient of the Brenntag source area in Area A. Time series data were available for only one monitoring well near the Beall source area (MW012). The residential wells selected exhibited detections of VOCs during previous rounds of sampling, and statistics for these wells were evaluated in the risk assessment (see Section 6.0). Trends in COPC concentrations were evaluated for each monitoring well using data for the entire period of record (2000 through 2002).

For this trend analysis, an increasing trend in concentration of COPCs at a plume boundary would be evidence that the plume at this location may be increasing in size. Decreasing concentrations at a plume boundary would be evidence that the plume may be decreasing in size at this location. Concentrations of COPCs at a plume boundary that neither increase nor decrease over time would suggest stable plume conditions (no dimensional change in plume). In addition, statistical testing for trends in groundwater quality from monitoring wells near source areas was important to establish or support the selection of some fate and transport model source term parameters.

Statistical testing for trends in groundwater quality from monitoring wells located within the core of the plume was important to support whether biodegradation processes such as reductive dechlorination are occurring. For example, a decreasing trend in parent compounds (PCE, TCE) coupled with an increasing trend in daughter products (cis-1,2-DCE and vinyl chloride) may support a conclusion that biodegradation processes are occurring at this location.

5.4.3.2 BIOCHLOR Model

The approach to modeling contaminant fate and transport at the LSGPS was simplified to the greatest degree possible without compromising the objectives of modeling. In doing so, simpler analytical models rather than one large and complex numerical model were used to simulate the migration of COPCs in groundwater. Use of analytical models rather than numerical models resulted in cost savings and rapid development of modeling scenarios. However, analytical model results can only be applied to meet the specific modeling objectives outlined in Section 5.4.1 (Anderson and Woessner 1992). The limitations of using a simplified analytical modeling approach at the LSGPS are more fully discussed in Section 5.4.7.

With this approach in mind, individual analytical models were developed for the portions of the COPC plume in Area A and Area B using BIOCHLOR version 2.2. Each model was independently constructed and calibrated using local site-specific lithologic, hydrogeologic, and chemical data specific to each area. To calibrate each model, important model parameters were systematically varied within acceptable ranges until the simulated concentrations of COPCs in groundwater at each area matched field COPC concentrations for the July and August 2002 RI groundwater sampling event. Each BIOCHLOR model was considered calibrated when a statistical regression analysis of model simulated COPC concentrations versus field COPC concentrations yielded a correlation coefficient greater than 0.80. Calibrated BIOCHLOR models presented in this RI provide one plausible interpretation of the existing field data; however, model solutions are not unique and other interpretations are possible.

The calibrated models for portions of the plume in Area A and Area B were subsequently used in a transient mode to simulate future COPC plume dimensions over time. The boundary of the model-simulated plume was defined by the 1.0 µg/L isocontour line; this isocontour interval is the smallest allowable interval that could be assessed using BIOCHLOR version 2.2. The 1.0 µg/L isocontour was then used to determine the maximum length and maximum width of the plume at various time intervals starting at year 1 of the model simulation. Model simulations were conducted for a total simulation time equal to 30 years.

The dimensions of the COPC plume were defined to be “increasing” if the simulated maximum length or maximum width of the plume continued to increase over time. The COPC plume was defined to be “decreasing” if the simulated maximum length and maximum width decreased over time. The plume was defined to be “stable” if the simulated maximum plume length and width did not change over a five-year simulation period. In addition, BIOCHLOR modeling results were compared to Mann-Kendall statistical trends; both as an internal check, and to ensure that the two independent analyses provided consistent results and conclusions.

5.4.3.3 Mixing Model

A groundwater-surface water mixing model was constructed using Microsoft Excel version 6.0 to evaluate the potential impact of COPCs in groundwater at the LSGPS on surface water quality in the Yellowstone River. Specifically, the mixing model was used to estimate the incremental increase in the concentration of COPCs in surface water in the Yellowstone River at low flow. Flow statistics that included the lowest mean monthly flow were generated using flow data from USGS surface water gauging station number 06214500 near Billings, Montana. Flow statistics used data for the entire 76 year period of record, from 1926 through 2002.

The groundwater-surface water mixing model incorporated July and August 2002 groundwater quality conditions from seven monitoring wells that were in adjacent to the Yellowstone River (MW305, MW118, MW120, MW124, MW125, MW126, and MW127). These groundwater quality data were used to establish the mass flux of COPCs across the entire LSGPS perpendicular to the direction of groundwater flow. Using this approach, the model instantaneously mixed the mass flux of COPCs from the LSGPS with surface water in the Yellowstone River. The mixing model assumed steady-state groundwater conditions and low-flow conditions in the Yellowstone River. The model is based on hydrogeologic water-balance relationships (Todd 1981). Details and equations for the mixing model are provided in Appendix L-7.

5.4.4 Site Conceptual Fate and Transport Model

A conceptual model describes the components of the groundwater flow system and is developed from regional, local, and site-specific data. Flow system components include parameters such as groundwater flow direction and gradient, aquifer thickness, water transmitting properties, and information related to source strength and activity. Development of a conceptual model is always necessary before constructing a computerized groundwater flow or contaminant transport model (EPA 1992c).

The conceptual model for the LSGPS was formulated to organize existing field data and literature-derived parameter values so that the groundwater flow system could be analyzed more readily. For the purpose of satisfying fate and transport modeling objectives, the conceptual model was simplified as much as possible; however, enough complexity was retained to adequately simulate groundwater system behavior and contaminant migration for the intended purposes of modeling (Anderson and Woessner 1992).

The conceptual model for the LSGPS includes two potential sources: one in Area A (Brenntag source area), the other in Area B (Beall source area). Facility operational history suggests the Beall source area may have been active as early as 1978 (Pentacore Resources, LLC 2001); a Detrex vapor degreaser was used to clean crude oil and asphalt tank trailers prior to repair. The drain field associated with the degreasing operation operated until 1991.

Detailed operational history from Brenntag could not be found, but it has been documented that Brenntag purchased the property in 1972. Based on this information, an assumption was made for the plume modeling conducted in Area A that the Brenntag source area may have been active beginning in 1972.

The assumptions above are for modeling purposes only; in fact, source activity may have actually began after 1972 for the plume in Area A; and after 1978, for the plume in Area B. Additional model assumptions incorporated in the site conceptual model are discussed in Section 5.4.5.

5.4.5 Assumptions and Model Input Parameters

Development of analytical transport models for the LSGPS involved incorporation of numerous simplifying assumptions related to lithology, water transmitting properties, groundwater flow rates and directions, chemical characteristics of COPCs, biodegradation rates; and the interpreted distribution of COPCs in groundwater. Important assumptions and BIOCHLOR model input parameters for Area A and Area B are summarized in Table 5-7.

Another set of assumptions was also used in the construction of the groundwater-surface water mixing model for the LSGPS (see Appendix L-7). These assumptions include:

- Flow statistics were generated using flow data from USGS station number 06214500 located on the Yellowstone River near Billings, Montana. The lowest mean monthly flow for the 76-year period of record was calculated by the USGS (USGS 2003), and equals 2,491 cfs. This flow rate was used to calculate the volume of water available for dilution on a daily basis in the Yellowstone River.
- Distance-weighted average groundwater concentrations from wells adjacent to the Yellowstone River in Area A and Area C from the July and August 2002 sampling event were assumed to be representative of the overall groundwater quality available for mixing.
- Less than detection limit values were set equal to the detection limit value for the purpose of all statistical analyses.
- Groundwater flux enters the river along the entire length of the LSGPS adjacent to the Yellowstone River.
- Volatilization of COPCs in surface water was not considered.

The mixing model assumptions listed above are conservative in that overall model results likely overestimate the overall impact of COPCs in groundwater on surface water resources in the Yellowstone River.

5.4.6 Results of Modeling

Mann-Kendall statistical tests for trends and the analytical fate and transport model BIOCHLOR were used to support conclusions about changes in dimensions of the COPC plume over time. A groundwater-surface water mixing model was used to perform an impact analysis of potential changes to surface water quality in the Yellowstone River. Detailed results for each model application are provided in the following sections.

5.4.6.1 Mann-Kendall Statistical Test for Trends

Time series data for 13 total wells including 8 monitoring wells (MW002, MW005, MW007, MW008, MW009, MW010, MW011, and MW012) and five residential wells (RW001, RW002, RW005, RW006, and RW007) were evaluated for trends using computerized Mann-Kendall statistical test software (WDNR 2001). Table 5-8 presents Mann-Kendall test results for selected groundwater monitoring wells in Area A for both the 80 and 90 percent confidence levels. Rationale for selection of these wells is provided in Section 5.4.3.1. Table 5-9 presents Mann-Kendall statistics for the selected monitoring well in Area B.

Table 5-10 summarizes Mann-Kendall statistics for selected residential monitoring wells RW001, RW002, RW005, RW006, and RW007 in the Lomond Lane area. These wells historically exhibited VOCs. Wells were statistically evaluated for trends in concentration over time as an aid in conducting the human health and ecological risk assessments for the LSGPS (see Section 6.0). Appendix L-1 provides detailed Mann-Kendall test results and other comparative statistics for groundwater quality data collected from each monitoring well. The results of Mann-Kendall statistical tests are described below.

Area A

Table 5-8 indicates that concentrations of PCE in monitoring well MW011 near the Brenntag source area appears to be increasing at the 80 percent confidence level. Also, there is no statistically valid decreasing trend in the concentrations of TCE and cis-1,2-DCE at this location. The concentration of VC at MW011 is decreasing at the 80 percent confidence level. Based on this analysis, the BIOCHLOR model for this portion of the plume in Area A assumes there is a continuing contribution of source in the subsurface.

Within the core portion of the plume downgradient of the Brenntag source area in Area A (MW007), there is a downward trend in the concentration of PCE and an upward trend in the concentration of degradation byproducts such as TCE and DCE at the 80 percent confidence level. There is no upward or downward trend in the concentration of VC. Comparable trends (decreasing concentrations of parent compounds coupled with increasing concentrations of daughter products) are also observed for some COPCs in monitoring well MW009. Trends in groundwater quality data from monitoring wells MW007 and MW009 suggest that biodegradation may be occurring within the plume near these wells (see Section 5.2.6).

The trend in concentrations of PCE, TCE, and VC near the leading edge of the portion of the plume downgradient of the Brenntag source area at monitoring well MW008 appears to be decreasing at the 80 percent confidence level. There is no upward or downward trend in the concentration of cis-1,2-DCE. The concentration of VC in monitoring well MW005 is decreasing. These data suggest that the downgradient edge of the VOC plume in this area may not be changing, or is possibly decreasing for some COPCs, at some locations.

Area B

Table 5-9 indicates that low-level concentrations of PCE near the Beall source area at monitoring well location MW012 appear to be decreasing at the 80 percent confidence level. The concentrations of TCE and cis-1,2-DCE at this location are not decreasing with time. Trend analysis is not possible for other portions of the plume in Area B due to limited available data. The BIOCHLOR model for the plume in Area B incorporates the trends at monitoring well location MW012 in development of the model source term parameters.

Residential Wells in the Lomond Lane Area

Table 5-10 provides Mann-Kendall statistics that suggest the concentrations of some COPCs in residential monitoring wells in the Lomond Lane area appear to be decreasing with time. However, exceptions include an increasing trend of TCE and cis-1,2-DCE in residential well RW005, and an increasing trend of TCE and cis-1,2-DCE in residential well RW006. It is not possible to draw a conclusion about plume dimensions from residential water supply wells; however statistics for residential monitoring wells were reviewed in preparing the human health and ecological risk assessments (Section 6.0).

5.4.6.2 Mixing Model

Model parameter input and simulated model output for the groundwater- surface water mixing model are summarized in Table 5-11 for low flow conditions. Model input and output screens are provided in Appendix L-2. The model-calculated dilution-attentuation factor for the LSGPS is equal to 4,515; concentrations of VOCs in groundwater are diluted by a factor of 4,515 by surface water flows in the Yellowstone River at low flow conditions. Model results also indicate that the incremental impact of COPCs in groundwater at the LSGPS on surface water quality in the Yellowstone River would not be detectable with current laboratory equipment at the lowest mean monthly flow for the period of record. For example, the simulated incremental increase of COPCs in surface water are equal to 0.0032 µg/L for

PCE, 0.00074 µg/L for TCE, 0.0016 µg/L for DCE, and 0.00018 µg/L for vinyl chloride. Mixing model results are used in the ecological risk assessment (Section 7.0).

5.4.6.3 BIOCHLOR Model

BIOCHLOR models were constructed and calibrated to summer 2002 groundwater quality conditions for portions of the plume in Area A and Area B. The BIOCHLOR model for the portion of the plume in Area A was calibrated using data from monitoring wells MW011, MW007, MW009, MW122, and MW126. The BIOCHLOR model for the portion of the plume in Area B was calibrated using data from monitoring wells MW201, MW012, MW204, and MW210. For the purpose of model calibration, these monitoring wells were assumed to be closest to the plume centerlines downgradient of the Brenntag and Beall source areas; in reality, they may not be located exactly on the plume centerline. The models were considered calibrated when a statistical regression analysis of model simulated concentrations versus field concentrations yielded a correlation coefficient greater than 0.80 for each model. Both BIOCHLOR models were successfully calibrated and calibrated model input parameters are provided in Table 5-7. Calibrated model parameter values were reviewed to ensure all values were within the range of commonly acceptable values published in the literature for similar geology and COPCs. In addition, parameter values are consistent with observed plume migration. Model input and output screens for the calibrated model simulations are provided in Appendix L-3 for the portion of the plume in Area A, and in Appendix L-4 for the portion of the plume in Area B.

Calibrated models were then run in a predictive mode; transient model simulations were conducted to evaluate groundwater conditions over time. During the transient simulations, the simulated distributions of COPCs in groundwater for Area A and Area B were used to evaluate the maximum plume length and width over time. For each transient model run, the simulated maximum length and width of each portion of the COPC plume was defined by the 1 µg/L isocontour interval, which was the minimum concentration reported by the BIOCHLOR model. Model input and output screens for the transient model simulations are provided in Appendix L-5 for the portion of the plume in Area A, and in Appendix L-6 for the portion of the plume in Area B.

BIOCHLOR model results are provided in Table 5-12 and Table 5-13 for Area A and Area B, respectively. Table 5-12 summarizes 30 years of transient model simulations for the portion of the plume in Area A downgradient of the Brenntag source area; a model-simulated time equal to 0 years corresponds to calendar year 1972. This is the assumed date that a simulated model source for Area A began; in reality, source activity in Area A may have begun at a later date. Model year 30 corresponds to calendar year 2002. Table 5-12 provides data regarding the maximum simulated length and width of the portion of the plume downgradient of the Brenntag source area as a function of time. Model results for the portion of the plume in Area A suggest that the downgradient edge of VOC plumes may have reached the Yellowstone River as early as 5 years after a source began (calendar year 1977). The VOC plumes appear to have reached a maximum length and width 10 years after a source began (calendar year 1982). Model results indicate that these maximum conditions continue throughout the 30-year transient simulation (calendar year 2002). BIOCHLOR model results support a conclusion that the maximum length and maximum width of the portion of the COPC plume downgradient of the Brenntag source area is not currently changing over time. This conclusion is corroborated by Mann-Kendall statistics that show stable trends in concentrations of COPCs in monitoring wells (1) nearest to the source area, and (2) at the lateral edges of this portion of the plume.

Table 5-13 summarizes 30 years of transient model simulations for the portion of the plume in Area B downgradient of the Beall source area. For this summary, a model-simulated time equal to 0 years corresponds to calendar year 1978, the assumed date that a simulated model source began in Area B; in reality, source activity in Area B may have begun at a later date. Model year 24 corresponds to calendar year 2002. Model year 30 corresponds to calendar year 2008. Table 5-13 provides data regarding the maximum simulated length and width of the TCE and DCE plume downgradient of the Beall source area. Model results suggest that the downgradient edge of this portion of the VOC plume had not yet stabilized 24 years after the source began (calendar year 2002). In addition, plume dimensions do not appear to have stabilized by the end of the 30-year model simulation (calendar year 2008). The transient model suggests this portion of the VOC plume currently appears to be increasing in length by about 10 ft/year. Therefore, BIOCHLOR model results support a conclusion that the portion of the COPC plume downgradient of the Beall source area is slowly increasing in size. This conclusion is corroborated by Mann-Kendall statistics for monitoring well MW012 that show concentrations of COPCs in monitoring wells nearest to the source area are not decreasing over time; there appears to be a continuing contribution of source in the subsurface.

5.4.7 BIOCHLOR Model Limitations

For this investigation, model parameters were selected based on field data where possible, or were based on published values for similar site conditions. Because the scope of modeling objectives was limited, no attempt was made to construct or calibrate a single large site-wide numerical groundwater flow or contaminant fate and transport model. Several screening-level models were used to evaluate the dimensions of COPC plume over time. Therefore, the type of modeling application described in this report is consistent with the characteristics of a “generic model” (Anderson and Woessner 1992). According to Anderson and Woessner (1992), generic models are used to analyze flow in simplified hydrogeological systems and may be useful for the purpose of screening.

The BIOCHLOR model for the portion of the plume in Area A was calibrated using July 2002 groundwater quality data from monitoring wells MW011, MW007, MW009, MW122, and MW126; the model for the portion of the plume in Area B was calibrated using data from monitoring wells MW201, MW012, MW204, and MW210. For the purpose of model calibration these monitoring wells were assumed to be located along the plume centerlines downgradient of the Brenntag and Beall source areas. The uncertainty as the exact concentrations of COPCs along the plume centerline affects the degree of model calibration. Therefore, model simulated results could be improved, assuming the model calibration could be improved by incorporating additional monitoring well data closer to the plume centerline.

Groundwater model development and calibration for Area A and for Area B was conducted using groundwater quality data from the July 2002 sampling event. Groundwater quality data from the October 2002 sampling event was not available at the time the modeling was conducted. Comparison of July 2002 and October 2002 groundwater quality data indicate there are variations in the concentrations of VOCs during these two sampling events; the downgradient edge of the portion of the plume in Area B in October 2002 extends further than in July 2002. Therefore, the existing BIOCHLOR model could be updated to reflect the most recent groundwater quality conditions.

Each model was independently constructed and calibrated using local site-specific lithologic, hydrogeologic, and chemical data specific to each area. During model construction and calibration, important model parameters (such as hydraulic conductivity and degradation rates for COPCs) were systematically varied within a range of acceptable values until the simulated concentrations of COPCs in groundwater monitoring wells at each area matched field COPC concentrations for the July 2002 RI groundwater sampling event. The BIOCHLOR models for the portions of the plume in Area A and Area B were considered calibrated when a statistical regression analysis of model simulated concentrations versus field concentrations in each area yielded a correlation coefficient greater than 0.80. Calibrated BIOCHLOR models presented in this RI provide one interpretation of the existing field data; however, model solutions are non-unique and other interpretations are possible.

The development of a mixing model using Excel 6.0 involved incorporating numerous assumptions and simplifications that likely have the effect of overestimating concentrations of COPCs in surface water for the Yellowstone River downgradient of the LSGPS area. For example, in calculating the distance weighted average concentration of COPCs in groundwater, all less than detection limit values were set equal to the detection limit for statistical purposes. Secondly, the lowest mean monthly flow in the Yellowstone River was used in the calculations. Therefore, the groundwater-surface water mixing model is considered to be conservative and protective from the standpoint of conducting both human health and ecological risk assessments.

Regardless of the lack of constructing and calibrating a large numerical site-wide model for the LSGPS, the results from screening models are adequate to satisfy the primary objectives of this report. Specifically, model results provide a reasonable estimate regarding whether COPC plumes at the LSGPS are increasing in size, decreasing in size, or are stable.

5.5 SUMMARY OF CONTAMINANT FATE AND TRANSPORT

The fate and transport of COPCs in Area A and Area B was evaluated through the following:

- Analysis of plume behavior through a geochemical characterization and review of patterns of electron donors and acceptors
- Hydrogeological data analysis
- Statistical analysis of historical analytical data collected from pre-RI monitoring wells and residential well samples
- Modeling of the transport and biodegradation of COPCs with BIOCHLOR and of the flux of contaminants into the Yellowstone River with a mixing model

5.5.1 Geochemical Characterization

Area A

The VOC plume in the alluvial aquifer extends from the upgradient Area A boundary and the Brenntag source area to the Yellowstone River. The core of the portion of the plume downgradient of the Brenntag source area (total VOC concentration greater than 100 µg/L) is relatively narrow and is generally less than 300 feet wide. This portion of the plume exhibits some lateral dispersion and is approximately 1,300 feet wide where it enters the river approximately 2,000 feet downgradient of the Brenntag source area. The bedrock aquifer downgradient from the Brenntag source area does not appear to be impacted by VOCs.

The portion of the plume downgradient of the Brenntag source area in Area A appears to be exhibiting Type I plume behavior where man-made carbon from released petroleum and possibly other compounds is the electron donor supply. Strong lines of geochemical evidence are present to support active reductive dechlorination including: (1) anaerobic conditions (dissolved oxygen less than 1.0 mg/L and strongly negative oxidation-reduction potential), (2) elevated methane, (3) depleted nitrate, (4) the presence of daughter products (DCE, VC, ethene, and ethane), and (5) hydrogen concentrations greater than 1.0 nM.

Area B

The highest concentrations of COPCs and the bulk of the COPC mass in the portion of the groundwater plume in Area B were detected east of Interstate Highway 90. This is the most significant portion of the plume in Area B. Low-level, site-wide contamination extends approximately 4,600 feet northwest to the Yellowstone River. TCE concentrations exceeding regulatory limits extend approximately 2,000 feet downgradient to monitoring wells along the BNSF Railroad corridor. With the exception of monitoring well MW210, COPC concentrations outside of the Beall source area do not exceed a total VOC concentration of 100 µg/L.

Type III plume behavior may be occurring within the Beall source area and downgradient to Lockwood Road where dissolved oxygen concentrations exceed 1.0 mg/L. Limited geochemical evidence of degradation is apparent in the Beall source area although the daughter product DCE is present and biogenic chloride appears to be present at concentrations 3 to 5 times greater than background. Evidence suggests that degradation may historically have taken place until available carbon sources were exhausted.

Area C

Detailed statistical modeling was not performed for the portion of the groundwater plume in Area C. Low-level concentrations of TCE and DCE are detected in Area C, northwest of the Interstate Highway 90 corridor.

5.5.2 Hydrogeological Data Analysis

The overall site gradient was 0.009 ft/ft with local hydraulic gradients in Area A estimated to be 0.0064; the hydraulic gradient in Area B is estimated to be 0.0052.

The hydraulic conductivity of the alluvial aquifer in Area A was estimated to be 0.295 ft/day from slug tests and 70 ft/day from the model calibration. The hydraulic conductivity of the alluvial aquifer in Area B based on slug testing was estimated to be 0.275 ft/day and 1.63 ft/day from model calibration. Due to the coarse-grained nature of sediments, slug test results likely underestimate the actual hydraulic conductivity; the calibrated model values for hydraulic conductivity are considered to be more appropriate for the LSGPS.

Based on the model calibrated hydraulic conductivity data, the site-specific seepage velocity in Area A was estimated to be about 1.5 ft/day or about 558 ft/year. In Area B, the seepage velocity was estimated to be about 0.0356 ft/day or about 13.0 ft/year. Larger seepage velocities may be present in Area B with increasing distance downgradient form the Beall source area.

5.5.3 Statistical Analysis of Groundwater Concentration Trends

Evaluation of monitoring well data within the Brenntag source area (MW011) indicates the concentrations of TCE and DCE are not decreasing over time at this location. In downgradient monitoring wells MW007 and MW009 (Area A), decreasing concentration trends of PCE and TCE coupled with increasing concentrations of daughter products, DCE and VC, were observed. These observations suggest biodegradation is active in the portion of the plume downgradient of the Brenntag source area. Decreasing trends in concentrations over time were present in one downgradient monitoring well (MW008) for both parent compounds and daughter products.

Monitoring well MW012 in the Beall source area exhibited no decreasing trend in the concentrations of TCE or cis-1,2-DCE over time. Concentrations of TCE and cis-1,2-DCE appear to be stable. Long-term monitoring well data were not available further downgradient; therefore, no statistical analyses were conducted at other locations in Area B.

5.5.4 Transport and Biodegradation Modeling

For the portion of the plume downgradient of the Brenntag source area in Area A, BIOCHLOR model results (assuming source activity since 1972) suggest that the downgradient edge of the VOC plume may have reached the Yellowstone River as early as model year 5 (calendar year 1977). This portion of the plume appears to have reached a maximum length and width by model year 10 (calendar year 1982), and model results suggest no significant changes in the dimensions of the plume have taken place after that time.

For Area B, BIOCHLOR model results support a conclusion that the portion of the plume downgradient of the Beall source area is slowly increasing in size.

The groundwater-surface water mixing model indicates that the incremental impact of COPCs in groundwater on surface water quality in the Yellowstone River at low flow would not be detectable with current laboratory equipment.

6.0 HUMAN HEALTH RISK ASSESSMENT

A complete baseline human health risk assessment (HHRA) (Section 6.0) and ecological risk assessment (ERA) (see Section 7.0) are being conducted under the RI. The *Risk Assessment Guidance for Superfund* (RAGS) Parts A through D (and draft RAGS Part E) were implemented for this HHRA, as detailed below.

6.1 PURPOSE AND SCOPE

The purpose of the HHRA is to provide risk managers with a basis for evaluating whether action is warranted to mitigate potential health effects from exposure to contaminants in soil, sediment, surface water, or groundwater at the site. Media samples were collected at appropriate locations across LSGPS as outlined in the RIWP (TtEMI 2002a) for use in characterizing baseline human health risks. The RAGS approach (EPA 1989; 2001c, d) was employed at LSGPS, including application of the basic four-step process:

- **Step 1 – Data Evaluation and Selection of COPC.** The data evaluation and selection of COPCs included an evaluation of spatial distribution of chemicals and identification of applicable data to carry forward in the remaining steps of the HHRA process. The spatial distribution of chemical concentrations was reviewed to identify patterns, including potential localization of chemicals and appropriate grouping of data subsets across the 580-acre LSGPS site. This information was incorporated into the selection of COPCs as well as the exposure assessment (Step 2). In addition, risk-based concentrations were used to screen maximum concentrations detected at the site to complete the COPC selection process.

- **Step 2 – Exposure Assessment.** The exposure assessment characterized the nature and magnitude of potential exposures to COPCs at the site. Specifically, it included preparing a description of the current and future exposure setting and land use (see Sections 1.3 and 3.5 of this RI report), identifying potential exposure scenarios and complete exposure pathways, identifying hypothetical exposure points, estimating exposure point concentrations, and estimating hypothetical chemical intakes.
- **Step 3 – Toxicity Assessment.** The toxicity assessment consisted of compiling toxicity values (slope factors, unit risk factors, reference doses, and reference concentrations) for COPCs and brief summaries of their bases.
- **Step 4 – Risk Characterization.** Site-related health risks and hazards were characterized using potential excess lifetime cancer risk estimates and hazard indices (HI) for adverse, non-cancer health effects associated with potential upper bound exposure to COPCs at the site. Cancer risks associated with exposure to COPCs classified as carcinogens were characterized as an estimate of the probability (excess risk) that an individual will develop cancer over a lifetime as a direct result of exposure to potential carcinogens (EPA 1989). This estimated excess risk was expressed as a unitless probability. For example, a cancer risk of 1×10^{-6} (equivalent to 1E-06 in scientific notation) indicates that an individual has a 1-in-1 million probability of developing cancer during a 70-year lifetime as a result of the assumed exposure conditions.

Uncertainties associated with the overall risk assessment were also identified. Further descriptions of each of the four steps are presented in Sections 6.2 through 6.5.

6.2 DATA EVALUATION AND IDENTIFICATION OF COPCS

Information on the evaluation of analytical data for use in the HHRA is summarized in this section. The data were validated before they were used in the HHRA (see Appendix H). Data used for the HHRA includes data collected during the RI field effort and also any validated analytical data deemed usable for risk assessment following EPA guidelines (EPA 1990). In particular, the most recent quarterly monitoring data for indoor air was determined to be appropriate for use in the baseline risk assessment for the LSGPS. The most recent quarterly surface water monitoring data and RI/quarterly monitoring data for groundwater were also determined to be appropriate for the HHRA.

As noted in Appendix H, with the exception of the rejected or qualified “R” data, the analytical data met EPA precision, accuracy, representativeness, comparability, and completeness criteria required for data usability in a risk assessment. The data provide the basis for selecting COPCs and estimating exposure point concentrations for the risk assessment.

6.2.1 Data Reduction

The approaches used for data reduction are consistent with the guidelines presented in RAGS Part A (EPA 1989). Analytical summary statistics and data results used to perform the risk assessments are presented in RAGS Part D format following EPA guidance (EPA 2001d). The following guidelines for data reduction were used to produce the data summaries for each medium.

- The COPCs for the HHRA include only VOCs at the LSGPS site. This is reflected in the data quality objectives for the site (TtEMI 2002a).
- If a chemical was not positively identified in any sample from a given medium (because it was reported as a nondetect and/or because of blank contamination), it was not addressed for that medium. No blank contamination was reported in any RI data from any medium.
- All data with "J" qualifiers were assumed to be positive identifications for that medium. "J" values are estimated concentrations reported below the minimum sample quantitation limit, or a value above the quantitation limit that has been qualified estimated due to laboratory or field quality control criteria out of limits.
- Duplicate samples from the same sampling location were considered as individual data points in summarizing the frequency of detection. The analytical results of duplicate samples will be included in summarizing the ranges of detected concentrations. However, prior to use of sample points in statistical analyses, duplicate results were averaged (see Appendix M-1).
- Rejected or "R"-qualified data was not used in the risk assessment, nor was unvalidated (pre-1999) data from historical investigations, or other historical data (as discussed for each media separately below).
- For each chemical detected in at least one sample in a given dataset and retained as a COPC, results reported as not detected were substituted with a value equal to one-half the sample quantitation limit as a proxy concentration for the statistical data analyses and exposure point concentration calculation.

Other data reduction rules were developed as needed for each media, and specific data evaluation issues are noted below.

6.2.1.1 Groundwater Data Reduction

Groundwater sampling results from the July 2002 quarterly monitoring and the RI sampling results from August 2002 were used in the HHRA. Although the RIWP and conceptual site model (TtEMI 2002a) initially indicated that groundwater data from residential and industrial wells will be used if construction, history, and integrity are considered sufficient for assessing reasonable maximum exposure risks, well construction details were not available for a majority of wells of this type at the LSGPS. Thus, only monitoring wells sampled during the July 2002 quarterly monitoring and RI comprised the data set to present a "snapshot in time" for risk assessment purposes.

To ensure that the exclusion of the residential and industrial well data would still be protective, data from groundwater samples collected from industrial wells and residential wells were compared to the RI dataset. This comparison ensured that well groupings were appropriate and that the range of concentrations observed in the industrial and residential wells were reflected in the RI dataset.

Monitoring Well Grouping

Two principal source areas were identified within the site (see Section 4.6.1 of this RI report) and data within these areas were grouped and evaluated separately. One source area was identified in Area A (Brenntag source area) and one source area was identified in Area B (Beall source area). No source area was identified in Area C.

Outside the source areas (nonsource areas), site-wide data were initially grouped by Area A, Area B, and Area C for convenience (as described in Section 1.3 of this RI report) because these groupings were readily available from the LSGPS database. In addition, this division separated the site data into three similarly sized geographic areas. Due to the low variability of COPC concentrations within each of these areas, no further geographic or other grouping of the data was considered necessary. Monitoring wells included in the RI dataset were grouped as follows (see Table 6-1):

- "Deep" groundwater samples from the two bedrock wells, MW128 and MW219, were evaluated separately, as wells completed into bedrock are different from the typical completions in the Lockwood area (which are situated in the shallower waterbearing unit). These data are summarized in Table M-2.1 of Appendix M.
- In Area A, a "source area" well grouping was established to allow for assessment of wells exceeding 10 times the groundwater TCE PRG (e.g., greater than 16 µg/L), which also exceeds the TCE maximum contaminant level (MCL, 5 µg/L). This resulted in wells MW007, MW008, MW009, MW011, MW101, MW102, MW104, MW116, MW121, MW122, and MW126 being included as the primary drivers for the elevated concentrations in this area. Monitoring wells MW004, MW005, MW117, and MW125 were also included in the Area A source area based on spatial proximity to wells located within the defined TCE plume area and PCE concentrations that exceeded 10 times the PCE PRG for tap water (e.g., greater than 6.6 µg/L). The summary statistics for all Area A "source area" wells are given in Table M-2.2 of Appendix M. Sampling locations for the Area A source area wells are shown on Figure 6-1 (see also Figures 2-1 and 2-2).

- In Area A, wells less than 10 times the groundwater TCE and PCE PRGs (e.g., less than 16 µg/L for TCE and less than 6.6 µg/L for PCE) were grouped, due to similarity in concentrations. The summary statistics for all Area A wells outside of the "source area" plume footprint are given in Table M-2.3 of Appendix M. Sampling locations included in the summary are shown on Figures 2-1 and 2-2.
- In Area B, a "source area" well grouping was established to allow for assessment of wells exceeding 10 times the groundwater TCE PRG (e.g., greater than 16 µg/L), which also exceeds the TCE MCL. This resulted in wells MW012, MW200, MW201, MW203, MW204, MW205, MW210, and MW212 being included as the primary drivers for the elevated concentrations in this area. No additional wells were included in the area B source area, based on PCE concentrations alone, as all Area B wells exceeding 10 times the PCE PRG (e.g., greater than 6.6 µg/L) also exceeded 10 times the TCE PRG. The summary statistics for all Area B "source area" wells are given in Table M-2.4 of Appendix M. Sampling locations for the Area B source area are shown on Figure 6-1 (see also Figures 2-3 and 2-4).
- In Area B, wells less than 10 times the groundwater TCE and PCE PRGs (e.g., less than 16 µg/L for TCE and less than 6.6 µg/L for PCE) were grouped, due to similarity in concentrations. Wells that fell outside the "source area" plume footprint were also included in the summary statistics for the remaining Area B wells. The summary statistics for all Area B wells outside of the source area are given in Table M-2.5 of Appendix M. Sampling locations included in the summary are shown on Figures 2-3 and 2-4.
- Because of a similarity in concentrations, all Area C shallow groundwater wells were grouped together. The summary statistics for all Area C wells are given in Table M-2.6 of Appendix M. Sampling locations included in the summary are shown on Figure 2-5.

The grouping into the Area A and Area B datasets facilitates protection of current and future receptors that might be located in a certain area that is less likely to be hydraulically influenced by one area versus another, for example, if a pumping well was producing shallow groundwater in a specific part of the neighborhood. Assessment of the areas as separate datasets does not imply that the plumes are discontinuous. Section 3.4.2 provides specific information regarding the hydrogeology of the shallow aquifer at LSGPS.

Groundwater data were reduced as follows:

- The only rejected ("R"-qualified) groundwater data were for four chemicals reported as nondetect by the laboratory, but qualified "R" during validation due to low instrument response for initial calibration standards. The results for the four affected chemicals (bromochloromethane, bromomethane, chloroethane, and chloromethane) were rejected in several samples from various sample delivery groups. Because of the poor initial calibration response, false negatives were possible for these four analytes in groundwater at LSGPS. These four chemicals were not, however, detected in any other media at LSGPS during the RI. Therefore, the rejection of these data has no impact on the HHRA at LSGPS.

6.2.1.2 Indoor Air Data Reduction

The indoor air database for the HHRA consisted of the most recent (February 2002) indoor air data collected at the LSGPS in the Lomond Lane area. The February 2002 data were validated by Portage Environmental, Inc. according to EPA Method TO15, EPA functional guidelines (EPA 1999b), and EPA Contract Laboratory Program statement of work document OLM04.2 (EPA 1999c).

As detailed in the *Comprehensive Indoor Air Sampling and Analytical Results Report* (CIASARR) (TtEMI 2002d), the indoor air data validator noted high percent differences between the initial calibration and continuing calibration for carbon tetrachloride, affecting 12 of the samples collected in February 2002. Carbon tetrachloride results for those 12 samples are qualified "J" (estimated); however, all J-qualified data were used in the HHRA.

The raw data, laboratory validation packages, and Region 8 Data Validation Report forms for the February 2002 indoor air sampling event are included in Appendix M of the CIASARR (TtEMI 2002d) and are not included in the present RI report. However, summary statistics in RAGS Part D Table 2 format are presented in Table M-2.7 of Appendix M, and tabulated data are presented in Appendix G. Locations of sampled residences are shown in Figure 6-3.

- Indoor air concentrations measured at the "background" location (542 Klenck) were not included in the Lomond Lane area-wide indoor air database for selection of COPCs or estimation of the exposure point concentration, as this background location was selected based on its location over groundwater concentrations below method detection limits. For conservatism and following EPA guidance (EPA 2001d, 2002c), the concentrations measured at 542 Klenck (the background location) were not subtracted from the exposure point concentration. Instead, the contribution of background concentrations to indoor air risk is discussed in the Uncertainty Analysis (Section 6.6).

- Indoor air concentrations measured in the samples from the two duplicates taken from 505 Lomond (LGSP-IA-08 and LGSP-IA-15) and 546 Lomond (LGSP-IA-12 and LGSP-IA-16) were treated separately in the summary statistics (Table M-2.7 of Appendix M) but averaged prior to calculation of the Lomond Lane area-wide indoor air exposure point concentration. Thus, (excluding the background location) 15 measurements were taken, representing 13 locations sampled in February 2002.

The statistical methods and site-specific considerations used to develop the community-wide exposure point concentration for indoor air are discussed in Section 6.3.2.2.

6.2.1.3 Surface Water and Sediment Data Reduction

Data were spatially grouped according to the surface water conveyance (either the Coulson irrigation ditch, AJ Gravel pond, or the wetland area near the Yellowstone River) in which they were collected, and tabulated individually in RAGS Part D standard Table 2 format (see Tables M-2.8 through M-2.11 of Appendix M). Sediment and surface water sample locations are shown on Figures 2-1 and 2-5.

Surface water and sediment data reduction issues included the following:

- The only rejected ("R"-qualified) surface water and sediment data were for four chemicals reported as nondetect by the laboratory, but qualified "R" during validation due to low instrument response for initial calibration standards. The results for the four affected chemicals (bromochloromethane, bromomethane, chloroethane, and chloromethane) were rejected in four samples of surface water and sediment at SW103, SW300, SW301, and SW303 in association with data delivery group B02050174. These four chemicals were not, however, detected in any other sample from these or any other media at LSGPS during the RI. Therefore, the rejection of this data has no impact on the HHRA at LSGPS.

- Six "J" (estimated) concentrations were reported in surface water and sediment for TCE and toluene but were used as if they were not qualified for purposes of risk assessment.

There were no detectable contaminants in the wetland area (Area A, sample location SW103) sediment, although contaminants were detectable in the surface water (see Table M-2.10 of Appendix M). None of the surface water or sediment data qualifiers impacted the risk assessment.

6.2.1.4 Surface and Subsurface Soil Data Reduction

Data were spatially grouped according to the area (Area A, B or C) in which they were collected, and tabulated individually in RAGS Part D standard Table 2 format (see Tables M-2.12 through M-2.15 of Appendix M). Area A had no detectable VOCs in any of the surface (0 to 2 ft bgs) samples; no surface soil samples were taken in Area C. Therefore, surface soil exposure pathways are complete only in Area B. Similarly, no detectable VOCs were reported in any Area C sample from 0 to 10 ft bgs. Therefore, subsurface exposure pathways were complete only in Areas A and B.

Soil data reduction issues included the following:

- The only rejected ("R"-qualified) soil data were for methyl ethyl ketone, which was reported as nondetect by the laboratory, but qualified "R" during validation because the instrument response for the initial calibration standard for methyl ethyl ketone was below the acceptable limit. When this occurs, false negatives are possible. However, methyl ethyl ketone is not known to have been released at the LSGPS and has not been detected in past investigations. Therefore, the rejection of these data has no impact on the HHRA at LSGPS.

- Several "J" (estimated) values were reported in soils from Areas A and B, but were used as if they were not qualified for purposes of risk assessment.

None of the soil data qualifiers impacted the risk assessment.

6.2.2 Identification of Chemicals of Potential Concern

Under the RAGS Part D paradigm, the maximum detected concentration of each chemical in a medium is evaluated to determine whether that detected chemical should be a COPC for the HHRA. COPCs were identified for each medium as described in the following subsections. To establish COPCs for the HHRA, the qualified (validated and usable) analytical database for the August 2002 RI effort and the most recent (July 2002) quarterly monitoring event was screened against EPA Region 9 PRGs (EPA 2002c) for residential land use, tap water, and ambient air, as appropriate. In this most recent update of the Region 9 PRGs (current as of October 2002), the PRGs for several chemicals, including TCE, were revised based on recent toxicological advances, as well as changes in exposure assumptions (EPA 2002d). Pursuant to Region 8 policy, the revised toxicity values for TCE (implemented in the October 2002 Region 9 PRG updates) have not met with peer review consensus and are subject to change. Therefore, the TCE PRGs effective prior to the October 2002 update, which are based on the previous TCE toxicity values, were retained pending finalization of the TCE peer review process. PRGs for the LSGPS COPCs are presented in the RAGS D Table 2s in Appendix M.

Concentrations of chemicals detected in water for areas with no direct measurement of indoor air concentrations were also compared to the target groundwater concentrations as listed in Table 2c of the *Draft Guidance for Evaluating the Vapor Intrusion to Indoor Air Pathway from Groundwater and Soils* (Subsurface Vapor Intrusion Guidance [SVIG; EPA 2002c]).

Region 9 PRGs were identified for the majority of chemicals detected in media at LSGPS. Surrogates were selected based on structural similarities for those chemicals lacking a PRG. Surrogates used in the selection of PRGs are noted in the footnotes to the RAGS Part D Table 2s where COPCs are selected. A Region 9 PRG or surrogate PRG was identified for all chemicals detected in media at LSGPS.

The objective of this step is to screen the available analytical data for the media of concern and identify the COPCs for each medium following RAGS protocols. For example, although numerous chemicals may be detected in soil or groundwater samples, they may be detected at a low frequency (e.g., less than 5 percent) and thus contribute very little to site-wide risks and therefore may be excluded as COPCs (EPA 1989). Where the maximum detected LSGPS concentration exceeded a PRG (and the COPC was detected in greater than 5 percent of the samples analyzed), the COPC was retained for that media and carried through the HHRA, taking into account the following screening criteria:

- Measured background concentrations were not used as a criterion for elimination of a COPC, per EPA guidance (EPA 1989). Chlorinated solvents are not naturally occurring and any occurrence would be expected to arise from an anthropogenic source. Therefore, all chemicals determined to be COPCs are included in the risk assessment. For indoor air, background concentrations were noted (see Table M-2.7 of Appendix M), but COPCs were not eliminated based on a background comparison.

The summary statistics for each media and COPC selection rationale were presented in a standard RAGS Part D Table 2 for each media (see Tables M-2.1 through M-2.20 in Appendix M). Each medium's COPCs are discussed briefly below.

6.2.2.1 Groundwater COPCs

Chemicals detected in groundwater and retained as COPCs for the quantitative HHRA are presented in Table 6-2. Summary data including all chemicals detected in groundwater, minimum and maximum detected concentrations, frequency of detection, and screening values are presented in RAGS Part D Tables (see Tables M-2.1 through M-2.6 in Appendix M). Summary tables for chemicals detected in groundwater and retained as COPCs for the groundwater to indoor air pathway are presented in Tables M-2.17 through M-2.20 of Appendix M (see Section 6.2.2.2 for a discussion of chemicals retained as COPCs for the groundwater-to-indoor-air pathway).

Deep Aquifer

Two deep wells (MW128 and MW219) were installed in bedrock at the LSGPS site. 1,2,4-trimethylbenzene, cis-1,2-DCE, and TCE were detected in groundwater samples collected from these wells; however, maximum concentrations were below the Region 9 PRGs for tap water. No chemicals were retained as COPCs in the bedrock wells.

Area A (Source Area)

Fourteen chemicals were detected at concentrations above the Region 9 tap water PRGs in groundwater samples collected from monitoring wells located within the source area in Area A as presented in Table 6-2 and Table M-2.2 of Appendix M. These chemicals were detected at a frequency of greater than 5 percent.

- Eight chlorinated organics (1,1,2-trichloroethane, chloroethane, chloroform, cis-1,2-DCE, methylene chloride, PCE, TCE, and VC) were detected at maximum concentrations exceeding their respective tap water PRGs. Concentrations of cis-1,2-DCE, methylene chloride, PCE, TCE, and VC also exceeded the MCLs. These eight chlorinated organics were retained as HHRA COPCs in the Area A source area.
- Two methylated benzenes (1,2,4-trimethylbenzene and 1,3,5-trimethylbenzene), ethylbenzene, and benzene were detected at maximum concentrations that exceeded their respective tap water PRGs. The maximum concentration of benzene (6.5 µg/L) also exceeded its MCL (5 µg/L). 1,2,4-Trimethylbenzene, 1,3,5-trimethylbenzene, ethylbenzene, and benzene were retained as HHRA COPCs in groundwater in the Area A source area.
- Xylene (total) and naphthalene were also detected at maximum concentrations that exceeded their respective tap water PRGs. Total xylenes and naphthalene were retained as COPCs in groundwater in the Area A source area.

Area A (Outside the Source Area)

Five chemicals, including four chlorinated organics (cis-1,2-DCE, PCE, TCE, and VC) and benzene were retained as HHRA COPCs as each was detected in groundwater samples from monitoring wells located outside the source area in Area A at a maximum concentration that exceeded its respective tap water PRG (see Table M-2.3 of Appendix M). Maximum detected concentrations of PCE, TCE, and VC also exceeded their respective MCLs.

Cis-1,2-dichloropropene and ethylbenzene were detected at a maximum concentration that exceeded their respective screening levels; however, because these chemicals were infrequently detected (detection frequency of less than 5 percent), they were not retained as HHRA COPCs.

Area B (Source Area)

Five chemicals, including four chlorinated organics (cis-1,2-DCE, PCE, TCE, and VC) and benzene were retained as HHRA COPCs as each was detected in groundwater samples from monitoring wells in the Area B source area at a maximum concentration that exceeded its tap water PRG (see Table M-2.4 of

Appendix M). Maximum detected concentrations of cis-1,2-DCE, PCE, and TCE also exceeded their respective MCLs.

Area B (Outside the Source Area)

Six chlorinated organics (1,2-dichloroethane, 1,3-dichloropropane, 2,2-dichloropropane, cis-1,3-dichloropropene, PCE, and TCE) were detected at maximum concentrations exceeding their respective tap water PRGs (see Table M-2.5 of Appendix M) or surrogate tap water PRGs. 1,2-Dichloroethane, PCE, and TCE were retained as HHRA COPCs. The maximum detected concentration of TCE (11 µg/L) also exceeded its MCL of 5 µg/L.

While maximum concentrations of 1,3-dichloropropane, 2,2-dichloropropane, and cis-1,3-dichloropropene exceeded their surrogate PRG screening values, these chemicals were not retained as HHRA COPCs for Area B (outside the source area) due to a lack of available toxicity factors. The possible effect of the elimination of these chemicals from the quantitative risk assessment is discussed in the Uncertainty Analysis (see Section 6.6).

Area C

Four chemicals, including three chlorinated organics (chloroform, PCE, and TCE) and benzene were detected in groundwater samples collected from monitoring wells in Area C at concentrations exceeding their tap water PRGs (see Table M-2.6 of Appendix M) and retained as HHRA COPCs. TCE was detected at a maximum detected concentration of 5.1 µg/L, which slightly exceeds the MCL of 5 µg/L.

6.2.2.2 Indoor Air COPCs

COPCs for indoor air in Area A were identified as presented in Table M-2.7 of Appendix M. Maximum detected concentrations of 1,2-dichloroethane, carbon tetrachloride, PCE, and TCE exceeded their ambient air PRGs. Concentrations of these chemicals also exceeded the target indoor air concentrations listed in Table 2c of the SVIG (EPA 2002c).

1,1-Dichloroethane, 1,1-dichloroethene, chloroethane, cis-1,2-DCE, and trans-1,2-dichloroethene were detected in indoor air at concentrations below their respective ambient air PRGs and target indoor air concentrations listed in Table 2c of the SVIG (EPA 2002c). However, as a conservative approach, these five chlorinated organics were retained as HHRA COPCs as they were also detected in groundwater at LSGPS (at a detection frequency greater than 5 percent).

For the other areas at the site, where measured indoor air concentrations were not available, modeling, using the *Johnson and Ettinger Model for Subsurface Vapor Intrusion Into Buildings* (EPA 2003d), was performed to estimate the concentration of chemicals in indoor air resulting from subsurface vapor intrusion. The most recent update to the EPA Johnson and Ettinger website (EPA 2003d) (http://www.epa.gov/superfund/programs/risk/airmodel/johnson_ettinger.htm) was April 7, 2003; however, the model does not yet incorporate all of the EPA SVIG (2002c) Appendix G defaults. EPA posted a note to the website above on March 13, 2003, indicating that updates to the model are forthcoming. As of the date of this submittal, however, EPA has not yet released the updated model.

For the groundwater-to-indoor-air pathway, chemicals detected in groundwater were compared to the target groundwater concentrations presented in Table 2c of the SVIG (EPA 2002c). Chemicals detected in groundwater at a concentration exceeding the target groundwater concentrations for vapor intrusion were retained as COPCs for the HHRA. Tables M-2.17 through M-2.20 of Appendix M present the SVIG target groundwater concentrations for the vapor intrusion pathway. Chemicals retained as COPCs for each area (outside the Area A source area) are discussed below.

- **Area A nonsource:** Three chemicals detected in groundwater were retained as COPCs for the groundwater to indoor air pathway as they exceeded their SVIG MCL-based target concentrations for groundwater. PCE (maximum concentration of 6.4 µg/L), TCE (maximum concentration of 11 µg/L), and VC (maximum concentration of 24 µg/L) were retained as COPCs in groundwater for the indoor air pathway (see Table M-2.17 of Appendix M).
- **Area B source:** Three chemicals detected in groundwater were retained as COPCs for the groundwater to indoor air pathway as they exceeded their SVIG MCL-based target concentrations for groundwater. cis-1,2,-DCE (maximum concentration of 1,380 µg/L), PCE (maximum concentration of 6.8 µg/L), and TCE (maximum concentration of 1,850 µg/L) were retained as COPCs in groundwater for the indoor air pathway (see Table M-2.18 of Appendix M).
- **Area B nonsource:** TCE was the only chemical detected in groundwater in this area that was above its SVIG MCL-based target concentration for groundwater. TCE was detected at a maximum concentration of 11 µg/L, which exceeded its SVIG MCL-based target concentration of 5 µg/L. TCE was retained as a COPC in groundwater for the indoor air pathway (see Table M-2.19 of Appendix M).
- **Area C:** TCE was the only chemical detected in groundwater in this area that was above its SVIG MCL-based target concentration for groundwater. TCE was detected at a maximum concentration of 5.1 µg/L, which slightly exceeded its SVIG MCL-based target concentration of 5 µg/L. TCE was retained as a COPC in groundwater for the indoor air pathway (see Table M-2.20 of Appendix M).

6.2.2.3 Surface Water and Sediment COPCs

As seen in Tables M-2.8 through M-2.11 of Appendix M, few surface water or sediment COPCs were identified. Specifically:

- Within the AJ Gravel pond (see Table M-2.8 of Appendix M), all three samples contained detectable PCE, TCE, VC and cis-1,2-DCE. Of these, all but the latter were reported at maximum concentrations that exceeded PRGs for tap water as well as (for TCE and VC) MCLs and WQB-7 criteria numeric standards for surface water (DEQ 2001b). As PCE, TCE, and VC exceeded their respective tap water PRGs, these chemicals were selected as HHRA COPCs in the AJ Gravel pond. Because of the gravelly nature of the pond, no sediment was collected during the RI.
- For surface water in the Coulson irrigation ditch (see Table M-2.9), one chemical was detected at a maximum concentration above the residential tap water PRG. Only one location (the maximum of 2.6 µg/L at SW300) exceeded the TCE tap water PRG. As noted in Section 6.5.3.7, although this results in one COPC for surface water, this pathway was qualitatively evaluated, given the magnitude of the single exceedance. For sediment in the Coulson irrigation ditch (see Table M-2.11), only toluene was detected, and the maximum concentration was below the PRG. Therefore, no sediment COPCs were identified for Coulson irrigation ditch.
- For the wetland area at sample location SW103 (see Table M-2.10), there were no sediment detections, but TCE surface water concentrations were reported in both the normal and duplicate samples taken at this location. However, the concentrations measured in the wetland area were below both the PRG for tap water as well as the WQB-7 criteria standard (set to the MCL of 5 µg/L). Therefore, no wetlands area COPCs were identified for the HHRA.

6.2.2.4 Surface and Subsurface Soil COPCs

As seen in Tables M-2.12 through M-2.15 of Appendix M, few soil COPCs were identified. Specifically:

- In Area A (see Table M-2.12), a screen of maximum detected soil concentrations (in the top 10 feet of soil) against Region 9 residential PRGs for direct contact indicated that no detected VOCs are COPCs.
- In Area B (see Table M-2.14), a screen of maximum detected soil concentrations (in the top 10 feet of soil) against Region 9 residential PRGs for direct contact indicated that no detected VOCs are COPCs.
- In Area C, no detectable chemicals were reported in any sample taken within 10 feet of the land surface. Therefore, there is no RAGS Part D standard Table 2 for Area C, and there are no COPCs for soil in Area C.

Therefore, while soil exposure pathways are hypothetically complete, no soil pathways were further quantitatively evaluated in the HHRA.

6.2.2.5 Yellowstone River COPCs

Concentrations of chemicals detected in sentinel wells along the length of LSGPS parallel to the Yellowstone River (MW118, MW120, MW124, MW125, MW126, MW127, and MW305) were used to model chemical concentrations in the Yellowstone River. Model concentrations of COPCs in the Yellowstone River were below the tap water PRGs and WQB-7 criteria surface water standard (see Section 5.4.3.3 for a description of the model used to predict COPC concentrations in the Yellowstone River). There are no COPCs for the Yellowstone River (see Table M-2.16 in Appendix M).

6.3 EXPOSURE ASSESSMENT

EPA identifies three components of an exposure assessment as follows: (1) characterizing the exposure setting, (2) identifying potential exposure pathways, and (3) quantifying exposures. The exposure setting characterization includes a discussion of current and future land use (see Sections 1.3 and 3.5). The identification of potentially complete exposure pathways includes a discussion of exposed populations, exposure pathways that may be complete, exposure scenarios, and hypothetical receptors. The quantification of exposures includes the quantitative characterization of exposure point concentrations and chemical intakes.

In the planning stages of the RI, the RIWP (TtEMI 2002a) included development of the dynamic conceptual site model. The conceptual site model was initiated as the planning tool that organized what was already known about the site before RI mobilization and helped the planning team identify what additional information must be gathered to make the decisions that will achieve the project’s goals. Use of the conceptual site model is ongoing, and an iterative approach (allowing the conceptual site model to evolve and mature as site work progresses and data gaps are filled) is required. Therefore, the conceptual site model (Figure 6-3) has been updated from the preliminary conceptual site model (previously Figure 9 of the RIWP; TtEMI 2002a) compiled before RI mobilization. This conceptual site model is the framework for understanding the receptors and exposure pathways included in the HHRA for the LSGPS.

Activity patterns for current and future receptors were evaluated under a reasonable maximum exposure scenario and under a central tendency exposure scenario based on current and future land and water uses.

As defined by EPA (1989), the reasonable maximum exposure scenario represents the maximum exposure that is reasonably expected to occur at a site, while the central tendency exposure scenario represents the reasonable average exposure expected to occur. Many exposure factors in the reasonable maximum exposure scenario are based on conservative "upper bound" estimates. The use of multiple conservative "upper bound" factors make the reasonable maximum exposure scenario protective of potential exposures. Exposure factors for both the reasonable maximum exposure and central tendency exposure scenarios are presented in Tables M-4.1.RME through M-4.7.RME for the reasonable maximum exposure scenario and Tables M-4.1.CTE through M-4.7.CTE for the central tendency exposure scenario (see Appendix M).

6.3.1 Exposure Pathways and Receptors

Discussed below are the media-based pathways of concern for the LSGPS (Section 6.3.1.1), with a brief introduction to the human receptors who may become exposed under both the reasonable maximum exposure and central tendency exposure scenarios (Section 6.3.1.2). A more detailed definition of the quantitative exposure parameters to facilitate dose estimates is provided for each receptor in Section 6.3.3.

6.3.1.1 Exposure Pathways

Mechanisms by which exposures may occur are called exposure pathways. EPA (1989) describes exposure pathways in terms of the following four primary components:

- A source and mechanism of chemical release
- A retention or transport medium (or media in cases involving transfer of chemicals among media)
- A point of human (receptor) contact with the contaminated medium (known as the exposure point)
- An exposure route (such as ingestion) at the contact point

All four of these components must be present for a potential exposure pathway to be considered complete and for exposure to occur. Below are the completed exposure pathways for the LSGPS, which correspond to the conceptual site model diagram (Figure 6-3) and to the pathways quantitatively evaluated in this HHRA, as shown on the RAGS Part D Table 1 (see Table M-1 of Appendix M).

Groundwater

The primary media known to be contaminated at the LSGPS is groundwater. Therefore, all complete groundwater-related exposure pathways are considered in this HHRA. In addition to the graphic presentation of the conceptual exposures shown in Figure 6-3, the RAGS Part D Table 1 (see Table M-1, Appendix M) provides the standardized information required at each Superfund site for each completed pathway and potentially exposed receptor.

As well water is currently available for non-potable use (irrigation, washing cars); an adult receptor was evaluated for direct contact with groundwater via the dermal and inhalation routes (outdoor air) during seasonal use. Similarly, an adolescent receptor (age 6 to 16) was evaluated via the dermal and inhalation (outdoor air) routes for seasonal use during recreational activities (swimming in wading pool, playing in sprinklers). This adolescent receptor was also assessed for incidental ingestion of groundwater during swimming activities.

Exposure routes for a resident (adult and child) not connected to the public water supply, who uses groundwater for whole-house use, include ingestion, inhalation (of vapors during showering or other household use), and dermal contact during bathing or showering. Inhalation exposures (from water table transfer upward through crawlspaces into indoor air) for a resident were evaluated in association with air separately (see below).

Current/future groundwater exposures for an industrial worker are limited to dermal exposure (direct contact) and inhalation (outdoor air) during routine cleaning activities (for instance, washing equipment, facility maintenance, and hand washing) at those facilities that are connected to the public water supply or that supply an alternate source of drinking water. Ingestion was not evaluated for this receptor as facilities connected to the public water supply or supplying an alternate source of drinking water do not use groundwater as a potable water source.

Exposure routes for an industrial worker working at a facility that is not connected to the public water supply (and who is not supplied with an alternate source of drinking water) include ingestion of tap water (assumes groundwater is used as a potable water source), and dermal and inhalation (outdoor air) exposure during routine cleaning activities (for instance, washing equipment, facility maintenance, and hand washing).

Although an occasional visitor to either a residence or an industrial facility was not quantitatively assessed for exposure to COPCs at LSGPS in the HHRA, exposure to a visitor is protected by evaluation of the more conservative full-time residential and industrial worker scenarios. Because a visitor would sustain only short-term exposure, as compared to the resident and industrial worker, any risks to a visitor would be expected to be less than those associated with these longer term exposures.

To address potential exposure during future excavation activities (up to 4 month duration) a future utility/construction worker contacting subsurface groundwater (during dewatering or emergency repair activities should groundwater seep into an excavation or trench) was considered. The utility worker was evaluated for exposure to chemicals in groundwater by incidental ingestion, dermal contact, and inhalation of vapors in outdoor air. As long-term utility repair/construction work is not currently ongoing, a current scenario was not considered for this receptor; the long-term “future”scenario is protective of a utility/construction worker conducting short-term emergency utility/construction repairs.

Surface Water and Sediment

Surface water and sediment may have become contaminated through groundwater-to-surface water discharge, via irrigation use, or via overland (runoff) migration. As noted in Section 6.2.2.3, only the AJ Gravel pond was identified as containing COPCs in surface water. Human exposure routes include incidental ingestion and dermal exposures during fishing and wading activities, as well as potential food-chain transfer through recreationally caught fish in the AJ Gravel pond. The inhalation pathway (volatiles in surface water to outdoor air) was not quantified as it was considered insignificant in these outdoor scenarios due to wind dispersion. No other exposures (to recreational users accessing ditches or the wetlands area near the Yellowstone River) are believed to be significant, based on the lack of COPCs identified for the sampled sediment and surface water from these other water bodies.

No worker exposure was assessed for surface water or sediment exposure, although it is acknowledged that mining of the AJ Gravel pond and periodic (approximately annual) dredging of the Coulson irrigation ditch occurs. During periodic collection of gravel at AJ Gravel pond, machinery is used to dredge gravel from AJ Gravel pond. A worker performing this type of activity was not quantitatively assessed in the HHRA, since collection of the gravel is done with machinery (i.e. steam shovels), with little or no direct contact of the worker with either the gravel from the pond or surface water. Similarly, no exposure of the workers assigned to dredging/clearing of the Coulson irrigation ditch was evaluated, because this work is also largely accomplished with machinery (i.e., backhoe) from the bank. A more conservative scenario (the recreational visitor scenario) instead accounts for exposure via direct contact routes (as described above) and uses an adolescent receptor with a smaller body weight. For these reasons, no additional assessment of a gravel pond worker or ditch maintainer was included.

Soils

Both surface soils (because of aboveground spills and leaks) and subsurface soils (because of leaching) are media for which exposure pathways were conceptually developed. However, the RI field effort indicates that surface and subsurface soil contamination is not widespread, as no soil COPCs have been identified for the LSGPS. Therefore, the soil pathways are (given the lack of COPCs) considered insignificant and no risks are concluded.

Air

Groundwater contamination may migrate from the subsurface (upward from the water table and capillary fringe) and accumulate in enclosed spaces. For residential assessments, measured indoor air concentrations have been recorded in the Lomond Lane area (see the CIASARR [TtEMI 2002d]). Current/future residential receptors in this sampling area were evaluated for exposure to COPCs in indoor via subsurface vapor intrusion.

In other areas of the site where no indoor air monitoring results were available, the Johnson and Ettinger model (EPA 2003d) was used to estimate indoor air concentrations resulting from subsurface vapor intrusion to indoor air from groundwater. A summary of the model and the site-specific model parameters are presented in Appendix M-3.

In addition, the volatilization of COPCs in groundwater to indoor air during bathing and showering was evaluated for current/future residential receptors not connected to the public water supply and assumes groundwater is used for whole-house use.

Volatilization of COPCs to outdoor air was modeled, using a box model approach, to also evaluate current and future use of groundwater for non-potable use such as car washing, lawn irrigation, or in wading pools. Appendix M-3 presents the model assumptions and equations used to model the volatilization of COPCs in groundwater to outdoor air.

For industrial exposures, outdoor use of industrial groundwater was assumed and modeled during appropriate activities. Similarly, utility/construction activities were modeled, using a box model approach, to evaluate groundwater intrusion and subsequent volatilization of contaminants into trench air. A summary of the box model, including the model assumptions and equations, used to estimate concentrations of COPCs volatilizing to outdoor air from groundwater is presented in Appendix M-3.

Biota

Ultimately, biota that are directly exposed to contaminated media (such as fish that live in a contaminated surface water body) may take up, or bioaccumulate, contaminants. This uptake can be important when contaminants transfer through the food chain to humans. Coulson irrigation ditch is used as a bait collection area, and since these fish are not for human consumption, this potential pathway is incomplete and was not quantified in this HHRA.

However, because the historic stocking of fish in the AJ Gravel pond and local use of the area, as well as the selection of surface water COPCs in the AJ Gravel pond (see Table M-2.8 of Appendix M), the AJ Gravel pond surface water was retained in the conceptual site model for quantitative evaluation in the HHRA. Concentrations of COPCs in surface water at AJ Gravel pond were modeled in a desktop food web model to estimate concentrations of COPCs in fish tissue to evaluate exposure via consumption of fish caught in AJ Gravel pond.

6.3.1.2 Human Receptors

For the exposure pathways and routes described above, particular receptors and scenarios are envisioned to characterize the long-term (chronic) risks due to contamination at the LSGPS, as described below. As part of both the data collection and exposure assessment phases, resident surveys and information from recent quarterly monitoring events (up to February 2002) were included, as appropriate, in establishing site-specific, protective assumptions for the HHRA. Residents and local workers were asked about their well use patterns, activity patterns, and structure construction details, which may assist in characterizing exposure and reducing uncertainty in the HHRA process.

Resident

A "typical" resident (following the EPA Region 9 recommended default exposure assumptions set forth in the PRG background document; EPA 2002d) is included in the HHRA to visualize how LSGPS contamination might affect Lockwood residents. Assumptions include 30 years of exposure (6 as a child, and 24 thereafter), 350 days per year. The details of the conservative assumptions used in the residential exposure scenario are outlined in Section 6.3.3.1. As noted in Section 6.3.3.1, where site-specific information warrants (such as from resident surveys and site reconnaissance), exposure assumptions were varied from the EPA Region 9 defaults for consideration of local exposure patterns.

Industrial Worker

A "typical" industrial worker (following the EPA Region 9 recommended default exposure assumptions set forth in the PRG background document; EPA 2002d) is included in the HHRA to assess how LSGPS contamination might affect Lockwood industrial/commercial workers. The details of the assumptions used in the industrial worker exposure scenario are outlined in Section 6.3.3.2. As noted in Section 6.3.3.2, where site-specific information warrants (such as from site reconnaissance and observations of work practices), exposure assumptions were varied from the EPA Region 9 defaults for consideration of local exposure patterns.

Utility/Construction Worker

A utility/construction worker (who might access subsurface soils and groundwater) scenario is included for use at the LSGPS. Exposure assumptions from other EPA regions and states and relevant information from the EPA *Exposure Factors Handbook* (EPA 1997d) were used to assess exposure of this receptor to COPCs at LSGPS. The site-specific depth to soil was assumed to include exposures to 10 feet bgs, as detailed in Section 6.3.2; however, since no soil COPCs were identified, only groundwater (ingestion, dermal contact, and inhalation exposures during trenching) exposures were quantified for this receptor (see Section 6.3.3.3).

Recreational User

A recreational user scenario (sometimes called a "site visitor" scenario) is included for use at the LSGPS to represent individuals who might access surface water bodies such as AJ Gravel pond. Following the precedents of other EPA regions and at other Superfund sites, many of the exposure assumptions for the recreational user and trespasser scenarios were taken from existing supplemental guidance (c.f., EPA 1995a). However, because the exposure assumptions developed for other regions may not reflect the climate- and location-specific needs of the Lockwood area, this scenario is site-specific, at least in part (see Section 6.3.3.4).

6.3.2 Exposure Point Concentrations

For each exposure scenario set forth in the RAGS Part D standard Table 1 (see Table M-1 in Appendix M) and each set of COPCs identified in each RAGS Part D standard Table 2 (see Tables M-2.1 through M-2.20 in Appendix M), an accompanying exposure point concentration table was prepared in RAGS Part D standard Table 3 format. If no COPCs were selected for a given medium, then no exposure point concentration table was prepared. Although potential surface soil, surface/subsurface soil, and sediment exposures were considered identified in the RAGS Part D standard Table 1 (see Table M-1 of Appendix M), no COPCs were identified for these three media at the LSGPS site. Therefore no surface soil, subsurface soil, or sediment exposure point concentrations were calculated.

Exposure point concentrations were calculated based on EPA guidance (EPA 2002b); for groundwater, the 95th percentile upper confidence limit on the arithmetic mean (UCL95) of the concentration was used as the exposure point concentration for the reasonable maximum exposure case, and the arithmetic mean concentration was used as the exposure point concentration for the central tendency exposure case. In the case of the reasonable maximum exposure, if the UCL95 exceeded the maximum detected concentration, the maximum detected concentration was used as the exposure point concentration. Further discussion of exposure point concentration development is presented in Appendix M-1. Exposure point concentrations for the reasonable maximum exposure and central tendency exposure scenarios are presented in Tables M-3.1.RME through M-3.8.RME and Tables M-3.1.CTE through M-3.8.CTE, respectively, of Appendix M. A summary of the exposure point concentrations is also included in Table 6.3.

6.3.2.1 Groundwater Exposure Point Concentrations

Groundwater exposure point concentrations were established upon critical review of the RI database. Upon review of the historical groundwater monitoring data and consultation among EPA, DEQ, and TtEMI, it was concluded that current groundwater data (July 2002 quarterly monitoring event and August 2002 RI) from RI sampling locations with sufficient characterization data (e.g., well completion information) would form the basis for the HHRA (DEQ 2002). As discussed previously in Section 6.2.1.1, residential well and industrial well data were not included in the in the data set evaluated in the HHRA. The final grouping of groundwater data for the HHRA was determined upon critical review of the RI database, as described in Section 6.2.1.1.

In areas where all residences are not currently connected to the public water supply, separate residential scenarios assuming (1) connection to the public water supply and (2) use of groundwater for whole-house use were evaluated. To aid the decision-making process, possible future use of unremediated and unattenuated, contaminated groundwater for whole-house use in areas currently served by the public water supply was assumed as a hypothetical reasonable maximum exposure scenario. An additional residential scenario involving the use of well-water for car washing, irrigation, or recreational (wading pool or sprinklers) was included to more closely represent the actual probable use of the wells at the present time. Similarly, for industrial well use, scenarios assuming the use of well water for facility cleaning activities (nonpotable use; for instance, washing equipment, facility maintenance, and hand washing) and also as a drinking water source should existing wells be used in the future (with no alternate drinking water supply) were evaluated.

Historically, residential and industrial wells within the LSGPS were monitored quarterly. Currently, residential wells serving homes that are not connected to the public water supply system within the LSGPS are monitored semiannually. Groundwater samples are collected and analyzed for the COPCs at the LSGPS. Residences where COPCs have exceeded the federal MCLs have been connected to the public water supply system. Residences that are not connected to the public water supply system and where COPC concentrations in their water supply well are near the MCLs receive an alternate drinking water supply. DEQ has confirmed that all businesses within the LSGPS site boundaries are either connected to the public water supply or provide an alternate drinking water supply for their employees.

To estimate risks associated with exposure to ground water at the LSGPS, exposure point concentrations representative of an reasonable maximum exposure scenario were calculated for each subarea (see Section 6.3.2 and Appendix M-1) using monitoring well data collected during the July 2002 quarterly monitoring and RI. A review of the data from residential and industrial wells sampled during quarterly monitoring (April, July, and October 2002) activities was conducted to ensure that the exposure point concentrations calculated from the RI data would not underestimate risks associated with groundwater use at specific locations. Residential and industrial well data was compared against the exposure point concentrations used in the risk calculations. Table 6-3 presents the analytical data for COPCs detected in the residential and industrial wells, MCLs, and the reasonable maximum exposure point concentrations (see also Tables M-3.1.RME through M-3.5.RME of Appendix M) for each subarea. The results for four wells (RW001, IW003, IW013, and IW016) exceeded the exposure point concentration in the corresponding LSGPS subarea. The uncertainty resulting from exclusion of the residential and industrial wells from the RI data set is discussed in Section 6.6.4.

Each of the exposure point concentrations for completed groundwater exposure pathways is presented in Tables M-3.1.RME through M-3.5.RME (reasonable maximum exposure scenario) and Tables M-3.1.CTE through M-3.5.CTE (central tendency exposure scenario) in Appendix M. In general, EPA's remediation decisions are based on reasonable maximum exposure scenarios, and central tendency exposure scenarios are presented only for comparison and informational purposes.

6.3.2.2 Indoor Air Exposure Point Concentrations

Residential indoor air concentrations were obtained from the indoor air sampling program (collected per the indoor air sampling and analysis plan; DEQ 2001a) database for the most recent (February 2002) sampling event as documented in the CIASARR (TtEMI 2002d).

To facilitate the evaluation of indoor air risks in a portion of Area A, the February 2002 data from each of the 13 locations sampled was evaluated (TtEMI 2002d). Concentrations measured in indoor air fluctuate seasonally, as demonstrated in the CIASARR (TtEMI 2002d), and no single location was consistently the "reasonable maximum exposure home" for all COPCs. Therefore, the February 2002 database (representing the most recent data set) was evaluated. Two indoor air sampling locations (345 Lomond Lane and 403 Lomond Lane) were thought to be in the centerline of the VOC plume; 345 Lomond Lane exhibited the highest indoor air concentrations in February 2002, and 403 Lomond Lane's indoor air concentrations were very similar between the two most recent (October 2001 and February 2002) events. Thus, since the highest indoor air concentrations were observed in the centerline of the plume in the cooler season, the most recent (February 2002) dataset was selected as representative of the indoor air concentrations for determining an exposure point concentration.

To assess indoor air risks for a portion of Area A, an exposure point concentration was derived using the entire February 2002 dataset. To ensure this approach was protective, the exposure point concentration was compared to the range of individual home measurements to ensure that the exposure point concentration did not significantly underpredict concentrations that had been measured in any given residence (see Section 6.6).

As discussed previously, in other areas of the site where no indoor air monitoring results were available, the Johnson and Ettinger model (EPA 2003d) was used to determine exposure point concentrations resulting from subsurface vapor intrusion to indoor air from groundwater. (See Appendix M-3 for a summary of the model.)

Risks resulting from exposure to COPCs in groundwater, which might volatilize to outdoor air, were evaluated using a box model approach (see Section 6.3.2.4 below).

6.3.2.3 Surface Water Exposure Point Concentrations

AJ Gravel pond was the only surface water body at LSGPS with both completed exposure pathways and detected concentrations of COPCs that exceeded the PRGs. Surface water concentrations were used for the AJ Gravel pond in the HHRA to derive the exposure point concentration for the recreational scenario. Exposure point concentrations in surface water were established for each water body for current and future exposures to the recreator via dermal and incidental ingestion exposure. Because of the (1) low concentrations of COPCs detected in surface water, and (2) open air location of the AJ Gravel pond, no inhalation exposure was considered during recreational use of the pond. It was assumed that normal volatilization processes would effectively dissipate VOC concentrations into outdoor air across the large surface water body, such that intermittent recreational exposures outdoors at a single point along a bank of a water body (or during wading) would be insignificant compared to dermal or ingestion exposure.

Three AJ Gravel pond samples from the RI sampling efforts were included in the AJ Gravel pond COPC selection (Table M-2.8) and exposure point concentration calculation. The resulting exposure point concentrations for this exposure area are presented in Table M-3.7.RME (for the reasonable maximum exposure scenario; see Table M-3.7.CTE for central tendency exposure point concentrations) in Appendix M. The average value was used as the central tendency exposure point concentration.

6.3.2.4 Outdoor Air Exposure Point Concentrations

Volatile COPCs may be released as vapors from groundwater during industrial use or during residential irrigation and recreational use. Outdoor air exposure point concentrations for VOCs that may be released from groundwater during utility and construction trenching were estimated through the use of a box model approach (which is presented in Appendix M-3).

For the purpose of developing exposure point concentrations in air (both indoor and outdoor), VOCs were identified on the basis of the methodology referenced in the EPA Region 9 PRGs (EPA 2002d): chemicals were considered to be volatile if they have a molecular weight of less than 200 grams per mole and a Henry's law constant greater than 1×10^{-5} atmosphere-cubic meter per mole. All COPCs identified in media at LSGPS are considered volatile.

6.3.3 Contaminant Dose Estimates

After the exposure point concentrations were calculated, they were combined with the receptor- and scenario-specific exposure assumptions set forth in RAGS Part D tables (see Tables M-4.1.RME through M-4.7.RME and Tables M-4.1.CTE through M-4.7.CTE of Appendix M) to estimate a daily dose for each COPC. In each case, doses were estimated as follows:

A chemical intake via the oral or dermal route, is expressed as milligrams of chemical per kilogram body weight per day (mg/kg-day) and is calculated for both reasonable maximum exposure (see Tables M-4.1.RME through M-4.7.RME of Appendix M) and central tendency exposure (Tables M-4.1.CTE through M-4.7.CTE of Appendix M) scenarios. EPA-derived exposure algorithms were used to estimate chemical intakes for each route of exposure. Equation 6-1 (EPA 1989) presents a generic equation for calculating chemical daily intake for the oral and dermal routes, as follows:

$$I = \frac{C \times CR \times EF \times ED}{BW \times AT} \tag{6-1}$$

where:

I = Intake, which is the amount of chemical at the exchange boundary (mg/kg-day)

C = Chemical concentration, which is the exposure point concentration (for example, µg/L for groundwater)

CR = Contact rate, which is the amount of contaminated medium contacted per unit of time or event and may be the ingestion rate or dermal contact rate (for example, 2 liters per day [L/day] for the ingestion rate of tap water)

EF = Exposure frequency, which is the rate at which the exposure occurs (normally expressed as a given number of days per year)

ED = Exposure duration, which is the number of years that a receptor comes in contact with the contaminated medium

BW = Body weight, which is the average body weight of the receptor over the exposure period (for example, 70 kilograms for the average U.S. adult)

AT = Averaging time, which is the period over which exposure is averaged (days). For cancer-related exposures, the averaging time is 25,550 days based on a lifetime exposure of 70 years (average life expectancy). For non-cancer-related exposures, the averaging time is equal to the exposure duration multiplied by the number of days in 1 year (or 365 days).

For the inhalation route, the generic intake equation can be adjusted to calculate the concentration in the inspired air such that:

$$Cadj \quad = \quad C \times ((EF \times ED) / AT) \tag{6-2}$$

where:

Cadj = Chemical intake concentration adjusted for a specific exposure scenario (mg/m^3)

C = Chemical concentration, which is the exposure point concentration (for example, mg/m^3 for indoor air)

EF = Exposure frequency, which is the rate at which the exposure occurs (normally expressed as a given number of days per year)

ED = Exposure duration, which is the number of years that a receptor comes in contact with the contaminated medium

AT = Averaging time, which is the period over which exposure is averaged (hours). For cancer-related exposures, the averaging time is 613,200 hours based on a lifetime exposure of 70 years (average life expectancy). For non-cancer-related exposures, the averaging time is equal to the exposure duration multiplied by the number of hours in 1 year (or 8,760 hours).

Exposure variable values used in the pathway-specific equations were based on a series of reported and assumed factors regarding current and potential land use patterns at the site. The HHRA exposure scenarios that required an exposure point concentration are depicted on the conceptual site model (Figure 6-3), and each scenario thus requires a RAGS Part D standard Table 4. Exposure variables used for this evaluation are consistent with EPA guidance. These exposure variable values were generally obtained from the following EPA risk assessment guidance:

- RAGS Volume 1 Human Health Evaluation Manual (Part A), Interim Final (EPA 1989)
- RAGS Volume 1 Human Health Evaluation Manual (Part B, Development of Risk-based PRGs), Interim (EPA 1991b)
- RAGS Volume 1 Human Health Evaluation Manual (Part E, Supplemental Guidance for Dermal Risk Assessment, Draft) (EPA 2001c)

If values were not available from these sources, EPA's *Exposure Factors Handbook* (1997d), recent updates by the EPA National Center for Environmental Assessment to the *Exposure Factors Handbook* or other sources (including professional judgment) were used. Exposure parameters were modified to take into account site-specific climate patterns, where appropriate.

The reasonable maximum exposure and central tendency exposure factors are described in the following text. Scenario-specific assumptions and exposure variables used to estimate chemical daily intakes for both the reasonable maximum exposure and central tendency exposure cases for each hypothetical receptor are included in the standard series (Tables M-4.1.RME through M-4.7.RME and Tables M-4.1.CTE through M-4.7.CTE, respectively) of exposure tables in Appendix M. Pathway-specific equations for estimating chemical daily intakes and exposure variables for each scenario (including specific references) are also included in the exposure parameter tables, which are presented in the standardized format specified by EPA (2001d).

In summary, the exposure parameters for each receptor are outlined below, with the rationale for each reasonable maximum exposure and central tendency exposure parameter.

6.3.3.1 Residents

Groundwater contamination extends into residential areas. A resident (adult and child) was evaluated to estimate risks to residents who reside over the groundwater plume at LSGPS. The current and future resident (adult and child) scenario evaluates risks to residential receptors connected to the public water supply; it assumes groundwater is not available for household use as a potable water source. The current and future resident adult and child (who are connected to the public water supply) were only quantitatively assessed for exposure to COPCs in indoor air.

A current and future resident (adult and adolescent) was evaluated to estimate risks from the use of well water for purposes other than a drinking water source, such as car washing, lawn irrigation, and recreation (i.e., wading pools and sprinklers).

Risks were also estimated for residential receptors who are not currently connected to the public water supply or who might use well water for whole-house use in the future. A current/future residential scenario assuming a groundwater well is used for potable use was evaluated. In this current/future scenario, exposure to COPCs via ingestion of, dermal contact with, inhalation of volatiles in groundwater during household use (showering/bathing), and inhalation of indoor air was evaluated.

Current/Future Resident Adult

Quantitative exposure parameters for this scenario include the following assumptions:

- Body weight: 70 kilogram (kg) (reasonable maximum exposure and central tendency exposure)
- Ingestion of groundwater (future only): 2 liters per day (reasonable maximum exposure); 1.4 liters per day (central tendency exposure)
- Dermal contact with groundwater (showering; future only): Exposure times of 0.58 hours per day (reasonable maximum exposure) and 0.25 hours per day (central tendency exposure); exposed skin surface area of 18,000 cm^2 (entire body)
- Inhalation of vapors in indoor air: Exposure time of 24 hours per day (reasonable maximum exposure and central tendency exposure)
- Inhalation of vapors from groundwater (during showering/bathing): Exposure time of 0.58 hours per day (reasonable maximum exposure) and 0.25 hours per day (central tendency exposure)
- Exposure frequency: 350 days per year (reasonable maximum exposure and central tendency exposure)
- Exposure duration: 24 years (reasonable maximum exposure) and 9 years (central tendency exposure)
- Averaging time, noncarcinogens: 8,760 days (reasonable maximum exposure) and 3,285 days (central tendency exposure); for the inhalation pathway, averaging times were expressed in hours (210,240 hours [reasonable maximum exposure] and 78,840 hours [central tendency exposure])
- Averaging time, carcinogens: 25,550 days (reasonable maximum exposure and central tendency exposure); for the inhalation pathway, averaging times were expressed in hours (613,200 hours [reasonable maximum exposure and central tendency exposure])

Current/Future Resident Adult (Car Washing/Lawn Irrigation Activities)

A resident adult was also evaluated for current and future intermittent exposure to groundwater from household use of nonpotable water during activities such as lawn irrigation and car washing. Dermal contact with and inhalation of vapors from groundwater were evaluated. Quantitative exposure parameters for an adult resident for this scenario include the following assumptions:

- Body weight: 70 kg (reasonable maximum exposure and central tendency exposure)
- Dermal contact with groundwater: Exposure time of 2 hours per day (reasonable maximum exposure) and 1.75 hours per day (central tendency exposure); exposed skin surface area of 5,700 cm^2 (hands, arms, and legs).
- Inhalation of vapors from groundwater: Exposure time of 2 hours per day (reasonable maximum exposure and central tendency exposure)
- Exposure frequency: 32 days per year (reasonable maximum exposure) and 16 days per year (central tendency exposure). Exposure frequency for this activity was estimated as occurring between mid-May to mid-September (16 weeks). For the reasonable maximum exposure scenario, one lawn watering event per week and one car wash per week were assumed. The central tendency exposure scenario assumed one lawn watering or car wash per week.
- Exposure duration: 24 years (reasonable maximum exposure) and 9 years (central tendency exposure)
- Averaging time, noncarcinogens: 8,760 days (reasonable maximum exposure) and 3,285 days (central tendency exposure); for the inhalation pathway, averaging times were expressed in hours (210,240 hours [reasonable maximum exposure] and 78,840 hours [central tendency exposure])
- Averaging time, carcinogens: 25,550 days (reasonable maximum exposure and central tendency exposure); for the inhalation pathway, averaging times were expressed in hours (613,200 hours [reasonable maximum exposure and central tendency exposure])

Current/Future Resident Child

Residential areas are encompassed within the boundaries of the LSGPS. A child residential receptor was considered for current and future on-site exposure. Quantitative exposure parameters for this scenario include the following assumptions:

- Body weight: 15 kg (reasonable maximum exposure and central tendency exposure)
- Ingestion of groundwater (future only): 1 liter per day (reasonable maximum exposure); 0.7 liter per day (central tendency exposure)

- Dermal contact with groundwater (bathing; future only): Exposure time of 1 hour per day (reasonable maximum exposure) and 0.33 hours per day (central tendency exposure); an age-adjusted skin surface area value (1,600 cm^2 per year per kg) was used to account for changes in a child's skin surface area based on changes in body weight over time (see Table 4.5.RME and Table 4.5.CTE of Appendix M for equations).
- Inhalation of vapors in indoor air: Exposure time of 24 hours per day (reasonable maximum exposure and central tendency exposure)
- Inhalation of vapors from groundwater (bathing, future only): Exposure time of 1 hour per day (reasonable maximum exposure) and 0.33 hour per day (central tendency exposure)
- Exposure frequency: 350 days per year (reasonable maximum exposure and central tendency exposure)
- Exposure duration: 6 years (reasonable maximum exposure and central tendency exposure)
- Averaging time, noncarcinogens: 2,190 days (reasonable maximum exposure and central tendency exposure); for the inhalation pathway, averaging times were expressed in hours (52,560 hours [reasonable maximum exposure and central tendency exposure])
- Averaging time, carcinogens: 25,550 days (reasonable maximum exposure and central tendency exposure); for the inhalation pathway, averaging times were expressed in hours (613,200 [reasonable maximum exposure and central tendency exposure])

Resident Adolescent (Swimming/Wading Pool)

A resident adolescent (age 6 to 16) was evaluated for intermittent exposure to groundwater during recreational activities such as swimming or playing in a wading pool or sprinklers. Incidental ingestion, dermal contact, and inhalation of vapors from groundwater were evaluated. Quantitative exposure parameters for this scenario include the following assumptions:

- Body weight: 39.4 kg (reasonable maximum exposure and central tendency exposure)
- Ingestion of groundwater (incidental): 0.05 liter per day (reasonable maximum exposure and central tendency exposure)
- Dermal contact with groundwater: Exposure time of 1 hour (reasonable maximum exposure) and 0.5 hour (central tendency exposure); exposed skin surface area of 14,530 cm^2 (entire body). The exposed skin surface area was calculated by averaging the 90th percentile for skin surface areas for male children age 6 to 16 as listed in the *Exposure Factors Handbook* (EPA 1997d). A male child was assumed as it results in a slightly higher surface area (as compared to 14,309 cm^2 for female children of the same age). This approach is conservative.
- Inhalation of vapors in outdoor air: Exposure times of 1 hour per day (reasonable maximum exposure) and 0.5 hours per day (central tendency exposure)
- Exposure frequency: 32 days per year (reasonable maximum exposure) and 16 days per year (central tendency exposure). The exposure frequency assumes activity between mid-May to mid-

September (16 weeks) and two events per week for the reasonable maximum exposure scenario and one event per week for the central tendency exposure scenario.

- Exposure duration: 10 years (reasonable maximum exposure and central tendency exposure)
- Averaging time, noncarcinogens: 3,650 days (reasonable maximum exposure and central tendency exposure); for the inhalation pathway, averaging times were expressed in hours (87,600 hours [reasonable maximum exposure and central tendency exposure])
- Averaging time, carcinogens: 25,550 days (reasonable maximum exposure and central tendency exposure); for the inhalation pathway, averaging times were expressed in hours (613,200 hours [reasonable maximum exposure and central tendency exposure])

6.3.3.2 Industrial Worker

Portions of the LSPGS are active industrial areas. The industrial worker was evaluated for on-site exposure assessment for current and future exposures. Based on site reconnaissance and worker interviews, current industrial facilities in this area use groundwater for various washing activities, either in open air yards or in large (18-wheeler) truck bays. In areas where businesses have been connected to the public water supply (or an alternate source of drinking water is supplied) and impacted groundwater is not currently used as a potable (drinking water) source, a current/future industrial worker was assessed for exposure to COPCs in groundwater via dermal contact with, and inhalation of, volatiles in groundwater during facility cleaning activities.

In areas not connected to the public water supply (or where an alternate source of drinking water is not supplied) and where private water wells may be used as a potable water source, a current/future industrial worker was also assessed for exposure to COPCs in groundwater via ingestion, in addition to the dermal and inhalation pathways.

Quantitative exposure parameters for this scenario include the following assumptions:

- Body weight: 70 kg (reasonable maximum exposure and central tendency exposure)
- Ingestion of groundwater (future only): 1.4 liter per day (reasonable maximum exposure) and 0.7 liter per day (central tendency exposure)
- Dermal contact with groundwater: 3,300 cm^2 exposed skin surface area (head, hands, and forearms; reasonable maximum exposure and central tendency exposure)
- Inhalation of vapors from groundwater in outdoor air: Exposure time of 4 hours per day (reasonable maximum exposure) and 2 hours per day (central tendency exposure)

- Exposure frequency: 250 days per year (reasonable maximum exposure and central tendency exposure; assumes a 5-day work week for 50 weeks of the year)
- Exposure duration: 25 years (reasonable maximum exposure) and 4.5 years (central tendency exposure; representative of the total continuous tenure with an employer for individuals 16 years old and over [U.S. Department of Commerce Statistical Abstract of the United States 1994]).
- Exposure time: 4 hours per day (reasonable maximum exposure) and 2 hours per day (central tendency exposure)
- Averaging time, noncarcinogens: 9,125 days (reasonable maximum exposure); 1,643 days (central tendency exposure); for the inhalation pathway, averaging times were expressed in hours (219,000 hours [reasonable maximum exposure] and 39,420 [central tendency exposure])
- Averaging time, carcinogens: 25,550 days (reasonable maximum exposure and central tendency exposure); for the inhalation pathway, averaging times were expressed in hours (613,200 hours [reasonable maximum exposure and central tendency exposure])

6.3.3.3 Utility/Construction Worker

A future site-specific utility worker/future construction worker scenario was used to assess exposures to contaminated current or future utility trench media (including intruding groundwater volatilizing into trench air) or to media during subsurface support construction. The utility/construction worker was assumed to be exposed during repair or replacement of subsurface utilities, during periods when groundwater may enter the 10-foot deep trench. This receptor was evaluated for exposure to COPCs via incidental ingestion of, dermal contact with, and inhalation of volatiles in groundwater.

Quantitative exposure parameters for this scenario include the following assumptions:

- Body weight: 70 kg (reasonable maximum exposure and central tendency exposure)
- Ingestion of groundwater (incidental): 0.05 liter per day (reasonable maximum exposure and central tendency exposure; represents the default value for the amount of water ingested during swimming)
- Dermal contact with groundwater: 3,300 cm^2 exposed skin surface area (head, hands, and forearms; reasonable maximum exposure and central tendency exposure)
- Exposure frequency: 120 days per year (reasonable maximum exposure) and 60 days (central tendency exposure). The reasonable maximum exposure frequency of 120 days per year is based on professional judgment and is thought to be representative of both (1) a one-time construction job (that is, exposure duration of one year), which includes outdoor tasks or maintenance activities that could take up to 120 days to complete, and (2) up to 120 one-day utility repairs in a year. For the central tendency exposure case, an exposure frequency of 60 days per year (or 2 months) was used.

- Exposure duration: 1 year (reasonable maximum exposure and central tendency exposure)
- Exposure time: 8 hours per day (reasonable maximum exposure and central tendency exposure)
- Averaging time, noncarcinogens: 124 days (reasonable maximum exposure) and 60 days (central tendency exposure); for the inhalation pathway, averaging times were expressed in hours (8,760 hours [reasonable maximum exposure and central tendency exposure])
- Averaging time, carcinogens: 25,550 days (reasonable maximum exposure and central tendency exposure); for the inhalation pathway, averaging times were expressed in hours (613,200 hours [reasonable maximum exposure and central tendency exposure])

6.3.3.4 Recreational Visitor

While it is acknowledged that adult anglers also use the pond, evaluation of an adolescent is more conservative, given that the smaller body weight results in an increased dose (relative to body weight) and thus higher calculated risk. Therefore, the evaluation of the adolescent is also protective of adults. Quantitative exposure parameters for this scenario include the following assumptions:

- Body weight: 39.4 kg (reasonable maximum exposure and central tendency exposure)
- Ingestion of surface water (incidental): 0.05 liter per day (reasonable maximum exposure and central tendency exposure)
- Ingestion of fish: 0.284 kilograms per meal (reasonable maximum exposure) and 0.113 kilograms per meal (central tendency exposure); the fish ingestion frequency was assumed to equal the exposure frequency (i.e., one fish meal per visit).
- Dermal contact with surface water: 8,156 cm^2 exposed skin surface area (reasonable maximum exposure and central tendency exposure)
- Exposure frequency: 32 days per year (reasonable maximum exposure) and 16 days per year (central tendency exposure). Exposure frequency for this activity was estimated as occurring between mid-May to mid-September (16 weeks). For the reasonable maximum exposure scenario, two visits to the pond per week were assumed. The central tendency exposure scenario assumed one visit per week.
- Exposure duration: 10 years (reasonable maximum exposure and central tendency exposure)
- Averaging time, noncarcinogens: 3,650 days (reasonable maximum exposure and central tendency exposure)
- Averaging time, carcinogens: 25,550 days (reasonable maximum exposure and central tendency exposure)

6.4 TOXICITY ASSESSMENT

Once the exposure assessment is complete and the exposure point concentrations are established, toxicity factors are applied following the recommended hierarchy. The most current toxicity information available was used to predict human health risks from exposure to contaminated LSGPS media.

The toxicity assessment for the HHRA included identification of toxicity values used to characterize non-cancer health effects and cancer risk, respectively. Tables M-5.1 through M-6.2 of Appendix M present toxicity values used for the site. Where route-specific toxicity values were not available, route-to-route extrapolations were used to derive toxicity values for organic compounds, consistent with EPA Region 9 conventions for route-to-route extrapolations (see Section 6.4.3).

6.4.1 Noncancer Toxicity Values

Per EPA Region 8 recommendations, the potential for noncancer health effects resulting from exposure to chemicals was assessed by comparing an exposure estimate (intake) with a reference dose or reference concentration for inhalation exposures. Reference doses and reference concentrations represent average daily intakes (expressed as mg/kg-day for reference doses and mg/m^3 for reference concentrations) that are expected to be without appreciable risk of adverse health effects to humans (including sensitive populations) during a lifetime of exposure (for chronic reference doses and reference concentrations) (EPA 1989).

Reference doses and reference concentrations are specific to the chemical, exposure route, and duration. Reference doses were available to assess oral and dermal exposures, while reference concentrations were used for inhalation exposures. For this assessment, oral reference doses were used to assess dermal exposures in the absence of route-specific dermal reference doses (EPA 1989). Oral reference doses are converted to dermal reference doses representing the toxicity of the absorbed dose by multiplying the oral reference doses by an appropriate gastrointestinal absorption factor (as recommended by EPA [2001c]), shown as the “oral absorption efficiency” in the RAGS Part D standard toxicity tables (Table M-5.1 of Appendix M). For the COPCs at LSGPS, adjustment by a gastrointestinal absorption factor was not recommended because organics are generally well absorbed by the gastrointestinal system (greater than 50 percent) (EPA 2001c). Reference doses and reference concentrations are also specific to the duration of exposure: chronic reference doses and reference concentrations are used to evaluate exposures occurring over periods of more than 7 years, and subchronic reference doses and reference concentrations are used to evaluate exposures occurring over periods of 2 weeks to 7 years. Since few subchronic reference doses and reference concentrations are available, this HHRA used reference doses and reference concentrations protective of chronic exposure for all scenarios, regardless of duration. The use of chronic reference doses and reference concentrations to assess risk to the construction worker (exposure duration of 120 days) was conservative.

The following were the primary sources of reference doses and reference concentrations, presented in order of preference:

- EPA's Integrated Risk Information System (IRIS), which is summarized in the Region 9 table of PRGs (EPA 2002f). IRIS is an on-line database that contains EPA-approved reference doses and reference concentrations (EPA 2003c). The reference doses and reference concentrations have undergone extensive review and are recognized as high-quality, agency-wide consensus information.
- Values recommended by the EPA National Center for Environmental Assessment, either listed in the Region 9 table of PRGs (EPA 2002c) as National Center for Environmental Assessment source values or as recommended by the Superfund Technical Support Center within EPA National Center for Environmental Assessment.
- Health Effects Assessment Summary Tables (HEAST), published by EPA (1997c).

Oral and inhalation reference doses were identified for the majority of COPCs at LSGPS. Oral and inhalation reference doses were not available for 1,3-dichloropropane; 2,2-dichloropropane; and cis-1,3-dichloropropene. Therefore, health effects related to exposure to these chemicals could not be quantitatively evaluated, which could lead to an underestimation of risks. The uncertainty associated with the lack of toxicity values for these chemicals is discussed in Section 6.6. Oral and inhalation reference doses for COPCs at LSGPS are presented in Tables M-5.1 and M-5.2 of Appendix M, respectively.

The noncancer reference concentrations are air concentrations expressed as mg/m^3, and are concentrations predicted to result in no adverse health effect in humans, even after chronic exposure to the reference concentration.

6.4.2 Cancer Toxicity Factors

EPA assigns a weight-of-evidence classification rating to chemicals. The weight-of-evidence rating qualitatively describes the likelihood that a chemical is a human carcinogen and is based on an evaluation of the available data from human and animal studies. Chemicals evaluated by EPA since the publication of the 1996 cancer guidelines, *Proposed Guidelines for Carcinogen Risk Assessment* (EPA 1996b), are evaluated using a weight-of-evidence narrative and a descriptor for classifying potential carcinogenicity to humans. One of the descriptors, "known/likely," "cannot be determined," and "not likely," is applied to each chemical. Chemicals evaluated by EPA before the publication of the 1996 guidelines were evaluated in accordance with the 1986 guidelines (EPA 1986). These chemicals were classified using an alphanumeric system in which the chemical was assigned to one of six groups: A, B1, B2, C, D, or E, defined as follows:

Group A:	Known human carcinogen
Group B1 and B2:	Probable human carcinogen
Group C:	Possible human carcinogen
Group D:	Not classifiable as to human carcinogenicity (inadequate or no evidence)
Group E:	Known human noncarcinogen

This earlier alphanumeric system is the one most commonly cited for the VOC COPCs encountered at the LSGPS, and therefore this classification is what is shown in the cancer toxicity value tables (Tables M-6.1 and M-6.2 of Appendix M) for consistency and completeness. EPA has developed slope factors for the oral route and unit risk factor values for the inhalation route for most of the chemicals in weight-of-evidence groups A and B. EPA (1989) defines a slope factor/unit risk factor as a plausible, upper bound estimate of the probability of an individual developing cancer per unit intake of a chemical. Dermal slope factors representing the toxicity of the absorbed dose, are derived by multiplying the oral slope factors by an appropriate gastrointestinal absorption factor (as recommended by EPA [2001c]), shown as the "oral absorption efficiency" in Table M-6.1 of Appendix M. Adjustment by a gastrointestinal oral absorption factor was not recommended for the COPCs at LSGPS, as organic chemicals are generally well-absorbed in the gastrointestinal tract (EPA 2001c).

The same selection hierarchy was used for the selection of carcinogenic toxicity factors (slope factors/unit risk factors) as was used for the noncarcinogenic toxicity factors.

Oral slope factors and inhalation unit risk factors were identified for the majority of carcinogenic (EPA weight-of-evidence groups A, B1, and B2) COPCs at LSGPS. Oral slope factors and inhalation unit risk factors were not available for 1,3-dichloropropane; 2,2-dichloropropane; and cis-1,3-dichloropropene; therefore cancer risks could not be determined for these chemicals. IRIS does not list a weight-of-evidence for these chemicals so it is uncertain as to whether they are carcinogens. A search of the

National Toxicology Program (NTP) Testing Information and Study Results Database (NTP 2003) did not show any studies conducted on these chemicals. The lack of cancer toxicity values for these chemicals may contribute to an underestimation of risk. The uncertainty associated with the lack of toxicity values for these chemicals is presented in Section 6.6. Oral and inhalation slope factors for carcinogenic COPCs at LSGPS are presented in Tables M-6.1 and M-6.2, respectively.

The unit risk factors for carcinogenic risk are expressed as a risk per cubic microgram per cubic meter, which is a risk per chronic air concentration.

6.4.3 Route-to-Route Extrapolation

For some LSGPS COPCs, toxicity values have been developed for oral exposure pathways, but not for inhalation. In these cases, route-to-route extrapolations were performed. This process involves using a toxicity value developed for one route of exposure (such as ingestion) and applying it to another (such as inhalation). Using this approach, it is assumed that the toxicity between the two pathways of exposure is identical. See Section 6.6.3 for a discussion of the uncertainty related to this assumption. When an oral reference dose was available but no inhalation reference concentration was available, the oral reference dose was adopted as the inhalation reference dose. An inhalation reference concentration was calculated by multiplying the inhalation reference dose by 70 kg and then dividing by 20 mg/m^3. This was the case for carbon tetrachloride, cis-1,2-DCE, and trans-1,2-dichloroethene. Similarly, when an inhalation reference dose was available but no oral reference dose was available, the inhalation reference dose was adopted as the oral reference dose. This was the case for chloroethane. The oral slope factor for carbon tetrachloride was route-extrapolated from the inhalation slope factor.

6.4.4 Toxicity Values for LSGPS COPCs

The toxicity values identified for each COPC at the LSGPS are summarized in the following sections. Oral reference dose values and inhalation reference concentration values are presented in Tables 5.1 and 5.2, respectively, of Appendix M. Oral cancer slope factors are presented in Table 6.1 of Appendix M; inhalation unit risk factors are presented in Table 6.2 of Appendix M. Appendix O contains detailed toxicity profiles for the COPCs identified for the HHRA.

6.5 RISK CHARACTERIZATION

Using the equations presented in RAGS Part A (EPA 1989), the risk characterization includes an estimate of the potential excess lifetime cancer risks and a HI characterizing the risk of adverse noncancer health effects attributable to potential exposure to COPCs in site media. The site cancer risk estimates and HIs for potential exposure to the COPCs are presented in Table M-7.1.RME through Table M-7.38.RME of Appendix M for the reasonable maximum exposure scenario for each complete exposure pathway (see Table M-7.1.CTE through Table M-7.38.CTE for the cancer risk estimates and HIs for the central tendency exposure scenario). The tables are in the standardized format specified by the RAGS, Part D (EPA 2001d).

The methodology employed for the risk characterization is presented in Sections 6.5.1 and 6.5.2. Section 6.5.3 summarizes the results of the risk assessment.

6.5.1 Noncancer Health Effects Calculation Methodology

The potential for exposure to result in noncancer adverse health effects was evaluated by comparing the intake estimate with a reference dose or reference concentration. When calculated for a single chemical, this comparison yielded a ratio termed the hazard quotient (HQ) as follows:

$$\textit{For oral and dermal exposure routes, pathway-specific HQ} = \frac{\textit{Intake (mg/kg-day)}}{\textit{RfD (mg/kg-day)}} \qquad (6\text{-}3)$$

and

$$\textit{for the inhalation exposure route, pathway specific HQ} = = \frac{\textit{Cadj (mg/m}^3\textit{)}}{\textit{RfC (mg/m}^3\textit{)}} \qquad (6\text{-}4)$$

Where:

Cadj = Chemical intake concentration adjusted for a specific scenario

RfC = Reference concentration

RfD = Reference dose

To evaluate the potential for noncancer, adverse health effects from exposure to multiple chemicals, the HQs for all chemicals were summed, yielding an HI as follows:

$$\textit{Pathway-specific HI} = \sum \textit{Pathway-specific HQ} \tag{6-5}$$

Pathway-specific HIs were then summed to estimate a total HI for each receptor as follows:

$$\textit{Total HI} = \sum \textit{Pathway-specific HI} \tag{6-6}$$

The total HI reflects an assumption, generally considered to be conservative, that the effects of the different chemicals are additive. When the total HI exceeds 1, further evaluation in the form of a segregation of HI analysis was performed to determine whether noncancer health hazards are a concern at the area (EPA 1989).

6.5.2 Cancer Risk Calculation Methodology

Cancer risks associated with exposure to COPCs classified as carcinogens were characterized as an estimate of the probability (excess risk) that an individual will develop cancer over a 70-year lifetime as a direct result of exposure to potential carcinogens. For example, a cancer risk of 1×10^{-6} indicates that an individual has a 1-in-1 million probability of developing cancer during a lifetime as a result of the assumed exposure conditions.

The National Oil and Hazardous Substances Pollution Contingency Plan (NCP) indicates that for known or suspected carcinogens, acceptable additional cancer risk falls within a range between one person in ten thousand ($1 \text{ x } 10^{-4}$) and one person in a million ($1 \text{ x } 10^{-6}$), known as the risk management range for this risk assessment. Risks below $1 \text{ x } 10^{-6}$ are considered to be insignificant. Risks above $1 \text{ x } 10^{-4}$ may indicate the need for further evaluation or remediation.

For COPCs classified as carcinogens, the cancer risks resulting from exposure to area COPCs were estimated using the following three steps:

First, to derive a cancer risk estimate for a single chemical and pathway, the chemical intake was multiplied by the chemical-specific slope factor as follows:

For the oral and dermal exposure routes, chemical-specific cancer risk (unitless) =

$$\text{Intake (mg/kg-day)} \times \text{SF (mg/kg-day)}^{-1} \qquad (6\text{-}7)$$

and

for the inhalation exposure route, chemical-specific cancer risk (unitless) =

$$\text{Cadj (mg/m}^{3}\text{)} \times \text{URF (mg/m}^{3}\text{)}^{-1} \qquad (6\text{-}8)$$

Where:

Cadj = Chemical intake concentration adjusted for a specific scenario

SF = Slope factor

URF = Unit risk factor

Second, to estimate the cancer risk associated with exposure to multiple carcinogens for a single exposure pathway, the individual chemical-specific cancer risks were assumed to be additive as follows:

$$\text{Pathway-specific Cancer Risk} = \sum \text{Chemical-specific Cancer Risk} \qquad (6\text{-}9)$$

Third, pathway-specific risks were then summed to estimate the total cancer risk as follows:

$$\text{Total Cancer Risk} = \sum \text{Pathway-specific Cancer Risk} \qquad (6\text{-}10)$$

6.5.3 Risk Assessment Results

The risk characterization combines the exposure assessment (and intake calculations) and toxicity evaluation, resulting in the final cancer risks and noncancer HI values for each receptor and pathway. For each receptor, a separate RAGS Part D standard Table 7 (the Table M-7 series in Appendix M) presents combined summaries of cancer and noncancer hazards for each receptor and exposure pathway. For the reasonable maximum exposure results, see Tables M-7.1.RME through M-7.38.RME of Appendix M; the central tendency results are presented in Table M-7.1.CTE through M-7.38.CTE of Appendix M.

Cancer risk probabilities were calculated for all carcinogenic COPCs for each receptor. Cancer risks for the on-site resident adult and child are reported separately; however in keeping with EPA Region 8 practice, a combined cancer risk value (accounting for exposure from age 1 to 6 years of age and 24 subsequent years of exposure) was determined to evaluate a long-term resident. In addition, noncancer health hazards, expressed as HQ for each COPC (resulting in a cumulative HI), were calculated. When evaluating noncancer health hazards, it is important to note that each COPC may act via a different toxicity mechanism to exert its effect on a different target organ. Therefore, cumulative noncancer hazards are most appropriately expressed for each mechanism or target organ (EPA 1989). At the same time, if a cumulative noncancer HI (the sum of all COPC-specific HQs) is less than the total acceptable noncancer hazard benchmark of unity (1.0) then, by definition, all organ-specific HIs are also below 1.0. If the noncancer HI for all COPCs was less than 1.0, then target organ-specific HIs were not calculated or discussed.

6.5.3.1 Area A Source Area

Cancer risks and noncancer health hazards for each receptor in the Area A source area are discussed below. Area A source area residents are connected to the public water supply. Businesses in the Area A source area are connected to the public water supply and/or supply an alternate source of drinking water.

Current/Future Resident Adult (with Public Water Supply) in Area A Source Area

As current residents in this area are using a public water supply rather than well water for potable use, only the risks associated with the inhalation COPCs in indoor air resulting from subsurface vapor intrusion for groundwater were quantified. In addition, possible shorter term exposures for current adult residents (see below) were also quantified.

For the adult residents currently living in the Area A source area, a reasonable maximum exposure cancer risk of 1×10^{-5} was calculated, which falls within the risk management range. For the long-term resident (childhood + adult exposure) a cancer risk of 2×10^{-5} was calculated, which is also within the risk management range. A noncancer HI of 0.5 was calculated, which is below the acceptable threshold of 1.0. Therefore, no unacceptable noncancer risks are presented for the current adult resident. Cancer risks and the noncancer HIs for the current resident adult are presented in Tables M-7.1.RME and M-9.1.RME of Appendix M.

A current/future resident adult was also evaluated for the risk associated with possible nonpotable uses of groundwater in the Area A source area, such as during car washing or lawn irrigation activities. Cancer risks for this additional activity were calculated at 5 x 10^{-5}, which is within the risk management range, and the noncancer HI was calculated to be 0.2, which is below the threshold of 1.0 for noncancer hazards. Therefore, no unacceptable noncancer risks are presented for the current adult resident. Cancer risks and the noncancer HIs for the current resident adult are presented in Tables M-7.3.RME and M-9.3.RME of Appendix M. Note that this pathway was included (despite representing exposures that are more likely to be acute and intermittent, not chronic, in nature) with conservative reasonable maximum exposure assumptions discussed in Section 6.3.3. Central tendency estimates (shown in Tables M-7.3.CTE and M-9.3.CTE of Appendix M) reflect lower risks of 9 x 10^{-6} for cancer and 0.1 for noncancer effects, given that the "average" adult resident may not sustain the exposure predicted for the reasonable maximum exposure case, which represents the "high end of individual risks."

Current/Future Resident Child and Adolescent Wader (with Public Water Supply) in Area A Source Area

Because residences are located in the Area A source area, a resident child (age 1 to 6 years) was also evaluated for exposure to COPCs in groundwater, since this subpopulation is expected to be sensitive based on the smaller body weight relative to estimated exposures. Similar to the resident adult, only inhalation via subsurface vapor intrusion of volatile chemicals from groundwater into indoor air was assessed for the current resident child; older children (adolescents) were evaluated for possible shorter-term exposures (see below).

For the child resident living in the Area A source area, a cancer risk of 3 x 10^{-6} was calculated, which falls within the risk management range. A noncancer HI of 0.5 was calculated, which is below the acceptable threshold of 1.0. Therefore, no unacceptable noncancer risks are presented for the current child resident. Cancer risks and noncancer HIs for the current resident child are presented in Tables M-7.2.RME and M-9.2.RME of Appendix M.

A current/future resident adolescent (age 6 to 16) was also evaluated for the risk associated with nonpotable uses of groundwater in the Area A source area, such as while playing in wading pools or sprinklers. Cancer risks were calculated at 7 x 10^{-5}, which is within the risk management range, and the noncancer HI was calculated to be 0.8, which is below the threshold of 1.0 for noncancer hazards. Therefore, no unacceptable noncancer risks are presented for the current/future resident adolescent. Cancer risks and noncancer HIs for the current resident adolescent playing in a wading pool are presented in Tables M-7.4.RME and M-9.4.RME of Appendix M. Note that this pathway was included (despite representing exposures that are more likely to be acute and intermittent, not chronic, in nature) with conservative reasonable maximum exposure assumptions discussed in Section 6.3.3. Central tendency estimates (shown in Tables M-7.4.CTE and M-9.4.CTE of Appendix M) reflect lower risks of 2 x 10^{-5} for cancer and 0.3 for noncancer effects, given that the "average" adolescent may not sustain the exposure predicted for the reasonable maximum exposure case, which represents the "high end of individual risks."

Current/Future Industrial Worker (with Public/Alternate Water Supply) in Area A Source Area

As industrial activities are conducted at locations throughout the LSGPS site, a current industrial worker in the Area A source area was evaluated for exposure to groundwater during facility cleaning activities, such as vehicle and equipment washing, and routine facility maintenance. The current industrial worker was evaluated for dermal contact and inhalation of volatiles from groundwater to outdoor air. The current industrial worker was assumed to use the public water supply (or an alternate source of drinking water) as a drinking water source. Cancer risks for this receptor were calculated to be 3 x 10^{-4}, which may indicate the need for further evaluation or remediation. The dermal pathway accounted for all of the excess cancer risk with an exposure route total of 3 x 10^{-4}. Cancer risk drivers were PCE (cancer risk of 2 x 10^{-4}) and VC (cancer risk of 5 x 10^{-5}).

The current industrial worker had a calculated HI of 2, which is above the noncancer hazard threshold of 1.0. Health hazards associated with the dermal pathway (HI of 2) contributed the majority of the risk, with PCE (HQ of 1) as the primary driver contributing 50 percent of the risk.

Tables M-7.5.RME and 9.5.RME of Appendix M presents the cancer risk and noncancer HIs for the current industrial worker in the Area A source area. Note that the reasonable maximum exposure case risks calculated for the current industrial worker in the Area A source area assume the conservative exposure durations and frequencies discussed in Section 6.3.3. The uncertainty associated with these assumptions is discussed further in Section 6.6.2. Risks associated with central tendency exposure were lower with a cancer risk of 2 x 10^{-5}, which is within the risk management range, and a HI of 0.3.

Future Resident Adult (without Public Water Supply) in Area A Source Area

To evaluate the possibility that, in the future, residents in the Area A source area may opt to use well water as a potable source for household use (i.e., drinking water and bathing) rather than continue using the public water supply, or if new wells are installed in the future, a future resident was evaluated for direct contact pathways including (1) ingestion of, and dermal contact with, COPCs in groundwater, and (2) inhalation of COPCs in indoor air resulting from the use of groundwater during showering. Risks associated with the inhalation of COPCs in indoor air resulting from subsurface vapor intrusion were also evaluated.

Although these risks are for future hypothetical residents (since all known current residents in the Area A source area obtain their whole-house use water from the public supply), no legally binding prohibitions on the use of existing wells or the installation of future groundwater wells are currently in place. Cancer risks for this receptor were 5 x 10^{-3}, which may indicate the need for further evaluation or remediation. For the long-term resident (childhood + adult exposure) a cancer risk of 8 x 10^{-3} was calculated, which may indicate the need for further evaluation or remediation. Risks for the resident child associated with direct contact with groundwater composed the majority of the risk (5 x 10^{-3}); PCE (1 x 10^{-3} cancer risk) and VC (4 x 10^{-3} cancer risk) for the ingestion pathway.

The HI for the resident adult was calculated to be 20, which is above the noncancer threshold of 1.0. Cis-1,2-DCE, PCE, and VC were the major contributors to the HI total as follows:

- Cis-1,2,-DCE: Ingestion pathway HQ of 6
- PCE: Ingestion pathway HQ of 5; dermal pathway HQ of 3
- VC: Ingestion pathway HQ of 2

Cancer risks and noncancer HIs for the future resident adult are presented in Tables M-7.6.RME and M-9.6.RME of Appendix M.

Since the (hypothetical) future resident adult noncancer HI was greater than 1.0, a target organ-specific analysis was conducted (see Table M-9.6.RME of Appendix M). Only the hematopoetic (or blood) system, which had an HI of 7, and the liver, which had an HI of 10, had a target organ HI greater than 1.0. This indicates that, if a future resident opted to use a shallow groundwater well for whole-house use in the Area A source area, both reasonable maximum exposure cancer and noncancer (and specifically, hematopoetic and liver effects) risks are above limits considered acceptable. Although the "average" adult resident may not sustain the exposure predicted for the reasonable maximum exposure case, which represents the "high end of individual risks," even the "average" or central tendency exposure risks (see Tables M-7.6.CTE and M-9.6.CTE of Appendix M) were 1 x 10^{-3} and 5 for cancer risk and noncancer hazard, respectively.

Future Resident Child (without Public Water Supply) in Area A Source Area

As with the hypothetical future resident adult above, a hypothetical future resident child was also evaluated for exposure to COPCs in groundwater, should well water be used for household purposes in the future. Children were assessed separately as they are expected to be a sensitive subpopulation, based on their size and body weight relative to the exposures estimated.

Cancer risks for the hypothetical future resident child living in a home with whole-house groundwater in use were 3 x 10^{-3}, which may indicate the need for further evaluation or remediation. Risks for the resident child associated with direct contact with groundwater comprised the majority of the risk (3 x 10^{-3}); with PCE (8 x 10^{-4} cancer risk for the ingestion and dermal pathways) and VC (2 x 10^{-3} cancer risk for the ingestion pathway) composing the majority of the excess cancer risk.

The HI for the future resident child was calculated to be 40, which is above the noncancer threshold of 1.0. Cis-1,2-DCE, PCE, TCE, and VC were the major contributors to the HI total as follows:

- Cis-1,2,-DCE: Ingestion pathway HQ of 10; inhalation pathway HQ of 1
- PCE: Ingestion pathway HQ of 10; dermal pathway HQ of 5
- TCE: Ingestion pathway HQ of 2
- VC: Ingestion pathway HQ of 5

Cancer risks and noncancer HIs for the future resident child are presented in Tables M-7.7.RME and M-9.7.RME of Appendix M.

Because this receptor's noncancer HI was greater than 1.0, a target organ-specific analysis was conducted (see Table M-9.7.RME of Appendix M). The liver (HI of 20) and hematopoetic (blood) system (HI of 20) both had target organ HIs above 1.0. These values indicate that, if a future resident opted to use a shallow groundwater well for whole-house use in the Area A source area, children (ages 1 to 6) living in the home would be predicted to have reasonable maximum exposure cancer and noncancer risks above limits considered acceptable. Although the “average” child resident may not sustain the exposure predicted for the reasonable maximum exposure case, which represents the “high end of individual risks,” even the “average” or central tendency exposure risks (see Tables M-7.7.CTE and M-9.7.CTE of Appendix M) were 5 x 10^{-4} and 9 for cancer risk and noncancer hazard, respectively.

Future Industrial Worker (without Pubic/Alternate Water Supply) in Area A Source Area

In addition to the pathways evaluated for the current industrial worker, a future worker in the Area A source area was evaluated for additional exposure via ingestion of groundwater, if well water from the Area A source area was used as a drinking water source. This assumes that the current public water supply (or an alternate supply of drinking water) would not be used for drinking water, and the shallow groundwater would be ingested by workers in the Area A source area. Installation of a new well in the Area A source area is not precluded.

If industrial workers ingested shallow groundwater in the Area A source area in the future, cancer risks for this receptor would be 2 x 10^{-3}, which may indicate the need for further evaluation or remediation. Risks associated with the ingestion (1 x 10^{-3}) and dermal pathways (3 x 10^{-4}) contributed the majority of the risk with PCE (7 x 10^{-4} cancer risk for the ingestion and dermal pathways), TCE (9 x 10^{-6} cancer risk for the dermal and ingestion pathway) and VC (9 x 10^{-4} cancer risk for the ingestion pathway) composing the majority of the excess cancer risk.

The HI for the future industrial worker was calculated to be 9, which is above the noncancer threshold of 1.0. Cis-1,2-DCE, TCE, and VC were the major contributors to the HI total as follows:

- Cis-1,2,-DCE: Ingestion pathway HQ of 3
- PCE: Ingestion pathway HQ of 3; dermal pathway HQ of 1
- VC: Ingestion pathway HQ of 1

Cancer risks and noncancer HIs for the future industrial worker are presented in Tables M-7.8.RME and M-9.8.RME of Appendix M.

Because this receptor’s noncancer HI was greater than 1.0, a target organ-specific analysis was conducted (see Table M-9.8.RME of Appendix M). The liver (HI of 6) and hematopoetic (blood) system (HI of 3) both had target organ HIs above 1.0. These values indicate that, if in the future, employers opted to provide their employees with drinking water from shallow wells in the Area A source area, workers would be predicted to have reasonable maximum exposure cancer and noncancer risks above limits considered acceptable. The “average” future worker may not sustain the exposure predicted for the reasonable maximum exposure case, which represents the “high end of individual risks.” The “average” or central tendency exposure risks (see Tables M-7.8.CTE and M-9.8.CTE of Appendix M) were lower; cancer risks (4×10^{-5}) were within the risk management range with a noncancer HI of 0.8.

Future Utility/Construction Worker in Area A Source Area

A future utility/construction worker was evaluated, should excavation or trenching activities be conducted in the Area A source area. During such activities, if groundwater seeped into an excavation or trench, a utility/construction worker could come into direct contact with COPCs in groundwater.

Cancer risks for this receptor were 7×10^{-6}, which is within the risk management range. The HI for the future utility/construction worker was calculated to be 1.0, which is at the threshold of 1.0 for noncancer effects. Because the total HI for this receptor was not greater than 1.0, no target organ HI was greater than 1.0. Cancer risks and noncancer HIs for the future industrial worker are presented in Tables M-7.9.RME and M-9.9.RME of Appendix M. Note that the reasonable maximum exposure case risks calculated for the current industrial worker in the Area A source area assume the conservative exposure durations and frequencies discussed in Section 6.3.3. The uncertainty associated with these assumptions is discussed further in Section 6.6.2.

The “average” future utility/construction worker may not sustain the exposure predicted for the reasonable maximum exposure case, which represents the “high end of individual risks.” The “average” or central tendency exposure risks (see Tables M-7.9.CTE and M-9.9.CTE of Appendix M) were lower; cancer risks (2×10^{-6}) were within the risk management range with a noncancer HI of 0.3.

6.5.3.2 Area A Nonsource Area

Cancer risks and noncancer health hazards for each receptor in the Area A nonsource area are discussed below.

Current/Future Resident Adult (with Public Water Supply) in Area A Nonsource Area

A current/future resident adult living in a home outside the Area A source area (i.e., the Area A nonsource area) connected to the public water supply was evaluated. Connection to the public water supply assumed that groundwater was not used as a drinking water source; therefore, for these Area A nonsource residents, with public water supply hookups, exposure was assessed only for exposure to COPCs in indoor air resulting from subsurface vapor intrusion. A shorter-term exposure scenario evaluated the nonpotable use of well water by a current resident (see below). However, several residents also have full well use (including ingestion), which was evaluated separately (see below).

For the adult residents connected to the public water supply who currently live in the Area A nonsource area, a reasonable maximum exposure cancer risk of 3×10^{-9} was calculated (inhalation of indoor air), which is considered to be insignificant. For the long-term resident (childhood + adult exposure) a cancer risk of 4×10^{-9} was calculated, which is considered to be insignificant. A noncancer HI of 0.00005, which is below the acceptable threshold of 1.0, was calculated for the resident adult. Therefore, no unacceptable cancer or noncancer risks are presented for the current/future adult resident who is connected to the public water supply. Cancer risks and noncancer HIs for the current resident adult associated with exposure to COPCs in indoor air are included in Tables M-7.13.RME and M-9.13.RME of Appendix M.

A current/future resident adult was evaluated for the risk associated with possible nonpotable uses of groundwater in the Area A nonsource area, such as during car washing or lawn irrigation activities. Cancer risks for this activity were calculated at 2×10^{-7}, which is considered to be insignificant, and noncancer HI was calculated to be 0.0007, which is below the threshold of 1.0 for noncancer hazards. Therefore, no unacceptable cancer or noncancer risks are presented for this current adult resident. Cancer risks and noncancer HIs for the current resident adult are presented in Tables M-7.10.RME and M-9.10.RME of Appendix M.

Current/Future Resident Child and Adolescent Wader (with Public Water Supply) in Area A Nonsource Area

For the child resident currently living at a residence in the Area A nonsource area that is connected to the public water supply, a reasonable maximum exposure cancer risk of 8×10^{-10} (inhalation of indoor air) was calculated, which is considered to be insignificant. A noncancer HI of 0.00005, which is below the acceptable threshold of 1.0, was calculated for the resident child. Therefore, no unacceptable cancer or noncancer risks are presented for the current/future child resident connected to the public water supply. Cancer risks and noncancer HIs for the current resident child associated with exposure to COPCs in indoor air are presented in Tables M-7.14.RME and M-9.14.RME of Appendix M.

A current/future resident adolescent (age 6 to 16) was evaluated for the risks associated with nonpotable uses of groundwater in the Area A nonsource area, such as while playing in wading pools or sprinklers. Cancer risks were calculated at 3 x 10^{-7}, which is considered to be insignificant, and a noncancer HI was calculated to be 0.003, which is below the acceptable threshold of 1.0 for noncancer hazards. Therefore, no unacceptable cancer or noncancer risks are presented for the current/future resident adolescent. Cancer risks and noncancer HIs for the current resident adolescent playing in a wading pool are presented in Tables M-7.11.RME and M-9.11.RME of Appendix M.

Current/Future Industrial Worker (with Public/Alternate Water Supply) in Area A Nonsource Area

A current/future industrial worker in the Area A nonsource area was evaluated for exposure to groundwater during facility cleaning activities, such as vehicle and equipment washing, and routine facility maintenance. This current/future industrial worker was evaluated for dermal contact and inhalation of volatiles from groundwater to outdoor air. This receptor was assumed to use the public water supply (or an alternate source of drinking water) as a drinking water source. Cancer risks for this receptor were calculated to be 8 x 10^{-7}, which is considered to be insignificant, and the noncancer HI was calculated to be 0.006, which is below the noncancer hazard threshold of 1.0. Therefore, no unacceptable cancer or noncancer risks are presented for the current/future industrial worker who uses the public water supply or an alternate source of drinking water. Tables M-7.12.RME and 9.12-RME of Appendix M presents the cancer risk and noncancer HIs for the current/future industrial worker in the Area A nonsource area.

Current/Future Resident Adult (without Public Water Supply) in Area A Nonsource Area

Because all residences in the Area A nonsource area are not currently connected to the public water supply and to address those residents that may opt to use well water or install a new well to serve as a source of potable water for household use (i.e., drinking water and bathing), a current/future resident (not connected to the public water supply) was evaluated for direct contact pathways including ingestion of and dermal contact with COPCs in groundwater, and inhalation of COPCs in indoor air resulting from the use of groundwater during showering. Risks associated with the inhalation of COPCs in indoor air resulting from subsurface vapor intrusion were also evaluated.

Cancer risks for this receptor were 4 x 10^{-5}, which is within the risk management range. For the long-term resident (childhood + adult exposure) a cancer risk of 7 x 10^{-5} was calculated, which is also within the risk management range. The HI for the resident adult was calculated to be 0.1, which is below the noncancer threshold of 1.0. Therefore, no unacceptable noncancer risks are presented for the current/future residential adult. Cancer risks and noncancer HIs for this current/future resident adult are presented in Tables M-7.13.RME and M-9.13.RME of Appendix M.

Current/Future Resident Child (without Public Water Supply) in Area A Nonsource Area

A current/future resident child was also evaluated for exposure to COPCs in groundwater in Area A nonsource area, at residences where groundwater is used (or may be used in the future) for whole-house use. Children were assessed separately since they are expected to be a sensitive subpopulation, based on their size and body weight relative to the exposures estimated.

Cancer risks for the current/future resident child living in a home with whole-house groundwater use were 3 x 10^{-5}, which is within the risk management range. The HI for this current/future resident child was calculated to be 0.3, which is below the noncancer threshold of 1.0. Therefore, no unacceptable noncancer risks are presented for the current/future residential child. Cancer risks and noncancer HIs for this current/future resident child are presented in Tables M-7.14.RME and M-9.14.RME of Appendix M.

Current/Future Industrial Worker (without Public/Alternate Water Supply) in Area A Nonsource Area

Because not all businesses in the Area A nonsource area are connected to the public water supply (and may not supply an alternate source of drinking water), a current/future worker was evaluated for additional exposure via ingestion of groundwater, if well water from the Area A nonsource area was used as a drinking water source. Dermal contact with groundwater and inhalation of volatiles in outdoor air were also evaluated.

If industrial workers ingested shallow groundwater in the Area A nonsource area, cancer risks for this receptor would be 1 x 10^{-5}, which is within the risk management range. The HI for the current/future industrial worker was calculated to be 0.06, which is below the noncancer threshold of 1.0. Therefore, no unacceptable noncancer risks are presented for the current/future industrial worker. Cancer risks and noncancer HIs for the future industrial worker are presented in Tables M-7.15.RME and M-9.15.RME of Appendix M.

Future Utility/Construction Worker in Area A Nonsource Area

A future utility/construction worker was evaluated should excavation or trenching activities be conducted in the Area A nonsource area. During such activities, should groundwater seep into an excavation or trench, a utility/construction worker could come into direct contact with COPCs in groundwater. Cancer risks for this receptor were 2×10^{-8}, which is considered to be insignificant. The HI for the future utility/construction worker was calculated to be 0.004, which is below the threshold of 1.0 for noncancer effects. Therefore, no unacceptable cancer or noncancer risks are presented for the future construction worker. Cancer risks and noncancer HIs for the future industrial worker are presented in Tables M-7.16.RME and M-9.16.RME of Appendix M.

6.5.3.3 Area B Source Area

Cancer risks and noncancer health hazards for each receptor in the Area B source area are discussed below. Area B source area residences and businesses are currently connected to the public water supply, or in the case of businesses, supply an alternate drinking water source.

Current/Future Resident Adult (with Public Water Supply) in Area B Source Area

A current resident adult in the Area B source area was evaluated for risks associated with the inhalation of COPCs in indoor air resulting from subsurface vapor intrusion from groundwater. In addition, possible shorter-term exposures for current adult residents were investigated (see below).

For the adult residents currently living in the Area B source area, a reasonable maximum exposure cancer risk of 5×10^{-7} was calculated, which is considered to be insignificant. For the long-term resident (childhood + adult exposure) a cancer risk of 6×10^{-7} was calculated, which is considered to be insignificant. A noncancer HI of 0.05, which is below the acceptable threshold of 1.0, was calculated for the resident adult. Therefore, no unacceptable cancer or noncancer risks associated with the inhalation of indoor air are presented for the current adult resident. Cancer risks and noncancer HIs associated with vapor intrusion to indoor air for the current resident adult are presented in Tables M-7.20.RME and M-9.20.RME of Appendix M.

A current/future resident adult was evaluated for the risk associated with possible nonpotable uses of groundwater in the Area B source area, such as during car washing or lawn irrigation activities. Cancer risks for this activity were calculated at 2 x 10^{-6}, which is within the risk management range. The noncancer HI was calculated to be 0.1, which is below the threshold of 1.0 for noncancer hazards. Therefore, no unacceptable noncancer risks are presented for the current/future adult resident. Cancer risks and noncancer HIs for the current/future resident adult are presented in Tables M-7.17.RME and M-9.17.RME of Appendix M.

Current/Future Resident Child and Adolescent Wader (with Public Water Supply) in Area B Source Area

A current resident child in the Area B source area was also evaluated for risks associated with the inhalation of COPCs in indoor air resulting from subsurface vapor intrusion from groundwater. A shorter-term recreational use (playing in a wading pool) scenario for an adolescent was also evaluated (see below). These scenarios assume public water supply for drinking water but outdoor seasonal recreational use of contaminated well water.

For the child residents currently living in the Area B source area, a reasonable maximum exposure cancer risk of 5 x 10^{-7} was calculated, which is considered to be insignificant. A noncancer HQ of 0.05, which is below the acceptable threshold of 1.0, was calculated for the resident child. Therefore, no unacceptable cancer or noncancer risks associated with the inhalation of indoor air are presented for the current child resident. Cancer risks and noncancer HIs associated with vapor intrusion to indoor air for the current resident adult are included in Tables M-7.21.RME and M-9.21.RME of Appendix M.

A current/future resident adolescent (age 6 to 16) was evaluated for the risk associated with nonpotable uses of groundwater in the Area B source area, such as while playing in wading pools or sprinklers. Cancer risks were calculated at 3 x 10^{-6}, which is within the risk management range. Noncancer HI was calculated to be 0.4, which is below the threshold of 1.0 for noncancer hazards. Therefore, no unacceptable noncancer risks are presented for the current/future resident adolescent. Cancer risks and noncancer HIs for the current/future resident adolescent playing in a wading pool are presented in Tables M-7.18.RME and M-9.18.RME of Appendix M.

Current/Future Industrial Worker (with Public Water Supply) in Area B Source Area

Because industrial activities are conducted at locations throughout the LSGPS site, a current industrial worker in the Area B source area was evaluated for exposure to groundwater during facility cleaning activities, such as vehicle and rack washing, and routine facility maintenance. The current industrial worker was evaluated for dermal contact and inhalation of volatiles from groundwater to outdoor air. This receptor was assumed to use the public water supply as a drinking water source (or use an alternate supply of drinking water). Cancer risks for this receptor were calculated to be 2 x 10^{-5}, which is within the risk management range. A noncancer HI was calculated to be 0.8, which is below the noncancer hazard threshold of 1.0. Therefore, no unacceptable noncancer risks are presented for the current industrial worker. Cancer risks and noncancer HIs for the current industrial worker are presented in Tables M-7.19.RME and M-9.19.RME of Appendix M.

Future Resident Adult in Area B Source Area

To evaluate the possibility that, in the future, residents in the Area B source area may opt to use well water as a potable source for household use (i.e., drinking water and bathing) rather than continue using the public water supply, a future resident was evaluated for direct contact pathways including ingestion of and dermal contact with COPCs in groundwater, and inhalation of COPCs in indoor air resulting from the use of groundwater during showering. Risks associated with the inhalation of COPCs in indoor air resulting from subsurface vapor intrusion were also evaluated.

Although these risks are for future residents are hypothetical (since all known current residents in the Area B source area obtain their whole-house use water from the public supply), no legally binding prohibitions on the use of existing wells or the installation of future groundwater wells are currently in place. Cancer risks for this receptor were 2 x 10^{-4}, which may indicate the need for further evaluation or remediation. For the long-term resident (childhood + adult exposure) a cancer risk of 4 x 10^{-4} was calculated, which may indicate the need for further evaluation or remediation. Risks for the adult resident associated with direct contact with groundwater composed the majority of the risk (2 x 10^{-4}); with TCE (2 x 10^{-4} cancer risk for the ingestion pathway) composing the majority of the excess cancer risk.

The HI for the resident adult was calculated to be 20, which is above the noncancer threshold of 1.0. Cis-1,2-DCE and TCE were the major contributors to the HI total as follows:

- Cis-1,2,-DCE: Ingestion pathway HQ of 4
- TCE: Ingestion pathway HQ of 8; inhalation of groundwater vapors pathway HQ of 1; dermal pathway HQ of 1

Cancer risks and noncancer HIs for the future resident adult are presented in Tables M-7.20.RME and M-9.20.RME of Appendix M.

Since the hypothetical future resident adult noncancer HI was greater than 1.0, a target organ-specific analysis was conducted (see Table M-9.20.RME of Appendix M). Only the hematopoetic (or blood) system, which had an HI of 5, and the liver, which had and HI of 10, had target organ HIs greater than 1.0. These values indicate that, if a future resident opted to use a shallow groundwater well for whole-house use in the Area B source area, both reasonable maximum exposure cancer and noncancer (and specifically, hematopoetic and liver) risks are above limits considered acceptable. The “average” or central tendency exposure cancer risk (see Tables M-7.20.CTE and M-9.20.CTE of Appendix M) was within the risk management range, at 5 x 10^{-5}; however, the noncancer HI of 3 was above the threshold of 1.0.

Future Resident Child in Area B Source Area

A hypothetical future resident child was also evaluated for exposure to COPCs in groundwater in Area B source area, should well water be used for household use in the future. Children were assessed separately since they are expected to be a sensitive subpopulation, based on their size and body weight relative to the exposures estimated.

Cancer risks for the hypothetical future resident child living in a home with whole-house groundwater in use were 1 x 10^{-4}, which is at the upper bound of the risk management range. The HI for the resident child was calculated to be 30, which is above the noncancer threshold of 1.0. Cis-1,2-DCE and TCE were the major contributors to the HI total as follows:

- Cis-1,2,-DCE: Ingestion pathway HQ of 9
- TCE: Ingestion pathway HQ of 20; inhalation of groundwater vapors pathway HQ of 2; dermal pathway HQ of 2

Cancer risks and noncancer HIs for the future resident child are presented in Tables M-7.21.RME and M-9.21.RME of Appendix M.

Since the hypothetical future resident child noncancer HI was greater than 1.0, a target organ-specific analysis was conducted (see Table M-9.21.RME of Appendix M). Only the hematopoetic (or blood) system, which had an HI of 10, and the liver, which had and HI of 20, had target organ HIs greater than 1.0. These values indicate that, if a future resident opted to use a shallow groundwater well for whole-house use in the Area B source area, both reasonable maximum exposure cancer and noncancer (and specifically, hematopoetic and liver) risks are above limits considered acceptable. Although the "average" child resident may not sustain the exposure predicted for the reasonable maximum exposure case, which represents the "high end of individual risks," even the "average" or central tendency exposure noncancer HI of 5 (see Tables M-7.21.CTE and M-9.21.CTE of Appendix M) was above the threshold of 1.0; however, the central tendency exposure cancer risk of 2 x 10^{-5} was within the risk management range.

Future Industrial Worker in Area B Source Area

In addition to the pathways evaluated for the current industrial worker, a future worker was evaluated for additional exposure via ingestion of groundwater, if well water from the Area B source area was used as a drinking water source. This assumes that the current public water supply (or other potable water source) would not be used for drinking water, and the shallow groundwater would be ingested by workers in the Area B source area.

If industrial workers ingested shallow groundwater in the Area B source area in the future, cancer risks for this receptor would be 1 x 10^{-4}, which is at the upper bound of the risk management range. The HI for the future industrial worker was calculated to be 7, which is above the noncancer threshold of 1.0. Cis-1,2-DCE and TCE were the major contributors to the HI total as follows:

- Cis-1,2,-DCE: Ingestion pathway HQ of 2
- TCE: Ingestion pathway HQ of 4

Cancer risks and noncancer HIs for the future industrial worker are presented in Tables M-7.22.RME and M-9.22.RME of Appendix M.

Since the hypothetical future industrial worker noncancer HI was greater than 1.0, a target organ- specific analysis was conducted (see Table M-9.20.RME of Appendix M). Only the hematopoetic (or blood) system, which had an HI of 2, and the liver, which had and HI of 5, had target organ HIs greater than 1.0. These values indicate that, if a future industrial worker opted to use a shallow groundwater well for use in the Area B source area, both reasonable maximum exposure cancer and noncancer (and specifically, hematopoetic and liver) risks are above limits considered acceptable. Although the "average" industrial worker may not sustain the exposure predicted for the reasonable maximum exposure case, which represents the "high end of individual risks," the "average" or central tendency exposure risks (see Tables M-7.22.CTE and M-9.22.CTE of Appendix M) were within the risk management range for cancer risks at 3×10^{-6} and below the noncancer threshold 0.4.

<u>Future Utility/Construction Worker in Area B Source Area</u>

A future utility/construction worker was evaluated, should excavation or trenching activities be conducted in the Area B source area. During such activities, should groundwater seep into an excavation or trench, a utility/construction worker could come into direct contact with COPCs in groundwater. Cancer risks for this receptor were 4×10^{-7}, which is considered to be insignificant, and noncancer HI was calculated to be 0.6, which is below the threshold of 1.0 for noncancer hazards. Therefore, no unacceptable cancer or noncancer risks are presented for the future construction worker. Cancer risks and noncancer HIs for the future industrial worker are presented in Tables M-7.23.RME and M-9.23.RME of Appendix M.

6.5.3.4 Area B Nonsource Area

Cancer risks and noncancer health hazards for each receptor in the Area B nonsource area are discussed below. One residence (RW003) in the Area B nonsource area uses well water; no businesses currently use well water for drinking. Both residential and industrial outdoor well water use occurs in the Area B nonsource area.

<u>Current/Future Resident Adult (with Public Water Supply) in Area B Nonsource Area</u>

For those residences connected to the public water supply, a current/future resident adult in the Area B nonsource area was evaluated for risks associated with the inhalation of COPCs in indoor air resulting from subsurface vapor intrusion from groundwater. In addition, possible shorter-term exposures for current/future adult residents were investigated (see below).

For the adult residents currently living in the Area B nonsource area, a reasonable maximum exposure cancer risk of 1×10^{-9} was calculated, which is considered to be insignificant. For the long-term resident (childhood + adult exposure) a cancer risk of 2×10^{-9} was calculated, which is considered to be insignificant. A noncancer HI of 0.0001, which is below the acceptable threshold of 1.0, was calculated for the resident adult. Therefore, no unacceptable cancer or noncancer risks are presented for this current/future adult resident. Cancer risks and noncancer HIs associated with vapor intrusion to indoor air for this current/future resident adult are included in Tables M-7.27.RME and M-9.27.RME of Appendix M.

A current/future resident adult was evaluated for the risk associated with possible nonpotable uses of groundwater in the Area B nonsource area, such as during car washing or lawn irrigation activities. Cancer risks for this activity were calculated at 1×10^{-8}, which is considered to be insignificant, and noncancer HI was calculated to be 0.0003, which is below the threshold of 1.0 for noncancer hazards. Therefore, no unacceptable cancer or noncancer risks are presented for the current/future adult resident. Cancer risks and noncancer HIs for the current/future resident adult are presented in Tables M-7.24.RME and M-9.24.RME of Appendix M.

Current Resident Child and Adolescent Wader (with Public Water Supply) in Area B Nonsource Area

A current resident child in the Area B nonsource area was also evaluated for risks associated with the inhalation of COPCs in indoor air resulting from subsurface vapor intrusion from groundwater, assuming no household use of groundwater as a potable water source. In addition, possible shorter-term exposures for a current adolescent resident were investigated (see below).

For the child resident currently living in the Area B nonsource area, a reasonable maximum exposure cancer risk of 4×10^{-10} was calculated, which is considered to be insignificant. A noncancer HI of 0.0001, which is below the acceptable threshold of 1.0, was calculated for the resident child. Therefore, no unacceptable cancer or noncancer risks are presented for the current child resident. Cancer risks and noncancer HIs associated with vapor intrusion to indoor air for the current/future resident child are included in Tables M-7.28.RME and M-9.28.RME of Appendix M.

A current/future resident adolescent (age 6 to 16) was evaluated for the risk associated with nonpotable uses of groundwater in the Area B nonsource area, such as while playing in wading pools or sprinklers.

Cancer risks were calculated at 2 x 10^{-8}, which is considered to be insignificant, and a noncancer HI was calculated to be 0.001, which is below the threshold of 1.0 for noncancer hazards. Therefore, no unacceptable cancer or noncancer risks are presented for the current child resident. Cancer risks and noncancer HIs for the current/future resident adolescent playing in a wading pool are presented in Tables M-7.25.RME and M-9.25.RME of Appendix M.

Current Industrial Worker (with Public/Alternate Water Supply) in Area B Nonsource Area

A current/future industrial worker in the Area B nonsource area was evaluated for exposure to groundwater during facility cleaning activities, such as vehicle and equipment washing, and routine facility maintenance. This receptor was assumed to use the public water supply (or an alternate supply of potable water) as a drinking water source. This current/future industrial worker was evaluated for dermal contact and inhalation of volatiles from groundwater to outdoor air. Cancer risks for this receptor were calculated to be 1 x 10^{-7}, which is considered to be insignificant, and a noncancer HI was calculated to be 0.002, which is below the noncancer hazard threshold of 1.0. Therefore, no unacceptable cancer or noncancer risks are presented for this current/future industrial worker. Tables M-7.26.RME and 9.26-RME of Appendix M presents the cancer risk and noncancer HIs for this current/future industrial worker in the Area B nonsource area.

Current/Future Resident Adult (without Public Water Supply) in Area B Nonsource Area

To evaluate those Area B nonsource area residents who are not connected to the public water supply or may opt to use well water as a potable source for household use (i.e., drinking water and bathing) rather than continue using the public water supply, a current/future resident was evaluated for direct contact pathways including ingestion of and dermal contact with COPCs in groundwater, and inhalation of COPCs in indoor air resulting from the use of groundwater during showering. Risks associated with the inhalation of COPCs in indoor air resulting from subsurface vapor intrusion were also evaluated.

Cancer risks for this receptor were 1 x 10^{-6}, which is at the lower bound of the risk management range. For the long-term resident (childhood + adult exposure) a cancer risk of 2 x 10^{-6} was calculated, which is within the risk management range. The HI for the resident adult was calculated to be 0.04, which is below the noncancer threshold of 1.0. Therefore, no unacceptable noncancer risks are presented for this future resident adult. The cancer risks and noncancer HIs for this resident adult are presented in Tables M-7.27.RME and M-9.27.RME of Appendix M.

Current/Future Resident Child (without Public Water Supply) in Area B Nonsource Area

A current/future resident child was also evaluated for exposure to COPCs in groundwater, should well water be used for household use in the future. Children were assessed separately since they are expected to be a sensitive subpopulation, based on their size and body weight relative to the exposures estimated.

Cancer risks for current/future resident child living in a home with whole-house groundwater in use were 7 x 10^{-7}, which is considered to be insignificant. The HI for the resident child was calculated to be 0.08, which is below the noncancer threshold of 1.0. Therefore, no unacceptable cancer or noncancer risks are presented for this resident child. Cancer risks and noncancer HIs for this resident child are presented in Tables M-7.28.RME and M-9.28.RME of Appendix M.

Future Industrial Worker (without Public Water Supply) in Area B Nonsource Area

In the future, should groundwater in the Area B nonsource area be used as a drinking water source (assumes an alternate drinking water supply is not provided), a future worker was evaluated for additional exposure via ingestion of groundwater.

If industrial workers ingest shallow groundwater in the Area B nonsource area, cancer risks for this receptor are 7 x 10^{-7}, which is considered to be insignificant, noncancer HQ was calculated to be 0.02, which is below the noncancer threshold of 1.0. Therefore, no unacceptable cancer or noncancer risks are presented for the current/future industrial worker. Cancer risks and noncancer HIs for this industrial worker are presented in Tables M-7.29.RME and M-9.29.RME of Appendix M.

Future Utility/Construction Worker in Area B Nonsource Area

A future utility/construction worker was evaluated should excavation or trenching activities be conducted in the Area B nonsource area. During such activities, should groundwater seep into an excavation or trench, a utility/construction worker could come into direct contact with COPCs in groundwater.

Cancer risks for this receptor were 3 x 10^{-9}, which is considered to be insignificant. The HI for the future utility/construction worker was calculated to be 0.002, which is below the threshold of 1.0 for noncancer effects. Therefore, no unacceptable cancer or noncancer risks are presented for the future construction worker. Cancer risks and noncancer HIs for the future industrial worker are presented in Tables M-7.30.RME and M-9.30.RME of Appendix M.

6.5.3.5 Area C

Cancer risks and noncancer health hazards for each receptor in Area C are discussed below. Because residents in Area C (such as RW027) use well water, whole-house exposures are possible for some residents, but not others (who are connected to the public water supply). Therefore, two separate analyses were conducted. Further, businesses in Area C use well water.

Current/Future Resident Adult (on Public Water Supply) in Area C

A current/future resident adult living in Area C was evaluated for risks associated with the inhalation of COPCs in indoor air resulting from subsurface vapor intrusion from groundwater, assuming no household use of groundwater as a potable water source. In addition, possible shorter-term exposures to groundwater from nonpotable use for current/future adult residents were investigated (see below).

For these adult residents living in Area C, a reasonable maximum exposure cancer risk of 8×10^{-10} was calculated, which is considered to be insignificant. For the long-term resident (childhood + adult exposure) a cancer risk of 1×10^{-9} was calculated, which is considered to be insignificant. A noncancer HI of 0.00007, which is below the acceptable threshold of 1.0, was calculated for the resident adult. Therefore, no unacceptable cancer or noncancer risks are presented for this current/future adult resident. Cancer risks and noncancer HIs associated with vapor intrusion to indoor air for this current/future adult are included in Tables M-7.34.RME and M-9.34.RME of Appendix M.

A current/future resident adult was evaluated for the risk associated with possible nonpotable uses of groundwater in Area C, such as during car washing or lawn irrigation activities. Cancer risks for these activities were calculated at 1×10^{-8}, which is considered to be insignificant. The noncancer HI was calculated to be 0.0002, which is below the threshold of 1.0 for noncancer hazards. Therefore, no unacceptable cancer or noncancer risks are presented for the current adult resident. Cancer risks and noncancer HIs associated with outdoor use of groundwater for the current/future resident adult are presented in Tables M-7.31.RME and M-9.31.RME of Appendix M. Note that this pathway was included (despite representing exposures that are more likely to be acute and intermittent, not chronic, in nature) with conservative reasonable maximum exposure assumptions discussed in Section 6.3.3.

Current Resident Child and Adolescent Swimmer (with Public Water Supply) in Area C

Similar to the resident adult above, a current/future resident child living in Area C was also evaluated for risks associated with the inhalation of COPCs in indoor air resulting from subsurface vapor intrusion from groundwater, assuming no household use of groundwater as a potable water source. In addition, possible shorter-term exposures for current an adolescent resident were investigated (see below).

For the child resident living in Area C whose only exposure is passive inhalation, a reasonable maximum exposure cancer risk of 2 x 10^{-10} was calculated, which is considered to be insignificant. A noncancer HI of 0.00007, which is below the acceptable threshold of 1.0, was calculated for the resident child. Therefore, no unacceptable cancer or noncancer risks are presented for this child resident. Cancer risks and noncancer HIs associated with vapor intrusion to indoor air for this current/future resident child are included in Tables M-7.35.RME and M-9.35.RME of Appendix M.

A current/future resident adolescent (age 6 to 16) was evaluated for the risk associated with nonpotable uses of groundwater in Area C, such as while playing in wading pools or sprinklers. Cancer risks were calculated at 2 x 10^{-8}, which is considered to be insignificant. The noncancer HI was calculated to be 0.0008, which is below the threshold of 1.0 for noncancer hazards. Therefore, no unacceptable cancer or noncancer risks are presented for the current/future adolescent playing in wading pool. Cancer risks and noncancer HIs for the current/future resident adolescent playing in a wading pool are presented in Tables M-7.32.RME and M-9.32.RME of Appendix M. Note that this pathway was included (despite representing exposures that are more likely to be acute and intermittent, not chronic, in nature) with conservative reasonable maximum exposure assumptions discussed in Section 6.3.3.

Current/Future Industrial Worker (with Public Water Supply) in Area C

As industrial activities are conducted at locations throughout the LSGPS site, a current industrial worker in Area C was evaluated for exposure to groundwater during facility cleaning activities, such as vehicle and rack washing, and routine facility maintenance. The current industrial worker was evaluated for dermal contact and inhalation of volatiles from groundwater to outdoor air. This receptor was assumed to use the public water supply as a drinking water source (or an alternate source of drinking water is supplied).

Cancer risks for this receptor were calculated to be 1 x 10^{-7}, which is considered to be insignificant. The noncancer HI for this current/future industrial worker was calculated to be 0.002, which below threshold of 1.0 for noncancer effects. Therefore, no unacceptable cancer or noncancer risks are presented for this industrial worker receptor. Cancer risks and noncancer HIs for the current/future industrial worker in Area C are presented in Tables M-7.33.RME and M-9.33.RME of Appendix M. Note that the reasonable maximum exposure case risks calculated for this industrial worker in Area C assume the conservative exposure durations and frequencies discussed in Section 6.3.3. The uncertainty associated with these assumptions is discussed further in Section 6.6.2.

Current/Future Resident Adult (without Public Water Supply) in Area C

For those residents in Area C that are not connected to the public water supply or who may opt to use well water as a potable source for household use (i.e., drinking water and bathing) rather than continue using the public water supply, a current/future resident was evaluated for direct contact pathways including ingestion of and dermal contact with COPCs in groundwater, and inhalation of COPCs in indoor air resulting from the use of groundwater during showering. Risks associated with the inhalation of COPCs in indoor air resulting from subsurface vapor intrusion were also evaluated.

Cancer risks for this receptor were 1 x 10^{-6}, which at the upper bound of the risk management range. For the long-term resident (childhood + adult exposure) a cancer risk of 2 x 10^{-6} was calculated, which is within the risk management range. The noncancer HI for the resident adult was calculated to be 0.03, which is below the noncancer threshold of 1.0. Therefore, no unacceptable noncancer risks are presented for the current/future adult resident in Area C. Cancer risks and noncancer HIs for this resident adult are presented in Tables M-7.34.RME and M-9.34.RME of Appendix M.

Current/Future Resident Child (without Public Water Supply) in Area C

A current/future resident child was also evaluated for exposure to COPCs in groundwater, should well water be used for household use in the future. Children were assessed separately as they are expected to be a sensitive subpopulation, based on their size and body weight relative to the exposures estimated.

Cancer risks for the resident child living in a home with whole-house groundwater use were 5 x 10^{-7}, which is considered to be insignificant. The noncancer HI for the resident child was calculated to be 0.06, which is below the noncancer threshold of 1.0. Therefore, no unacceptable cancer or noncancer risks are presented for the child resident. Cancer risks and noncancer HIs for the future resident child are presented in Tables M-7.35.RME and M-9.35.RME of Appendix M.

Current/Future Industrial Worker (without Public/Alternate Water Supply) in Area C

To evaluate those workers who work at facilities that are not connected to the public water supply and where no alternate source of drinking water is supplied, a worker was evaluated for additional exposure via ingestion of groundwater in Area C. This assumes that the public water supply would not be used (and no alternate drinking water supplied) for drinking water, and the shallow groundwater would be ingested by workers in Area C.

Cancer risks for this receptor were 5×10^{-7}, which is considered to be insignificant. The noncancer HI for the future industrial worker was calculated to be 0.01, which is below an acceptable hazard threshold of 1.0 for noncancer effects. Therefore, no unacceptable cancer or noncancer risks are presented for the industrial worker. Cancer risks and noncancer HIs for the current/future industrial worker are presented in Tables M-7.36.RME and M-9.36.RME of Appendix M.

Future Utility/Construction Worker in Area C

A future utility/construction worker was evaluated should excavation or trenching activities be conducted in Area C. During such activities, should groundwater seep into an excavation or trench, a utility/construction worker could come into direct contact with COPCs in groundwater.

Cancer risks for this receptor were 2×10^{-9}, which is considered to be insignificant. The noncancer HI for the future utility/construction worker was calculated to be 0.001, which is below the acceptable hazard threshold of 1.0 for noncancer effects. Therefore, no unacceptable cancer or noncancer risks are presented for the future utility/construction worker. Cancer risks and noncancer HIs for the future industrial worker are presented in Tables M-7.37.RME and M-9.37.RME of Appendix M.

6.5.3.6 Surface Water – AJ Gravel Pond

A current/future adolescent recreator was evaluated for the risk associated with possible recreational uses of surface water in AJ Gravel pond, such as during dermal exposure or incidental ingestion of surface water during wading or ingestion of contaminated fish tissue. Cancer risks for this activity were calculated at 1 x 10^{-5}, which is within the risk management range. Risks associated with ingestion of fish tissue composed the majority of the risk (1 x 10^{-5}). The noncancer HI was calculated to be 0.2, which is below the acceptable hazard threshold of 1.0 for noncancer hazards. Therefore, no unacceptable noncancer risks are presented for the current/future adolescent recreator. Cancer risks and noncancer HIs for the current/future recreator adolescent are presented in Tables M-7.38.RME and M-9.38.RME of Appendix M.

6.5.3.7 Coulson Irrigation Ditch Surface Water

As noted in Section 6.2.2.3 above, TCE was retained as a COPC in surface water at the Coulson irrigation ditch. TCE was detected in one sample only at a concentration (2.6 µg/L) that exceeded the Region 9 PRG (1.6 µg/L); the remaining samples were all below the Region 9 PRG. All detected concentrations were below the MCL of 5 µg/L. The Coulson irrigation ditch area is not an area expected to be frequented by recreational visitors. Hypothetically, if the Coulson irrigation ditch water was consumed at a rate of 2 L per day for 30 years, the maximum projected risk would be 1.6 x 10^{-6}, which is within the risk management range. However, because of the nature of the drainage ditch, it is unlikely that Coulson irrigation ditch water would be consumed or that a receptor would have prolonged, chronic contact with surface water in the Coulson irrigation ditch. Reportedly, the ditch is dredged 1 to 2 times per year, however, there is minimal direct contact with surface water because of the machinery used to perform the work. For these reasons, direct contact with surface water in the Coulson irrigation ditch should present minimal risk based on this infrequent (not chronic) intermittent exposure.

6.5.4 Risk Characterization Summary

In all, Table 6-4 presents the risks by receptor and by medium (and subarea, where applicable).

The human health risk assessment concluded that the following scenarios are within the risk management range or considered to be insignificant risks under the NCP:

- Resident adults and children in each of the subareas, assuming only indoor air inhalation exposures as a result of subsurface vapor intrusion.
- Resident adults who use contaminated well water to wash cars or irrigate their lawn in each of the subareas.

- Resident adolescents who recreate with contaminated well water in kiddie pools or sprinklers in each of the subareas.

- Recreators who fish from or wade/dip their arms in the AJ Gravel pond.

- Utility/construction workers in any of the subareas.

- Industrial workers in Area A nonsource, Area B source, Area B nonsource, and Area C subareas who use the public water supply or are supplied an alternate source of drinking water.

- Resident adults and children in Area A nonsource. Area B nonsource, and Area C subareas who use groundwater as a potable water source for whole-house use and/or drinking water source.

- Industrial workers in Area A nonsource. Area B nonsource, and Area C subareas who use groundwater as a potable water source for interior use and/or drinking water source.

Those scenarios and receptors that had cancer risks that may indicate the need for further evaluation or remediation are as follows:

- Industrial workers in Area A source subarea that use groundwater as a potable water source for interior use and/or drinking water source but may spend 4 hours of each workday in contact with Area A source subarea groundwater.

- Resident adults and children in Area A source and Area B source subareas that use contaminated groundwater for whole-house use, including bathing, drinking, and washing.

- Industrial workers in Area A source and Area B source subareas that use contaminated groundwater for unrestricted workplace use, including drinking and washing.

6.6 UNCERTAINTY ANALYSIS

The objectives of the uncertainty analysis, defined by EPA (EPA 1989; and 1992a), are as follows:

- Provide to the appropriate decision-makers a summary of those factors that significantly influence the risk results, evaluate their range of variability, and assess the contribution of these factors to the underestimate or overestimate risk.
- Discuss the data underlying the assumptions that most significantly influence risk, to highlight the strengths and weaknesses of the HHRA results.

Virtually every step in the HHRA process requires numerous assumptions, all of which contribute to uncertainty in the risk evaluation. In the absence of empirical or site-specific data, assumptions are developed based on best estimates of data quality, exposure parameters, and dose-response relationships. To assist in the development of these estimates, EPA provides guidelines and standard default exposure factors to be used in HHRAs (EPA 1989; 1991b). The use of these standard factors is intended to promote consistency among risk assessments where assumptions must be made. However, their usefulness in accurately predicting risk depends on their applicability to the site-specific conditions. It is likely, therefore, that the net effect of all the assumptions yields a conservative estimate of total risk. However, the uncertainty analysis does not change the quantitative risk characterization results presented in Section 6.5.3. Table 6-5 presents a graphical summary of the uncertainties discussed in the following sections.

6.6.1 Data Evaluation and COPC Selection Uncertainties

The following are some of the uncertainties related to the data evaluation and reduction approaches (Section 6.2.1) used in this HHRA.

- Between the time that the data quality objectives for the RI were determined and the RI field mobilization occurred, EPA Region 9 updated some of the screening-level PRGs for LSGPS COPCs. In isolated cases, where a dilution was required in the analysis, the VOC detection limits were not always below the current (October 1, 2002) Region 9 residential PRG for tap water. However, all analyses were conducted with up-to-date and appropriate analytical methods and, therefore, only a slight underestimate may have possibly resulted in these few isolated cases.

- Various types of data qualifiers are attached to analytical data by either the laboratory conducting the analyses or by the person performing data validation. A common data qualifier in data packages is the “J” qualifier. Data qualified with a J are estimated concentrations reported below the minimum confident sample quantitation limit (SQL), also known as the practical quantitation limit, or are estimated because quality assurance parameters were out of range. In this HHRA, all data qualified with a J were used the same way as positive data that did not have the qualifier. The use of J-qualified data as the reported concentration in the case of the LSGPS HHRA is believed to result in an overestimation of the actual concentration and thus, actual risk.

- The dataset for a particular chemical generally will contain some samples with positive results and others with nondetect results. The nondetect results are usually reported as SQLs or practical quantitation limits. The chemical may be present at a concentration just below the reported SQL, or it may not be present in the sample at all. In this HHRA, if a chemical was reported as a nondetect in a sample, it was assumed to be present at one-half the SQL for that sample. Measure of degree of over- or under-estimation cannot be determined.

- Once proxy values were assigned to nondetect results, the data distribution was evaluated to determine the best approach to assessing exposure point concentrations representative of an area at an exposed population’s point of exposure. To calculate these exposure point concentrations, a certain decision tree was employed (see Appendix M-1), in the spirit with EPA’s most recent draft guidance on calculating the concentration term (EPA 2002b). When an assumption about the normality of a dataset is made, potential uncertainty is introduced, as the “true” distribution of data (assuming infinite sample size) is unknown. Measure of degree of over- or under-estimation cannot be determined.

- Grouping of data (related both to the data reduction of Section 6.2.1 and to exposure assessment; see also Section 6.6.2) into either depth strata (for soil) or horizontally defined subareas (for groundwater) introduced some uncertainty, as either the depth (vertical) or horizontal boundaries of the group affect ultimately the sample size for each subset. For the LSGPS HHRA, however, the data grouping was selected to ensure that the reasonable maximum exposure case was conservatively represented; slight changes in horizontal grouping from one subarea to another are not major sources of uncertainty, given the relatively homogeneous data sets used in the LSGPS HHRA.

- Overall characterization of the nature and extent relied on the RI and data of known quality. Although other data may be available (in particular, unvalidated data of unknown quality from the source area on Brenntag), these data would need to be subjected to EPA data validation processes and usability determinations before they would be appropriate for inclusion in the HHRA. At the same time, at least one of the samples from the Brenntag source area indicated higher concentrations than recorded in any other RI sample. Overall, however, the effect on projected risks in the Area A source area may be negligible if these data were to be validated, found usable, and included in the risk assessment, in large part due to the statistical approaches used to determine exposure point concentrations across a larger area (see Appendix M-1).

In addition to the mechanical data reduction and statistical approaches, uncertainty was introduced in the LSGPS HHRA when the COPC selection process (Section 6.2.2) was conducted.

- For the indoor air data set, contaminants that were detected in underlying groundwater were retained in air for conservative assessment of cumulative risks from all pathways and all media, even if concentrations were below the soil vapor intrusion guidance screening concentrations (EPA 2002c). This measure should have overestimated risks by conservatively retaining COPCs.

- Even if contaminants met the criteria for elimination in a dataset based on frequency of detection alone (minimum 20 samples, with frequency less than 5 percent detection), if they were degradation products of either TCE or PCE, they were conservatively retained for cumulative assessment. This measure should have overestimated risks by conservatively retaining COPCs.

- For selection of soil COPCs, residential land use was assumed, even in currently industrial or commercial properties. Although conservative, this ensured that the COPC set was complete if future residential redevelopment were to occur. This measure should have overestimated risks by conservatively retaining COPCs. Similarly, for selection of groundwater COPCs for ingestion, dermal, and other whole-house exposures, the residential tap water PRG was used as the screening criterion. For industrial or commercial properties, this would conservatively retain COPCs that may not strictly be of concern if less rigorous use of water is actually in place. This measure should have overestimated risks by conservatively retaining COPCs.

- For selection of surface water COPCs, the comparison criteria were also the residential tap water PRG. However, in some cases, the WQB-7 criteria is a higher (i.e., less conservative) value than the tap water PRG. This measure should have overestimated risks by conservatively retaining COPCs.

- For screening the indoor air vapor intrusion pathway (i.e., the concentration of chemicals in air based solely on the home's presence above contaminated groundwater), conservative screening concentrations in groundwater were applied to the subareas (Area A nonsource, Area B source/nonsource, and Area C) where (a) no indoor air monitoring has been conducted, and (b) the measured concentrations in the Lomond Lane area (Area A source area) are not thought to be representative, based on differences in soil type and/or depth to water. However, reliance upon any screening value assumes that the underlying assumptions are valid. Based on professional judgment and review of the boring logs for these areas, the soil vapor intrusion guidance (EPA 2002c) appears to present conservative screening toxicity values for this purpose, but the exact magnitude of the over- or underestimation cannot be predicted.

- Since maximum concentrations (regardless of distribution of the underlying data sets) were used to select COPCs, the COPC selection process is conservative overall.

6.6.2 Exposure Assessment Uncertainties

Significant site-specific uncertainties associated with the exposure assessment for LSGPS include:

- A source of uncertainty associated with the exposure assessment was the use of 100 percent gastrointestinal availability in converting oral toxicity factors to dermal toxicity factors (as shown in Tables M-5.1 and M-6.1 of Appendix M). Although this assumption thus inherently treats chemicals as being equally toxic via the dermal route as if the chemicals, the magnitude of the uncertainty associated with this assumption is unknown for the LSGPS COPCs.

- The assumption that the chlorinated solvents detected in the LSGPS groundwater adheres to the empirical equations set forth in RAGS Part E (EPA 2001c) is consistent with the state of the science. However, EPA noted in RAGS Part E that, for halogenated chemicals, the predicted partitioning coefficient values used may have been underestimated, as the equation developed for that prediction was derived almost entirely of hydrocarbons with a relatively constant ratio of molar volume to molecular weight (EPA 2001c). Because chlorinated chemicals have a lower ratio of molar volume relative to their molecular weight than hydrocarbons (because of the relatively weighty halogen atom), the partitioning coefficient correlation will tend to underestimate permeability coefficients for halogenated organic chemicals. To address this problem, EPA is exploring a new partitioning coefficient correlation based on molar volume and the logarithmic value of the octanol-water partition coefficient (EPA 2001c). The magnitude of the uncertainty associated with this assumption is unknown for the LSGPS COPCs.

- Concentrations of chemicals in groundwater may exhibit seasonal variations. At this site, the RI monitoring well data (from August and October 2002) were used to calculate groundwater exposure point concentrations. The actual risk for groundwater pathways may have been slightly underestimated or overestimated for each chemical due to seasonal variations and ongoing degradation processes.

- As per EPA Region 8 preference and recent developments in attempting to ascertain the scientific inputs necessary to adjust inhalation rates by age-dependent body size and weight, the inhalation pathway was not adjusted for body weight (and possible changes in inhalation rate) of children. In this manner, a direct comparison to lifetime “acceptable” concentrations (reference concentrations) was possible, given adjustment for the fraction of a lifetime for which exposure occurred. Because of scientific limitations, this approach (despite its own inherent uncertainty) was determined to be the most conservative yet accurate approach for the inhalation pathway.

- The exposure assessment used readily available, standardized reasonable maximum exposure parameters wherever possible, and cited heavily from RAGS guidance (EPA 1989, 2001c) and the *Exposure Factors Handbook* (EPA 1997d). These estimates are expected to conservatively overpredict risks for most people, and rather predict risks for the "high end of individual risks" as intended by the reasonable maximum exposure scenario (EPA 1992a). To assess the "average" exposed person, estimates using central tendency exposure factors were also presented in this HHRA. These exposure factor assumptions also contain uncertainty, as even the "average" person is not easily characterized.

- Where default or otherwise standardized reasonable maximum exposure or central tendency exposure parameters were unavailable or inappropriate, site-specific parameters were assigned, based largely on professional judgment or specialized site-specific knowledge about the LSGPS. However, no "one size fits all" assumption best fits many of these scenarios. For example, although the HHRA assumed a current industrial worker may use contaminated groundwater up to half of every work day of his or her tenure (such as for the reasonable maximum exposure current industrial worker scenario), this may be unlikely for most Lockwood area businesses. Further, the HHRA conservatively assumed a "worst-case" assumption for each reasonable maximum exposure parameter based on professional judgment; for example, in the case of the industrial workers, workers were not assumed to wear gloves. In the case of residential recreational scenarios, such as the residential wading pool user, the entire body (including the entire head and face) was assumed to remain saturated with contaminated groundwater, although in reality, it is a fact that a child playing in a wading pool could not remain submerged for the entire play session. Instead, to err on the side of conservatism, these site-specific assumptions were made to account for the potential that significant splashing (in the case of the adolescent wading pool recreator) or worker failure to use any form of protective clothing should occur.

Finally, the concept of reasonable maximum exposure was used to develop exposure doses for all of the receptors that were evaluated. Reasonable maximum exposure is defined as the "maximum exposure that is reasonably expected to occur at the site" (EPA 1989). Several significant variables that determine the exposure doses for the reasonable maximum exposure are based on upper-bound (typically 90th to 95th percentile or greater) estimates. Therefore, where reasonable maximum exposure assumptions are employed, the calculated exposure dose for any given chemical which results from the integration of all of these variables represents an upper-bound estimate of the probable exposure dose. The use of these upper-bound exposure parameters, coupled with conservative estimates of toxicity, in turn, will yield risk results that represent an upper-bound estimate of the carcinogenic risk and noncarcinogenic hazard index. A similar compounding effect of the conservative assumptions is present for the central tendency exposure case, which is intended to represent the "average" case (EPA 1992a).

6.6.3 Toxicity Assessment Uncertainties

For a risk to exist, there must be significant exposure to the COPCs and the COPCs must be toxic at the predicted exposure levels. The toxicological uncertainties primarily relate to the methodology by which slope factors and reference doses are developed. In general, the methodology currently used to develop slope factors and reference doses is conservative and likely results in an overestimation of human toxicity (EPA 1989). These and other factors are discussed in the following bullets.

- Although there is evidence to suggest some carcinogens may exhibit thresholds, cancer slope factors are developed assuming there is no safe level of exposure to any chemical proven or suspected to cause cancer. This approach implies that exposure to even a single molecule of a chemical may be associated with a finite risk, however small. The assumption is that even if relatively large doses of a pollutant were required to cause cancer in laboratory animals (i.e., much higher than a person would ever likely be exposed to over a lifetime), these exposure doses can be linearly extrapolated downward many orders of magnitude to estimate cancer slope factors for humans. A significant uncertainty for the carcinogens is whether the cancer slope factors accurately reflect the carcinogenic potency of these chemicals at low exposure concentrations. The calculated slope factor is used to estimate an upper-bound lifetime probability of an individual developing cancer as a result of exposure to a particular level of a carcinogen. Therefore, the cancer slope factors developed by EPA are generally conservative and represent the upper-bound limit of the carcinogenic potency of each chemical. The actual risk posed by each chemical is unknown, but is likely to be lower than the calculated risk and may even be as low as zero (EPA 1989). The conclusion is that these toxicity assumptions typically result in an overestimation of carcinogenic risk.

- The assumption that all carcinogens (whether Groups A, B1, B2, or C, in the original EPA classification scheme) can cause cancer in humans is also conservative. Only those chemicals classified as Group A carcinogens by EPA are unequivocally considered human carcinogens. The other three classes are probable (Groups B1 and B2) or possible (Group C) human carcinogens. In this LSGPS HHRA, all “probable” or “possible” carcinogens were given the same weight in the toxicity assessment (and consequently in the estimation of risk) as known human carcinogens. This assumption most likely overestimates actual carcinogenic risk to humans.

- In the development of reference doses, it is assumed that a threshold dose exists below which there is no potential for adverse health effects to the most sensitive individuals in the population. The reference dose is typically derived by applying several uncertainty factors of 10 each to a no-observed-adverse-effects level (NOAEL) or a lowest-observed-adverse-effects level (LOAEL) determined from a dose-response study in animals. An additional modifying factor of up to 10, which accounts for a qualitative professional assessment of additional uncertainties in the available toxicity data, also may be applied (EPA 1989). The final degree of extrapolation for the chemicals detected at the LSGPS ranged from 30 to 3,000 (see Table M-5.1 of Appendix M) and, therefore, result in probable overestimates of risk by up to the same degree. In general, the calculated reference dose is likely to be overly protective, and its use probably results in a moderate to high overestimation (approximately equates to an order of magnitude) of the potential for noncarcinogenic risk.

- Route-to-route extrapolation was employed for some LSGPS COPCs lacking toxicity factors. Inhalation of toxicity factors for carbon tetrachloride, chloroethene, cis-1,2-DCE, trans-1,2-dichloroethene, or PCE is were route extrapolated from oral toxicity factors; this approach presupposes that inhalation of theses chemicals is as hazardous as ingesting it, and that the effects would be exerted in the same manner. Similarly, the an oral toxicity factor for chloroethane was route-extrapolated from an inhalation toxicity value. In reality, the route of exposure can result in different effects or differing degrees of effect. Therefore, the uncertainty associated with the use of route-to-route extrapolation may over- or underestimate risk to an undetermined degree.

- Upon consultation with EPA Region 8 toxicologists and the Superfund Technical Support Center, surrogate toxicity data were not recommended for use for chemicals lacking toxicity factors (1,3-dichloropropane; 2,2-dichloropropane, and cis-1,3-dichloropropene). Because these three chemicals were detected in only one sample each in the Area B nonsource area (see Table M-2.5 of Appendix M), the magnitude of this data gap in relation to the overall risks presented in the LSGPS HHRA is believed to be insignificant. The magnitude of the underestimation of risk associated with this assumption is unknown.

- IRIS currently lists ethylbenzene in the weight-of-evidence class D group, indicating a lack of animal and human data as the basis. While the EPA Region 9 PRG tables list an inhalation slope factor (and route-extrapolated oral slope factor) derived by the National Center for Environmental Assessment, the IRIS carcinogen classification was given precedence. Carcinogenic risks for ethylbenzene were not quantified for this risk assessment. This approach would result in an underestimation of risk only if consensus was reached between the agencies such that ethylbenzene was determined to be a carcinogen in the future.

- Considerable controversy surrounds the ongoing toxicity reassessment of TCE under way at EPA. Although some EPA regions have adopted the interim toxicity values for use in risk assessments, the official position of EPA Region 8 is to await finalization of the IRIS profile for TCE formally approving the toxicity value change. At this time, the scientific controversy results in a wide range of candidate toxicity values that may ultimately be part of the IRIS profile; however, even the EPA cancer classification for TCE has not yet been finalized. Because of this controversy, EPA Region 8 has determined that the previous IRIS values (current prior to 1994 when the TCE reassessment was initiated) should be used for the LSGPS HHRA. As there exists a range of candidate toxicity values, the net effect of this decision is unknown; however, if more conservative toxicity factors are selected for the final reassessment, then the use of less conservative toxicity values in this HHRA could result in an underestimation of risk.

- The toxicity assessment of PCE is also recent (EPA 2001e), but similarly includes some scientific discrepancies, even in the determination of EPA cancer classification. According to the Superfund Technical Support Center, the PCE weight-of-evidence classification is on the C-B2 continuum, and thus its conservative assessment as if it was a known human carcinogen is conservative. Further, EPA (2001e) notes that recent studies have detected a significant range between the 5th and 95th percentile cancer slope factor that spans almost one order of magnitude. EPA concluded the currently recommended inhalation toxicity factors are the best, yet conservative estimates of the statistical cancer potency for PCE in the human population. The magnitude of the uncertainty associated with this assumption is unknown.

- The toxicity assessment of VC included a unique approach to its derivation, resulting in cancer toxicity values assuming adulthood-only exposure versus exposure throughout a lifetime, including childhood. Although the appropriate (either adult-only or entire-lifetime) toxicity values were applied to the appropriate scenarios, it is unlikely that the “real” toxicity of VC follows such a bright line. The magnitude of the overestimation of risk associated with this assumption is unknown.

- The individual isomers of xylene (m-xylene, o-xylene, and p-xylene) were not individually assessed in the HHRA because of a lack of isomer-specific data. However, “total xylenes” were instead assessed, and this should quantitatively account for the detection of the individual isomers, and was selected to reflect the more toxic combination of the isomers as detailed in IRIS. Within any given sample, however, this assumption may slightly overestimate or underestimate sample-specific individual isomer risks.

- Although dermal risks for PCE in groundwater were elevated for some scenarios, it is important to realize that the HHRA predicted dermal uptake based on equations presented in RAGS Part E (EPA 2001c) that may not appropriately or precisely model the actual uptake of any given chemical. In particular, as noted in Section 6.4.3.1, PCE is not rapidly absorbed through the skin, especially when compared to inhalation exposure. Therefore, if constant contact with PCE is not maintained (such as skin dampened with PCE-containing groundwater inside a glove or swimsuit), these risks are likely to be overestimated.

As discussed in the bulleted items above, a conservative approach was adopted to address the toxicity assessment uncertainties; therefore resulting in a conservative HHRA, with respect to the toxicity assessment.

6.6.4 Risk Characterization Uncertainties

As with all risk assessments, uncertainty in the risk characterization is a compendium of all the previous uncertainties (relating to data reduction, COPC selection, and the exposure and toxicity assessments). The culmination of these uncertainties is manifested in the final point estimates presented in Section 6.5. However, consistent with risk characterization guidance, such point estimates are required to give risk managers a set of comparable risks for agency prioritization. Of particular interest in the LSGPS are the following uncertainties:

- The assumption that all cancer risks are additive is a precaution, although in some cases, chemicals may have a specific organ that is targeted in the carcinogenic process. Similarly, although noncancer effects were assumed cumulative if the same target organ or system was involved, rate-limiting or potentiation steps may be present in the human body that do not strictly result in additive cancer risks or noncancer hazards. Therefore, the exact magnitude or direction (whether an overestimate or underestimate) cannot be determined.

- Although homes in the Lomond Lane area (above the Area A source area) contained COPCs measured in indoor air linked to the LSGPS groundwater plume, a local background location (542 Klenck), which was remote from the LSGPS plume, also showed indoor air concentrations of these COPCs. Many COPC concentrations were within U.S. background levels for homes across the nation, indicating that contribution of nonplume sources (such as from household cleaners, building materials, or ambient outdoor air) is more likely than plume sources. However, consistent with the soil vapor intrusion guidance (EPA 2002c) and EPA background guidance, no subtraction of "background" risks from the risks in Section 6.5 was performed.

- A monitoring well (MW202) believed to be upgradient of the LSGPS recently (as of October 2002) exhibited detectable concentrations of some LSGPS contaminants. Further monitoring is planned to determine whether the LSGPS plume extends to that well. Regardless, no background was established for groundwater contaminants at the LSGPS, since the contaminants present are manmade and linked to release(s) at the LSGPS per EPA guidance.

- Exposure point concentrations were calculated based on the 95 percent upper confidence limit from the RI data set. These exposure point concentrations were used to estimate risks for residential and industrial worker receptors resulting from direct contact with groundwater. No residential well or industrial well data was included in the HHRA data set. From the information presented in Table 6-3, only three wells (RW001, IW003, and IW016) located in the Area A nonsource area and one well (IW013) located in Area C had TCE concentrations that exceeded its respective exposure point concentration for the subarea. If these actual TCE concentrations are substituted for the TCE exposure point concentration to determine cumulative cancer risks and noncancer HIs for a resident and/or industrial worker in the Area A nonsource area or Area C, as appropriate, the resulting risks to the resident and/or industrial worker are essentially unchanged and are still within the risk management range or considered to be insignificant risks.

In addition, risk managers at the LSGPS will ultimately determine the final "risk management" range for the LSGPS, which may range (per the NCP) between 1×10^{-6} and 1×10^{-4} for cancer risks. The determination of this risk characterization bright line will be documented in the forthcoming record of decision for the LSGPS.

6.6.5 Uncertainty Summary

A graphical summary of the uncertainties of the LSGPS HHRA is presented in Table 6-5. Overall, conservative measures were used to address the uncertainties in the HHRA; thus, the HHRA is expected to overestimate actual risks to receptors in the LSGPS area rather than underestimate risks.

7.0 ECOLOGICAL RISK ASSESSMENT

As part of the LSGPS Superfund process, the RI/FS phase also includes an ERA. Section 7.1 summarizes the overall approach to ERA at Superfund sites. The Superfund process includes a statement of the ecological "problem" facing a Superfund site; this information is set forth in the ERA problem formulation (Section 7.2), which includes a description of the ecological setting (Section 7.2.1) and introduction to the ecological components of the conceptual site model(Section 7.2.2). This underlying problem formulation helped establish the data quality objectives for the LSGPS RI (set forth in the RIWP; TtEMI 2002a) that shaped the sampling plan contained in the RIWP. The outcome of the ERA process is the determination (at the first scientific management and decision point) whether the RI sampling indicates any risks (above desktop screening levels) to ecological receptors and requires further evaluation.

7.1 INTRODUCTION AND APPROACH

A screening-level ERA was conducted for the LSGPS according to EPA guidance (EPA 1997b). Step 1 of EPA guidance consists of the initial site visit, problem formulation process, and screening-level toxicity evaluation. The problem formulation phase (see Section 7.2) is conducted to evaluate whether (1) significant wildlife habitat exists at the site or the site is used by wildlife (see Section 7.2.1) and (2) actual or potential complete exposure pathways to ecological receptors exist (see Section 7.2.2). In accordance with EPA guidance, an initial site visit was conducted in April 2002, with results characterized in Section 7.2.1. The screening-level toxicity evaluation is embodied in the selection of ecological screening toxicity values and the COPC selection process (see Section 7.3).

At the conclusion of Step 1 of EPA guidance, Step 2 begins and exposures are quantified (see Section 7.4) and risks are calculated, based on the ecological effects evaluation (Section 7.5). Based on these findings from Step 2, risk managers (including EPA, DEQ, and other natural resource trustee agencies) reach a scientific/management decision point to determine whether a full, site-specific ecological risk assessment is necessary. At LSGPS, this ERA process was concluded at the screening-level scientific/management decision point, based on the findings set forth in Section 7.6.

7.2 PROBLEM FORMULATION

Problem formulation (as applied here) is the process for generating and evaluating preliminary hypotheses about why effects may occur due to Superfund contamination (EPA 1998). The problem formulation includes the development of an ecological conceptual model that illuminates relationships between a stressor and assessment endpoint, or between several stressors and assessment endpoints.

To establish these hypotheses, the ERA considers each of the key elements that factor into the ecological setting at the LSGPS, including habitat availability, contaminant fate and transport, receptor classes present, and complete exposure pathways. Each of these elements is discussed below. The final product of the problem formulation (Section 7.2.2.3) is a set of established assessment endpoints that adequately reflect management goals and the ecosystem they represent.

7.2.1 Ecological Setting

Key elements suggested as useful for framing the ecological setting are set forth in various EPA guidance (EPA 1992b, 1997b) and are found in various places within this RI report. General site data, including geographical location (including coordinates and maps), size, and land use are presented in Section 1.3 of this RI report. The overall site history for LSGPS is presented in Sections 1.4 and 1.5 of this RI report. Geological information is presented in Section 3.2 of this RI report. Groundwater discharge and surface water bodies are introduced in Sections 3.3 and 3.4 of this RI report, with pertinent habitat detailed below in Section 7.2.1.2.

A detailed contamination profile (including nature and extent of contamination) is introduced in Section 4.0 of this RI report. In summary, the primary class of chemicals detected during the RI sampling at the LSGPS were chlorinated VOCs, which thus became the focus of the ERA. However, the full contamination profile (including all detectable contaminants in all media sampled during the RI) was screened specifically for ecological effects, as detailed in Section 7.3. Among the information unique to the ERA are the following habitat descriptions. Because of the urban nature of the LSGPS, no vegetation or wildlife inventory has been conducted to date. Figure 1-2 of this RI report shows the location of all major features (including ponds, wetlands, ditches, and the Yellowstone River) relevant to the urban residential and industrial facilities in Lockwood.

7.2.1.1 Upland Habitat

Results of the field reconnaissance survey in April 2002 indicated that very limited habitat exists in the urban LSGPS area, with the exception of the wetland area and wetlands area to the west, urban conveyance habitat offered by the Coulson irrigation ditch, and the manmade habitat at the AJ Gravel pond, in the northernmost portion of the LSGPS. More attractive ecological habitat is present in and along the banks of the Yellowstone River, rather than in the urban (residential and commercial) Lockwood area. These wetland/aquatic habitats are discussed in Section 7.2.1.2.

7.2.1.2 Wetland/Aquatic Habitat

The LSGPS has frontage along the southeastern bank of the Yellowstone River in Yellowstone County, Montana. Riparian vegetation has been described as being highly variable in the Yellowstone River Basin, and may include marsh, meadow, shrublands, and forest communities (Knight 1994 as cited in

USGS 1999). In addition to this natural habitat, three urban, localized wetland/aquatic habitat areas are provided on the LSGPS itself, as detailed below. Because of the proximity of nearby, better-quality habitat along the Yellowstone River riparian corridor, it is unlikely that wildlife would preferentially access the smaller, limited LSGPS wetland and aquatic habitats described below. At the same time, the proximity of the Yellowstone River may allow for serendipitous forage from other surface water bodies (such as the smaller, limited LSGPS wetland, ditch, and pond described above), when they might otherwise be considered too small and less than ideal habitat.

To evaluate the three urban, localized habitats within the LSGPS, surface water and sediment samples were collected in the wetland habitats during the RI. These samples were screened to identify ecological COPCs by comparing analytical results to appropriate ecological screening values for each medium. The data evaluation and COPC selection process is detailed for each medium in Section 7.2. In addition to the three localized habitats within the LSGPS, consideration of the site proximity to the Yellowstone River was included in the ecological input to the conceptual site model, as discussed below. Figure 1-2 of this RI report shows the location of each of these features, including ponds, wetlands, ditches, and the Yellowstone River.

Wetlands Area, West LSGPS along Cerise Road

During the April 2002 site reconnaissance effort, a wetland area (including presence of cattails in standing waters within a former chute channel along Cerise Road) was identified. This area is described in Section 3.3.5. Because of the potential for wildlife ingestion and forage on this apparently residential property, the wetland area surface water and sediment (at station SW103) was sampled. Wetland areas at the LSGPS are depicted in Figure 1-2 of this RI report.

Ditch: The Coulson Irrigation Ditch

The Coulson irrigation ditch is introduced in Section 3.3.2 of this RI report. This ditch is dry in much of its reach nearest to the main headgate at the river unless water is flowing. During April 2002 site reconnaissance, the majority of the Coulson irrigation ditch was observed for evidence of ecological receptors and to provide input to the ecological components of the conceptual site model.

The LSGPS area Coulson irrigation ditch, like many streams of the eastern Yellowstone River basin plains, is dominated by grasses, rushes, and sedges. Other findings of the April 2002 survey included observations of raccoon, deer, and rabbit tracks in the moist sediment of the Coulson irrigation ditch bottom immediately west of the LWSD, just south of Cerise Road. To the west of the Auto Auction property, water cover appeared to exist either as natural seeps or where rainfall accumulates. Up to 24 inches of water cover was apparent behind the L&J Machine Shop (101 Cerise Road), where location SW300 was sampled. Populations of minnows and other bait-size fish are known to exist north of Island Park Road where the Coulson irrigation ditch parallels Lomond Lane, immediately downstream of the Cozy Court Trailer Park (232 Lomond Lane) in the Lomond Lane area. During the April 2002 site reconnaissance visit, a local resident was observed trapping minnows in the Coulson irrigation ditch for subsequent use in catch-and-release recreational fishing. Samples were collected from the sediment and surface water in the Coulson irrigation ditch as described in Section 4.5.3.

Once the Coulson irrigation ditch angles eastward, the presence of surface water continued in the ditch in April 2002 (and during the RI) across the LSGPS, despite the fact that the Coulson irrigation ditch was not actively receiving river water via the headgate at the Yellowstone River. Therefore, groundwater recharge to the ditch is likely in this area (see Section 3.4.7 of this RI report). The nature of the habitat is similar, with the most prevalent high-water marks observed generally 2 to 3 feet above the ditch bottom.

Pond: The AJ Gravel Pond

Characterization of both piscivorous wildlife risks and human fish ingestion risks (see Section 6.0) are of interest, based on the habitat afforded by (and human use of) the AJ Gravel pond. The AJ Gravel pond was stocked with fish in 1998, including largemouth bass (*Micropterus salmoides*), smallmouth bass (*Micropterus dolomieu*), "ling" (burbot, *Lota lota*), sunfish (such as *Lepomis cyanellus*), and catfish (such as the channel catfish; DEQ 1998). The stocked fish were caught elsewhere and released into the AJ Gravel pond. The owners of the property structured the 2-acre pond, which is 20 feet deep, to include two 30-foot-deep trenches on the sides of the pond for fish to retreat during historic dewatering activity (DEQ 1998). Residents report that the pond has been a "catch and release" fishing pond. Fish sizes ranged up to 18 inches in the summer of 1998, and juvenile fish (approximately 15 to 20 at a time) had been observed near the banks. In addition to aquatic pond habitat, diving ducks have been observed (DEQ 1998).

Fish in the AJ Gravel pond were reported to be lethargic in December 1998. Behavioral changes reported included failure of the fish to rise to the surface at normal feeding times (DEQ 1998). Normal seasonal metabolic slowdown may explain such reported behavioral changes, as the aquatic metabolism will slow down during winter as the water gets colder. In addition, some prey (such as insects) are less abundant in the winter months, resulting in less fish actively feeding at the surface. Historic water quality data, collected in September 1998, indicated that concentrations of 0.11 mg/L TCE and 0.661 mg/L PCE, which were compared (in consultation with Montana Fish, Wildlife and Parks) to bluegill (also in the sunfish family) lethality benchmarks of 44 mg/L TCE and 13 mg/L PCE (DEQ 1998). Therefore, it was quickly determined that historic concentrations would not pose a risk of mortality to the fish. Ideally, this ERA would include screening against comprehensive aquatic benchmarks that also account for other more subtle effects on aquatic species, including not only mortality but also narcosis (i.e., lethargy), which can result from exposures to high concentrations of other VOCs; however, no aquatic toxicity values that assessed narcosis were available (see Section 7.5). Lethargic fish may become prey more often than nonexposed fish, resulting in increased feeding on contaminated fish by piscivorous wildlife and avian species.

The substrate of the AJ Gravel pond is gravel, as the name implies, which had been mined for commercial sale prior to the hiatus on dewatering when contamination was detected in 1998. Therefore, the habitat present is manmade (or man-altered), and little true sediment is present in accessible areas of the pond. Thus, no sediments were collected from the AJ Gravel pond. However, surface water of the AJ Gravel pond was sampled, as described in Section 4.5.1.

Yellowstone River

The Yellowstone River was introduced in Section 3.3.1 of this RI report. Because the Yellowstone River may receive contaminated groundwater from the LSGPS, it is of particular interest because of its status as a sensitive environment. Specifically, the Yellowstone River is a body of water designated by the State of Montana for protection of aquatic life. The Yellowstone River is expected to provide superior habitat relative to nearby urban wetland, ditch, and pond habitat. Although the particular stretch of the Yellowstone that borders the LSGPS is a part of the urban Billings landscape, the river's upper reaches sustain cutthroat trout and other important fish species, which, in turn, sustains wildlife species ranging from grizzly bears to otters. Downriver, cutthroat populations give way to blue-ribbon brown and rainbow trout fisheries. Bald eagles inhabit the upper stretches of the Yellowstone (including Yellowstone County, where the LSGPS is located). The Yellowstone River Basin near Billings is a part of the Northwestern Great Plains ecoregion (described in USGS 1999).

7.2.2 Ecological Conceptual Site Model

The conceptual site model is an integral part of the RI process. The conceptual site model is the planning tool that organizes what is already known about the site prior to RI mobilization, and helps the planning team identify what additional information is required to make the decisions that will achieve the project's goals (EPA 2001b). As noted in the RIWP, where the conceptual site model was first introduced (see Figure 9 of TtEMI 2002a), use of the conceptual site model is ongoing, and an iterative approach (allowing the conceptual site model to evolve and mature as data gaps are filled) is required. Therefore, the conceptual site model discussed herein (see Figure 6-3 in Section 6.0) is the revised conceptual site model that takes into account findings of the RI effort.

Because the sources and release mechanisms contributing to present-day contamination are discussed elsewhere in this RI report and were discussed in the RIWP (TtEMI 2002a), those data are not repeated here. Instead, the information that is important to the ecological portions of the conceptual site model is highlighted below.

7.2.2.1 Contaminant Fate and Transport

The contaminant fate and transport mechanisms in place at LSGPS are the same for all media. However, the main unique "fate and transport" consideration in the ERA process is the fact that uptake into food items (plant material or invertebrate prey, for primary consumers) into the food chain is a potentially significant pathway. Specifically, where ecological receptors might forage from contaminated media at LSGPS, food chain transfer must be considered, particularly for bioaccumulative compounds. This possibility is investigated in Section 7.3.2.1 for surface water, which ultimately became the media of focus for the ERA at the LSGPS (see Section 7.3).

7.2.2.2 Receptor Classes and Representative Receptors

To describe potential risks to the LSGPS ecosystem, a simplified discussion of the relationships between the various feeding guilds likely to access the LSGPS is needed. The receptors include those potentially exposed through direct contact with, and ingestion of, VOC-contaminated media. These receptors were organized based on feeding guild, as discussed below. Available information about the regional ecology was reviewed to develop the conceptual relationships between feeding guilds necessary to identify and analyze dietary pathways.

Professional judgment was used to assign each receptor to a single feeding guild, even though some receptors may occupy different feeding niches concurrently, depending on age, dietary requirements, available food, reproductive status, and season.

Primary Consumers

Numerous primary consumers representing several guilds and communities are expected at the site. These receptors occupy the second trophic level. Primary consumers include herbivorous wildlife, soil and benthic invertebrates, and detritivores (i.e., consumers of decaying matter, such as bottom feeders).

Terrestrial herbivorous mammals (e.g., mice, voles, rabbits, squirrels, and deer) and birds (e.g., ring-necked pheasant, wild turkey, doves) feed on terrestrial vegetation, such as grasses, seeds, and fruit. These receptors also may be exposed through ingestion of contaminated drinking water, and through dermal contact with, and incidental ingestion of, contaminated sediment and soil.

Soil infauna (earthworms) and other terrestrial invertebrates may be exposed through direct contact with contaminated soils.

Where perennial water cover occurs (such as in the habitats described in Section 7.2.1.2), aquatic animals that may be exposed to surface water and sediment include:

- Benthic invertebrates, such as insect larvae, odonates, pelecypods, and decapods
- Zooplankton, such as Crustacea and Copepoda
- Various aquatic-dependent life stages of amphibians (e.g., tadpoles)
- Herbivorous fish (e.g., minnows)

These primary consumers may subsequently become prey to secondary consumers.

Secondary Consumers

Receptors occupying secondary or intermediate positions in the LSGPS food web occur at the site. True omnivores expected at the LSGPS site include small mammals, raccoons, turtles, and frogs. Omnivorous fish (such as catfish) also exist, primarily in the larger habitat afforded by the AJ Gravel pond and the Yellowstone River. These receptors may be exposed through (1) direct ingestion of contaminated water,

(2) incidental ingestion of contaminated soil and sediment, and (3) indirect ingestion of contaminated food.

Insectivorous and omnivorous birds, such as swallows, the red-winged blackbird, and the American robin, may be directly exposed through ingestion (drinking, preening after bathing) of contaminated water, and indirectly through ingestion of mature aquatic and terrestrial insects, or ingestion of earthworms and other soil invertebrates. Exposure potential for mid-level consumers will be dominated by two pathways: (1) direct ingestion of contaminated media, and (2) direct ingestion of contaminated animal matter.

Top-Level Consumers

Top-level carnivores expected in the vicinity of the LSGPS site include birds of prey (such as the observed red-tailed hawk and various owl species, as well as piscivorous birds such as the bald eagle), foxes, and snakes. Few or none of these receptors may actually be present, because of the human activity in the Lockwood area. Further, although the size of the AJ Gravel pond does not qualify it as ideal attractive habitat for bald eagle forage, it is possible that an eagle would consider opportunistic feeding in this area. Carnivorous fish (including bass, sunfish, and burbot) are expected to occur both in the AJ Gravel pond (due, in part, to human stocking for catch-and-release recreational fishing) as well as in the Yellowstone River adjacent to the site.

Rare, Threatened, and Endangered Species

Although no rare, or threatened and endangered species that potentially occur within Yellowstone County are definitively known to occur at the LSGPS, the possibility of home range overlap was considered. According to the U.S. Fish and Wildlife Service (USFWS), the special status (rare, threatened, endangered, or other special status) species include the bald eagle (federally listed and threatened), black-footed ferret (federally listed and endangered), mountain plover (proposed threatened), and black-tail prairie dog (a candidate species under consideration for listing)(USFWS 2002). The latter three special status species are known to occur within Yellowstone County but are extremely unlikely to occur specifically at the LSGPS site based on a lack of habitat in this urban area. However, because the bald eagle's range is large, it is possible that an eagle may opportunistically feed on or near the LSGPS, such as from the AJ Gravel pond. Therefore this special status receptor was included in a more detailed literature search for avian toxicity information (see Section 7.5) and risk characterization (see Section 7.6).

7.2.2.3 Assessment and Measurement Endpoints

For the ERA, part of the conceptual site model includes a description of how a given stressor (in the case of LSGPS, the stressor is a contaminant) might affect ecological components of the environment (EPA 1997b). The importance of particular ecological endpoints and subsequently identified measurement endpoints follows both the *Ecological Risk Assessment Guidance for Superfund* (EPA 1997b) as well as EPA's most recent guidance on assessment endpoints (EPA 2002e). Specifically, EPA (1997b) defines assessment endpoints as explicit expressions of the actual environmental values (e.g., ecological resources) that are to be protected. The assessment endpoint is then measured with a measurement endpoint, which in the case of ecotoxicity, is generally the measurable biological response to a contaminant that can be related to the valued characteristic chosen as the assessment endpoint (EPA 1997b). In the case of the LSGPS ERA, the measurement endpoints (toxic responses) are being used to make inferences about risks to the assessment endpoints. Additional details regarding the toxicity of the chlorinated solvents are available in the toxicity assessment (Section 7.5).

Consistent with EPA (1997b) examples and the general assessment endpoints guidance (EPA 2002e), the relevant generic assessment endpoints encompassed in the LSGPS ERA were selected as they relate to protection of local organisms. The general assessment endpoints may include protection of organisms in an assessment population or community (1) against kills (mass mortality), (2) against gross anomalies, and (3) of survival, fecundity, and growth. In the case of the LSGPS, where special status species are present, EPA advocates an emphasis on the latter category, which is an endpoint that applies particularly to special status (including threatened and endangered) species, bald and golden eagles, and migratory birds (EPA 2002e).

Depicted in Table 7-1 are the assessment and measurement endpoints reflected in the ERA for the LSGPS.

Assessment Endpoint Selection

Based on the review of historical information and observations made during the April 2002 site visit and during the RI field effort, the following assessment endpoints were chosen to evaluate potential risk to ecological communities in the LSGPS study areas.

1. Protection of carnivorous, omnivorous, and herbivorous mammals that may ingest contaminated food (i.e., prey, insects/invertebrates, plants), surface water, and

soil/sediment from potentially lethal, reproductive, systemic, or general adverse toxic effects of COPCs.

2. Protection of carnivorous, omnivorous, and herbivorous birds that may ingest contaminated food (i.e., invertebrates and plants), surface water, and soil/sediment from potentially lethal, reproductive, systemic, or general adverse toxic effects of COPCs.
3. Protection of amphibians/reptiles exposed to contaminated surface water and soil/sediment from potentially lethal, reproductive, systemic, or general adverse toxic effects of COPCs.
4. Protection of the aquatic community in LSGPS surface water bodies from potential adverse toxic effects of COPCs.
5. Maintenance of the benthic macroinvertebrate community structure and function in the perennial reach of the Coulson irrigation ditch.
6. Maintenance of soil (invertebrate and plant) communities' structure and function.

Null Hypotheses

For each of the assessment endpoints identified, a corresponding null hypothesis was formed to capture the relationship between the assessment endpoint and their predicted responses when exposed to COPCs. The testable (null) hypotheses identified for ecological receptors near the LSGPS site are shown in Table 7-1 in association with their corresponding assessment and measurement endpoints.

Measurement Endpoints and Measurement Endpoint Receptor Species Selection Rationale

Measurement endpoints and receptor species were chosen to test the null hypotheses and, therefore, evaluate the assessment endpoints. Screening-level media concentrations based on food chain accumulation studies (Sample and others 1996) were the "benchmarks" used to evaluate risk to wildlife that may access on-site (terrestrial) areas of the site as well as on-site aquatic areas at the LSGPS.

Screening-level benchmarks were used for the LSGPS ERA and therefore, the measurement endpoint receptor species represented within the benchmark calculations are the LSGPS species being assessed. As described in Section 7.3, Sample and others (1996) derived the benchmarks for assessment of effects on eight representative mammalian wildlife species:

- Short-tailed shrew (*Blarina brevidauda*)
- Little brown bat (*Myotis lucifugus*)
- Meadow vole (*Microtus pennsylvanicus*)

- White-footed mouse (*Peromyscus leucopus*)
- Eastern Cottontail (*Sylvilagus floridanus*)
- Mink (*Mustela vison*)
- Red fox (*Vulpes fulva*)
- White-tailed deer (*Odocoileus virginianus*)

In addition, for some contaminants, effects on 11 representative avian wildlife species were considered. Unfortunately, as of the Sample and others (1996) publication, toxicity data for bird species was lacking for all of the site-specific LSGPS VOC COPCs. For this reason, a toxicity assessment data gap was addressed in Section 7.5.2, with additional evaluation of an avian carnivore (the bald eagle), which would be the most highly exposed of the feeding guilds due to its trophic level and the bioaccumulation potential of the LSGPS chlorinated VOC COPCs. Dietary exposure of the bald eagle (the target receptor species chosen to fill the avian screening-level data gap) to contaminants was estimated by a food chain effects model (described in Section 7.4.2) and was compared with existing and newly compiled toxicity data for birds (see Section 7.5.2).

The wildlife species chosen by Sample and others (1996) are appropriate for screening at the LSGPS because they are widely distributed in the United States and provide a representative range of body sizes and diets. These receptors were also selected for use in screening the LSGPS RI data because (1) they (or similar species in the same feeding guilds) can be reasonably expected to occur at the site; (2) natural history information, required to assess risk to such wildlife, is available; and (3) a moderately high exposure potential exists based on high ingestion rate-to-body weight ratios, particularly for the smaller species evaluated by Sample and others (1996). Although it is assumed that this array of receptors conservatively protects wildlife in these guilds, no information on the comparative species-specific toxicity of VOCs is available to determine the sensitivity of these species. Further descriptions of the chosen wildlife species and chemicals are provided in the source (Sample and others 1996) report.

Because of a complete lack of toxicity data relevant to amphibians and reptiles, no specific measurement endpoint receptor species were identified. This data gap is discussed further in Section 7.7.3. The aquatic benchmarks (see Section 7.3.1) also take into account a wide array of freshwater species (including fish, invertebrates, and aquatic plants), and therefore a specific species has not been identified as the single measurement endpoint species. Similarly, the terrestrial community benchmarks (see Section 7.3.3.3) are based on available EPA TERRETOX (EPA 2003a) database compilations, and include several plant and invertebrate species. Therefore, a specific species has not been identified as the measurement endpoint species for the soil communities.

7.2.2.4 Exposure Pathways

If a source has had a release to a site media to which receptors are exposed, then an exposure pathway is complete. Discussed below are the media-based ecological pathways of concern for the LSGPS. The conceptual basis for the investigation of these pathways was initially set forth in the RIWP (TtEMI 2002a).

Groundwater

No ecological exposures to groundwater exist at the LSGPS; however, see discussion below relating to surface water and sediment. The groundwater-surface water interface is an important ecological risk assessment medium of interest.

Surface Water and Sediment

Ecological exposures may include direct and indirect contact of semiaquatic or aquatic species with surface water contamination in the wetland/aquatic habitats discussed in Section 7.2.1.2. Terrestrial ecological receptors (wildlife) would be expected to access the irrigation ditches, ponds, and the river for drinking water (ingestion) exposure.

Soil

Ecological exposures to soils include ingestion (for wildlife) and direct contact (for plants and invertebrates). For birds and mammals, two potentially complete soil exposure pathways are expected to be significant (EPA 2000): (1) incidental ingestion of soils during feeding, grooming and preening; and (2) ingestion of food contaminated as a result of the uptake of soil contaminants (see Biota section below). Soil pathways not included are soil particulate inhalation and dermal contact, because of the difficulty associated with quantitatively assessing these pathways, as described below.

Inhalation of particulates is not included in the ecological conceptual site model since respirable particles (>5 micrometers [µm]) are most likely ingested as a result of mucocilliary clearance (ingestion from respiratory tract to digestive tract) rather than being inhaled (Witschi and Last 1996 as cited in EPA 2000) and thus already accounted for in the soil ingestion pathway for ecological receptors. In addition, at equal exposure concentrations, inhalation of contaminants associated with dust particles is expected to contribute less than 0.1 percent of total risk compared to oral exposures (EPA 2000); therefore, dust inhalation is not included in the ecological conceptual site model for LSGPS.

Birds and mammals may also be exposed to contaminants in soils via dermal contact. However, current ecological exposure information is insufficient to evaluate dermal exposure for the LSGPS VOC contaminants in various soil matrices, or to predict possible rates of absorption for many species. For most contaminants, dermal exposure is expected to contribute less than 1 percent to 11 percent of the total risk compared to oral exposures (EPA 2000). For the LSGPS contaminants, in particular, their volatile nature indicates that dermal absorption would not occur, as volatiles would most likely migrate into air before permeating fur, scales, and skin, based on vapor pressures and other chemical properties. In addition, RI soil data (screened in Section 7.3.1) does not indicate (for near surface soil samples collected to 4 ft bgs) that COPCs are significant contributors to ecological risks from surface soil or near surface soil concentrations.

Air

For ecological receptors, burrowing animals could be exposed to relatively high concentrations of VOCs in their burrows via inhalation, provided COPCs are present in the near surface soils (EPA 2000). However, site reconnaissance does not indicate the widespread presence of burrowing receptors, and the RI soil data (screened in Section 7.3.1) does not indicate (for near surface soil samples collected to 4 ft bgs) that COPCs are significant contributors to ecological risks from surface soil or near surface soil concentrations.

Biota

Ultimately, biota that are directly exposed to contaminated media (such as earthworms that live in contaminated near surface soils, or sediment invertebrates in the Yellowstone River) may take up, or bioaccumulate, contaminants. This uptake can be important when contaminants transfer through the food web to higher trophic level consumers (such as omnivorous mammals and birds who feed on the earthworms, or to fish who consume sediment invertebrates).

For most media at the LSGPS (such as soil and sediment), VOC concentrations detected in the RI are below concentrations that would be expected to accumulate significantly in biota. Nonetheless, soil concentrations were conservatively screened in Section 7.3.1 to ensure protection of herbivores and omnivores. For the LSGPS, the primary complete pathway for biota in the ERA is through surface water to prey fish uptake and subsequent transfer of contaminants to upper trophic levels. These topics are explored further in Sections 7.3.1.2 and 7.4.1.

7.3 DATA EVALUATION AND IDENTIFICATION OF ECOLOGICAL COPCS

As with the HHRA (Section 6.2), the RI data were validated and confirmed usable for risk assessment. In the case of the ERA, the conceptual site model framework was used to focus on the media of interest in areas where ecological habitat was available. Thus, the primary media of interest were surface water and sediment in the Coulson irrigation ditch, AJ Gravel pond, the wetland area and wetlands area on the western edge of the LSGPS, and the eastern banks of the Yellowstone River. Surface water and sediment are discussed in Sections 7.3.1 and 7.3.2, respectively.

Because of a lack of habitat, soil was not a primary media for evaluation in the ERA, since contaminants were detected only in active process areas either on Brenntag's active industrial property (Area A) or on Beall Trailers' active industrial property (Area B). For completeness, however, an ecological soil screening was evaluated with RI sampling data in the source areas, as discussed in Section 7.3.3.

The complete derivation of the screening-level benchmarks used in Section 7.3 of this RI report is detailed in the 217-page Sample and others (1996) source, and is not repeated here. In summary, Sample and others (1996) presented NOAEL- and LOAEL-based toxicological benchmarks for assessment of effects of 85 chemicals on 9 representative mammalian wildlife species and 11 avian wildlife species. The wildlife species were chosen because they are widely distributed and provide a representative range of body sizes and diets. Further descriptions of the chosen wildlife species and chemicals are also provided in the source (Sample and others 1996) report. The NOAEL-based benchmarks represent values believed to be nonhazardous for the listed wildlife species; LOAEL-based benchmarks represent threshold levels at which adverse effects are likely to become evident (Sample and others 1996). These benchmarks consider contaminant exposure through oral ingestion of contaminated media (including food chain transfer); however, exposure through inhalation and/or direct dermal exposure are not considered in the screening-level benchmarks. This is consistent with Section 7.2.2.4 and the ecological conceptual site model for the LSGPS.

7.3.1 Surface Water

Surface water COPCs were ultimately selected for the LSGPS ERA by screening against surface water ecological screening toxicity values for direct contact (see Section 7.3.1.1), along with a consideration of bioaccumulation potential (see Section 7.3.1.2) and receptor sensitivity (see Section 7.3.1.3).

7.3.1.1 Direct Contact COPCs

To select ecological COPCs, the maximum concentration of each detected chemical in each surface water body[1] was first compared to the lowest available EPA or U.S. Department of Energy (DOE) benchmarks. No state water quality benchmarks protective of aquatic life (DEQ 2001b) were available for the LSGPS contaminants detected in water. The two (EPA and DOE) sources of information used were noted to be the lowest (most conservative) of the available benchmarks compiled regularly by the California Environmental Protection Agency (CalEPA), as of December 2001 (CalEPA 2001). All original values were confirmed in the appropriate DOE or EPA documents.

Although they were cited in the recent CalEPA compilation as being the lowest available benchmarks, screening values lower than the EPA (1996a) screening toxicity values for PCE, TCE, and toluene were identified for use at the LSGPS to screen for ecological COPCs in surface water. All three of the CalEPA-cited EPA (1996a) values were as calculated in Suter and Mabrey (1994, as cited in EPA 1996a); however, alternate screening toxicity values were calculated by the original authors in 1996 (see Suter and Tsao 1996), after publication of the EPA (1996a) values. Because of this, the secondary chronic Tier II values (Suter and Tsao 1996) were ultimately used for PCE, TCE, and toluene because they were more conservative than the EPA (1996a) values cited by CalEPA, and were based on more recent data. Each screening toxicity value for PCE, TCE, and toluene is thus based on protection of multiple endpoints (including a 20 percent reduction in recruit abundance[2], calculated as an "EC20") and several different organisms, as follows (from Suter and Tsao 1996):

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[1] The evaluation of the Yellowstone River itself was best approached through a conservative, worst-case evaluation of contaminant groundwater loading into the Yellowstone River. The LSGPS plume has reached the river, as contamination has been detected in the shallow waterbearing unit in "sentinel" RI wells installed along the Yellowstone River bank. Because of this, a detailed evaluation of the groundwater impact to Yellowstone River concentrations was conducted (see Section 5.4).

[2] Typically, recruitment is defined by researchers as the survival of individuals that settle to an arbitrarily chosen point in time (Pile and others 1996). Juveniles must grow to an adult "settler" stage and survive through a post-settlement time period defined by the researcher to be considered recruits (Pile and others 1996). Thus, recruitment is important in population structure, because it is linked to survival to the adult stage.

- The PCE screening toxicity value included protection from chronic exposure via an embryo-larval test on fathead minnows (*Pimephales promela*) and 28-day tests on *Daphnia magna*. The test EC20 value for fish (also fathead minnows) and were also protective of *Daphnia magna*. Aquatic plant (*Selenastrum capricornutum*) cell number and chlorophyll A production are also protected at the screening toxicity value concentration (Suter and Tsao 1996).

- The TCE screening toxicity value included protection of fathead minnows, flagfish (*Jordanella floridae*), and bluegill. Protection of daphnids was based on an EPA estimate for *Daphnia pulex* (Suter and Tsao 1996).

- The toluene screening toxicity value included protection of the fathead minnow, bluegill, and guppy (*Poecilia reticulata*). *Chlorella vulgaris* was used in 10-day tests to ensure the chronic value was also protective of aquatic plants (Suter and Tsao 1996).

The DOE (1997) values for cis-1,2-DCE and vinyl chloride were thus used, with their bases noted as follows:

- The cis-1,2-DCE ecological screening toxicity value for surface water is protective only of direct contact for aquatic life, and is the secondary chronic value. During the development of this value, DOE noted that toxicity reference values were not available for piscivorous wildlife, and therefore, the screening toxicity value cannot be assumed to protect wildlife (DOE 1997). See resulting additional evaluation (using the piscivore-protective values of Sample and others 1996) for cis-1,2-DCE in Section 7.3.1.2.

- The VC ecological screening toxicity value for surface water was noted by DOE to be protective of piscivores and bioaccumulation (via river otter dietary ingestion) and protective against a LOAEL (DOE 1997).

7.3.1.2 Bioaccumulation Potential and Indirect (Food Chain) COPCs

The detected surface water contaminants each have some propensity to accumulate in animal tissue from surrounding or ingested media, based on the fact that their bioconcentration factors exceed 1. Therefore, for completeness, consideration of the bioaccumulative potential of each of the detected chemicals was made.

To evaluate bioaccumulation and the indirect exposures via the food chain, a separate screening of the maximum detected surface water concentrations was conducted (see Table N-2.5) for protection of piscivorous wildlife (i.e., mammals who eat fish) using the lowest NOAEL-based water concentration for piscivores listed in Table 12 (Appendix D) of Sample and others (1996). This imparted protection to hypothetical mink and river otters that might drink the contaminated water and feed on fish that had accumulated the detected contaminants. As shown in Table N-2.5, for the chlorinated solvents, only the

AJ Gravel pond data were screened, because these detected concentrations were the highest of the water bodies sampled (comparison data available in Tables N-2.2 through Table N-2.4)[3]. Because all measured concentrations fell below the piscivore screening toxicity values, no ecological COPCs were concluded on this basis, and mammals who eat fish (even from the AJ Gravel pond, assuming maximum detected concentrations) are protected.

However, this bioaccumulative screen did not take into account the different sensitivity of avian (bird) receptors that might feed on contaminated prey. At the time the Sample and others (1996) piscivore benchmarks for the chlorinated solvents were compiled, no avian toxicity data was available. Therefore, an updated literature search to confirm the availability of avian toxicity data was deemed prudent. Section 7.3.1.3 details the approach for evaluating these potentially sensitive receptors. See also Section 7.7.1 for important uncertainties related to the screening approach.

7.3.1.3 Sensitive Receptor COPCs

The screening described in Section 7.2.1 protected aquatic life (fish, aquatic insects, and plants) as well as piscivorous wildlife and terrestrial wildlife that might drink from the AJ Gravel pond. However, as noted above, the surface water screening through Table N-2.5 did not account for the potential different sensitivity of avian receptors. Because of this potential, and based on potential presence of special status species (see special status species discussion in Section 7.2.2.2) in this feeding guild, an additional evaluation was performed.

The evaluation of potentially sensitive avian receptors (such as the bald eagle) was completed with an approach parallel to that of Sample and others (1996) used in deriving the mammalian piscivore screening values. Therefore, the chlorinated solvents detected in the AJ Gravel pond surface water were retained as COPCs for further evaluation of risks to piscivorous birds.

Details of the approach to ensure avian piscivores are protected is explained in Sections 7.4 (avian exposure assessment), 7.5 (avian toxicity assessment), and 7.6 (risk characterization), with the uncertainty in the approach (and data gaps remaining) discussed in Section 7.7.

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[3] Although toluene was detected in the Coulson irrigation ditch, it was not detected in the other surface water bodies. Further, the toluene maximum concentration (0.35 µg/L, as shown in Table N-2.2) was hundreds of times lower than the piscivorous mammal screening value of 764 µg/L given in Sample and others (1996).

7.3.2 Sediment

Sediment was sampled in the Coulson irrigation ditch (see Table N-2.6), where only one contaminant, toluene, was detected. Toluene (which ranged in concentration from 0.19 to 0.34 mg/kg) was compared to sediment benchmarks of 6.45 and 128 mg/kg for fish and daphnids, respectively, assuming 1 percent total organic carbon (Jones and others 1996) and using the lowest chronic values of Suter and Tsao (1996). The maximum toluene concentration in sediment is well below the most conservative sediment benchmark of 6.45 mg/kg.

Sediment in the Yellowstone River was not sampled during the RI. Sediment sampling within the Yellowstone River was not planned based on two fate and transport considerations unique to the VOCs present as well as the site-specific setting. First, actual river bank VOC contamination in sediments is unlikely, given the presence of the steep cut bank at the river bank, which results both in enhanced volatilization of contaminants if they were to occur at those banks as well as a lack of accessible sampling locations. Second, potentiometric maps (based on upgradient groundwater) show that groundwater contamination discharging to the Yellowstone River is likely doing so below the surface of the water, beneath the actual riverbed itself. Even if subsurface groundwater discharges at some point within the Yellowstone River bed, the fast flow of the river, volatile nature of the contaminants, and the extremely low concentrations of VOCs modeled to reach the Yellowstone (see Section 5.4) indicate this medium is unlikely to be contaminated by VOCs.

Sediment in the AJ Gravel pond was not sampled, as readily accessible sediments were buried deep beneath the cobbles and gravel of the pond, such that exposure to sediment is likely to be a negligible pathway for most receptors. Instead, surface water concentrations (see Section 7.3.1.1) were measured and screened to establish COPCs for the AJ Gravel pond.

Based on the contaminant screening for Coulson irrigation ditch sediments, no ecological COPCs were identified, and therefore, no ecological risk is attributed to sediment contamination in the LSGPS ERA.

7.3.3 Soil

As expected (based on conceptual site model predictions of contaminant fate and transport), for those releases (in source areas) that may have occurred as a result of surface spills or leaks, the volatile nature and short half-lives of the contaminants in question has resulted in little residual contamination in the biologically active soil depth (typically 0 to approximately 6 to 8 inches bgs). Based on the increased depth from the land surface and less active volatilization and less chemical degradation (such as via photolysis), where ecological exposures may occur up to 24 inches (such as via burrowing), the concentrations measured in the 0-to-2-foot bgs interval were also of interest.

For completeness, soil was screened in the ERA, despite a current lack of habitat in the urban and industrial locations sampled. The biologically active depth was of primary interest for terrestrial wildlife in these areas, and included protection for shallow burrowers, such as short-tailed shrews, which burrow only within a few inches of the ground surface. Other burrowing receptors (such as rabbits, which may access depths of up to 24 inches) may not be widespread in these industrial urban properties, based on a lack of suitable habitat. However, rabbits have been observed on the Brenntag property (DEQ 2003). Although shallow burrows (such as for shrews, mice, and rabbit warrens) were conservatively screened (see Section 7.3.3.2), deeper burrows (such as for prairie dogs, which may extend 10 feet or deeper) are not currently or likely to be present in the future.

7.3.3.1 Biologically Active Soil Depth COPCs

The biologically active soil depth, for purposes of this ERA is considered to be 0 to about 8 inches bgs. Three samples on Beall Trailers' active industrial property (Area B) contained detectable VOC contamination in approximately the top 6 to 8 inches of soil, as shown in Table N-2.7 of Appendix N.

To verify that these concentrations would not be above ecological screening levels if these currently industrial properties were returned to viable ecological habitat in the future, the lowest available (most conservative) wildlife screening levels developed by Sample and others (1996) were applied. The lowest protective risk-based concentration in food (forage and/or prey) was selected as a proxy for a protective risk-based concentration in soil, since the magnitude of soil contamination (only three detections) and lack of current habitat indicate that soil contamination is a minor contributor to ecological risk as the LSGPS. Essentially, it was assumed that wildlife could eat soil contaminated with the same concentration allowed in their food, for purposes of this screen. As such, the lowest NOAEL-based food value for any of the terrestrial wildlife listed in Table 12 (Appendix D) of Sample and others (1996) was the most conservative screening toxicity value for soil. LOAEL-based screening toxicity values are also shown, for comparison.

These screening toxicity values protect against the critical effect (which, for these chemicals, was hepatotoxicity) for all evaluated terrestrial wildlife. Sample and others (1996) set these effects to protect herbivores, omnivores, and carnivores (including piscivores), as represented by the little brown bat, short-tailed shrew, white-footed mouse, meadow vole, mink, cottontail rabbit, red fox, river otter, and whitetail deer. The critical (lowest and most conservative) screening toxicity values were used, as follows:

- The PCE (listed in Sample and others 1996 as 1,1,2,2-tetrachloroethylene) ecological screening toxicity value of 2.82 mg/kg is the lowest protective food concentration among the wildlife indicators studied, and was designed to be protective of all evaluated wildlife including the cottontail rabbit.

- The TCE ecological screening toxicity value of 1.39 mg/kg is the lowest protective food concentration among the wildlife indicators studied, and was designed to be protective of all evaluated wildlife including the short-tailed shrew.

No concentrations in the biologically active depth (0 to 8 inches bgs) exceeded the NOAEL-based wildlife-protective values. Please see also Section 7.7.1 for a discussion of uncertainties related to the COPC selection process.

7.3.3.2 Burrowing Soil Depth COPCs

For purposes of this ERA, the burrowing depth in the soil is considered to be 0 to 2 feet bgs. Thus, in addition to screening the biologically active soil depth (0 to 8 inches bgs), all samples in the 0 to 2 ft bgs interval on the Beall property (Area B) were also screened, as shown in Table N-2.8 of Appendix N. All maximum concentrations fell below the ecological screening toxicity values protective of wildlife (described in Section 7.3.3.1 above). The maximum detection of 2.1 mg/kg TCE detected at sample location, SS208, at a depth of 0 to 2 feet bgs exceeded the lowest NOAEL-based screening value of 1.39 mg/kg (protective of the most sensitive mammalian guild, the mammalian insectivores, as represented by the short-tailed shrew) but did not exceed the LOAEL-based screening value. Therefore, it is unlikely that concentrations of TCE would result in effects, as the maximum detected concentration is nearly ten times lower than the LOAEL-based screening value.

Based on the contaminant screening for soils, no ecological COPCs were identified, and therefore, soils are not evaluated further in the LSGPS ERA.

7.3.3.3 Community Effects Screen

The data shown in Table N-2.7 of Appendix N was compared to known effects levels for contaminants in the biologically active depth (0 to 8 inches bgs) to assess effects on the structure and function of terrestrial plant and soil invertebrate communities. Based on an updated search of TERRETOX (EPA's terrestrial toxicity database), 50 records were available for the two detected contaminants, PCE and TCE. Based on this TERRETOX search (EPA 2003a), the following may be concluded.

PCE

Three discrete terrestrial community studies were identified in the TERRETOX database (EPA 2003a) for PCE, and all investigated plant effects limited to lettuce (*Latuca sativa*) and tobacco (*Nicotiana tabacum*). Neither effects on population (measured in biomass) nor reproduction (germination) were recorded below 1,000 mg/kg. Therefore, the maximum LSGPS soil concentration in the biologically active depth (0.4 mg/kg) is well below any level of concern for the soil communities studied to date. No information was available to characterize soil invertebrate toxicity related to PCE exposure. Qualitatively, sub-part-per-million concentrations are not expected to result in invertebrate toxicity; although this remains a data gap, it is believed to be insignificant.

TCE

Eight discrete terrestrial community studies were identified for TCE in the TERRETOX database (EPA 2003a). Earthworm mortality (in *Eisenida fetida*) was reported in two references (but based on the same underlying laboratory study with filter paper) as 105 ug/cm^2; however not enough information was provided in the underlying study to convert these data to soil units of mg/kg. Of all the vegetation studies, the primary usable information indicated that growth effects concentrations (where 50 percent of plants would be affected), or EC50, are greater than 1,000 mg/kg for at least 14 different types of plants (Pestemer and Ausberg 1989 as cited in EPA 2003a), as confirmed by other researchers. Therefore the maximum LSGPS soil concentration in the biologically active depth (0.058J mg/kg) is well below any level of concern for the soil communities studied to date.

7.4 EXPOSURE ASSESSMENT

Because COPCs were identified at the AJ Gravel pond and because of a lack of ecological screening toxicity values appropriate for piscivorous avian species (see Section 7.2.1.3), Step 2 of EPA guidance (EPA 1997b), consisting of an exposure evaluation and risk characterization, was conducted. To evaluate potential ecological risk to piscivorous birds, food chain modeling was completed for the AJ Gravel pond. Food chain models assess the exposure of higher-level receptors to COPCs in ingested media. Protection of a piscivorous bird, the bald eagle, from effects on survival, fecundity, and growth was identified as the assessment endpoint (see Table 7-1) for the food-chain modeling at the LSGPS for three reasons. First, the bald eagle is a threatened species known to occur in Yellowstone County (see special status species discussion in Section 7.2.2.2). Second, the AJ Gravel pond lies adjacent to the Yellowstone River, contains appropriate prey (fish), thus providing potential (though less than ideal) forage for the bald eagle. Third, the bald eagle dietary modeling uses conservative parameters to protect individuals of the population and is therefore considered to be a conservative model. Estimated site doses were calculated using maximum surface water concentrations and estimated concentrations of chemicals in tissue. Tissue concentrations were calculated using conservative bioaccumulation factors for fish and site-specific surface water concentrations.

7.4.1 Exposure Point Concentrations

Based on the screening-level data gap for establishing screening values for piscivorous birds, concentrations of water (for ingestion) and in food (prey, assumed to be a diet of 100 percent fish from the third trophic level) were needed. Because (1) surface water concentrations in the AJ Gravel pond were the highest measured from surface water bodies during the RI, and (2) the size, depth, location, and food abundance in the AJ Gravel pond makes it marginal habitat for piscivorous birds, the maximum AJ Gravel pond surface water data was used as the water ingestion EPC.

In addition, food (fish) EPCs were calculated using the equation:

$$C_{Fish} = BAF_{TL3} * C_{Water}$$

where:

C_{Fish}	Concentration in trophic level 3 fish (µg/kg)
BAF_{TL3}	Trophic level 3 bioaccumulation factor (L/kg), taken from Table 3 of Sample and others (1996)
C_{Water}	Maximum concentration in AJ Gravel pond water (µg/L), taken from Table N-2.1

The bioaccumulation factor is the ratio of the concentration of a contaminant in tissue (mg/kg) to its concentration in water (mg/L), where both the organism and its prey are exposed, and is expressed as L/kg. Bioaccumulation factors were predicted by Sample and others (1996) by multiplying the bioconcentration factor for the contaminant by the appropriate food chain multiplying factor (see Table 3 of Sample and others 1996). The bioconcentration factors for the detected contaminants in surface water were obtained from Sample and others (1996), which were based on EPA (1995c as cited in Sample and others 1996) octanol-water partition coefficients. These bioconcentration factors were higher (i.e., more conservative) than the WQB-7 criteria bioconcentration factors as appropriate for the screening calculations.

Sample and others (1996) noted that mink, belted kingfisher, great blue heron, and osprey consume 100 percent trophic level 3 fish (EPA 1995b as cited in Sample and others 1996). Bald eagles would be expected to have a similar type of diet, such as when "pirating" food from osprey. Therefore, for the EPC calculations for prey of the piscivorous bird, the trophic level 3 food chain multiplying factor (appropriate for the octanol-water partition coefficient of the chemical) was applied as appropriate. Although this is an oversimplification of the bald eagle diet (which is known to contain other food items), fish assumed to be contaminated in the AJ Gravel pond provide a "worst case" dietary composition for evaluation of the LSGPS contaminants.

The food (in this case, fish) EPCs calculated for fish in the AJ Gravel pond are shown in Table N-3 of Appendix N.

7.4.2 Contaminant Dose Estimates

For upper trophic level organisms, such as the piscivorous/carnivorous bald eagle, exposure is assessed by quantifying the daily dose of ingested contaminated food items and ingested media. COPC daily dose ingested depends on (1) the COPC concentration in food items and media, (2) the measurement endpoint receptor's trophic level (i.e., top carnivore), (3) the trophic level of animal food items (i.e., prey), and (4) the measurement endpoint receptor's food and media ingestion rates (EPA 1999a). The complexity of the daily dose equation depends on (1) the number of food items in a measurement receptor's diet, and (2) the trophic level of each prey item and of the measurement endpoint receptor. The daily dose of the COPC ingested by a measurement endpoint receptor, considering all food items and media ingested, was calculated using the following generic equation:

$$D = (IR_F * C_i * P_i * F_i) + (IR_M * C_M * P_M)$$

where

D	Daily dose of COPC ingested (mg COPC/kg BW-day)
IR_F	Measurement endpoint receptor food item ingestion rate (kg/kg BW-day)
C_i	COPC concentration in *i*th food item (mg COPC/kg)
P_i	Proportion of *i*th food item that is contaminated; may be equivalent to area use factor (unitless)
F_i	Fraction of diet consisting of food item *i* (unitless)
IR_M	Measurement endpoint receptor media ingestion rate (kg/kg BW-day [soil or sediment] or L/kg BW-day [water])
C_M	COPC concentration in media (mg/kg [soil or sediment] or mg/L [water])
P_M	Proportion of ingested media that are contaminated (unitless)

The daily dose of COPC ingested by a measurement endpoint receptor was determined by summing the contributions from each contaminated medium and food item, and setting all proportions to unity. This approach assumes that 100 percent of the measurement receptor's diet (total daily mass of food items ingested, and all water consumed) is contaminated. In the case of the LSGPS bald eagle, no incidental soil or sediment ingestion was assumed, since LSGPS sediments only contain trace detectable sediment toluene (see Table N-2.6). This approach is inherently conservative, because it assumes the piscivorous bird forages exclusively from the AJ Gravel pond each day, when in reality, the size and attractiveness of this area would not support long-term eagle forage.

Relevant inputs to the equation were obtained from the *Wildlife Exposure Factors Handbook* (EPA 1993). Although males and females have most of the same characteristics, females are usually larger. Adult males measure about 3 feet from head to tail and weigh 7 to 10 pounds. Adult females can weigh as much as 14 pounds. Male bald eagles have wingspans of about 6½ feet, while a female bald eagle may have a wingspan of 8 feet.

The bird's diet is composed mostly of fish, but it will also prey on rodents, ducks, snakes, and carrion. For the LSGPS assessment, however, since the primary contaminated medium is surface water and since fish would be expected to accumulate the contaminants more than other classes of prey, a diet of 100 percent fish was assumed, and the F term was set to unity. For daily consumption rates, 0.12 kg/kg-day was assumed (equivalent to a consumption rate of 360 grams of fish per bird per day), based on two studies of free-flying eagles cited in the *Wildlife Exposure Factors Handbook* for adult eagles of both sexes (EPA 1993). For water consumption, 0.036 L/kg-day water intake was assumed, based on an average of the adult male and female estimates (EPA 1993).

Based on the fish EPCs given in Table N-3 and the maximum water concentrations given in Table N-2.6, the bald eagle's daily dose was calculated, as shown in Table N-4.

7.5 TOXICITY ASSESSMENT

The screening-level toxicity assessment included the compilation of screening-level toxicity benchmarks for each of the relevant LSGPS media. Because the screening-level toxicity benchmarks were used in the COPC selection, they are introduced in Section 7.3.

In addition to the screening-level toxicity benchmarks, however, additional effort was made to update the toxicity assessment for chlorinated solvent effects reported in the literature since 1995 and to attempt to fill data gaps in the screening-level toxicity database (such as the lack of toxicity values for toluene for direct sediment contact). Therefore, this section summarizes the general ecological toxicity associated with the chlorinated solvents (Section 7.5.1) and the additional literature search approach and findings (Section 7.5.2). Details of the screening-level benchmarks used are included in Appendix O.

7.5.1 Ecological Effects of Chlorinated Solvent Exposure

As with most target analytes, toxicity of the chlorinated solvent COPCs has been studied in laboratory mammals, with a wide range of effects noted (see Section 5.3 of this RI report for a brief summary of mammalian toxicity). Aquatic species have also been studied sufficiently to provide a database for key laboratory species. Avian species, however, have been little studied with regard to their sensitivity to chlorinated solvents. Each of the following sections highlights the potential effects of exposure considered in the LSGPS ERA.

During the ecological effects review, as noted in Section 7.2.2.3, particular care was taken to evaluate toxicological effects that could result in ecologically relevant impacts (EPA 1998) In the case of the LSGPS ERA, anticipated toxic responses are the measurement endpoints being used to make inferences about risks to the assessment endpoints described in Section 7.2.2.3. Where toxicity data are limited, however, (such as for VOC effects on avian species), less can be said about protection of the measurement endpoint because toxic effects being used as the measurement endpoint are not well characterized.

Thus, toxic impacts on survival (which may be reduced by direct lethality or by sublethal effects that diminish survival probabilities), fecundity (the production of viable young), and growth (increased mass or length) of some proportion of the animals or plants in an assessment population or community are the basic measurement endpoints for nonhuman organisms (EPA 2002e). In addition, species may be protected at the organism level (not as simply part of a population or community), and includes those species that are protected by statute: (a) Endangered and threatened species (i.e., those listed by the USFWS or the National Marine Fisheries Service as in danger of extinction under the Endangered Species Act), (b) Bald eagles and golden eagles, which are protected by the Bald Eagle Protection Act, and (c) nearly all birds in the U.S., including their eggs and nests, which are protected by the Migratory Bird Treaty Act (EPA 2002e).

As recommended in the most recent general assessment endpoints guidance (EPA 2002e), when a list of potential assessment endpoints has been developed, it may be necessary to review the list and reduce it to the set that is important to the decision. Because of the limitations of time and resources, it is often advisable to limit the list of assessment endpoints to those that are most relevant and susceptible. In the case of the LSGPS, for example, the assessment endpoints of the ERA are in contrast to the HHRA for the LSGPS, which focused primarily on the carcinogenic effects of the chlorinated solvents. Mammalian studies have shown that many of the chlorinated solvent COPCs do result in various types of cancer; however, on the ecological scale, lifetime carcinogenicity studies may be less relevant to the survival of wildlife than more imminent, noncancer effects. Carcinogenicity, included in “deformities, lesions, or tumors in animals” among the possible “gross anomaly” endpoints as used in EPA (2002e), was not an endpoint considered in the LSGPS ERA, since more sensitive endpoints were available for assessment.

7.5.1.1 Ecological Effects Summary for PCE

The general mammalian effects of PCE were summarized in Section 5.1.3 of this RI report. No avian toxicity data were available (see also Sections 7.5.2.1 and 7.7.3) to assess PCE in this ERA.

PCE is moderately toxic to aquatic organisms. Toxicity values are reported in the range of greater than 4.0 mg/L to 46 mg/L (EPA 2003b). The 96-hr LC_{50} values for freshwater fish species are 4.0 to 8.43 mg/L for the American flagfish (*Jordanella floridae)*, 13 mg/L for the bluegill sunfish (*Lepomis macrochirus)*, and 13.4 to 21.4 mg/L for the fathead minnow (*Pimephales promelas)*.

No information was available to assess the toxicity of PCE to amphibian or reptile species. Although limited, some information was available to assess the toxicity of PCE to earthworms and plants (see Section 7.3.3.3).

7.5.1.2 Ecological Effects Summary for TCE

The general mammalian effects of TCE were summarized in Section 5.1.3 of this RI report.
Avian toxicity to TCE has not been widely studied; however, the literature search conducted for the LSGPS ERA (Section 7.5.2.1) did identify a mixtures study that found possible avian effects on reproductive performance, egg quality, embryo toxicity, and immune function. When injected into avian embryos (eggs), TCE has been found to be teratogenic, with targeted effects on the heart (see Section 7.5.2.1).

TCE has been rather widely studied in freshwater invertebrate and fish species. The most sensitive species and ecologically relevant effects were summarized in the aquatic benchmarks discussed in Section 7.3.1.

No information was available to assess the toxicity of TCE to amphibian or reptile species, with the exception of the single LC50 value identified for a non-native salamander species (see Section 7.5.2.3). Although limited, some information was available to assess the toxicity of TCE to earthworms and plants (see Section 7.3.3.3).

7.5.1.3 Ecological Effects Summary for Cis-1,2-DCE

Although DCE is composed of two isomers, only cis-1,2-DCE was detected in the media relevant for ecological exposure (i.e., soil, sediment, and surface water) and therefore cis-1,2-DCE is the primary isomer of interest for the ERA. Unfortunately, little information is available to assess the ecological toxicity of this contaminant (see also Sections 7.5.2 and 7.7.3). No information was available to assess the toxicity of DCE to amphibian or reptile species. No information was available to assess the toxicity of DCE to earthworms or plant species.

Only one freshwater toxicity study was located in AQUIRE (EPA 2003b) for the mixed isomers of 1,2-DCE. Buccafusco and others (1981 as cited in EPA 2003b) conducted static acute toxicity tests using young of the year bluegill (*Lepomis macrochirus*). The 96-hr LC_{50} was 140 mg/L for bluegill. No other information on the effects of cis-1,2-DCE on freshwater aquatic species is available.

7.5.1.4 Ecological Effects Summary for VC

The general mammalian effects of VC were summarized in Section 5.1.3 of this RI report. No avian toxicity data were available (see also Sections 7.5.2.1 and 7.7.3) to assess VC in this ERA. No information was available to assess the toxicity of VC to amphibian or reptile species. No information was available to assess the toxicity of VC to earthworms or plant species.

VC does not typically persist in surface water long enough to be assessed in aquatic toxicology tests. Four studies were identified in AQUIRE (EPA 2003b) for VC (under the synonym chloroethene), representing effects studied in ciliates (single-celled protozoa), mosquitoes, water fleas, pouch snails, western mosquitofish, and northern pike. Concentrations of 388 mg/L are fatal to northern pike; all other AQUIRE data were less relevant, because they were related either to general accumulation data in these other species or general population changes in ciliates (EPA 2003b).

7.5.2 Literature Search Approach and Findings

Based on specific data gaps identified in the ERA process at the LSGPS, an expanded toxicity data literature search was conducted to (1) identify avian toxicity reference values for all of the LSGPS COPCs, (2) identify a mammalian cis-1,2-DCE toxicity reference value, and (3) identify amphibian/reptile toxicity reference values.

7.5.2.1 Avian Toxicity Reference Value Literature Search

To evaluate the literature relevant to bird species, an online search of the open scientific literature was conducted. Of 130 records identified for the search terms “bird and solvent,” only a handful were relevant to the LSGPS COPCs. Those that were found to discuss either PCE, TCE, cis-1,2-DCE, or VC are summarized below.

A study on 1,1-dichloroethylene in a chick model was identified (Goldberg and others 1992), but no studies with 1,2-DCE were identified. A TCE study in an avian model (Loeber and others 1988 as cited in Goldberg and others 1992) confirmed TCE as an avian teratogen, with significantly increased numbers of malformations and embryonic death in treated groups compared with controls (Loeber and others 1988 as cited in Goldberg and others 1992). TCE-treated embryos had more than 3 times as many congenital heart deformities compared to control groups, establishing TCE as a cardiac teratogen in an avian model (Loeber and others 1988 as cited in Goldberg and others 1992). However, the relevance of the exposure route employed in this study (involving injection of TCE into eggs) is questionable, since the high concentrations of TCE used in the study are not expected to result within the eggshell based on low-level environmental exposure of avian parents.

Effects on reproductive performance, egg quality, embryo toxicity, and immune function in broiler breeder hens were assessed based on a drinking water mixture containing TCE and other contaminants (arsenic, cadmium, lead, and benzene). This study (Vodela and others 1997a,b) conducted with TCE concentrations of 0.65 mg/L and 6.5 mg/L TCE, identified a linear relationship between increasing concentration of the chemical mixture in drinking water and decreasing body weight of hens. The low concentration of the chemical mixture significantly decreased egg production and egg weight, increased percentage embryonic mortality, and suppressed immune function (Vodela and others 1997a,b). However, the observed effects cannot be attributed to the TCE concentration alone, and may be related either to the effects of the mixture as a whole or the presence of any other contaminant or combination thereof (including 0.8 mg/L arsenic, 1.3 mg/L benzene, 5.1 mg/L cadmium, and 6.7 mg/L lead). Conservatively, the effects could be entirely attributed solely to TCE and the concentration of 0.65 mg/L TCE adopted as a drinking water benchmark for avian species. No LOAEL-to-NOAEL conversion factor was applied, because of the inherent conservatism already present when assuming the mixture's effects were solely due to TCE. Assuming a typical chicken body weight of 1.5 kg (EPA 1988b as cited in Sample and others 1996), and using the allometric equation for water consumption from the Wildlife Exposure Factors Handbook (Equation 3-15 of EPA 1993), this is the equivalent to a final avian TCE toxicity reference value of 0.034 mg/kg-day.

No other toxicity values could be identified for avian species in the scientific literature. Therefore, the toxicity of PCE, cis-1,2-DCE, and VC to avian species remains a data gap.

7.5.2.2 Mammalian Cis-1,2-DCE Toxicity Reference Value Literature Search

No toxicity reference value was available for cis-1,2-DCE (see Section 7.7.3), as confirmed through a check of studies compiled in IRIS; no toxicological profile (which reviews mammalian effects) has been compiled by ATSDR for 1,2-DCE. The Superfund Health Risk Technical Support Center had no additional information on the toxicity of 1,2-DCE.

7.5.2.3 Amphibian/Reptile Toxicity Reference Value Literature Search

The open literature (including online toxicity search engines such as TOXLINE) was searched for solvent effects on amphibians and reptiles. The AQUIRE database was checked to evaluate whether aquatic amphibious species may have been evaluated for these chemicals. In addition, the search encompassed recent compilations such as *Ecotoxicology of Amphibians and Reptiles* (Sparling and others 2000), review articles (such as Campbell and Campbell 2002), and relevant papers from annual Society of Environmental Toxicology and Chemistry meetings for the past three years.

Of all the LSGPS COPCs in surface water, these references and the AQUIRE database (EPA 2003b) contained only one relevant amphibian study. AQUIRE reported TCE LC_{50} information for a salamander species, the Mexican axolotyl (*Ambystoma mexicanum*), at 32-56 mg/L over 24 hours or a mean of 48 mg/L over 48 hours (Sloof 1982 as cited in EPA 2003b). However, this species is not indigenous to Montana and therefore the relevance of the sensitivity of this species to native LSGPS amphibians is uncertain. Qualitatively, the maximum concentration of 0.41J mg/L TCE detected in LSGPS surface water is orders of magnitude below this single amphibian LC_{50} and therefore, no risk of mortality would be suspected. No amphibian benchmarks were available for PCE, cis-DCE or VC, and therefore a toxicity data gap was concluded (see Section 7.7.3).

No reptile toxicity data were located for the LSGPS chlorinated COPCs. Reptiles as a group have been largely neglected in toxicological studies. Overall, the reptilian toxicological literature has major gaps and >90 percent of all published papers deal only with whole body and/or tissue residue analysis (Sheffield and others 2000). Since no reptile benchmarks were available for PCE, TCE, cis-DCE, or VC, a toxicity data gap was concluded (see Section 7.7.3).

7.6 RISK CHARACTERIZATION

Section 7.6.1 presents the risk characterization summary using the screening-level toxicity benchmarks shown in Tables N-2.1 through N-2.8. Section 7.6.2 presents the risk characterization for the bald eagle. HQs (see Section 7.6.2) were calculated for COPCs by comparing site doses with toxicity reference values derived from the original literature (see Section 7.5.2).

7.6.1 Summary of Screening

As described in the COPC selection process outlined in Section 7.3, the LSGPS ERA was focused at the screening-level stage, because of the following:

- All surface water concentrations at the LSGPS were below conservative direct contact and food-chain-protective aquatic benchmarks (see Tables N-2.1 through N-2.5).
- All sediment concentrations at the LSGPS were below sediment benchmarks (see Table N-2.6).
- All soil concentrations at the LSGPS were below soil benchmarks (see Sections 7.3.3.2 and 7.3.3.3). Soil contamination was detected only in the top 2 feet of soil in the Beall Trailers industrial property; no contaminants were detected in the top 2 feet of soil in the Brenntag area. The COPC screen discussed in Section 7.3.3.2 concluded that risks were acceptable.

7.6.2 Bald Eagle Hazard Quotients

As shown in Table N-4, the available toxicity reference values were applied. Based on the conservative daily intake where 100 percent of the eagle's diet was assumed to be contaminated fish from the AJ Gravel pond contaminated with the maximum detected concentrations of VOCs, and that 100 percent of the eagle's daily water intake was contaminated at the maximum VOC concentration, an HQ of 1 was concluded. Therefore, since a hazard quotient greater than unity was not calculated, no unacceptable risks to bald eagles are concluded.

7.6.3 Risk Characterization Conclusions

The results of the screening-level ERA for the LSGPS indicate the site does not pose unacceptable risk to ecological receptors. Based on the ERA, no action is recommended for risk management to address ecological risk at the LSGPS.

7.7 UNCERTAINTY ANALYSIS

Below are the primary uncertainties associated with the LSGPS ERA, which relate to the problem formulation, COPC selection, exposure assessment, and toxicity evaluation.

7.7.1 Uncertainty in Problem Formulation and COPC Selection

Conceptual models formed during the problem formulation represent one of the most important sources of uncertainty in an ERA (EPA 1998). As noted by EPA (1998):

> If important relationships are missed or specified incorrectly, the risk characterization may misrepresent actual risks. Uncertainty arises from lack of knowledge about how the ecosystem functions, failure to identify and interrelate temporal and spatial parameters, omission of stressors, or overlooking secondary effects. In some cases, little may be known about how a stressor moves through the environment or causes adverse effects. Multiple stressors are the norm and a source of confounding variables, particularly for conceptual models that focus on a single stressor.

Although attempts to minimize these sources of uncertainty were maximized at the LSGPS, oversimplification of ecological relationships is inherent in every ERA. In the case of the LSGPS, the COPCs are not widely investigated in ecological receptors, leading to considerable assumptions about how these COPCs may affect ecosystems.

Uncertainty in the screening approach includes the use of screening-level benchmarks which, although thought to be conservative and scientifically valid, may not represent every species present at the LSGPS. Unfortunately, toxicity data are not available for most species and, therefore, no real alternative to such a generalized screening approach is an option.

Where no screening-level benchmark exists (see also Section 7.7.3), COPC selection may not give a full picture of all ecological stressors due to this lack of information. However, the LSGPS ERA is as complete as possible given the current state of the science.

7.7.2 Uncertainty in Exposure Assessment

As with the HHRA, uncertainty is inherent in each of the exposure assumptions made in the ERA. Because a screening-level approach was used, the exposure assessment assumptions are inherent in the screening-level media benchmarks developed by Sample and others (1996). For the bald eagle assessment, the most conservative assumptions (i.e., 100 percent forage from the maximally contaminated area) were assumed. This resulted in an overestimate of risks.

Overall, the lack of quantitative assessment of inhalation and dermal exposures is a data gap in the exposure assessment, largely due to an absence of relevant exposure information in wildlife species. However, based on the conceptual site model for the LSGPS, these exposures are not believed to be significant.

As an example, the available inhalation data indicate that much higher concentrations of TCE would need to be present in soil to present a risk. Carlsen (1996) reviewed risks to burrowing animals from TCE inhalation, and noted that the available literature for TCE indicated that concentrations of 733 mg/m^3 were required to result in avoidance by rats (Goldberg and others 1964 as cited in Carlsen 1996), and 3,813 mg/m^3 was required to result in weight loss in dogs (Prendergast and others 1967 as cited in Carlsen 1996). The subparts per million concentrations present in LSGPS soils (see Table N-2.8) would not result in burrow concentrations even approaching these levels of concern, and therefore, the inhalation pathway is insignificant, even for burrowing receptors.

8.0 CONCLUSIONS AND RECOMMENDATIONS

This section presents the conclusions of the RI performed at the LSGPS, identifies data gaps and recommendations to address the data gaps, and provides a brief description of the FS process.

8.1 CONCLUSIONS

The objectives of the RI at the LSGPS were to delineate the nature and extent of contamination at the site, evaluate fate and transport of contamination, and evaluate risks to human health and the environment. These objectives have generally been met; exceptions and data gaps are explained in this section and recommendations are provided. The RI identified two sources of groundwater contamination within the site: the Brenntag source area and the Beall source area. Portions of the groundwater within and downgradient of these sources contain chemicals at concentrations greater than both federal and state regulatory standards and pose unacceptable risks to human health. Soil contamination at the site does not pose unacceptable risks to humans. Soil, groundwater, surface water, or sediment contamination does not pose an unacceptable risk to ecological receptors at the site or within the Yellowstone River. Conclusions for the LSGPS RI program are described further in the following sections.

8.1.1 Nature and Extent of Contamination

Concentrations of COPCs above SSLs were reported in subsurface soil samples taken from sample locations associated with the Brenntag source area. The limits of soil contamination at the Brenntag source area have not been fully defined and additional work is being proposed to address this data gap (Section 8.2). A preliminary estimate of the volume of contaminated soil is 35,500 cubic yards. Concentrations of COPCs above SSLs were also reported in subsurface soil samples taken from locations associated with the Beall source area. The limits of soil contamination at the Beall source area have not been fully defined and additional work is being proposed to address this data gap (Section 8.2). Because of the limited width of potentially contaminated soil in this area features, the volume of contaminated soil has not been estimated.

The highest concentrations of VOC groundwater contamination at the LSGPS have been reported in the portions of the plume within and downgradient of the Brenntag and Beall source areas. In addition, lower level concentrations of VOC groundwater contamination have been reported throughout the site. This contamination has generally been delineated on the southern and eastern boundaries by samples with no COPCs above laboratory detection limits. The southern boundary is also delineated by the alluvial aquifer/bedrock boundary. Some additional work is being proposed to better define the plume boundary in some locations (Section 8.2). The plume is delineated on the western and northern edges by the Yellowstone River where the alluvial aquifer is considered to discharge to the river.

The only surface water samples with COPC concentrations greater than regulatory criteria were located within the AJ Gravel pond. COPCs in the AJ Gravel pond are likely the result of the discharge of contaminated groundwater to surface water in the pond. No sediment samples exhibited COPC concentrations above SSLs. COPCs detected in Coulson irrigation ditch water were below regulatory criteria; contaminated groundwater likely discharges into the ditch during nonirrigation flow periods. The greatest volume of groundwater discharge to Coulson irrigation ditch appears to occur downstream of surface water station SW300 during nonirrigation flows. When there is sufficient water, contaminated surface water in Coulson irrigation ditch flows off-site to the east. When irrigation flows are present in the ditch, ditch water levels are elevated above those in adjacent groundwater and little or no groundwater discharge to the ditch is expected.

8.1.2 Contaminant Fate and Transport

Contaminant fate and transport was investigated by (1) evaluating geochemical and hydrological data, and statistical analysis of groundwater quality data, (2) development and calibration of contaminant transport models using BIOCHLOR, and (3) developing a groundwater-surface water mixing model to estimate mass flux of COPCs to the Yellowstone River.

Geochemical characterization of the Brenntag source area indicates that active reductive dechlorination is taking place in the source area and in the portion of the plume immediately downgradient. The portion of the plume within and downgradient of the Brenntag source area appears to be exhibiting Type I plume behavior; man-made carbon from released petroleum and possibly other compounds act as the electron donor supply. Geochemical characterization of the Beall source area indicates limited evidence for active reductive dechlorination. Low-level concentrations of TCE and DCE remain relatively constant in the aquifer northwest of the Interstate Highway 90 corridor. The Beall source area exhibits Type III behavior; dissolved oxygen concentrations exceed 1.0 mg/L. Historically, some biodegradation may have occurred because the DCE daughter product is present. Data suggests that biodegradation may have taken place until available natural and man-made carbon was exhausted.

Hydrogeologic data were evaluated to quantify groundwater flow gradients and directions, to assess water transmitting properties of the aquifer, to estimate the velocity of groundwater flow, and to estimate the transport of COPCs in groundwater. Based on the model calibrated hydraulic conductivity data, the site-specific seepage velocity in Area A was estimated to be about 1.5 ft/day or about 558 ft/year. In Area B, the seepage velocity was estimated to be about 0.0356 ft/day or about 13.0 ft/year.

Mann-Kendall statistical tests for groundwater concentration trends were used to support conclusions about changes in the size of the plume over time. The maximum length and width of the portion of the contaminant plume near the Brenntag source area appears to be stable. The maximum length and width of the portion of the contaminant plume near the Beall source area appears to be increasing. In addition, data from wells installed prior to the RI support a conclusion that biodegradation is active in portions of the plume downgradient of the Brenntag source area.

For the portion of the plume in Area A downgradient of the Brenntag source area, BIOCHLOR model results (assuming a release date of 1972) suggest that PCE and DCE in the downgradient edge of this portion of the plume may have reached the Yellowstone River as early as model simulation year 5 (calendar year 1977). PCE and DCE concentrations appear to have reached stable conditions by model year 10 (calendar year 1982); no significant changes in the length or width of this portion of the plume have taken place after that time. For the portion of the plume in Area B downgradient of the Beall source area, BIOCHLOR model results support a conclusion that this portion of the plume is slowly increasing in size.

A groundwater-surface water mixing model was used to assess the impact of groundwater contamination on surface water quality in the Yellowstone River. Conservative or worst-case model results indicate that the incremental impact of COPCs in groundwater on surface water quality in the Yellowstone River would not be measurable at the lowest mean monthly flow in the river for the period of record.

8.1.3 Human Health and Ecological Risk Assessments

Complete baseline human health and screening-level ecological risk assessments were conducted under the RI program. Media samples were collected across the LSGPS for use in characterizing risks to human health and the environment. The conclusions of the human health and ecological risk assessments are summarized in the following sections.

8.1.3.1 Baseline Human Health Risk Assessment

Cancer risks and noncancer hazard indices were calculated for an array of current and future residential and industrial exposures at the LSGPS. Both the reasonable maximum exposure and a central tendency exposure scenario were considered for each receptor selected for quantitative assessment.

The human health risk assessment concluded that the following scenarios are within the risk management range or considered to be insignificant risks under the NCP:

- Resident adults and children in each of the subareas, assuming only indoor air inhalation exposures as a result of subsurface vapor intrusion.
- Resident adults who use contaminated well water to wash cars or irrigate their lawn in each of the subareas.
- Resident adolescents who recreate with contaminated well water in kiddie pools or sprinklers in each of the subareas.
- Recreators who fish from or wade/dip their arms in the AJ Gravel pond.

- Utility/construction workers in any of the subareas.
- Industrial workers in Area A nonsource, Area B source, Area B nonsource, and Area C subareas who use the public water supply or are supplied an alternate source of drinking water.
- Resident adults and children in Area A nonsource. Area B nonsource, and Area C subareas who use groundwater as a potable water source for whole-house use and/or drinking water source.
- Industrial workers in Area A nonsource. Area B nonsource, and Area C subareas who use groundwater as a potable water source for interior use and/or drinking water source.

Those scenarios and receptors that had cancer risks that may indicate the need for further evaluation or remediation are as follows:

- Industrial workers in Area A source subarea that use groundwater as a potable water source for interior use and/or drinking water source but may spend 4 hours of each workday in contact with Area A source subarea groundwater.
- Resident adults and children in Area A source and Area B source subareas that use contaminated groundwater for whole-house use, including bathing, drinking, and washing.
- Industrial workers in Area A source and Area B source subareas that use contaminated groundwater for unrestricted workplace use, including drinking and washing.

Central tendency or more “average” risks were lower than those projected for the high-end reasonable maximum exposure scenarios summarized above. Important uncertainties in these reasonable maximum exposure conclusions were summarized in the baseline human health risk assessment. After consideration of these findings and identification of applicable or relevant and appropriate requirements for the LSGPS, remedial action objectives may be proposed to address these potential human health risks.

8.1.3.2 Screening-Level Ecological Risk Assessment

The ecological risk assessment included a detailed screening of all detected chemicals in each medium sampled at the LSGPS specifically for ecological effects. The most conservative available ecological screening benchmarks were employed, with an updated toxicity effects literature search in an attempt to close data gaps identified in the screening process. The LSGPS ecological risk assessment found:

- All surface water concentrations at the LSGPS (including the maximum detected concentrations present in the AJ Gravel pond as well as modeled concentrations in the Yellowstone River) were below conservative direct contact and food-chain-protective aquatic benchmarks.

- All sediment concentrations at the LSGPS were below sediment benchmarks.

- All soil concentrations at the LSGPS were below soil benchmarks.

- A conservative desktop food model was employed to evaluate top-level avian carnivores, such as the bald eagle, because of that species' special status and possible home range overlap with the LSGPS. Based on an assumption that 100 percent of the eagle's diet was contaminated fish from the AJ Gravel pond (contaminated with the maximum detected concentrations of VOCs in any surface water body at the LSGPS), and that 100 percent of the eagle's daily water intake was contaminated at the maximum VOC concentration, a hazard quotient of 1 was concluded. Therefore, since a hazard quotient greater than unity was not calculated, no unacceptable risks to bald eagles were concluded.

The results of the ecological risk assessment for the LSGPS indicates the site does not pose an unacceptable risk to ecological receptors. Based on these findings, no action is required to address ecological risk at the LSGPS.

8.2 RECOMMENDATIONS

After a review of all data collected during the RI, several data gaps have been identified. To address these data gaps, additional work is recommended at the LSGPS and is described in the following sections.

8.2.1 Area A

Some data gaps have been identified and additional work is recommended at the site to address these data gaps. The following additional work is recommended at the Brenntag source area and the portion of the plume downgradient of the Brenntag source area:

- Groundwater samples collected from the Brenntag source area monitoring wells indicate a release of an unknown type of petroleum hydrocarbon in the vicinity of wells MW101, MW102, and MW104 that is commingled with chlorinated compounds. Other organic compounds, such as ethanol and methanol, that are stored at the Brenntag facility may have also been released and may act as significant sources of carbon. Groundwater samples were analyzed for VOCs only and cannot be used for identification or quantification of other product types such as petroleum hydrocarbons. Quantification of the available carbon mass in the source area is necessary to evaluate attenuation and reductive dechlorination processes.
- The installation and sampling of monitoring wells PT001 through PT004 was conducted by SECOR in December 2001. The information was provided to TtEMI in the fall of 2003 after RI

activities were completed. Soil and groundwater data from monitoring well PT002 indicate that residual NAPL may be present. Additional borings and sampling should be conducted in the vicinity of monitoring well PT002 to evaluate the extent of the soil contamination and NAPL. Installation of additional soil borings to the west of PT002 on Keller Transport property and to the east of PT002 toward the tank farm area should also be considered.

- A portion of the plume in Area A daylights or discharges into the AJ Gravel pond. The flux of this contaminant mass cannot be estimated since the pond depth is not known. The bottom of the AJ Gravel pond needs to be profiled in order to calculate the mass of COPCs entering the pond and estimate the contaminant mass loss from the groundwater to surface water.
- Surface water samples collected from Coulson irrigation ditch during the RI, indicate that groundwater containing COPCs is entering the ditch at station SW300 and elsewhere downstream to Klenck Lane (LSGPS north boundary). In order to estimate the mass flux of contamination into the Coulson irrigation ditch and the potential for transport downstream, an accurate bottom profile of the ditch should be made.

8.2.2 Area B

The following additional work is recommended at the Beall source area, the portion of the plume downgradient of the Beall source area, and in other locations in Area B:

- The limits of source material at the Beall source area have not been fully defined. Additional source material is likely found beneath the oil-water separator and associated steam clean bay drainage pipe, based on groundwater COPC concentrations in this area. Because of the limited width of potentially contaminated soil beneath these features, the volume of this source area is difficult to estimate. Additional subsurface investigation of this area may further delineate this source but it may be difficult to locate contaminated soil without removal and inspection of the oil-water separator and piping. Investigation by excavation of the oil water separator and steam clean bay drain lines should be conducted to delineate the location of the potential soil contamination source beneath these features.
- Monitoring well MW202, the background monitoring well at the Beall Trailers facility, contained detectable cis-1,2 DCE (0.22 µg/L) during the October 2002 sampling event. A new upgradient monitoring well should be installed further east of MW202 in order to obtain background water quality conditions at the facility.
- Monitoring well MW219, the bedrock well at the Beall source area, contained detectable VOCs during both the August and October 2002 sampling events. The sample collected from the October event contained 12 µg/L of TCE (above the regulatory limit of 5 µg/L) and 1.8 µg/L of DCE. The delineation of the VOC plume both vertically and horizontally within the bedrock aquifer has not been completed and should be performed by the installation of up to five additional bedrock monitoring wells.
- The delineation of the higher concentrations of VOCs in the portion of the plume downgradient of U.S. Highway 87 East is likely incomplete. Total VOC concentrations in monitoring well MW210 (214.07 µg/L) are higher in comparison to wells located along the downgradient (northwest side) of the Beall facility. Total VOC concentrations in monitoring well MW210 was 47 to 88 percent higher than in monitoring wells MW203 (64.43 µg/L), MW204 (114 µg/L),

MW205 (65 µg/L), and MW206 (25.96 µg/L) indicating a possible slug of VOC contamination has moved from the Beall source area and may be present in the residential area northwest of U.S. Highway 87 East. Two to three additional alluvial aquifer monitoring wells should be completed in the residential area northwest of the Beall source area and northeast of monitoring well MW210.

- TCE concentrations detected in groundwater samples from monitoring wells MW207 and MW209 exceed the MCL/WQB-7 criteria at concentrations of 28 and 5.4 µg/L. DCE was also detected in both monitoring wells and PCE was detected in monitoring well MW207. The northeastern boundary of the plume in the vicinity of monitoring wells MW207 and MW209 remains undefined and is tentatively defined by residential wells located east of MW207 and MW209. Additional monitoring wells should be considered for installation northeast of monitoring wells MW207 and MW209 to further define the plume boundaries.

8.2.3 Area C

No additional work is recommended in Area C.

8.3 FEASIBILITY STUDY PROCESS

The FS is a continuation of the RI in the RI/FS process under CERCLA (EPA 1988). After the site has been characterized in the RI process, the FS is used to develop and evaluate remedial alternatives that can be used to remediate certain areas at a site that are identified as posing an unacceptable risk to human health or the environment. The development of remedial alternatives requires the following:

(1) identifying federal and state, chemical- and location-specific applicable or relevant and appropriate requirements;

(2) identifying remedial action objectives;

(3) identifying potential treatment, resource recovery, and containment technologies that will satisfy these objectives;

(4) screening the technologies based on their effectiveness, implementability, and cost; and

(5) assembling technologies and their associated containment or disposal requirements into alternatives for the contaminated media at the site.

Typically, a range of remedial alternatives are developed during the FS process varying primarily in the extent to which they rely on long-term management of the site contamination. The upper bound of the range is an alternative that would eliminate the need for long-term management (including monitoring) at the site. The lower bound is an alternative that involves treatment as a principal element, but some long-term management of portions of the site that did not constitute “principal threats” would be required (EPA

1988). Once potential alternatives have been developed, certain alternatives may be screened out to reduce the number of options that are analyzed in detail in order to minimize the resources dedicated to evaluating options that are less promising.

Should existing site data be insufficient to adequately evaluate remedial alternatives, treatability tests may be necessary during the FS to evaluate a particular technology on specific site contaminants. Generally, treatability tests involve bench-scale testing to gather information to assess the feasibility of the technology.

Once sufficient data are available, remedial alternatives are evaluated in detail with respect to nine evaluation criteria that the EPA has developed to address the statutory requirements and preferences of CERCLA (EPA 1988) and the NCP. The results of the detailed analysis are summarized and presented so that an appropriate remedy consistent with CERCLA and the NCP can be selected.

In general, the results of the FS then lead to the selection of a remedy or remedies, preparation of a record of decision document; development of the remedial design, and implementation of the remedial action.

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TABLES

FIGURES