DRAFT
RISK ASSESSMENT FOR THE
SIEMENS WATER TECHNOLOGIES CORP.
CARBON REACTIVATION FACILITY
PARKER, ARIZONA

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LIST OF ABBREVIATIONS

ACGIH	American Conference of Governmental Industrial Hygienists
AEGL	Acute exposure guideline level
AIHA	American Industrial Hygiene Council
APC	Air pollution control equipment
AZMET	Arizona Meteorological Network
ATSDR	Agency for Toxic Substances and Disease Registry
CALEPA	California Environmental Protection Agency
CFM	Cubic feet per minute
CRIT	Colorado River Indian Tribes
CRSSJV	Colorado River Sewage System Joint Venture
DRE	Destruction and removal efficiency
E	Exponent in the presentation of numerical results (e.g., 3E-4 = 3×10^{-4})
HEAST	USEPA Health Effects Assessment Summary Tables
HHRAP	Human Health Risk Assessment Protocol published in 2005 by USEPA
IH	Industrial hygiene
IRAP	Industrial Risk Assessment Program
IRIS	USEPA Integrated Risk Information System
ISCST3	Industrial Source Complex Short-Term 3 air model
NAAQS	National Ambient Air Quality Standard
NESHAPs	National Emission Standards for Hazardous Air Pollutants
NIOSH	National Institute on Occupational Safety and Health
NPDES	National Pollutant Discharge Elimination System
NWS	National Weather Service
OSHA	Occupational Safety and Health Administration
PCDDs/PCDFs	Polychlorinated dibenzo-p-dioxins and polychlorinated dibenzo furans
PDT	Performance Demonstration Test
PEL	Permissible exposure limit
PM	Particulate matter
PM10	Particulate matter less than 10 microns in diameter
PM2.5	Particulate matter less than 2.5 microns in diameter
POTW	Publicly owned treatment works
ppm	parts per million
RCRA	Resource Conservation and Recovery Act
REL	Acute inhalation reference exposure level
RfC	Inhalation reference concentrations
RfD	Non-cancer reference dose
PRG	Preliminary remediation goals
QA	Quality assurance
SWT	Siemens Water Technologies Corp.
TEF	Toxic equivalency factors
TEQs	Toxic equivalents
2,3,7,8-TCDD	2,3,7,8-Tetrachlorodibenzo-p-dioxin

LIST OF ABBREVIATIONS (Continued)

2,3,7,8-TCDF   2,3,7,8-Tetrachlorodibenzofuran
TIC   Tentatively identified compound
TOE   Total organic emissions
TWA   Time-weighted-average
USEPA   U.S. Environmental Protection Agency
USGS   U.S. Geological Survey
VOC   Volatile organic compound
WHO   World Health Organization
WQS   Water quality standards

EXECUTIVE SUMMARY

The Siemens Water Technologies Corp. facility (SWT facility) is a carbon reactivation plant located within the 269,000 acre Colorado River Indian Tribes ("CRIT") Reservation just outside of the Town of Parker in La Paz County, Arizona. The facility is located in an industrial park established by CRIT on Tribal land and is operated pursuant to a lease between the company and CRIT. The facility reactivates spent carbon which has been previously used to remove pollutants from water and air. The spent carbon is reactivated by heating it to very high temperatures under controlled conditions in a carbon reactivation furnace. The newly reactivated carbon product is then reused as an activated carbon product.

A human health and ecological risk assessment of the facility was conducted as part of the facility's permitting activities for the carbon reactivation furnace under the Resource Conservation and Recovery Act permitting regulations at 40 CFR §270.10. A risk assessment is a scientific study that can help evaluate risks associated with exposure to chemicals in the environment. This risk assessment represents one of the final steps in a process that has extended over a seven year period beginning with the U.S. Environmental Protection Agency's (USEPA's) request to develop a Risk Assessment Workplan. The risk assessment was conducted by a team of scientists and engineers from independent consulting firms with expertise in risk assessment, toxicology, environmental engineering and air dispersion modeling.

This risk assessment was performed according to a USEPA-approved Risk Assessment Workplan ("Workplan") developed in 2003, updated by agreement with the USEPA to include elements of more recent 2005 USEPA guidance for risk assessments of waste combustion facilities. The USEPA approvals were received prior to the initiation of this study which included evaluations of potential human health and ecological risks associated with both furnace stack air emissions and fugitive air emissions from spent carbon unloading. The assessment also included evaluations of potential risks associated with exposure to the facility's effluent discharge to the Colorado River Sewage System Joint Venture publicly owned sewage treatment plant and with exposure to airborne chemicals in the workplace at the facility.

The risk assessment used a large amount of site-specific data, including but not limited to:

- comprehensive testing of emissions from the furnace stack, with analysis for site-specific chemicals of potential concern;
- data on spent carbon characteristics, the facility configuration, and facility operations;
- local land use and demographic information;
- water resources data available from the U.S. Geological Survey and the U.S. Bureau of Reclamation; and
- meteorological data from Parker, Arizona.

In the absence of site-specific information, health-protective default values recommended by the USEPA were used. Chemical-specific toxicological data and chemical properties for the compounds selected for evaluation were obtained from the USEPA or from other public health agencies, organizations or databases primarily recommended by the USEPA. In addition, many mathematical models developed by the USEPA and presented in the Agency's guidance documents were applied to perform the risk assessment calculations. Overall, the models and input data used in the risk assessment are expected to provide conservative (i.e., health protective) estimates of potential risks.

Potential risks from stack emissions into the air were evaluated for over 170 compounds selected for detailed assessment based on a comprehensive performance demonstration test (PDT) approved in advance by the USEPA and conducted at the facility by an independent testing firm. The PDT involved several days of stack gas sampling and sophisticated chemical analysis. The list of chemicals selected for evaluation included compounds that were detected in stack emissions and also over 80 compounds that were not detected but were included in the calculations to ensure that risks would not be underestimated. Stack emission rates for the selected compounds were calculated based on either PDT results, proposed permit limits or, for a few chemicals, long-term average chemical feed rates and a conservative value for the furnace's destruction and removal efficiency. Potential risks from fugitive air emissions were evaluated for 21 compounds selected for evaluation based on their concentrations in spent carbon, the number of deliveries and amounts delivered to the facility, chemical toxicity, and volatility. Air dispersion and deposition modeling was conducted using a model developed and approved by the USEPA to allow calculation of chemical concentrations in air and deposition rates onto the earth's surface within a 154 square mile study area surrounding the facility. The mathematical equations used to calculate the fate and transport of each chemical in the environment, environmental concentrations for each chemical, and human exposures and risks, were based on current USEPA guidance and solved using the Industrial Risk Assessment Program software.

Human Health Risk Assessment

The stack emissions human health risk assessment calculated exposures for several different types of individuals who could hypothetically be exposed to emissions from the plant: adult and child residents, adult and child farmers, adults and children assumed to eat fish caught from the Colorado River or the Main Drain, and a nursing infant. In risk assessment terminology, these groups of individuals are known as "receptors". Each adult or child receptor was assumed to be exposed through a variety of pathways (e.g., the adult farmer receptor was assumed to be exposed via inhalation, soil ingestion, homegrown produce ingestion, and ingestion of home-raised or locally-raised beef, pork, poultry, and eggs). Each adult receptor was also conservatively assumed to be the mother of a breast-fed infant with the potential for transmission of chemicals from the mother through nursing. The fugitive emissions human health risk assessment evaluated inhalation exposures for adult and child residents, and adult and child farmers.

A variety of risk evaluations were performed in the human health risk assessment, as summarized below:

- Chronic long-term excess lifetime cancer risks from stack emissions were found to be at least five times lower than the USEPA's combustion risk assessment target level of 1×10^{-5} (one in 100,000) over a 70-year lifetime when all compounds were included. The excess lifetime cancer risks were reduced to 50 or more times lower than the target risk level when just one compound (that was not detected in the stack gases and has not been received at the facility in spent carbon) was removed from the analysis. Excess lifetime cancer risks due to inhalation of fugitive emissions were at least 200 times below the USEPA target risk level. The excess lifetime cancer risks would remain below the USEPA target risk level even if stack and fugitive emissions were considered together.

- Chronic long-term non-cancer effects from exposure to stack and fugitive emissions were predicted not to occur with a large margin of safety. Calculated exposures were at least 25 times lower and 250 times lower, respectively, than the conservative non-cancer target level used by USEPA for combustion sources, which is a hazard index value of 0.25.

- An analysis of short-term acute inhalation exposures showed that adverse acute effects would not occur with a large margin of safety at assessed residential locations and also at maximum impact points beyond the facility boundary.

- The calculated air and soil concentrations for residential receptors were determined to be below conservatively-derived preliminary remediation goals that have been developed by USEPA Region 9.

Ecological Risk Assessment

An ecological risk assessment was also conducted to evaluate potential effects of stack emissions on selected representative ecological receptors within the facility area. The ecological analysis evaluated potential impacts to wildlife that was considered to be at greatest risk based on habitat use, exposure potential, ecological significance, and population status. The habitat types that were considered consisted of creosote bush scrub, agricultural areas, riparian corridors and backwaters, the Colorado River, and the Main Drain. The species selected for evaluation consisted of aquatic life, plants, the badger, Gambel's quail, the great horned owl, the burrowing owl, the southwestern willow flycatcher, the double-crested cormorant, the Yuma clapper rail and mule deer. Potential risks were evaluated by comparing calculated concentrations or exposures to toxicity reference values (TRVs) derived to be protective of these receptor groups. The TRVs were obtained from a variety of sources, including the USEPA, the State of Arizona, ecological databases and the published literature.

The calculated environmental concentrations and exposures to animals and birds were not only below the TRVs but also below the conservative ecological target risk level specified by USEPA Region 9 for this project (i.e., a hazard index value of 0.25). These results indicate that adverse ecological effects from exposure to stack emissions are not expected to occur for the evaluated receptors. Concentrations in surface water and sediment were

found to be more than 800 times lower than the 0.25 target hazard index level. Concentrations in plants ranged from just below the 0.25 target level to more than 400 times lower than the 0.25 target level. Exposures to selected bird species were found to be at least five times lower than the 0.25 target level. Finally, exposures to the evaluated mammal species were determined to be at least 5,000 times below the 0.25 target level.

Wastewater Discharge from the Facility to the Wastewater Treatment Plant

The risk assessment also evaluated the potential incremental impact of the facility's wastewater effluent on chemical concentrations discharged from the publicly owned treatment plant into the Main Drain. The analysis also evaluated potential fish tissue concentrations and associated potential human health fish ingestion risks in the Main Drain downstream of the treatment plant's discharge point. This evaluation focused on 19 compounds selected based on measurements obtained from the facility's effluent discharge.

This evaluation showed that the incremental contribution of the facility's effluent on the treatment plant discharge and the Main Drain does not pose unacceptable risks to either aquatic life or human health. The modeled discharge concentrations were below or equivalent to the most stringent applicable state water quality standards and criteria and the treatment plant's discharge permit limits for all evaluated compounds. Semi-annual toxicity tests performed on the treatment plant's discharge since 2000 have consistently shown no toxicity to aquatic organisms. Additionally, potential risks due to ingestion of fish caught from the Main Drain associated with the incremental contribution of the SWT facility effluent were all below USEPA target risk levels for both cancer and non-cancer effects.

Worker Evaluation of Fugitive Emissions

The risk assessment included an evaluation of workplace air concentrations associated with spent carbon unloading using methods consistent with those adopted by the U.S. Occupational Safety and Health Administration and the National Institute of Occupational Safety and Health. This analysis compared modeled on-site ambient air concentrations for the 21 selected compounds due to fugitive emissions, to workplace permissible exposure limits. The worker evaluation indicated that ambient air concentrations due to fugitive emissions during spent carbon unloading would not exceed occupational exposure limits within the property boundary. These results were supported by many years of industrial hygiene measurements, which have predominantly shown air concentrations of regulated chemicals to be either below quantitation limits or typically 100 or more times below the occupational standards and criteria.

Conclusion

In conclusion, the risk assessment presented in this document demonstrates that, using conservative assumptions, the potential risks associated with air emissions from the Siemens Water Technologies Corp. carbon reactivation furnace and from spent carbon unloading are below regulatory and other target risk levels for both human health and ecological receptors. Additionally, the incremental contribution of the facility effluent on

the wastewater treatment plant discharge and the Main Drain does not pose unacceptable risks to either aquatic life or human health. Finally, fugitive emissions during spent carbon unloading do not exceed occupational exposure limits in ambient air at the facility.

RISK ASSESSMENT

1.0 INTRODUCTION

The Siemens Water Technologies Corp. facility (SWT facility) is a carbon reactivation plant located within the 269,000 acre Colorado River Indian Tribes (CRIT) Reservation in La Paz County, Arizona. The facility, formerly known as Westates Carbon-Arizona, Inc., is located just outside the Town of Parker in an industrial park owned by CRIT and is operated pursuant to a lease between the company and CRIT. The facility reactivates spent carbon, which has been previously used to remove pollutants from water and gases by heating it to very high temperatures under controlled conditions. The newly reactivated carbon product is then reused as an activated carbon product.

Activated carbon is used in treatment equipment to remove impurities from water, air and food. For example, activated carbon is widely used as a component of air pollution control systems (Cooper and Alley 2002). For carbon systems to remain effective, the carbon must be replaced regularly. Once carbon begins to approach its capacity to adsorb or filter impurities, it is recycled. Applications for activated carbon systems include improving the taste and quality of drinking water, treating industrial wastewater, purifying materials used in production processes (including foods and medicines), controlling air emissions, and decontaminating groundwater at environmental cleanup sites.

Spent carbon arrives at the facility in a variety of containers, including barrels, drums, bulk truck units and bulk bags. Spent carbon is accepted from a variety of sources, many of which are Fortune 500 companies as well as state and federal agencies, including the U.S. Environmental Protection Agency (USEPA). On average, as of the date of this study, about two-thirds of the spent carbon received at the facility is not classified as a hazardous waste under the U.S. Resource Conservation and Recovery Act (RCRA). The remaining one-third is classified as a hazardous waste because it has been used to treat materials that are classified as hazardous under RCRA (e.g., air and water at environmental cleanup sites that has been treated with spent carbon).

This document presents a human health and ecological risk assessment for the facility. A risk assessment is a scientific study that can help evaluate risks associated with exposure to chemicals in the environment. This risk assessment was conducted as one component in the facility's RCRA permitting process. It is one of the final steps in a process that has extended over a seven year period beginning with the USEPA's request to develop a Risk Assessment Workplan in 2001.

The risk assessment was conducted by a team of scientists and engineers with expertise in risk assessment, toxicology, environmental engineering and air dispersion modeling. CPF Associates, Inc. began working on this project in 2001, and prepared the Risk Assessment Workplan as well as this risk assessment. CPF is a Washington, D.C.-based scientific and health consulting firm with expertise in performing risk assessments for a variety of different types of waste treatment technologies, including combustion facilities. CPF also provided project management over all contractors and consultants who contributed to the risk assessment. Focus Environmental, Inc. provided the emission rates used in this risk

assessment, and engineering expertise related to facility operations. Focus has provided engineering and environmental services to SWT over the duration of this project, including both managing the Performance Demonstration Test at the facility and preparing the recent RCRA Part B permit application. Focus provides environmental engineering and regulatory compliance services, and has extensive expertise in the engineering and testing of combustion facilities. ToxServices, Inc. assisted with the compilation of human health toxicological criteria and performed quality assurance of risk assessment calculations and inputs. ToxServices is a scientific consulting firm with expertise and experience in providing toxicology, regulatory, and risk assessment consulting services to certification and testing laboratories, private industry, and the federal government. Air dispersion and deposition modeling was performed by TRC. TRC provides environmental permitting, engineering, and compliance testing services for energy-related companies as well as a wide range of industrial clients in the U.S. and internationally, and possesses expertise in the development, application and evaluation of air modeling for a wide variety of emission sources. MACTEC assisted in the performance of the ecological risk assessment. MACTEC is a consulting firm that provides engineering, environmental and remedial construction services to public and private clients worldwide, and possesses in-depth expertise in ecological and habitat evaluations and the performance of ecological risk assessments.

Biographies of the study participants are provided in Appendix A. All of the above study participants are independent of Siemens Water Technologies Corp.

1.1   Project History

In 1990 and 1991, the SWT facility (then known as Westates Carbon-Arizona, Inc.) negotiated a lease agreement with CRIT and obtained the necessary permits to locate the facility in an industrial park on the CRIT Reservation. Before construction began, an environmental assessment was completed and a "Finding of No Significant Impact" was approved by the Bureau of Indian Affairs. The facility's RCRA Part A permit application was submitted in August 1991, in accordance with RCRA requirements. The facility has been operating since August 1992 under a variety of regulatory programs, including the Part A interim status regulations at 40 CFR Part 265 and USEPA regulations under the Clean Air Act's Benzene National Emission Standards for Hazardous Air Pollutants (NESHAPs) (Subpart FF of 40 CFR Part 61). The facility is also subject to regulations issued by the Occupational Safety and Health Administration (OSHA).

A RCRA Part B permit application was originally submitted to USEPA in November 1995 that discussed an existing carbon reactivation furnace (RF-1) and a future carbon reactivation furnace (RF-2). In February 2007, an amended Part B application was submitted to USEPA for RF-2, since the older furnace (RF-1) had been shut down (Focus 2007).

To provide a historical context for this project, a chronology of risk assessment actions and other related events leading up to this report is provided below:

- August 2001: USEPA Region 9 requested that SWT prepare a performance demonstration test (PDT) plan and a risk assessment workplan as part of the process for completing its review of the RCRA facility permit application (USEPA 2001a). The review of this permit application is being conducted in accordance with the requirements for a Miscellaneous Unit under Subpart X of 40 CFR Part 264. In its August letter, USEPA identified a variety of requirements for the risk assessment workplan and the human health and ecological risk assessments.[1]

- November 2001: A site visit to the facility and facility area was conducted by CPF.

- January 2002: Meetings were held with SWT, USEPA, CRIT, CPF and Focus.

- January and April 2002: Additional site visits were conducted.

- April 2002: An open house providing information about the SWT facility, the PDT, and the risk assessment process was held in Parker.

- June 2002: The first version of the Working Draft Risk Assessment Workplan ("Workplan") was submitted to USEPA (CPF 2002).

- March 2003: Comments on the Workplan were received from USEPA (USEPA 2003a).

- May 2003: A revised Workplan was submitted to USEPA incorporating USEPA's comments (CPF 2003a).

- September 2003: Additional comments on the Workplan were received from USEPA (USEPA 2003b).

- November 2003: The Workplan was finalized and submitted to USEPA (CPF 2003b).

- November 2003: The Performance Demonstration Test (PDT) Plan for the carbon reactivation furnace was submitted to USEPA (Focus 2003).

- March 2005: USEPA provided conditional approval of the Workplan and the PDT Plan (USEPA 2005a).

- March 2006: The PDT, which included measurement of stack emissions during facility operations, was conducted at the facility by Focus.

- June 2006: The PDT report was submitted to USEPA (Focus 2006).

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[1] Risk assessments conducted for combustion sources to date have rarely included a full-scale ecological risk assessment such as that requested by USEPA for this project.

- February 2007: The facility's revised and updated RCRA Part B permit application was submitted to USEPA (Focus 2007).

- April 2007: USEPA provided approval to use the PDT air emissions test data in the risk assessment and to perform the risk assessment calculations using the Industrial Risk Assessment Program (IRAP) software (USEPA 2007a).

As suggested in the chronology, the risk assessment and PDT are closely inter-related elements in the RCRA permit process.   The relationship between these two activities is shown in Figure 1.

During the preparation of the Workplan, review and input was solicited not only from USEPA Region 9, but also from CRIT and other stakeholders. Many comments were received during this process and were incorporated into the final Workplan. In addition, USEPA conducted public outreach for this project and held consultations with CRIT (USEPA 2005c). For example, in January 2004, USEPA issued a public notice in the Parker Pioneer and mailed a notice to the facility's stakeholder mailing list inviting public comment on the Workplan. As part of this effort, copies of the Workplan were placed in the Parker Public Library and the CRIT Library in Parker (USEPA 2004d).

Figure 1-1

**Flow Chart of the Facility RCRA Permit Process for
the Performance Demonstration Test and the Risk Assessment**
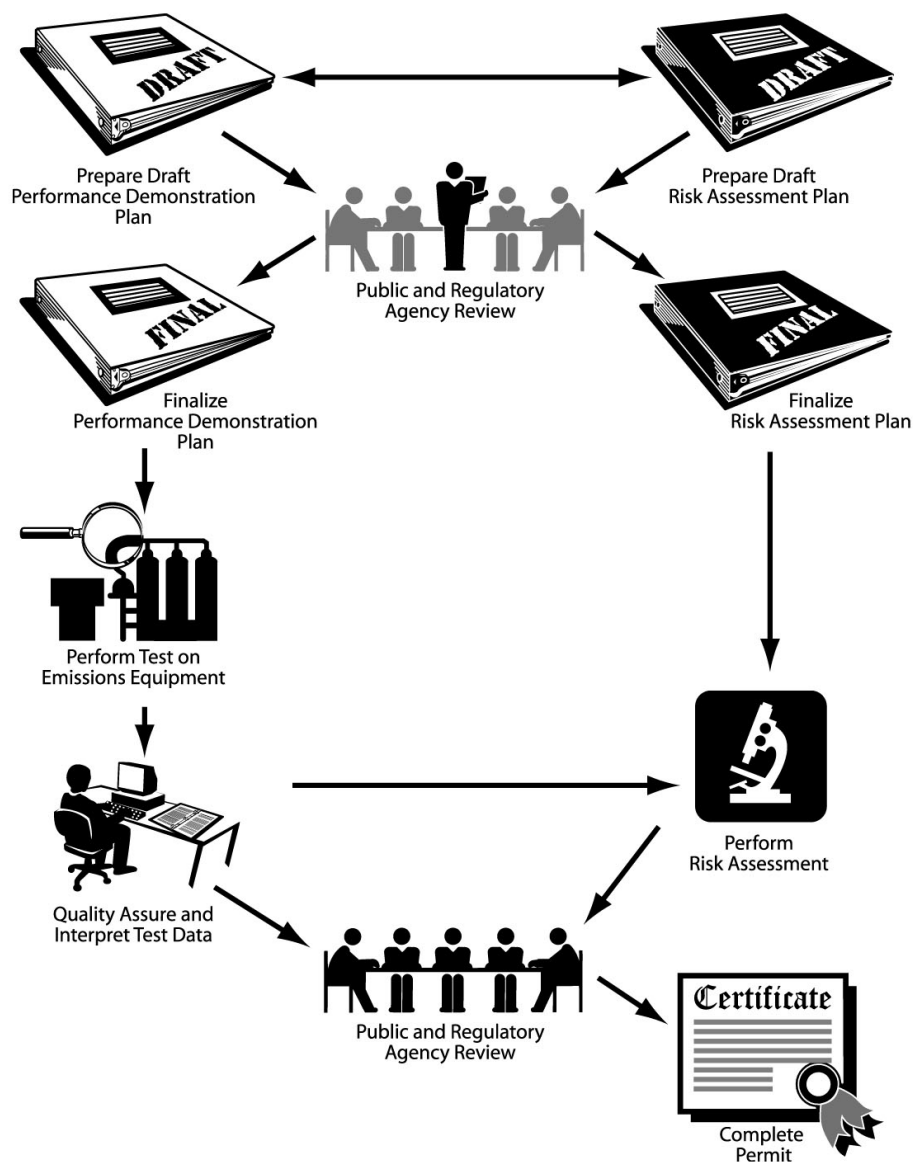
1.2   The Risk Assessment Process

The 2003 Risk Assessment Workplan provided a critical roadmap that was followed during the conduct of this risk assessment. The Workplan described the approaches that would be used to perform the facility risk assessment and it included detailed instructions on a wide variety of risk assessment elements (for example, methods for selecting chemicals for evaluation, performance of air dispersion and deposition modeling, and compilation of toxicological criteria). The Workplan was previously submitted to and approved by USEPA, and can be provided upon request.

In the several years since the Workplan was prepared, there have been some changes to USEPA risk assessment guidance and methods, most notably USEPA's publication in 2005 of a revised Human Health Risk Assessment Protocol (HHRAP) for Hazardous Waste Combustion Facilities. This guidance incorporates many important updates to USEPA's methods, particularly revisions to the fate and transport modeling equations and chemical-specific input parameters. To reflect this newer information, the risk assessment relied to a large extent on the more recent 2005 HHRAP. To facilitate consistency with the 2005 guidance, and as approved in advance by USEPA (2007a), a publicly available software program called IRAP, programmed by Lakes Environmental specifically to reflect USEPA's 2005 HHRAP, was used to perform most of the risk assessment calculations. This software has been widely used in the U.S. (e.g., most USEPA Regions and several states) and among its benefits are reliance on quality-assured programmed calculations, readily available USEPA-specified chemical-specific data, and the ability to address the large number of compounds required to be evaluated in this project. The IRAP program only includes the approaches specifically provided in HHRAP, however, and thus it is limited in its ability to address non-routine risk assessment elements. As a result, while the Workplan provided the primary roadmap for this project, in some cases modifications were made both to reflect HHRAP and to accommodate the capabilities of the IRAP program. This approach was approved for this project in advance by USEPA (2007a).

The Workplan also described a process for requesting site-specific information from CRIT for consideration in the risk assessment. SWT followed this procedure as required. Where information was not received or not available, this project relied on site-specific information available at the time the risk assessment was performed (e.g., information from published reports, publicly accessible information on the internet, contacts with local officials and site visits).

Overall, this risk assessment analyzed specific sets of assumptions that are, collectively, expected to overestimate potential risks. The risk assessment, therefore, calculates the potential for risks to occur under specific assumptions and does not calculate actual human health or ecological impacts.

1.3   Report Organization

The remainder of this document presents the risk assessment of the facility. The following topics are covered:

- A brief introduction to the facility area
- An overview of the risk assessment process
- Presentation of the human health risk assessment
- Presentation of the ecological risk assessment
- A brief summary of quality assurance procedures
- A listing of references cited in this document
- Appendices with supporting information

2.0   FACILITY AND AREA DESCRIPTION

The Workplan provided a detailed discussion of both the facility vicinity and facility operations. Additionally, the RCRA Part B permit application (Focus 2007) provides a comprehensive discussion of the facility including, for example, equipment and operations, and health and safety procedures. Rather than repeat this information here, the reader is referred to the Workplan and the RCRA Part B application which can be provided upon request. For general reference, a few of the figures from the Workplan are shown below, specifically Figure 2-1 which shows the facility location, Figure 2-2 which presents a map of the CRIT Reservation, Figure 2-3 which presents photographs of the facility area and surrounding landscape, Figure 2-4 which is an aerial photograph of the facility, and Figure 2-5 which illustrates a habitat map for the facility area.

Figure 2-1
Facility Location

Facility

N

Scale
~1 km   0
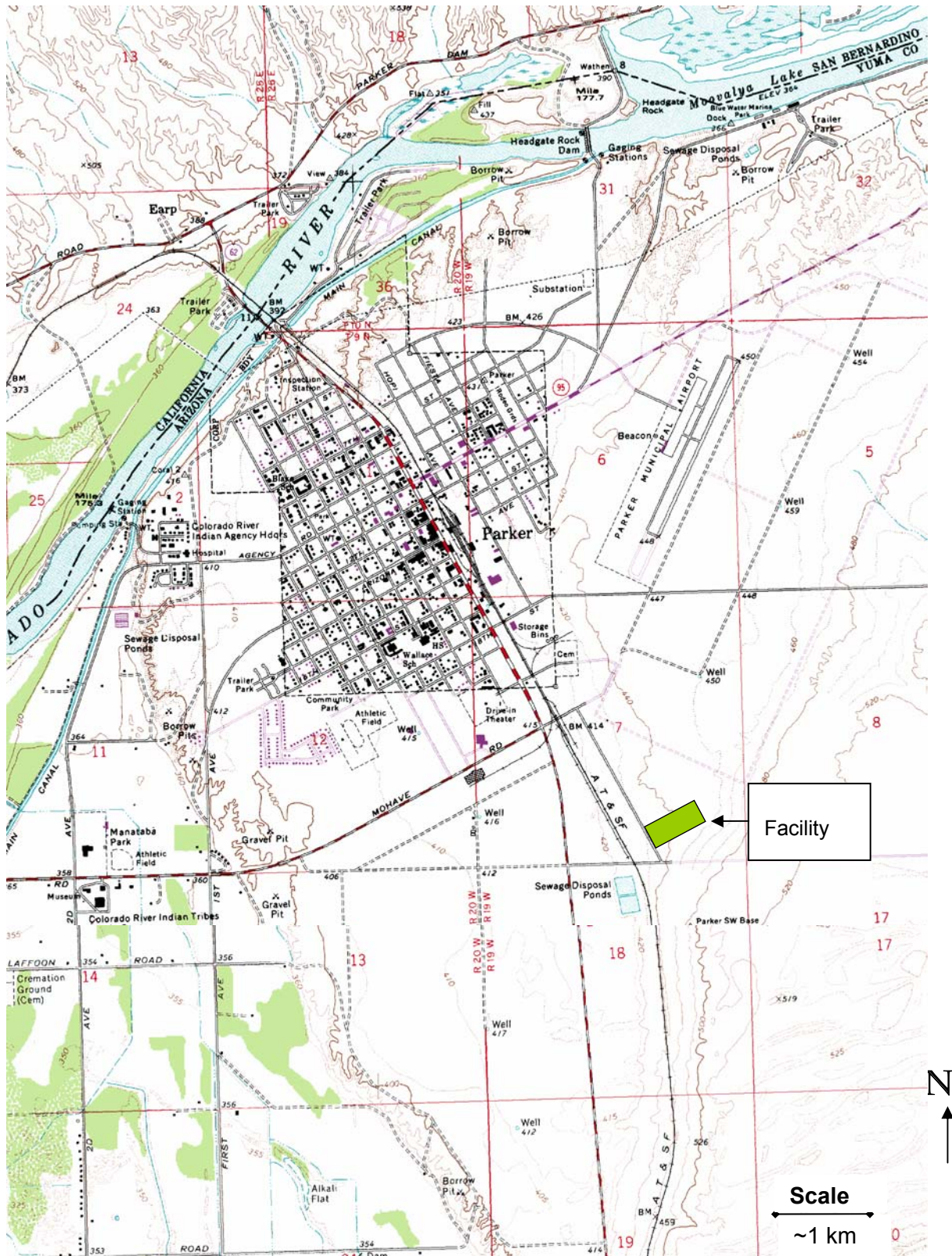
Figure 2-2

Colorado River Indian Tribes Reservation Map

Facility location

5   0   5 **Miles**

Reservation Boundary
Colorado River
Main Roadways
Roadways
AZ & CA State Line
AZ & CA Railroad
Township & Range Lines
Section Lines
Levee Roadway
Town of Parker

CRIT Planning Department
John Scott, Jr., GIS Technicia
September 20, 2001

Colorado River Indian Tribes
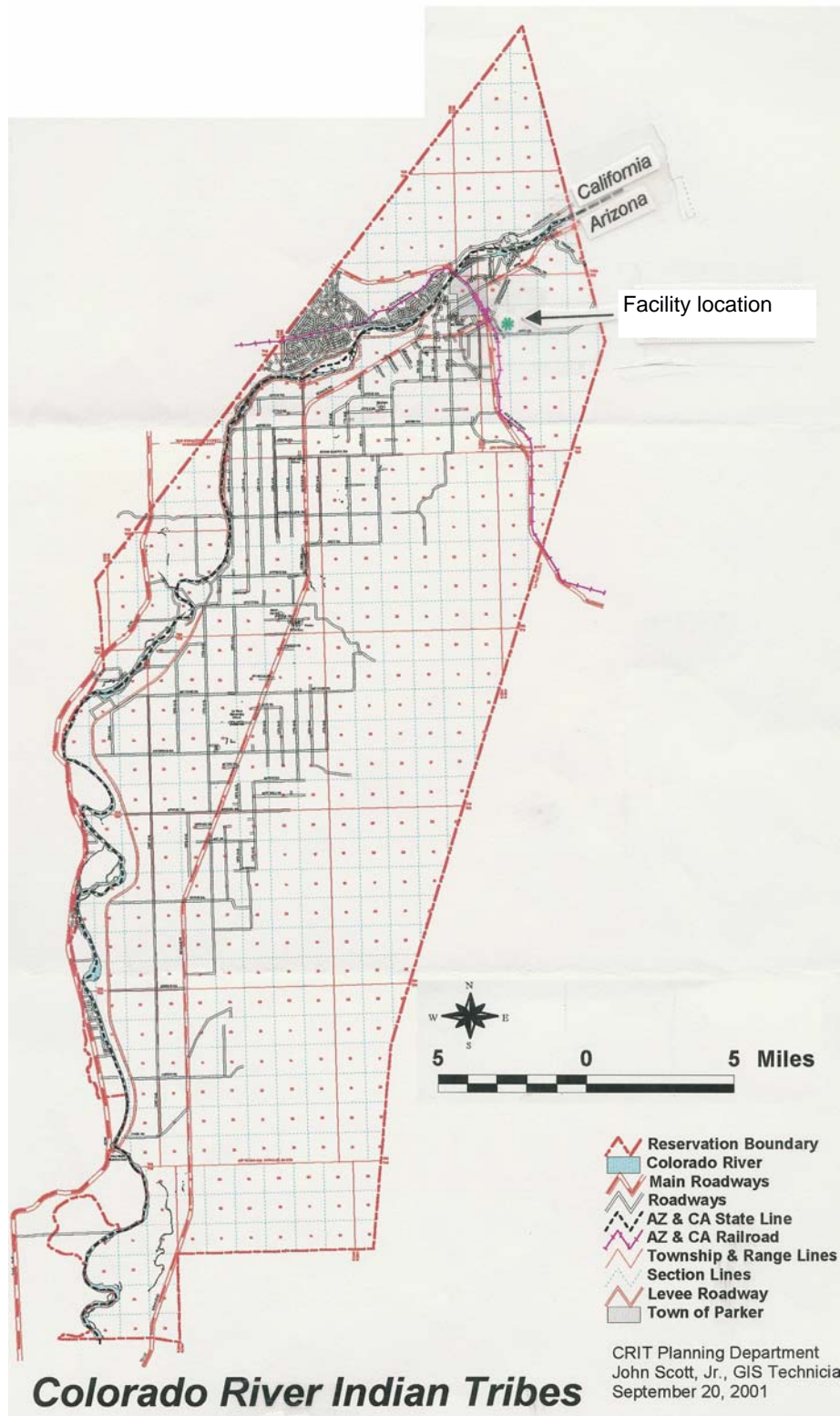
Figure 2-3
Landscape in the Facility Area

Figure 2-4
Aerial View of the Facility

Figure 2-5
Habitat Map - USGS Topography

Colorado River

Town of Parker

Facility Location

PARKER VALLEY

Legend

✖ Facility Location

☐ Study Area (20 km)

Habitat Types:

🟧 Agricultural   ☐ Creosote Bush Scrub

🟩 Riparian   ☐ Developed

🟦 Water   (Residential/Commercial/Industrial)

N

0 500 1,000 2,000
Meters

Nevada

California   Arizona

Prepared by BRP   Checked by ALF
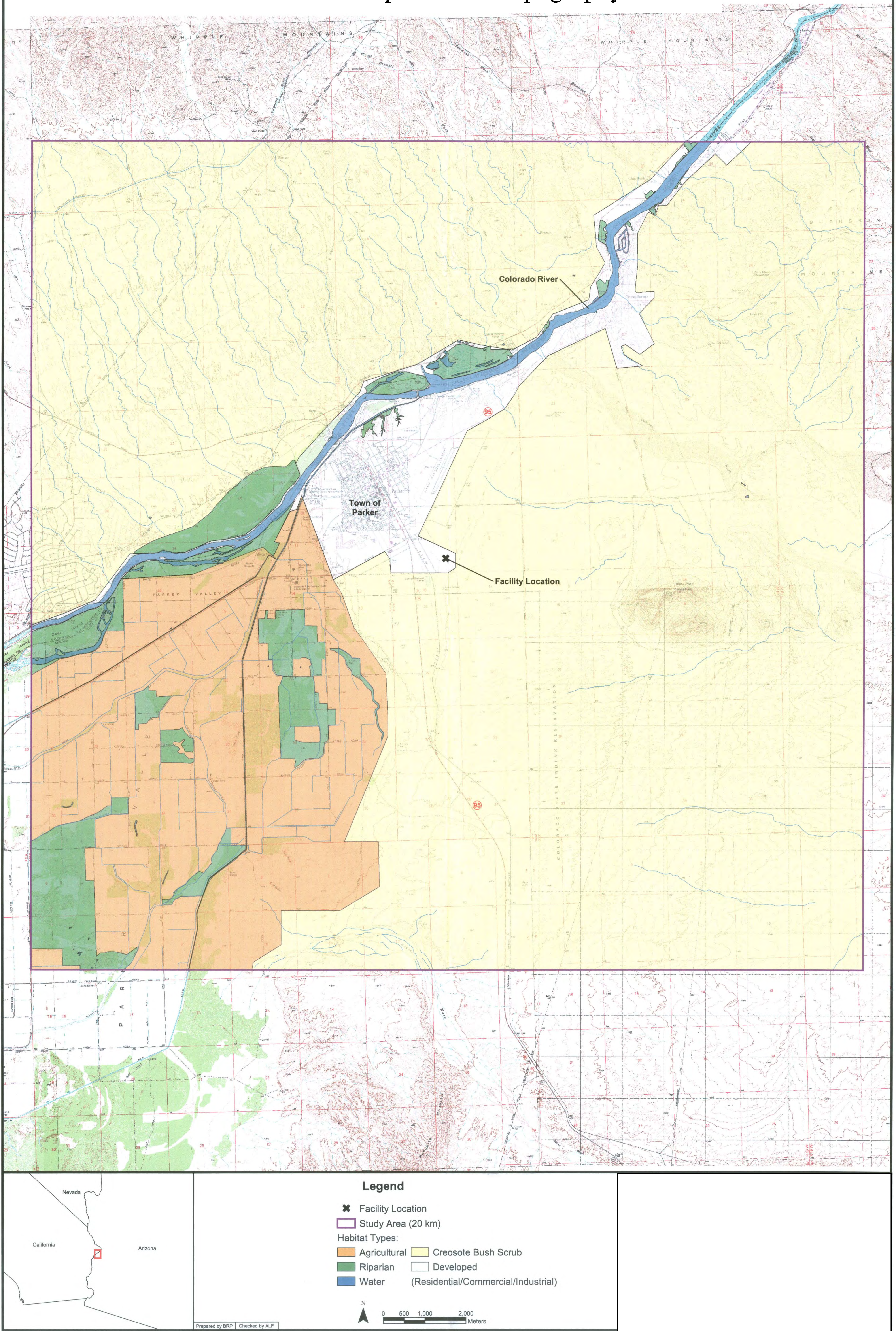
3.0   RISK ASSESSMENT OVERVIEW

This remainder of this report summarizes the methods used to conduct the human health and ecological risk assessment, and presents the risk assessment results. As noted in the Workplan, the human health and ecological portions of the risk assessment share some common elements. These common elements are chemical emission rates, air dispersion and deposition modeling and fate and transport modeling used to calculate exposure concentrations in environmental media such as soil, plants and surface water. Elements that are unique to each analysis include the inputs and methods used to calculate exposures and chemical-specific toxicity criteria.

The human health and ecological portions of the risk assessment relied on a variety of regulatory guidance documents in addition to the methods described in the Workplan, as shown in Figure 3-1. In addition to relying on these guidance documents, the risk assessment used a large amount of site-specific data, including but not limited to:

- comprehensive testing of emissions from the furnace stack, with analysis for site-specific chemicals of potential concern
- data on spent carbon characteristics, the facility configuration, and facility operations
- local land use and demographic information
- water resources data available from the U.S. Geological Survey and the U.S. Bureau of Reclamation
- meteorological data from Parker, Arizona.

The basis for each site-specific value used in the analysis is provided in this report. In the absence of site-specific information, health-protective default values recommended by the USEPA were used.

Figure 3-1

**Overview of Risk Assessment Process
and Guidance Documents**

Emission Rates

Guidance Documents:
** Human Health Risk Assessment Protocol for Hazardous
   Waste Combustion Facilities (USEPA 2005)*
* Risk Burn Guidance (USEPA 2001)
* Working Draft Risk Assessment Protocol (2003)

↓

Air Dispersion and Deposition Modeling

Guidance Documents:
** Human Health Risk Assessment Protocol for Hazardous
   Waste Combustion Facilities (USEPA 2005)*
* Guideline on Air Quality Models (USEPA 2005)
* Working Draft Risk Assessment Protocol (2003)

↓

Exposure Concentrations

Guidance Documents:
** Human Health Risk Assessment Protocol for Hazardous
   Waste Combustion Facilities (USEPA 2005)*
* Working Draft Risk Assessment Protocol (2003)

Human Health Risk Assessment

Guidance Documents:
** Human Health Risk Assessment Protocol for
   Hazardous Waste Combustion Facilities (USEPA
   2005)*
* Risk Burn Guidance (USEPA 2001)
* Exposure Factors Handbook (USEPA 1997)
** Superfund Risk Assessment Guidance (USEPA
   1989)*
* Working Draft Risk Assessment Protocol (2003)

Ecological Risk Assessment

Guidance Documents:
** Guidelines for Ecological Risk Assessment
   (USEPA 1998)*
** Screening Level Ecological Risk Assessment
   Protocol for Hazardous Waste Combustion
   Facilities (USEPA 1999)*
** Ecological Risk Assessment Guidance for
   Superfund: Process for Designing and
   Conducting Ecological Risk Assessments
   (USEPA 1997)*
* Working Draft Risk Assessment Protocol (2003)

4.0   HUMAN HEALTH RISK ASSESSMENT

This section presents the human health risk assessment for the carbon reactivation facility. The key steps in this assessment, consistent with USEPA guidance and the U.S. National Academy of Sciences, consist of:

- Hazard Identification
- Exposure Assessment
- Risk Characterization
- Discussion of Uncertainties

Figure 4-1 provides a flow chart of the human health risk assessment process for stack and fugitive emissions, each step of which is described below. It should be noted that all of the algorithms used to calculate environmental concentrations, exposures and potential risks associated with stack and fugitive air emissions beyond the property boundary were based entirely on HHRAP, and implemented using the IRAP software. In addition, separate discussions are provided below to address several issues identified for supplemental consideration by USEPA Region 9 or raised by the community during the Workplan development stage of this project, specifically evaluation of potential risks from exposure to airborne chemicals in the workplace from fugitive emissions and evaluation of the potential contribution of the facility's effluent on discharges from the Colorado River Sewage System Joint Venture (CRSSJV) sewage treatment plant.

4.1   Hazard Identification

The Hazard Identification presents the selection of chemicals for evaluation as well as the toxicity data for each selected chemical. This section focuses on the selection of compounds for the stack emissions risk assessment. Selection of compounds for the fugitive emissions analysis is presented later in this report (Section 4.3.2).

4.1.1   Selection of Chemicals of Potential Concern for Stack Emissions

The approach for selecting chemicals of potential concern (COPC) for quantitative evaluation in the human health risk assessment of stack air emissions was outlined in the Workplan. This approach specified that chemicals would be selected based on a variety of factors:

- Compounds would be selected from the list of constituents analyzed for during the PDT. As requested by USEPA, compounds analyzed for but not detected in the PDT were included in the evaluation, in addition to detected compounds. The PDT was approved in advance by the USEPA and conducted in March 2006 by an independent testing firm. It included comprehensive testing of the facility for site-specific chemicals of potential concern under operating conditions intended to overestimate emissions. The results of the PDT are presented in a comprehensive report prepared by Focus (2006).

Figure 4-1

**Flow Chart of the Human Health
Risk Assessment Process
for the Carbon Reactivation Facility**

Hazard Identification:
Selection of Chemicals
Toxicity Characterization

Exposure Assessment:
Quantification of Emission Rates
Air Dispersion/Deposition Modeling
Population Analysis
Identification of Exposure Pathways
Calculation of Environmental Concentrations
Calculation of Human Exposures

Risk Characterization:
Stack Emissions
  Long-Term Cancer Risks
  Long-Term Non-Cancer Risks
  Short-Term Inhalation Risks
  Margin of Exposure for PCDDs/PCDFs
  Infant Exposure to PCDDs/PCDFs
  Evaluation of Lead
Fugitive Emissions
  Long-Term Cancer Risks
  Long-Term Non-Cancer Risks
  Short-Term Inhalation Risks

Discussion of Uncertainties:
General Review of Uncertainties

Discussion of Additional Topics
      (e.g., Dioxin-Like PCBs
      Unidentified Organics
      Tentatively Identified Compounds)

- Tentatively identified compounds (TICs) in the PDT results would be considered for inclusion as chemicals for detailed evaluation.[2]

- Compounds that could potentially be present in spent carbon, even if they were not analyzed for during the PDT, would be considered for evaluation. A list of compounds that could be in spent carbon was compiled in the Workplan.

Application of this selection approach resulted in the identification of over 225 compounds for detailed evaluation in the human health risk assessment, including more than 100 compounds that were not detected in the PDT and also all detected TICs. Table 4.1-1 summarizes the list of selected compounds and indicates the basis for each compound's inclusion in the risk assessment.

4.1.2   Toxicity Characterization

The toxicity characterization followed the methods laid out in the Workplan, as described below.

4.1.2.1   *Chronic Health Effects Criteria*

The toxicity data used to evaluate chronic, long-term risks includes oral cancer slope factors and inhalation unit risk factors for predicting excess lifetime cancer risks and oral reference doses (RfDs) and inhalation reference concentrations (RfCs) for predicting the potential for long-term non-cancer effects. These toxicity data were compiled for each selected compound either directly from HHRAP's chemical-specific database (which is included in the IRAP software) or from the toxicity data sources recommended by HHRAP. Appendix B presents the chronic toxicity data compiled for compounds not already addressed in HHRAP that were used in the calculation of potential risks. Of the more than 200 compounds selected for evaluation, chronic toxicological criteria were not available from USEPA's recommended sources for 49 compounds. These compounds are discussed in the uncertainty section of this risk assessment.

As noted in the Workplan and HHRAP, mixtures of PCDDs/PCDFs were evaluated using toxic equivalency factors (TEFs) which relate the toxicity of each 2,3,7,8-congener to the toxicity of 2,3,7,8-TCDD, the most well-studied and most toxic congener among the PCDDs/PCDFs.[3] In this system, the TEF for 2,3,7,8-TCDD is 1.0 and the other congeners have TEF values ranging from 1.0 to 0.00001. For example, the TEF for 2,3,7,8-TCDF is 0.1, which means that the potential toxicity of 2,3,7,8-TCDF is considered to be 10 times

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[2] A TIC is a compound that is not specifically targeted for an analysis but which is detected. This means that while it can be seen in a laboratory analysis, its identity and concentration cannot be determined with certainty without further analytical investigation.

[3] Polychlorinated dioxins and furans are a class of chemicals known as polychlorinated dibenzodioxins (PCDDs) and polychlorinated dibenzofurans (PCDFs), sometimes referred to as dioxins and furans. There are 75 PCDDs and 135 PCDFs, with each individual compound referred to as a congener. Only 7 of the 75 PCDD congeners and 10 of the 135 PCDFs are considered to be toxic; these are compounds with chlorine molecule substitutions at the 2, 3, 7, and 8 positions on the compound. In this document, the mixture of polychlorinated dioxins and furans are referred to as "PCDDs/PCDFs".

lower than that for 2,3,7,8-TCDD. To apply the TEF concept, the TEF of each congener present in a mixture is multiplied by its respective concentration or exposure and the products are summed to obtain the total TCDD toxic equivalents (TEQs) of the mixture. The TEFs are incorporated into the IRAP software consistent with USEPA (2005b) specifications.

4.1.2.2   *Acute Health Effects Criteria*

In addition to long-term toxicity data, the potential for short-term acute effects from stack emissions to air were evaluated using acute reference air concentrations. These concentrations, representing the short-term level in air above which adverse effects may occur, are provided in HHRAP and programmed into the IRAP software for many compounds. For compounds not addressed in HHRAP, acute reference air concentrations were derived from the published literature following HHRAP guidance. Appendix B presents the acute inhalation toxicity data compiled for compounds not already addressed in HHRAP. Among the more than 200 compounds selected for consideration in this study, 17 did not have acute inhalation toxicity criteria. Compounds without human health toxicity criteria are discussed in the uncertainty section of this study.

4.2   Stack Emissions Exposure Assessment

The next major step in the risk assessment is the stack emissions exposure assessment, which consists of the following elements:

- Quantification of stack air emissions
- Air dispersion and deposition modeling
- Population analysis
- Identification of exposure pathways
- Evaluation of environmental concentrations
- Calculation of human exposures

These elements of the exposure assessment were discussed in the Workplan and are described below.

4.2.1   Stack Emission Rates

4.2.1.1   *Long-Term Emission Rates*

One of the most important inputs to a combustion source exposure assessment is the chemical emission rate. Emission rates should reflect releases associated with actual facility operations, however, in this risk assessment assumptions were made that were designed to be more conservative than actual facility operating conditions. These assumptions included using PDT test results, which were measured under operating conditions intended to overestimate actual facility emissions, using proposed permit limits for compounds which had lower measured levels from the PDT, and including many compounds that were not detected in the PDT. As a result, the emission rates used in this assessment are expected to overestimate potential risks as compared to actual facility emissions.

The stack emission rates were calculated by Focus and are listed in Table 4.2-1 along with an indication of the basis for each value. In general, as noted above, emission rates were based on either the PDT results, proposed permit limits or, for a few chemicals that could be present in spent carbon but were not measured during the PDT, long-term average chemical feed rates and a conservative destruction and removal efficiency (DRE) of 99.99%.[4] Emission rates derived from the PDT measurements were calculated as described in the Workplan, based on the arithmetic average of results across the three test runs and using one-half the detection limit for non-detect results, consistent with standard risk assessment practice. Emission rates for the combustion gases sulfur dioxide and nitrogen dioxide were based on results from a miniburn test conducted in April 2005 since these were not measured during the PDT. Appendix C presents the detailed PDT test results used by Focus to calculate emission rates for this risk assessment.

Emission rates for mercury were identified in the PDT for three forms of mercury - particulate phase divalent mercury, vapor phase divalent mercury and vapor phase elemental mercury - as required for the USEPA (2005b) risk calculations. The speciation of mercury was determined by analyzing the separate components of the mercury sampling train. As recommended in USEPA (2001c), it was assumed that the particulate matter and front half rinse results represented divalent particulate mercury, the acidified impinger solution result represented divalent vapor phase mercury, and the potassium permanganate solution result represented elemental vapor mercury. The PDT results indicated a mercury breakdown for the stack emissions as 0.5% particulate phase divalent mercury, 19.8% vapor phase divalent mercury and 79.7% vapor phase elemental mercury.

4.2.1.2   Upset Scaling Factors

As discussed in the Workplan, consistent with USEPA (2005b) guidance, upset conditions were considered in this risk assessment. This was to be accomplished by adjusting the stack emission rates upwards by an upset scaling factor according to the equation below:

$$ER_{RA} = ER_{SE} * USF \qquad \text{(Equation 4-1)}$$

where

ER_{RA} = emission rate for input to risk assessment (g/sec),
ER_{SE} = emission rate based on stack emissions (g/sec), and
USF = upset scaling factor (unitless).

A scaling factor was developed using data provided by SWT for the carbon reactivation facility. SWT identified upset conditions that have the potential to affect stack emission rates, and compiled data on historical upsets at the facility that occurred for these conditions during 2001 and 2002. Based on the upset data, which are summarized in Table 4.2-2, the scaling factor was calculated according to HHRAP methods to be 1.02. The HHRAP method for deriving the scaling factor assumes that emissions increase by a factor of 10 for the

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[4] The DREs measured in the PDT averaged more than 99.997% (Focus 2006).

percentage of operating time under upset conditions. The factor of 10 was based on a default approach for nonhazardous waste incinerators presented by the California Air Resources Board (1990) in which emissions were assumed to increase by a factor of 10 during upsets. The 1.02 scaling factor calculated for this project has a negligible numerical impact on the long-term stack emission rates, and thus the emission rates already shown in Table 4.2-1 were used, without adjustment according to Equation 4-1, to characterize long-term stack emissions.

As noted in the Workplan, the upset scaling factor does not reflect startup or shutdown conditions for the reactivation furnace stack because, under these conditions, emissions associated with spent carbon will not occur. During startup, there is no spent carbon in the reactivation furnace. Startup procedures involve increasing the temperature of the reactivation furnace and afterburner over a period of roughly 33 hours using natural gas only. Spent carbon is not introduced into the multiple hearth furnace until temperatures have reached their required levels. As a result, upset emissions associated with spent carbon do not occur during start up conditions. Shut down procedures involve shutting off spent carbon feed to the furnace and waiting until all spent carbon has been cleared from all hearths before starting to cool down the furnace. The amount of time needed to clear the furnace hearths of spent carbon is approximately 42 minutes. After all spent carbon is cleared from the furnace, temperatures in the furnace are slowly lowered to ambient temperature over a period of roughly 32 hours. Since the required high temperatures are maintained in the furnace, and the air pollution control equipment is continuously operated until all spent carbon is cleared, upset emissions associated with spent carbon do not occur during normal shut down conditions.

4.2.1.3   *Short-Term Emission Rates*

In addition to long-term emission rates, short-term emission rates were also considered in the acute inhalation risk analysis. The short-term emission rates were intended to reflect a one-hour period of time rather than a long-term, multi-year time period. Two sets of short-term emission rates were evaluated, one assuming no upset condition occurs during the one-hour period evaluated, and the other assuming an upset does occur during that one hour.   The set of emission rates shown in Table 4.2-1 were used to calculate inhalation risks for the non-upset acute analysis. The risks associated with the upset condition were then calculated by increasing the acute results for the non-upset condition by a factor of 10, which assumes that an upset occurs for the entire 1-hour period evaluated.

4.2.2   Air Dispersion and Deposition Modeling

Air dispersion and deposition modeling is required in order to calculate chemical concentrations and ultimately human exposures from stack emissions. This modeling was performed according to a protocol included in the Workplan. The air dispersion model used was the most recent version of the Industrial Source Complex Short-Term model available from the USEPA (ISCST3, Version 02035). This model was developed and approved by USEPA. The remainder of this section summarizes the modeling performed using ISCST3 for this project. Appendix D describes the modeling work in greater detail.

The general application of modeling results in the risk assessment is outlined in Table 4.2-3 and, as described in the Workplan, was organized as follows:

- Long-term chronic risks were calculated using annual average modeling results. Annual average ambient air concentrations and annual average deposition rates were used to calculate concentrations in a variety of environmental media relevant to the risk assessment, with calculations performed using the IRAP software which incorporates USEPA (2005b) methods.

- Short-term acute inhalation risks were calculated using 1-hour average modeling results, also using the IRAP software.

Facility and meteorological input data used in the modeling are described in Appendix D. Facility-specific inputs were based on actual operating data (e.g., stack height, exhaust gas temperature, exhaust gas exit velocity) while meteorological inputs were based on surface air data collected by the Arizona Meteorological Network (AZMET) in Parker and upper air data (e.g., mixing heights) obtained from measurements collected at the National Weather Service (NWS) station at Flagstaff Pulliam Airport.

Both dry and wet deposition are important components in the facility's risk assessment. The risk assessment therefore considered four possible sources of deposition, consistent with USEPA (2005b) guidance:

- Dry deposition of particles,
- Wet deposition of particles,
- Dry deposition of gases, and
- Wet deposition of gases.

Wet and dry deposition modeling of particles requires information on the size distribution of emitted particles from the stack. The particle size distribution was based on test data collected from the facility stack during the PDT (see Appendix D). Consistent with USEPA (2005b) guidance, the particle size distribution was treated in two different ways in the ISCST3 model. A mass-weighted particle size distribution was used to represent emissions of metals (except mercury) that would form particles in the reactivation unit combustion area. A surface area-weighted size distribution was used to reflect organic compounds and mercury that most likely exit the combustion area as gases and then adsorb onto the surface of already-formed particles.

As outlined in USEPA (2005b) guidance, the ISCST3 model runs provided nine different types of outputs that were used in the risk calculations, as follows:

- Ambient air concentrations of mass-weighted particles
- Ambient air concentrations of surface area-weighted particles
- Ambient air concentrations of gases
- Dry deposition of mass-weighted particles

- Dry deposition of surface area-weighted particles
- Wet deposition of mass-weighted particles
- Wet deposition of surface area-weighted particles
- Dry deposition of gases
- Wet deposition of gases

These air and deposition modeling results were calculated across the modeling domain study area indicated in the Workplan, a 20 km-by-20 km square study area (154 square miles) with the facility stack at its center (see Figure 4-2). Modeling results were calculated at each of more than 4,000 receptor grid points beyond the facility property boundary within the modeling domain. A fine receptor grid was used with grid points evenly spaced at 100 m (328 foot) intervals out to 3 km from the facility. A coarse grid was used from 3 km to 10 km, with points evenly spaced at 500 m (1,600 foot) intervals. A description of the receptor grids is also provided in Appendix D.

The air dispersion and deposition modeling was performed using a unitized (1 g/sec) emission rate. The model outputs are thus referred to as "unitized" values, expressed in units of $\mu g/m^3$ per 1 g/sec for air concentrations and g/m^2-year per 1 g/sec for deposition rates. Chemical-specific concentrations and deposition rates may be obtained by multiplying the unitized results by the chemical-specific emission rates, a standard risk assessment step that occurs in the IRAP software.

The annual average unitized modeling results for this project are illustrated in several isopleth[5] figures provided in Appendix E, with one figure for each of the different types of air concentrations and for each of the different dry deposition model outputs (i.e., vapor, particle mass weighted, and particle surface area weighted). An evaluation of the unitized modeling results showed that roughly 99% of the total wet plus dry deposition at any given receptor point was due to dry deposition, which is not surprising in an area that receives less than 6 inches of rain per year. Isopleths of unitized wet deposition rates were, therefore, not prepared, not only because of the negligible contribution of wet deposition to the total deposition rates, but also because the unitized wet deposition rates were too small to be plotted using the IRAP software.

Several specific receptor locations were identified for evaluation in the risk assessment by examining the unitized modeling results across specified types of land use areas. For example, annual average air concentrations and deposition rates were used to evaluate long-term chronic risks for residential assessment locations. Accordingly, the annual average unitized modeling results within areas currently used for residential assessment purposes within the Town of Parker and within the CRIT Reservation area with access to irrigation water were examined, and the maximum annual average impact locations in both areas were selected for detailed evaluation. One-hour average air concentrations were used to evaluate short-term acute inhalation risks in residential areas, at locations used for other purposes (e.g., commercial), and also undeveloped areas. Thus, the 1-hour average unitized modeling results were also examined to identify maximum impact locations within residential areas of

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[5] An isopleth is a line that connects points of equal amounts of a quantity, such as an air concentration or a deposition rate.

Figure 4-2

Siemens Water Technologies Corp. Carbon Reactivation Facility, Parker, Arizona Risk Assessment Study Area
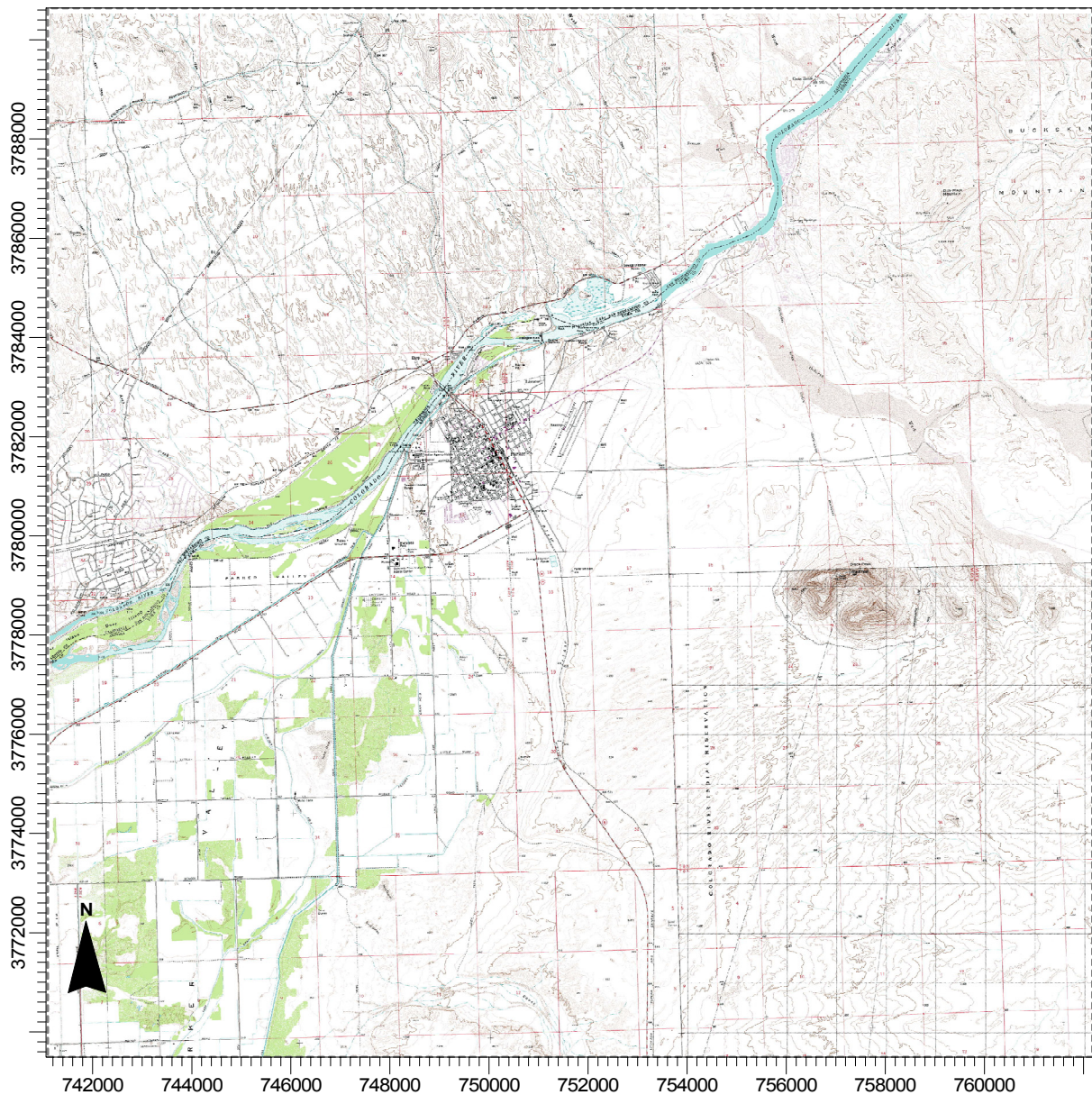
Note: The x and y axes display UTM coordinates (universal tranverse mercatur grid system coordinates) in meters.

SCALE: 1:129,086

0 _____ 4 km

IRAP-h View - Lakes Environmental Software

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the Town of Parker and the CRIT Reservation area with access to irrigation water, at locations used for non-residential purposes, and at the maximum impact point beyond the property boundary. Table 4.2-4 lists all of the receptor point locations selected for evaluation for both the chronic and acute stack emissions risk assessment. Figure 4-3 shows these locations overlain on a topographical map of the area.

4.2.3   Population Analysis

The next step in the exposure assessment involved identifying populations in the facility area through demographic and land use data, and information on population activity patterns. Local information was obtained for this project through site visits, contacts with local officials, published reports, and publicly available local descriptive information on the internet.[6]

4.2.4   Identification of Exposure Pathways

The next exposure assessment step was the selection of a set of exposure pathways for evaluation in the risk assessment. This list of pathways was selected based on site-specific information on land use, USEPA (2005b) default exposure pathways, USEPA's (2001a) request that the risk assessment consider exposure due to subsistence fishing, hunting and agriculture, and the available options programmed into the IRAP software.

A variety of local information regarding home produce gardens and locally raised animals was received from the La Paz County Agricultural Extension Office (Masters 2007). A few residents in the facility area may raise the following types of animals – beef cattle, pigs, chickens, lamb and goat. Some of these animals are raised by children as part of the local 4H program, and these animals are required to be sold rather than used as a household food source. There are no large beef farms within the modeling domain, dairy cows are not raised at all in the local area, and there are no commercial animal slaughter facilities in the Parker area. Based on communications with colleagues, Masters (2007) estimated that at most 10% of a resident's diet of animal products would be obtained from locally raised animals. For residents who might butcher their own locally raised animals, it was estimated that no more than 20% of a person's annual animal products diet would come from locally raised animals. Some residents in the study area cultivate home gardens, but because of the dry, hot climate, there is only a limited portion of the year during which produce may be grown. Based on communications with colleagues, Masters (2007) estimated that no more than 20% of a person's annual produce ingestion was likely to be obtained from homegrown produce in the project study area.

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[6] Local sources of information relied on for the project included, but were not limited to: USGS (2005, 2006a, 2006b, 2007), Williams (2007a, 2007b), Tunnel (2007), Jones (2007), Weiss (2007a, 2007b), Addiego (2007), SCS (1986), Milliken (2007), USBR (2007), USDOI (2000), AZDC (2005), and Masters (2007).

Figure 4-3
Receptor Point Locations Evaluated in the
Stack Emissions Risk Assessment
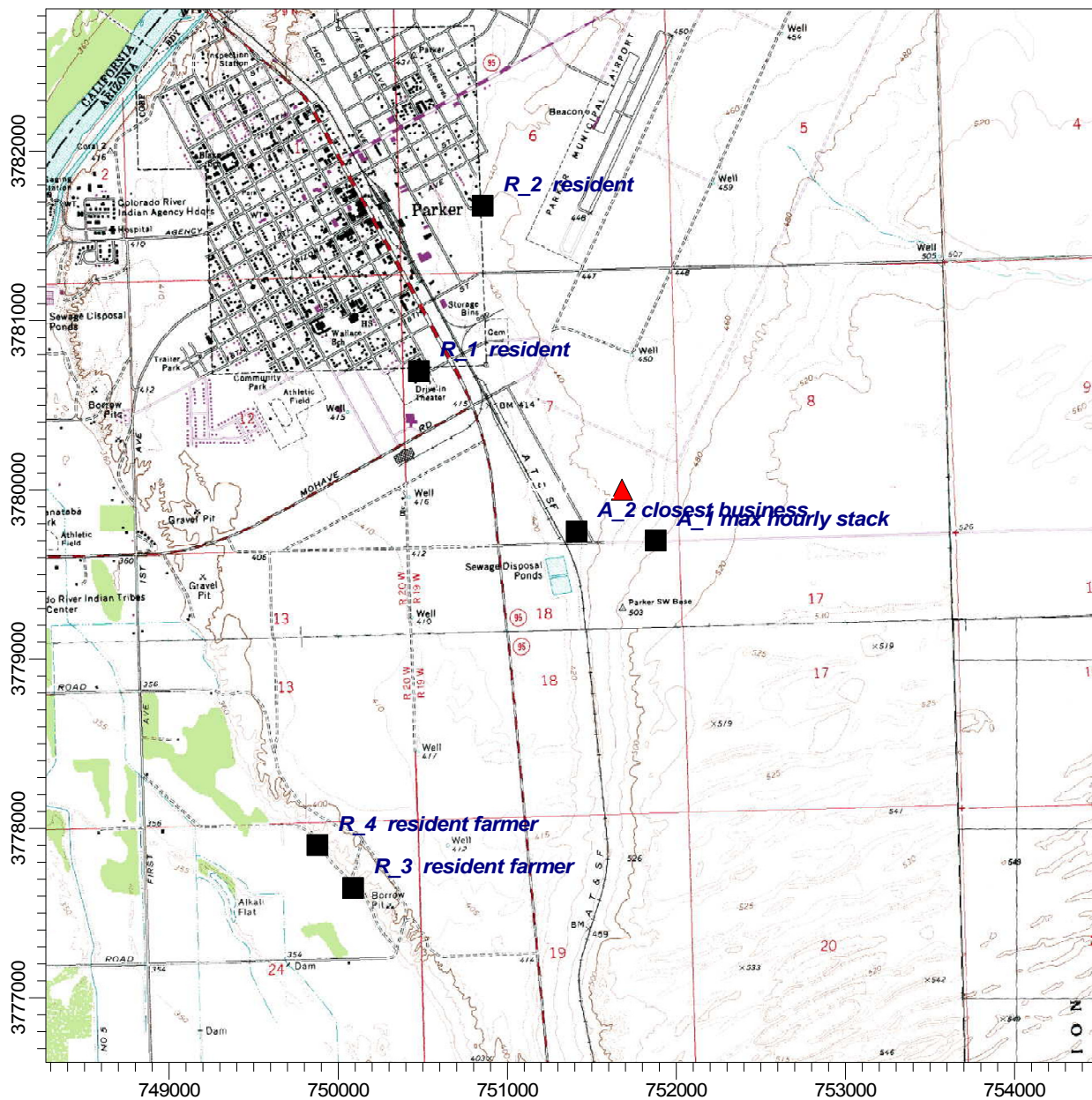
COMMENTS:

COMPANY NAME:

MODELER:

SCALE: 1:37,982

0 ⊢━━━━━━━┤ 1 km

DATE:

PROJECT NO.:

IRAP-h View - Lakes Environmental Software

Fishing occurs in the facility area, but details on where people routinely fish, how often people fish, and how much locally caught fish is ingested were not available at the time this project was performed.[7] Hunting also occurs in the facility area for a variety of animals, including mule deer.[8]

Another important factor affecting the selection of exposure pathways was the capabilities of the IRAP software, which directly reflects HHRAP methods. The IRAP software is programmed with all of USEPA's default exposure pathways which consist of inhalation of air, and ingestion of soil, produce, beef, chicken, eggs, fish, dairy milk, and pork.

Based on the available information at the time this assessment was performed in conjunction with the options available in the IRAP software, all of the USEPA (2005b) default exposure pathways except for dairy milk ingestion were retained for evaluation. Potential exposures associated with ingestion of venison, lamb and goat meat were evaluated in the uncertainties section of this report.

Table 4.2-5 identifies the exposure pathways and receptors that were selected for quantitative evaluation in this risk assessment using the IRAP software. As can be seen, this assessment addressed exposures to several different types of individuals (referred to as "receptors") who could hypothetically be exposed to stack air emissions from the facility: adult and child residents, adult and child farmers, adults and children assumed to eat fish caught from the Colorado River or the Main Drain, and a nursing infant conservatively assumed to be the child of each of the adult receptors, with the potential for transmission of chemicals from mother's breast milk.

4.2.5   Calculation of Environmental Concentrations

The next step in the exposure assessment was the calculation of chemical concentrations in each environmental medium of interest. These are referred to as exposure point concentrations. For example, concentrations were calculated in soil, homegrown produce, fish, animal products, and human breast milk. All the equations used to calculate environmental concentrations were based on HHRAP and are programmed into the IRAP software.

Many input parameters are required in order to calculate environmental concentrations using the USEPA (2005b) fate and transport modeling equations. These include numerous chemical-physical properties describing each compound and its behavior in the environment. Although USEPA (2005b) identified these properties for over 200 compounds in HHRAP (and all are included in IRAP), there were many additional compounds selected for evaluation in this risk assessment, based on the PDT results, for which these same types of chemical-physical properties needed to be compiled. Appendix F presents the properties that were compiled for these additional compounds and a listing of data sources for each value.

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[7] www.azgfd.gov/h_f/where_fish_southwest.shtml.
[8] www.azgfd.gov/h_f/hunting_units_43a.shtml and hunting_units_44a.shtml.

A variety of environmental parameters that are not chemical-specific are also needed to calculate environmental concentrations (e.g., rainfall, waterbody characteristics, animal feed ingestion rates). These parameters were, in most cases, based on USEPA-specified default values. A few of the inputs are required to be site-specific and these were obtained or derived from locally-available information. In addition, the default values for some of the inputs were refined with site-specific information where possible. Table 4.2-6 summarizes the site-specific input parameters used to calculate environmental concentrations in this risk assessment, along with the basis for each value. Other than these site-specific values, all other inputs were based on USEPA's (2005b) recommended default values.

The risk assessment calculated environmental concentrations for a variety of hypothetical receptors in the facility area. As noted above in Table 4.2-4, several receptor point locations identified from the unitized ISTST3 modeling results were evaluated. The default methods used to calculate environmental concentrations for these receptor points were extremely conservative, in that the calculations implausibly assume homegrown produce, home-raised animals and the animal's locally-obtained feed all come from a single receptor point, rather than averaged across the acreage necessary to grow large quantities of produce or crops, and to raise animals. These hypothetical receptor scenarios were complemented by the addition of four area-based residential receptors. Two of these area-based receptors were evaluated using as inputs unitized modeling results averaged across the Town of Parker and across the CRIT Reservation area with access to irrigation water and within the modeling domain (i.e., the receptors were not located at any single point). Similarly, the unitized modeling results averaged across waterbody and watershed areas for the Main Drain and the Colorado River within the modeling domain were used to evaluate two fish ingestion pathway receptors. These two waterbodies were selected based on input received from local officials and USEPA Region 9 during the Workplan preparation period of this project, although the extent of fishing in the Main Drain may be extremely limited (Masters 2007). Table 4.2-7 summarizes all the receptors evaluated in the stack emissions risk assessment, including both receptors located at specific points as well as receptors evaluated based on area-wide modeling results.

4.2.6   Calculation of Human Exposures

The last exposure assessment step is the calculation of human exposures in the facility area for each pathway. These calculations relied on the methods laid out in USEPA (2005b), which are programmed into the IRAP software. The types of information used to calculate exposures include rates of exposure for each pathway (e.g., food ingestion rates, soil ingestion rates), the fraction of ingestion of particular food types from locally-raised produce or animal products, and data on body weight, exposure frequency (i.e., days/year exposed) and exposure duration (i.e., total years exposed). As noted above, the exposure rates addressed both children and adults, consistent with current USEPA (2005b) guidance. A few of the exposure parameters were refined based on site-specific information received from Masters (2007), specifically the fraction of homegrown produce ingested by a resident was assumed to be 20% and the fraction of home-raised beef, pork, poultry and eggs ingested by a farmer was assumed to be 20%. All other exposure parameters were based on USEPA health-protective default values, including the default assumption of subsistence fishing.

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4.3   Fugitive Emissions Exposure Assessment

This section of the report includes an exposure assessment of potential fugitive air emissions associated with the carbon reactivation facility. The Workplan described a variety of processes involving spent carbon at the facility that have the potential for fugitive particulate and volatile organic compound (VOC) emissions. The reader is referred to Section 4.3.1 of the Workplan for this discussion. In general, potential fugitive emissions from activities involving spent carbon are reduced through standard work practices, facility design, and air pollution control (APC) devices. At no time other than during unloading is spent carbon exposed directly to the ambient environment. In addition, the intrinsic highly adsorptive nature of spent carbon results in very low partitioning of contaminants from the carbon to the atmosphere.

4.3.1   Potential Fugitive Emission Source Selected for Evaluation

Based on the review of the potential for fugitive air emissions from activities involving spent carbon presented in the Workplan, the activity expected to have the highest potential impacts associated with fugitive air emissions from spent carbon was identified for evaluation in this study. This activity is spent carbon unloading at the outdoor hopper (H-1). The outdoor hopper is an enclosed three-walled building with a fixed roof located on a concrete containment area. It has heavy long plastic sheeting on the front where spent carbon is unloaded. The hopper has an air exhaust system which filters collected air from inside the structure through a fabric filter baghouse and carbon adsorption system. A hand-held water spray system is also used at H-1 during unloading if needed to minimize potential dust emissions from dry spent carbon and to facilitate transfer of the spent carbon from the hopper through the piping system to the spent carbon storage tanks.

Based on data collected at the facility from 2005 and 2006, between 82%-86% of the spent carbon received at the facility annually is unloaded into the outdoor hopper from a variety of different bulk container types (e.g., roll-off containers, slurry trucks). The remainder is unloaded indoors inside the spent carbon storage and warehouse building into hopper H-2 (e.g., drums, supersacks). Hopper H-2 is also equipped with an air exhaust system, which directs collected air to the same baghouse and carbon adsorber as the outdoor hopper.

There are two general types of spent carbon received at the facility: wet carbon (referred to as "aqua carbon") which has been used for water treatment and is roughly 50% moisture content by weight, and dry carbon (referred to as "vapor carbon") which has been used for air treatment and is roughly 10% moisture content by weight. Data from 2005 and 2006 show that approximately 42%-46% of the spent carbon unloaded at the outdoor hopper is wet while about 54%-58% of the unloaded spent carbon is dry.

4.3.2   Selection of Chemicals for Evaluation

The next step in the fugitive emissions analysis was the selection of chemicals of potential concern to be evaluated. This selection process considered data on each compound's concentration in spent carbon, the frequency and magnitude of spent carbon deliveries

containing both volatile and inorganic compounds, each organic compound's tendency to volatilize into ambient air during unloading, and the potential toxicity of the compound. Table 4.3-1 presents a summary of this information for those compounds received in spent carbon at the facility from 2003-2006, based on the facility's Toxics Release Inventory reporting.[9]

The compounds listed in Table 4.3-1 were then ranked for a variety of factors that could be associated with potential risks in order to select chemicals of potential concern. Compounds were ranked in the following categories:

- Number of deliveries over the 4-year 2003-2006 period
- Total pounds delivered over the 4-year 2003-2006 period
- Potential volatility (based on concentration and Henry's law constant)
- Potential for acute inhalation health effects (based on chemical concentration and acute reference air concentration),
- Potential for chronic non-cancer health effects (based on chemical concentration and chronic inhalation reference air concentration),
- Potential for chronic cancer risks (based on chemical concentration and inhalation cancer unit risk factor)
- Identification of compounds that are known human carcinogens

Compounds were selected for evaluation for the fugitive emissions analysis if they ranked in the top five of any category or are classified as a known human carcinogen by the USEPA, the International Agency for Research on Cancer, or the U.S. National Toxicology Program. The top five ranking results, as well as the 21 selected compounds of potential concern for detailed evaluation, are shown in Table 4.3-2.

4.3.3   Calculation of Fugitive Emission Rates

Calculation of emission rates is the next step after the selection of chemicals for evaluation. In this study, fugitive air emission rates were calculated using mathematical modeling. The emission rates are combined with air dispersion modeling results to calculate potential ambient air concentrations, and associated inhalation risks. This section describes the emission modeling methods for both fugitive organic vapors as well as dusts and inorganic compounds that may be present in dust. The fugitive emission modeling did not take into account the air exhaust system employed at the outdoor hopper, an approach that is expected to overestimate potential emission rates.

4.3.3.1   Fugitive Organic Vapor Emissions

Organic compound emissions during spent carbon unloading at the outdoor hopper were calculated using two mathematical modeling methods developed for USEPA (USEPA 1997, 2004a). Conceptually the approach was based on a pore space gas model developed to

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[9] The Toxics Release Inventory (TRI) Report for 2003-2006 was provided to CPF by M. McCue, Director of Plant Operations, Siemens Water Technologies Corp. May 2007.

calculate organic emissions from dumping of petroleum-contaminated soil onto piles (this model was developed by Radian for USEPA 1997). The Radian model calculates an emission rate by assuming that a portion of the chemical concentration within the air-filled pore space of the dumped material is released to the atmosphere during unloading.

Two sets of calculations were performed to address the two different types of spent carbon unloaded at the outdoor hopper (i.e., aqua carbon and vapor carbon). These types of spent carbon were evaluated separately because their characteristics vary (e.g., moisture content, types of containers unloaded).

Chemical concentrations within the air-filled pore space of spent carbon were calculated using a method outlined by USEPA (2004a), based on work by Johnson et al. (1990) and Johnson and Ettinger (1991), which mathematically partitions the total concentration of a compound into sorbed, aqueous, and vapor phases. The partitioning is modeled by taking into account chemical-specific properties as well as properties of the material, as follows:

$$C_s = \frac{\left(H'*C_{sp}*BD\right)}{\left(Ew + Koc*foc*BD + H'*Ea\right)}$$ (Equation 4-2)

where

C_s	=	chemical concentration in air-filled pore spaces (g/cm^3),
H'	=	Henry's law constant (unitless),
C_{sp}	=	concentration in spent carbon (g/g),
BD	=	bulk density (g/cm^3),
Ew	=	water-filled porosity of spent carbon (unitless),
Koc	=	organic carbon:water partition coefficient (cm^3/g),
foc	=	fraction organic carbon in spent carbon (unitless), and
Ea	=	air-filled porosity of spent carbon (unitless).

Chemical emission rates associated with spent carbon unloading at the outdoor hopper during the workday were then calculated based on the Radian model methodology (USEPA 1997) as follows:

$$ER = \frac{\left(C_s*Vol*HR*Exc\right)}{AT}$$ (Equation 4-3)

where

ER	=	chemical emission rate (g/sec),
Vol	=	volume of air pore space within spent carbon per hour during unloading (cm^3/hr),
HR	=	hours unloading per workday (4 hrs),
Exc	=	pore gas to atmosphere exchange constant (unitless), and
AT	=	averaging time (25,200 seconds per 7-hour period between

7 AM – 2 PM when unloading activities occur).[10]

The volume of air within spent carbon during an unloading event was calculated as follows:

$$Vol = \frac{(Ea * Q * 1{,}000)}{BD} \qquad \text{(Equation 4-4)}$$

where

Vol	=	volume of air pore space within spent carbon per hour during unloading (cm^3/hr),
Q	=	amount of spent carbon unloaded per unloading event per hour (kg/hr), and
1,000	=	conversion factor (1,000 g/kg).

The amount of spent carbon unloaded per hour (Q) was calculated based on data specific to this facility, including an analysis of spent carbon containers' capacities, approximate unloading times per container type, and the average amount of spent carbon, by container type and container capacity, unloaded during 2005 and 2006. The amount unloaded per unloading event per hour was calculated as follows:

$$Q = \frac{Mass_{sp}}{Hrs_{sp}} \qquad \text{(Equation 4-5)}$$

where

$Mass_{sp}$ =	average mass of spent carbon unloaded per event (2,975 kg aqua spent carbon or 1,783 kg vapor spent carbon), and
Hrs_{sp} =	average unloading duration per container (0.77 hours for aqua spent carbon containers or 0.55 hours for vapor spent carbon containers).

The scenario-specific input parameters for these modeling equations are presented in Table 4.3-3. The values for these parameters were based on spent carbon data from the facility, where available, or from the published literature (e.g., Kleineidam et al. 2002). Note that several of the parameter values vary for the two different types of spent carbon unloaded at the outdoor hopper (vapor or aqua spent carbon). Table 4.3-4 presents the chemical-specific input parameters used in the modeling equations to calculate emission rates. Table 4.3-5 presents the calculated organic compound chemical emission rates for each selected chemical of potential concern.

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[10] Personal communication with M. McCue, Director of Plant Operations, May 7, 2007.

4.3.3.2   Fugitive Dust and Inorganic Compound Emissions

Emission rates of dust and inorganic compounds during spent carbon unloading were calculated using a screening-level emission factor equation presented by USEPA (2006) that calculates dust emission rates from batch drop operations. This model was developed based on test results for a variety of materials used in a variety of industries, such as the coal and quarrying industries. The fraction of particles less than 75 microns in diameter (known as "silt content" in soil science) in the tested materials ranged from 0.44%-19%. Analyses of dry spent carbon from the facility show a silt content of roughly 0.5% (i.e., passing through a 200-mesh sieve screen).[11] This means that spent carbon has a silt content at the low end of the range of tested materials used to develop the USEPA emissions model, and thus it is likely to have a lower potential to generate dust emissions than the model predicts. As a result, the dust emission rates calculated using USEPA's emission factor are likely to be overestimated.

Dust emission rates were calculated only for vapor spent carbon unloaded at the outdoor hopper, since dust emissions will not occur during unloading of the water-saturated aqua carbon. In addition to total dust emissions, emission rates for different particle size categories were calculated using USEPA's default particle size multipliers. The particle sizes evaluated were selected for consistency with comparison benchmark particulate matter concentrations that are available. Accordingly, emission rates for inhalable particles less than or equal to 10 microns (i.e., PM10) were calculated for comparability to the National Ambient Air Quality Standards (NAAQS) set under the Clean Air Act and workplace exposure limits. Emission rates for PM2.5 were also calculated for comparability to the PM2.5 NAAQS.

The emission factor equation presented by USEPA (2006) is as follows:

$$E = (k * 0.0016)\frac{\left(\dfrac{U}{2.2}\right)^{1.3}}{\left(\dfrac{M}{2}\right)^{1.4}} \qquad \text{(Equation 4-6)}$$

where

     E   =   emission factor (kg particulate matter/megagram batch drop material),

     K   =   USEPA default particle size multiplier (0.35 for PM10, 0.053 for PM2.5),

     U   =   mean wind speed (2.38 m/sec, based on Parker, AZ data), and

     M   =   material moisture content (10% for vapor spent carbon).

The particulate matter emission rate was then calculated as follows:

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[11] Spent carbon analytical report provided by Siemens Water Technologies Corp., Activated Carbon Laboratory, Los Angeles, CA. July 17, 2007.

$$ER_{PM} = E * Q * conv \qquad \text{(Equation 4-7)}$$

ER$_{PM}$ = emission rate of particulate matter (g/sec),
Q = amount of spent carbon unloaded per unloading event per
hour (kg/hr), and
conv = conversion factor (megagram/1,000 kg * 1,000 g/kg * hr/3,600 sec).

Chemical-specific emission rates for inorganic compounds were then calculated by multiplying the particulate matter emission rate by the chemical concentration in the vapor spent carbon, as follows:

$$ER_{cpd} = ER_{PM10} * C_{sp} \qquad \text{(Equation 4-8)}$$

ER$_{cpd}$ = inorganic compound emission rate (g/sec),
ER$_{PM10}$= emission rate of PM10 particles (g/sec), and
C$_{sp}$ = concentration in spent carbon (g/g).[12]

Inorganic compound emission rates were calculated from the inhalable PM10 particle size category emission rate (i.e., ER$_{PM10}$) for comparability to occupational exposure limits and for the inhalation risk assessment.

The scenario-specific input parameters and calculated dust emission rates are presented in Table 4.3-6. Table 4.3-7 presents the calculated inorganic compound chemical emission rates for each selected chemical of potential concern.

4.3.4   Air Dispersion Modeling for Fugitive Emissions

Air dispersion modeling was conducted using the ISCST3 model to calculate ambient air concentrations associated with fugitive emissions during spent carbon unloading. Appendix D describes the details of the modeling performed for the fugitive emissions source. As described in the Workplan, fugitive emissions from the hopper were treated in ISCST3 as a volume source, with dimensions defined by the hopper building, and were modeled using a unitized (i.e., 1 g/sec) emission rate. The emission source was assumed to be "on" every day for the 7-hour period during 7 AM - 2 PM, based on the period of time during typical facility operations that spent carbon may be unloaded at the outdoor hopper.[13] The meteorological data used to model the fugitive emissions source were identical to the data used to model dispersion of stack emissions (e.g., 2001-2005 Arizona Meteorological Network data from Parker). The set of off-site receptor grid points used for stack emissions modeling was also applied for the fugitive emissions modeling.

The ISCST3 model calculated unitized annual average modeling results (to evaluate chronic long-term risks) and 1-hour average modeling results (to evaluate short-term acute inhalation

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[12] For the inorganic compounds evaluated, total spent carbon concentrations were assumed to reasonably reflect the concentrations that would be solely associated with the solid phase.

[13] Personal communication with M. McCue, Director of Plant Operations, May 7, 2007.

risks) at all of the modeled off-site receptor locations beyond the property boundary. Since the modeling was performed using a unitized emission rate, the resulting ISCST3 air concentrations were expressed in units of μg/m^3 per 1 g/sec. Chemical-specific concentrations were then calculated using the IRAP software by multiplying the unitized results by the chemical-specific emission rates.

The specific locations addressed in the fugitive emissions risk assessment were identified by examining the unitized ISCST3 modeling results across specified types of land use areas. The annual average unitized modeling results within areas currently used for residential assessment purposes within the Town of Parker and within the CRIT Reservation with access to irrigation water were examined, and the maximum annual average impact locations in both areas were selected for detailed evaluation. The 1-hour average unitized modeling results were examined to identify maximum impact locations within residential assessment areas of the Town of Parker and the CRIT Reservation with access to irrigation water, at locations used for non-residential purposes, and at the maximum impact point beyond the property boundary. In addition to these locations, the receptor locations selected earlier for the stack emissions risk assessment were also evaluated. Table 4.3-8 lists all of the receptor point locations selected for evaluation for both the chronic and acute fugitive emissions inhalation risk assessment. Figure 4-4 shows these locations overlain on a topographical map of the area.

4.3.5   Identification of Exposure Pathways

The next step in the fugitive emissions analysis was the selection of exposure pathways for evaluation. As explained in the Workplan, the most important exposure pathway for this type of emissions source is direct inhalation and, accordingly, this risk assessment focused on the inhalation pathway of exposure.

4.3.6   Calculation of Environmental Concentrations

Chemical concentrations in ambient air were calculated, as described above, by multiplying the unitized results by the chemical-specific emission rates. This calculation was performed using the IRAP software for all the selected inorganic and organic compounds at the evaluated receptor locations. The organic compound emission rates used in this calculation were, however, based only on the vapor carbon values; since these emission rates were higher than for aqua spent carbon, this will tend to overestimate air concentrations and associated risks.

4.3.7   Calculation of Human Exposures

Inhalation exposures were calculated using the IRAP software. These calculations rely on the modeled ambient air concentrations, inhalation rates, and data on body weight, exposure frequency (i.e., days/year exposed) and exposure duration (i.e., total years exposed). Exposures due to inhalation were calculated using the HHRAP default assumptions for both an adult and a child.

Figure 4-4
Receptor Point Locations Evaluated in the
Fugitive Emissions Risk Assessment
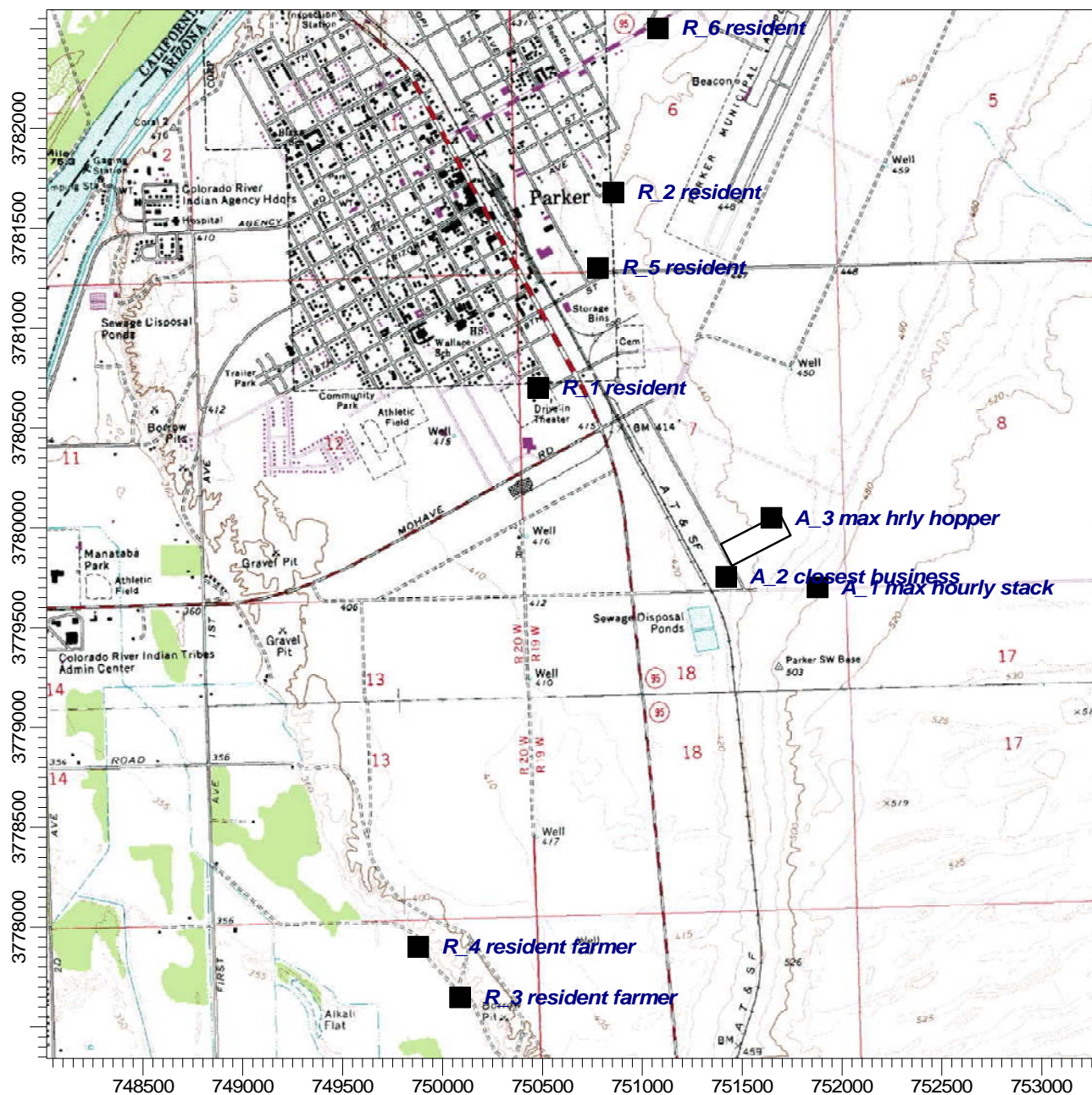
COMMENTS:

COMPANY NAME:

MODELER:

SCALE: 1:32,047

0 ⎯⎯⎯⎯⎯ 1 km

DATE:

PROJECT NO.:

IRAP-h View - Lakes Environmental Software

4.4   Risk Characterization

This section of the report presents the risk characterization, in which potential risks associated with both stack and fugitive emissions are addressed. As described earlier, the stack emissions risk assessment was a multiple exposure pathway analysis, whereas the fugitive emissions risk assessment addressed only the inhalation pathway of exposure.

4.4.1   Stack Emissions

4.4.1.1   Chronic Long-Term Risks

Chronic long-term risks associated with stack emissions were calculated according to the HHRAP methods and using the IRAP software to perform the calculations. Both excess lifetime cancer risks and the potential for non-cancer effects were evaluated. This was accomplished by combining exposures with toxicity values for cancer and non-cancer effects.

Excess Lifetime Cancer Risks

Cancer risks reflect the upper bound probability that an individual may develop cancer over a 70-year lifetime under the assumed exposure conditions. The risks are referred to as "upper bound" because they are unlikely to be underestimated and, in fact, may range from as low as zero to the upper bound value. Cancer risks were calculated, by the IRAP program, separately for each chemical and summed across chemicals for each exposure pathway. Risks were also added across pathways for hypothetical population groups that were evaluated (e.g., adult and child resident, adult and child farmer). The cancer risks were evaluated relative to the USEPA (1998a) target risk level of 1E-5 (which is equivalent to 1×10^{-5}). A cancer risk of 1×10^{-5} means that an individual could have, at most, a one in 100,000 chance of developing cancer over a 70-year lifetime under the evaluated exposure conditions. In comparison, each person in the U.S. has a background risk of developing cancer over a lifetime of about one in three.

The excess lifetime cancer risks are shown in Table 4.4-1. The detailed results for each exposure pathway and receptor are provided in Appendix G. As can be seen in this table, results are presented for the following three groups of evaluated chemicals:

- *Group 1 - All detected compounds.* This group includes 95 compounds that were detected in the PDT in addition to several compounds that were not measured during the PDT but which were evaluated based on emission rates derived from feed rates.
- *Group 2 - All evaluated compounds, both detects and compounds that were not detected, except for benzidine.* This group includes 177 compounds, 82 of which were not detected in the PDT. This group does not include benzidine which was not detected in the PDT in stack gases and for which there is no evidence from waste profile reports and analytical spent carbon data that it has ever been accepted

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in spent carbon received at the facility. [14] Benzidine was singled out because it was found to be a significant risk driver, accounting for more than 95% of the total cancer risk when included in the risk calculations.

- *Group 3 - All evaluated compounds.* This group includes 178 compounds, of which 83 were not detected in the PDT, including benzidine.

The risks are also presented for three general categories of human receptors who could hypothetically be exposed to stack air emissions:

- *Resident receptors.* These receptors include residential assessment locations in the Town of Parker and assume exposure occurs via inhalation, soil ingestion and homegrown produce ingestion.
- *Farmer receptors.* These receptors include residential assessment locations assumed to have access to irrigation water and assume exposure occurs via inhalation, soil ingestion, homegrown produce ingestion, and ingestion of home- or locally-raised beef, poultry, eggs, and pork.
- *Fish ingestion.* These receptors are assumed to fish in either the Main Drain or the Colorado River with exposures occurring only as a result of fish ingestion. These risks may be added to any of the evaluated residential receptors.

The additional (i.e., excess) lifetime cancer risks for Group 1, all detected compounds, ranged from 1E-9 (one in one billion) for the fish ingestion pathway, to 6E-8 (six in one hundred million) for resident receptor R_2. These results were more than 100 times lower than the 1E-5 target cancer risk level.

The risk results for Group 2, all detected and non-detected compounds except benzidine, were slightly increased above Group 1, while still well below the target level. Excess lifetime cancer risks calculated for Group 2 ranged from 1E-9 (one in one billion) for the fish ingestion pathway, to 2E-7 (two in ten million), again for resident receptor R_2. These results are 50 or more times lower than the 1E-5 target cancer risk level.

For Group 3, which added the non-detected compound benzidine to the risk calculations, excess lifetime cancer risks increased for all the residential receptors but did not change for the fish ingestion pathway. The highest cancer risk result was 2E-6 (two in one million) for the resident receptor R_2, five times below the 1E-5 target cancer risk level. As noted above, when benzidine was included in the risk calculations for the resident and farmer receptors, it accounted for more than 95% of the total cancer risks, even though this compound was not detected in the PDT, and there is no evidence from waste profile reports and analytical spent carbon data that it has ever been accepted in spent carbon received at the facility. If fish ingestion risks were added to the evaluated resident and farmer receptor results, all the excess lifetime cancer risks would still remained below the target risk of 1E-5.

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[14] Benzidine was used in the past mostly to produce dyes, however, it has not been produced for sale in the U.S. since the mid-1970's. Major U.S. dye companies no longer make benzidine-based dyes, and benzidine is no longer used in medical laboratories or in the rubber or plastics industries (ATSDR 2001).

Although all the calculated excess lifetime cancer risks were below the target level, the results were examined to identify the dominant compounds accounting for the majority of the risks. This evaluation focused on Group 1 (all detected compounds) and Group 2 (all compounds except benzidine) because, as noted above, benzidine was not detected in the PDT but dominated the risk assessment results when included in the calculations. The dominant compounds affecting these risk assessment results are described below:

- For the resident receptors, the dominant compound in Group 1 was cadmium, accounting for over 90% of the total risk mostly due to direct inhalation. Cadmium was conservatively evaluated in this risk assessment using an emission rate based on a proposed permit limit that was more than 30 times higher than the emission rate measured during the PDT. This means that the risks calculated for cadmium in this analysis are expected to be overestimated due to the emission rate by at least a factor of 30.

- For the farmer receptors, the dominant Group 1 compounds were cadmium and PCDDs/PCDFs, accounting for roughly 75% and 23% of the total risks, respectively. The most important exposure pathway for PCDDs/PCDFs was beef ingestion. PCDDs/PCDFs also accounted for most of the calculated fish ingestion cancer risks. As with cadmium, PCDDs/PCDFs were evaluated in this risk assessment using emission rates based on a proposed permit limit. The measured PCDD/PCDF emission rates during the PDT, which was performed using spiked feed to maximize the production of combustion by-products such as PCDDs/PCDFs, were roughly four times lower than the values used in this risk assessment. Even with emission rates conservatively based on proposed permit limits, the cancer risks due to stack emissions for all detected compounds were well below the target risk level of 1E-5.

- The dominant compounds in Group 2 for the resident receptors included cadmium in addition to arsenic and beryllium, primarily due to inhalation exposure. Arsenic and beryllium were not detected in the PDT but were evaluated in the risk assessment using emission rates based on permit limits. The use of permit limits as a basis for emission rates for cadmium, arsenic and beryllium is expected to greatly overestimate potential risks, by more than an order of magnitude.

- For the farmer receptors, the dominant compounds in Group 2 still included cadmium and PCDDs/PCDFs, in addition to arsenic and beryllium. PCDDs/PCDFs continued to account for almost all of the calculated fish ingestion risks.

Potential Non-Cancer Effects

The potential for non-cancer health effects was evaluated by comparing calculated exposures with non-cancer oral reference doses (RfDs) and inhalation reference concentrations (RfCs), consistent with USEPA (2005b). A hazard quotient was calculated for each chemical, using the IRAP program, by dividing its exposure by its reference dose or reference air concentration. The hazard quotients for each pathway were added across all chemicals, as an initial evaluation step, regardless of the type of health effect endpoint, to produce what is called a hazard index. Hazard index results were evaluated against the USEPA (1998a)

target level of 0.25. This target hazard index level is quite conservative; in many other environmental regulatory programs the target hazard index level is 1.0.

A hazard index summed across all compounds, not taking into account the type of health effects associated with each compound, is a conservative first step in evaluating the potential for non-cancer effects. If the hazard index for all compounds is above a value of one (1), this indicates that the hazard index values should be recalculated for groups of compounds having similar types of health effects or the hazard quotient values for those compounds producing a hazard index above one should be examined in more detail. If the hazard index for compounds with similar types of health effects is below one, then adverse health effects are not expected to occur. Even if the hazard index for compounds with similar types of health effects is above one, this does not automatically mean that adverse health effects will occur (for example, because of the safety factors that are incorporated in the non-cancer reference doses and reference air concentrations). Rather, this type of result means that there is an increased chance that health effects might occur. In this case, further research should be conducted to evaluate the potential for public health effects.

The non-cancer hazard index values for stack emissions (summed across all compounds regardless of type of health effect) are shown in Table 4.4-1. These values ranged from 0.003 to 0.01, were essentially the same for all three groups of compounds (Groups 1, 2 and 3), and were 25 or more times lower than the target level of 0.25. If the hazard index results were recalculated for groups of compounds having similar types of health effects, rather than all compounds, the resulting values would be even lower and still well below the target level.

The dominant compounds affecting the hazard index results were chlorine, for the resident and farmer receptors, mostly due to inhalation, and methyl mercury for the fish ingestion pathway. Chlorine was evaluated in this risk assessment using an emission rate based on a proposed permit limit that was much higher than the results measured during the PDT. The permit limit-based chlorine emission rate was roughly 20 times higher than the emission rate measured in the PDT, even though many chlorine-containing compounds were spiked into the feed during the PDT. Similarly, mercury was evaluated in this risk assessment using a permit limit-based emission rate that was about 15 times higher than the measured PDT emission rate. These results indicate that the non-cancer results due to stack emissions were not only below the target level using emission rates conservatively based on proposed permit limits, but would be even lower if measured PDT emission rates were used.

Summary

These results show that additional lifetime cancer risks from long-term exposure to stack emissions are well below regulatory target risk levels and that non-cancer health effects are not expected to occur from long-term exposures to stack emissions in residential areas near to the reactivation facility.

4.4.1.2   Margin of Exposure for PCDDs/PCDFs

The USEPA has not developed a non-cancer reference dose for PCDDs/PCDFs. As an alternative, a margin of exposure approach developed by USEPA was applied to compare the calculated doses in the risk assessment to typical background U.S. exposure levels (USEPA 2005b). This analysis is consistent with USEPA's (2001a) request that a margin of exposure analysis be conducted to assess PCDDs/PCDFs. Following the USEPA (2005b) protocol, in this analysis, the maximum PCDD/PCDF toxic equivalent (TEQ) average daily dose predicted for an adult receptor in the risk assessment associated with stack emissions was compared to a typical background level of 1 pg TEQs/kg-day.   This analysis showed that the highest calculated average daily PCDD/PCDF TEQ dose to an adult (8E-5 pg/kg-day for farmer receptor R_3) was well below the typical background level.

4.4.1.3   Infant Exposure to PCDDs/PCDFs

The USEPA has not developed risk assessment methods to quantitatively evaluate the potential risks to a breast-fed infant from exposure to PCDDs/PCDFs. In this study, infant exposures to PCDDs/PCDFs were evaluated as an adjunct to the adult exposure scenarios evaluated for stack emissions. Hypothetical infant exposures were evaluated following the approach presented in USEPA (2005b), which is programmed into the IRAP software. In this method, the average daily dose to PCDDs/PCDFs, expressed as 2,3,7,8-TCDD toxic equivalents (TEQs), from breast milk ingestion is calculated and then compared to a comparison background level for a nursing infant. The comparison level used in this analysis was an average infant intake level of 60 pg/kg-day for 2,3,7,8-TCDD TEQs based on USEPA (2005b). It is very important to recognize, however, that the method specified for use in this risk assessment is a default regulatory approach; it does not reflect actual knowledge of the potential health effects, if any, of short-term exposure via breast-milk ingestion on an infant.

The calculated average daily doses from breast milk ingestion are shown in Table 4.4-2 for each adult receptor evaluated. These doses ranged from 0.0002 - 0.002 pg TEQs/kg-day, more than 10,000 times lower than the target intake level. These results indicate that potential exposure to PCDDs/PCDFs by a nursing infant would be far below background levels.

4.4.1.4   Acute Short-Term Risks

Facility Operating Conditions Under Non-Upset Conditions

The potential for short-term acute inhalation risks associated with stack emissions was also evaluated in the risk assessment, consistent with USEPA (2005b) methods. This was accomplished using the IRAP software, by comparing modeled short-term, 1-hour average air concentrations with the acute reference air concentrations in a manner similar to the evaluation of non-cancer risks. The evaluation addressed not only the maximum impact point for hourly concentrations beyond the facility boundary, but also receptors located in residential and non-residential land use areas.

The air concentrations used to evaluate acute risks were conservatively based on the highest 1-hour average air concentration calculated for each specified receptor location and compound out of a total of 43,800 hours evaluated by the ISCST3 model (i.e., 5 years of hourly meteorological data from 2001-2005 from Parker were used). The concentrations for the remaining 43,799 hours were lower than those used in this analysis.

An acute hazard quotient was calculated in the IRAP program by dividing each chemical's modeled 1-hour average air concentration by its acute reference concentration. Quotients below one are not expected to result in health effects. Quotients above one indicate an increased chance that mild transient adverse health effects might occur (e.g., eye irritation) or a clearly defined objectionable odor associated with the specific compound being evaluated might be noticed, although these may still be unlikely to occur because safety factors are incorporated in the acute reference air concentrations.

Table 4.4-3 summarizes the results of the acute inhalation analysis using the stack emission rates shown in Table 4.2-1. The detailed results are provided in Appendix H. As the summary table shows, the hazard quotients, which were calculated for each chemical individually, ranged from less than 1E-10 to 0.08. These values were all well below the target level of one, by factors of 12 or more times. If the hazard quotients for the individual compounds were added together for groups of compounds having similar types of health effects (e.g., respiratory), the combined results would still be well below a target level of one.

Upset Conditions

Acute inhalation risks were also evaluated assuming an upset condition occurred for 1 hour at the facility, during which emissions were assumed to increase by ten times as recommended in HHRAP. As noted earlier, the factor of 10 increase is based on a 15-year old conservative regulatory default assumption for nonhazardous waste combustors. The potential acute hazard quotients under this scenario would be ten times higher than those shown in Table 4.4-3, with values ranging from <1E-10 to a maximum of 0.8 occurring at the maximum 1-hour average impact point (i.e., location A_1 where there is no residential or commercial land use). If the hazard quotients for the individual compounds were added together for groups of compounds having similar types of health effects (e.g., respiratory), the combined results would still be below a target level of one.

The highest hazard quotients for all evaluated receptor locations under upset conditions were due to arsenic, nitrogen dioxide, chlorine, and sulfur dioxide, with values at the maximum impact point (A_1) of 0.8 for arsenic, 0.4 for nitrogen dioxide, 0.09 for chlorine, and 0.07 for sulfur dioxide, and at the closest business location (A_2) of 0.2 for arsenic, 0.4 for nitrogen dioxide, 0.09 for chlorine and 0.07 for sulfur dioxide. The results for arsenic and chlorine were calculated using emission rates based on proposed permit limits that were much higher than the results measured during the PDT. The measured arsenic emission rate from the PDT was over 30 times lower than the emission rate used in this risk assessment, while the measured chlorine emission rate was roughly 20 times lower than the emission rate used in this risk assessment (and chlorine was spiked into the feed during the PDT). These differences in evaluated versus measured emission rates indicate that the acute hazard

quotients for arsenic and chlorine under both non-upset and upset conditions, are expected to be overestimated by more than a factor of 10.

The acute toxicity criteria for the compounds with the highest hazard quotients were all based on acute reference exposure levels from the California Environmental Protection Agency, which lists mild respiratory irritation as the health effects endpoint for chlorine, nitrogen dioxide and sulfur dioxide and lists reproductive/developmental effects (based on reduced fetal weight in mice) for arsenic. Hazard quotients may be added together to evaluate potential risks for multiple compounds, but only for groups of compounds having similar health effects endpoints. In this case, the sum of all hazard quotients grouped for compounds with similar health effects endpoints remains below the target level of 1.0.

Summary

These results indicate that short-term health effects are not expected to occur in areas near to the reactivation facility as a result of inhalation exposure to stack emissions, either under conservatively evaluated long-term conditions or under hypothetical upset conditions..

4.4.1.5   Evaluation of Lead

USEPA (2005b) recommends that lead be evaluated in a combustion source risk assessment initially by comparison with a soil benchmark level of 400 mg/kg in soil. If the calculated soil concentration exceeds the benchmark, USEPA recommends that additional evaluation of potential blood lead levels be performed using the Integrated Uptake Biokinetic Model (IEUBK). In this study, the lead soil concentrations at the evaluated receptor locations, due to stack emissions, were calculated to range from 6E-6 mg/kg to 3E-4 mg/kg, more than one million times lower than USEPA's target level, indicating that no further evaluation of lead was warranted.

4.4.1.6   Comparison to Risk-Based Standards and Criteria

Consistent with the Workplan, the risk assessment also compared the calculated environmental concentrations to available standards and criteria. Specifically, the highest annual average modeled air concentrations associated with stack emissions at a residential receptor were compared with the NAAQS and USEPA Region 9 risk-based preliminary remediation goals (PRGs). Similarly, the maximum annual soil concentrations modeled at a residential assessment receptor were compared with USEPA Region 9 risk-based PRGs for residential soil. Concentrations calculated in surface water were also compared to ambient water quality criteria in the ecological risk assessment section of this report.

The results of this comparison, presented in Appendix I, showed that all the modeled air concentrations were far below both the NAAQS and the very conservatively derived risk-based PRGs. The modeled soil concentrations were also found to be far below the risk-based residential soil PRGs.

4.4.2   Fugitive Emissions

4.4.2.1   Chronic Long-Term Risks

Chronic long-term risks associated with fugitive emissions during spent carbon unloading were calculated by combining the inhalation exposures with toxicity values for cancer and non-cancer effects according to the HHRAP methods described in USEPA (2005b), using the IRAP software to perform the calculations. This methodology is the same as that described above for evaluating chronic risks from stack emissions. The fugitive emissions analysis evaluated only the inhalation pathway of exposure, as described above in the selection of pathways section.

The results of the chronic inhalation risk assessment for both cancer risks and non-cancer health effects are shown in Table 4.4-4. The detailed results for each compound evaluated are provided in Appendix J. The additional (i.e., excess) lifetime cancer risks ranged from 2E-9 (two in one billion) to 5E-8 (five in one hundred million); these results were 200 or more times lower than the 1E-5 target cancer risk level. The non-cancer hazard index values (summed across all compounds regardless of type of health effect) ranged from 0.0004 to 0.001; these values were 250 or more times lower than the target level of 0.25. If the hazard index results were calculated for groups of compounds having similar types of health effects, rather than all compounds, the resulting values would be even lower and still well below the target level of 0.25.   If the fugitive emissions risk results were added to those calculated for stack emissions, the combined results would still be below both the cancer and non-cancer target risk levels.

These results show that additional lifetime cancer risks in residential assessment areas near the reactivation facility, from long-term inhalation exposure to fugitive emissions from spent carbon unloading, individually or in combination with risks from stack emissions, are well below the regulatory target cancer risk level. Similarly, the results show that non-cancer health effects are not expected to occur from long-term inhalation exposure to fugitive emissions in residential assessment areas near the reactivation facility, individually or in combination with stack emissions.

4.4.2.2   Acute Short-Term Risks

The potential for short-term acute inhalation risks associated with fugitive emissions was also evaluated in the risk assessment. This was accomplished by comparing predicted short-term, 1-hour average air concentrations with acute reference air concentrations. The methodology described above for evaluating acute risks from stack emissions was also used to evaluate fugitive emissions.

Table 4.4-5 summarizes the results of the acute inhalation analysis for fugitive emissions. The detailed results for the selected chemicals are provided in Appendix K. As this table shows, the hazard quotients, which were calculated for each chemical individually, ranged from less than 1E-9 to 0.02 at the maximum off-site impact point (A_3). These values were all well below the target level of one, by factors of 50 or more times. If the hazard quotients for the individual compounds were added together for groups of compounds having similar

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types of health effects (e.g., respiratory), the combined results would be even lower, and still well below a target level of one. Moreover, if the acute results from fugitive and stack emissions for compounds emitted from both sources were added together at the evaluated receptor locations, the results would still be well below the target level.

These results indicate that short-term health effects are not expected to occur in areas near to the reactivation facility as a result of inhalation exposure to fugitive emissions during spent carbon unloading at the outdoor hopper, individually or in combination with risks from stack emissions.

4.4.2.3   Evaluation of Particulate Matter

The potential for health effects to occur as a result of fugitive particulate matter emissions was also evaluated. This analysis compared maximum off-site particulate matter (PM) concentrations to the NAAQS for PM10 and PM2.5. NAAQS are established by USEPA for criteria pollutants, including PM10 and PM2.5, and impose ambient air quality concentration standards which are determined by USEPA to be protective of public health with an adequate margin of safety. The current PM10 NAAQS is a 24-hour average of 150 $\mu g/m^3$, while the current PM2.5 NAAQS includes both a 24-hour average of 35 $\mu g/m^3$ and an annual average of 15 $\mu g/m^3$.

The maximum off-site annual average concentration of PM2.5 was calculated by multiplying the PM2.5 emission rate (see Section 4.3.3.2) by the maximum off-site unitized annual average concentration (which occurred at the property boundary where there is no residence). The resulting annual average concentration was 2.5E-3 $\mu g/m^3$, more than 6,000 times lower than the NAAQS. Maximum off-site 24-hour average PM10 and PM2.5 concentrations were calculated by multiplying the emission rates by the maximum off-site unitized 1-hour average air concentration (which also occurred at the property boundary), and also by a scaling factor of 0.4 to convert from a maximum 1-hour concentration to a maximum 24-hour concentration (USEPA 1992). The resulting PM10 and PM2.5 maximum 24-hour average concentrations were 0.6 $\mu g/m^3$ and 0.09 $\mu g/m^3$, respectively, 250 or more times lower than their respective NAAQS. This evaluation indicates that potential off-site impacts of particulate matter emissions associated with spent carbon unloading at the outdoor hopper will be protective of human health.

4.4.2.4   Comparison to Risk-Based Standards and Criteria

This part of the risk assessment compares the calculated ambient air concentrations associated with fugitive emissions to available standards and criteria. Specifically, the highest annual average modeled air concentrations at a residential assessment receptor were compared with NAAQS and USEPA Region 9 risk-based PRGs. The results of this comparison, presented in Appendix L, showed that all the modeled air concentrations were below both the applicable NAAQS and the very conservatively derived risk-based PRGs.

4.4.3   Wastewater Discharge from the Facility to the Joint Venture

4.4.3.1   Introduction

Wastewater discharged from the reactivation facility is transported via an underground pipe to the Colorado River Sewage System Joint Venture (CRSSJV) publicly owned treatment works (POTW). The reactivation facility effluent is regulated under an industrial wastewater discharge permit granted to SWT from the CRSSJV in accordance with the Clean Water Act.

The CRSSJV is a primary wastewater treatment plant that serves both the Town of Parker and the Colorado River Indian Tribes, a service population of approximately 5,000 people (USEPA 2001b). Roughly 18% of the water entering the POTW originates from the reactivation facility. Flow rate data from 2006 show a discharge rate from the POTW of about 709,000 gallons of water per day, with the reactivation facility contributing roughly 129,000 gallons per day to this amount. The remaining water entering the POTW comes from other businesses (e.g., Custom Metal Finishing, as indicated in USEPA 2001c) and households in the service area. The CRSSJV discharges the treated water to the Main Drain discharge canal, which begins slightly upstream of the CRSSJV discharge point and travels more than 10 miles in a south-southwesterly direction through the CRIT Reservation before discharging into the Colorado River. The amount of water flowing through the Main Drain substantially increases as it moves downstream due to the addition of water overflow from irrigation canals and seepage from adjacent agricultural land.

The CRSSJV performs semi-annual priority pollutant sampling of its discharge water, in addition to daily sampling for a variety of constituents, including metals, biological oxygen demand, pH and total suspended solids. Chronic aquatic toxicity tests are also conducted using raw CRSSJV effluent every 6 months on water fleas and fathead minnows.

4.4.3.2   Evaluation of Reactivation Facility Discharge

As requested by USEPA and described in the Workplan, a screening-level modeling analysis was conducted to evaluate the potential incremental contribution of the reactivation facility's effluent on chemical concentrations discharged from the CRSSJV into the Main Drain.

The incremental concentrations at the CRSSJV discharge were calculated using mathematical modeling. The calculated incremental concentrations were then compared to ambient water quality criteria in conjunction with a review of the CRSSJV semi-annual effluent toxicity testing results. In addition, potential fish tissue concentrations and associated potential human health fish ingestion risks were evaluated in the Main Drain at a location downstream of the CRSSJV discharge point where fishing was assumed to occur.

4.4.3.3   Evaluation of Reactivation Facility Incremental Impact to CRSSJV Discharge

Incremental chemical concentrations in the CRSSJV discharge due to effluent from the carbon reactivation facility were calculated in a series of six steps which are discussed below:

- Compile chemical concentrations in effluent and select compounds for evaluation
- Calculate total, dissolved and particulate concentrations in facility effluent
- Calculate incremental facility concentrations resulting from treatment at the CRSSJV
- Repartition concentrations at outfall between total, dissolved and particulate phases
- Compile ambient water quality standards and criteria for selected compounds
- Compare incremental facility concentrations to water quality standards

Compile Chemical Concentrations in Effluent and Select Compounds for Evaluation

Measurements of compounds in the reactivation facility effluent were compiled using data collected over the past two years (2005-2006) and provided to CPF by SWT.[15] Table 4.4-6 presents the data that were compiled.

All detected compounds, even if detected only once, were selected for evaluation. For these 19 detected compounds, the minimum and maximum detected levels were identified. Average concentrations were also calculated if there were at least three detected sample concentrations and the majority of reported results were detects. Table 4.4-6 indicates the compounds selected for analysis and the summary concentration data for each compound.

Calculate Total, Dissolved and Particulate Concentrations in Facility Effluent

The concentrations of each compound in the facility effluent in dissolved and particulate phases were calculated from the total measured concentrations according to a screening-level model provided by USEPA (1985):

$$C_{dissolved} = \frac{C_{total}}{[1 + (K_{dsw} * TSS * 1E-6)]} \qquad \text{(Equation 4-9)}$$

and

$$C_{particulate} = C_{total} - C_{dissolved} \qquad \text{(Equation 4-10)}$$

where

$C_{dissolved}$	= dissolved concentration in water (μg/L),
C_{total}	= total concentration in water (μg/L),
$C_{particulate}$	= chemical concentration on suspended solids in water (μg/L),
K_{dsw}	= suspended solids:water partition coefficient (L/kg),
TSS	= total suspended solids concentration (7 mg/L; average in facility effluent), and
1E-6	= conversion factor.

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[15] Data provided by M. McCue, Director of Plant Operations, Siemens Water Technologies Corp. May 2007.

The suspended solids:water partition coefficients were identified from recommended USEPA sources (USEPA 2005b, USEPA 2004b, USEPA 1996, and Baes et al. 1984). The partition coefficient was selected taking into account the average pH in the reactivation facility effluent (8.1) for those compounds for which the partition coefficient is pH-dependent (arsenic, barium, cadmium, chromium III and selenium), as described in USEPA's Soil Screening Guidance (USEPA 1996).

The results of these calculations are shown in Table 4.4-7. As indicated in this table, these calculations were performed using two sets of effluent concentrations in order to allow evaluation of both acute and chronic water quality impacts. The maximum single measured value (24-hour or less composite measurement) was conservatively used to model short-term concentrations for comparison to acute criteria or standards. The average concentration was used to model longer-term concentrations for comparison to chronic criteria or standards. Note that long-term concentrations could not be calculated for a number of compounds due to the large percentage of results that were non-detects.

Calculate Incremental Facility Concentrations Resulting From Water Treatment

The change in facility-related concentrations at the CRSSJV was calculated using a mass balance approach, taking into account both the effect of water treatment (particulate and organics removal) and the effect of water flow into the CRSSJV from other sources, as follows:

$$C_{CRSSJVoutfall} = C_{effluent} * (1-RE) * Q_{facilityeffluent} / Q_{CRSSJVoutfall} \qquad \text{(Equation 4-11)}$$

where

$C_{CRSSJV\ outfall}$ = incremental concentration at CRSSJV outfall (μg/L),
$C_{effluent}$ = concentration in facility effluent (μg/L),
RE = removal efficiency (98%),
$Q_{facility\ effluent}$ = water flow rate into CRSSJV (129,465 gpd), and
$Q_{CRSSJV\ outfall}$ = water flow rate at CRSSJV outfall (708,541 gpd).

The removal efficiency at the CRSSJV was determined from the treatment plant's discharge records for 2005 which showed 98% removal of biological oxygen demand (BOD) and 98% removal of suspended solids. Annual average flow rates for the reactivation facility effluent and the CRSSJV were determined from measurements collected at both locations throughout 2006. As noted above, incremental concentrations at the CRSSJV outfall were calculated separately using short-term and long-term reactivation facility effluent concentrations. Table 4.4-8 presents the calculated concentrations at the CRSSJV outfall due to the incremental contribution from the reactivation facility's effluent.

Repartition Concentrations at Outfall Between Total, Dissolved and Particulate Phases

The concentrations of each compound in the CRSSJV outfall, due to the reactivation facility effluent, were repartitioned between dissolved and particulate phases using the same methodology shown above. The total concentrations in the CRSSJV outfall due to the reactivation facility effluent were calculated by summing the dissolved and particulate phase results shown in Table 4.4-8. These total concentrations were then repartitioned between dissolved and particulate phases taking into account the average pH and suspended solids levels measured at the CRSSJV outfall (7.0 and 3 mg/L, respectively). The resulting concentrations, presented separately for acute and chronic evaluation, are shown in Table 4.4-9.

Compile Ambient Water Quality Standards and Criteria For Selected Compounds

The next step in this evaluation involved compiling Arizona ambient water quality standards (WQS) and the CRSSJV discharge limits for the selected compounds. Arizona WQS for the Colorado River were assumed to be applicable to the CRSSJV outfall, based on similar treatment by USEPA in the CRSSJV's National Pollutant Discharge Elimination System (NPDES) permit. Table 4.4-10 presents the applicable standards and criteria for the selected compounds.

Compare Incremental Reactivation Facility Concentrations to Water Quality Standards

Table 4.4-11 presents the comparison of modeled incremental facility-related concentrations at the CRSSJV outfall to available water quality standards. The results of this screening-level effluent modeling analysis showed that the modeled concentrations in the CRSSJV discharge associated with the reactivation facility effluent were below the most stringent applicable State water quality standards and criteria and the CRSSJV discharge permit limits for all evaluated compounds except selenium. The modeled short-term selenium concentration was below the most stringent acute WQS, however, the modeled long-term average selenium concentration (2.4 µg/L) was marginally above the most stringent chronic criterion (Arizona's chronic WQS of 2 µg/L; the current USEPA ambient water quality criterion for selenium is 5 µg/L). This small difference is well within the bounds of uncertainty associated with the screening-level modeling evaluation and indicates that the modeled result is essentially equivalent to the WQS. Note that the calculated concentration at the outfall was based on the average concentration of selenium in the reactivation facility effluent over the past two years, whereas the effluent concentrations appear to be decreasing over time.

4.4.3.4   CRSSJV Effluent Toxicity Testing

The modeling results described above can be put into context by examining chronic toxicity testing results from the CRSSJV, which provide a more direct evaluation of potential aquatic toxicity of the treatment plant's discharge. Chronic toxicity testing is required to be performed semi-annually on effluent from the CRSSJV. These tests are conducted in January and July, each representing six (6) days of flow-weighted effluent composite samples. Test organisms are the water flea, *Ceriodaphnia dubia* and the fathead minnow,

Pimphales promelas. Toxicity endpoints are survival and reproduction for *C. dubia* and survival and growth for *P. promelas*. The tests are conducted according to USEPA protocols (USEPA 2002a, 2002b) and include the full range of quality assurance required by the guidelines. Among the many tests conducted from 2001 through 2006, there has been no statistically significant difference between control samples and samples with 100% effluent. On the basis of these tests, it may be concluded that the whole effluent from the CRSSJV possesses no toxicity to aquatic organisms.

4.4.3.5 *Potential Fish Ingestion Risks for the Main Drain*

The uptake of chemicals from the Main Drain into fish and associated potential human health risks from fish ingestion were also addressed, as requested by USEPA. The compounds evaluated in the fish ingestion risk analysis were those for which average concentrations were calculated at the CRSSJV outfall, due to the incremental contribution from the reactivation facility's effluent. The fish ingestion pathway was evaluated at a downstream location on the Main Drain where fishing may occur and where water flow rate measurements are routinely collected by the U.S. Geological Survey (USGS). In December 2001, USEPA's Region 9 GIS Center prepared a map for a public meeting that displayed three fishing locations on the Main Drain (about 7, 12, and 15.5 miles downstream of the CRSSJV outfall) (USEPA 2001c). The evaluated location in this study was the middle location, which was the only one of the three with detailed water flow rate and drainage canal dimension data (USGS Station # 9428508).

The methods specified in HHRAP were used to calculate fish tissue concentrations, fish ingestion intakes by people assumed to regularly eat fish caught from the Main Drain, and potential excess lifetime cancer risks and the potential for non-cancer health effects. Potential exposures and risks were evaluated for both an adult and a child assumed to regularly ingest fish caught from the Main Drain. In the absence of site-specific data, it was conservatively assumed that 100% of the fish eaten by a person every year for many years would be caught only from the Main Drain (i.e., USEPA's HHRAP default assumption for a subsistence fisher receptor).

Two modifications to USEPA's default methods were incorporated into the calculations to reflect more refined information. USEPA's default selenium bioconcentration factor included in HHRAP was updated to reflect more recent information which shows that diet is an important route of selenium exposure to fish (USEPA 2004c). Older concepts of selenium bioaccumulation assumed that uptake occurred primarily from water. Accordingly, a bioaccumulation factor (BAF) based on field studies (409 L/kg) was developed to reflect the importance of diet to selenium uptake to fish.[16] In addition, the fish ingestion intake for arsenic was adjusted to reflect the fraction of arsenic present in the inorganic form in fish, since most arsenic in fish is present in the non-toxic organic form

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[16] The bioaccumulation factor (BAF) for selenium used in both the stack emissions risk assessment and in this calculation was based on the geometric mean of 12 values reported in dry tissue weight from field studies (USEPA 2004c), adjusted to wet tissue weight following USEPA's HHRAP methodology (assuming a fish moisture content of 0.8 per USEPA (1999) Ecological Risk Screening Protocol). The resulting BAF was 409 (L/kg wet weight). This compares with USEPA's HHRAP default value of 129 (L/kg wet weight), which was based on the geometric mean of 12 laboratory values.

(ATSDR 2005). Field measurements of arsenic in freshwater fish show the fraction inorganic as 0.01-0.125 (ATSDR 2003, USEPA 2003c). The State of Arizona uses a fraction of 0.1 for inorganic arsenic in calculating the State ambient water quality criterion for arsenic for fish consumption.[17] In this analysis, the Arizona value of 0.1 was thus used to adjust the fish ingestion arsenic intakes.

Table 4.4-12 presents the results of the Main Drain fish ingestion risk analysis, and shows all of the input parameters, and exposure and risk calculation equations that were used. Both the excess lifetime cancer risks and the non-cancer hazard quotient values were below USEPA's target health benchmarks. The excess lifetime cancer risks were calculated to be 3E-7 for an adult subsistence fisherman and 4E-8 for a child subsistence fisherman, both assumed to obtain 100% of the fish ingested solely from the Main Drain. These results are 30 or more times below USEPA's target cancer risk level of 1E-5. The compound accounting for essentially all of the cancer risk was arsenic, based on a calculated dissolved concentration in the Main Drain of 0.033 µg/L which is more than 50 times lower than background levels in the Colorado River in the Parker area.[18] The total hazard index values, based on the sum of all hazard quotients regardless of their potentially differing health effects endpoints, were 1E-2 for both an adult and a child, more than 20 times lower than USEPA's target level of 0.25 and 100 times lower than the more common regulatory target level of 1.0.

4.4.3.6   Summary

Based on the evaluation presented above, it can be concluded that the incremental contribution of the facility effluent on the CRSSJV outfall and the Main Drain does not pose unacceptable risks to either aquatic life or human health. The modeled concentrations in the CRSSJV discharge associated with the reactivation facility effluent are below the most stringent applicable State water quality standards and criteria and the CRSSJV discharge permit limits for all evaluated compounds except selenium. The modeled short-term selenium concentration was below the most stringent acute water quality standard (WQS), however, the modeled long-term average selenium concentration (2.4 µg/L) was marginally above the most stringent chronic criterion (Arizona's chronic WQS of 2 µg/L; the current USEPA ambient water quality criterion for selenium is 5 µg/L). This small difference is well within the bounds of uncertainty associated with the screening-level modeling evaluation and indicates that the modeled result is essentially equivalent to the WQS. More importantly, semi-annual toxicity tests performed on the CRSSJV effluent have consistently shown no toxicity to aquatic organisms. Additionally, potential risks due to ingestion of fish caught from the Main Drain associated with the incremental contribution of the facility effluent were all below USEPA target risk levels for both cancer and non-cancer effects.

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[17] Personal communication. Email from S. Pawlowski, Arizona Department of Environmental Quality, to S. Foster, CPF Associates, Inc. May 29, 2007.

[18] The average dissolved arsenic concentration measured in the Colorado River below Parker Dam is 2.1 ug/L, based on 2000-2005 data from USGS Station #09427520.

4.4.4   Worker Health and Safety Evaluation

As indicated in the Workplan, a risk analysis was conducted using methods consistent with those adopted by OSHA and NIOSH in which workplace air concentrations were compared to workplace permissible exposure limits. The worker analysis focused on spent carbon unloading at the outdoor hopper, the activity expected to have the highest potential impacts associated with fugitive air emissions from spent carbon (as described in the review of activities presented in the Workplan). This activity was evaluated using both modeled on-site air concentrations and available employee industrial hygiene air measurements.

It should be noted, however, that the facility has a well-developed worker health and safety program operating in compliance with OSHA. This program includes training, medical monitoring, industrial hygiene sampling and use of personal protective equipment.
For further information on worker health and safety at the facility, the reader is referred to the detailed discussion provided in Section 4.4.4 of the Workplan and the discussion of the personnel training program and procedures used to prevent hazards at the facility in the RCRA Part B permit application (Focus 2007).

4.4.4.1   Modeled On-Site Concentrations

Ambient air concentrations for the worker scenario were calculated using the emission rates already described above for the fugitive emissions source in conjunction with ISCST3 modeling results. The dispersion modeling of this emission source was identical to that described above for stack emissions with two modifications. First, the ISCST3 air dispersion model was run for a set of on-site receptor locations (rather than off-site receptors), evenly spaced at 50 foot increments, to evaluate the on-site occupational scenario. Second, 8-hour average unitized modeling results were calculated (instead of annual and 1-hour averages) in order to evaluate concentrations relative to 8-hour average occupational exposure limits. Appendix D describes the air dispersion modeling in more detail.

The maximum 8-hour average modeling result occurred at the location closest to the hopper (about 10 m or 33 feet north of the hopper) for all five years of modeled meteorological data (2001-2005 datasets). The 8-hour average unit concentrations at this location ranged from 8,586 ug/m^3 per 1 g/sec (2001 meteorological data) to 16,426 ug/m^3 per 1 g/sec (2003 meteorological data). All other 8-hour average concentrations were lower than these maximum values. Chemical-specific concentrations on site were then calculated by multiplying the unitized maximum result (16,426 ug/m^3 per 1 g/sec) by the chemical-specific emission rates. The fugitive emission rates, and the methods used to calculate them, were presented earlier in this report.

4.4.4.2   Evaluation of Modeled Air Concentrations

Table 4.4-13 lists the modeled maximum 8-hour average air concentrations on site for the fugitive emissions source and compares these concentrations to available occupational 8-hour average exposure limits. The occupational exposure limits included Occupational Safety and Health Administration (OSHA) permissible exposure limits (PELs), National

Institute on Occupational Safety and Health (NIOSH) reference exposure limits (RELs) and, if NIOSH RELs were not available, American Conference of Governmental Industrial Hygienists (ACGIH) threshold limit values (TLVs).

As can be seen from Table 4.4-13, the modeled on-site maximum 8-hour average air concentrations for the evaluated chemicals were all below the available occupational exposure limits. The modeled concentrations were from 5 to more than 1,000,000 times lower than the corresponding occupational exposure limits. If the results were evaluated collectively, by summing the ratios of concentration to exposure limit across all compounds, the combined results would still be below the exposure limits. The highest result, having modeled concentrations 5-50 times lower than its occupational exposure limit, was for 1,3-butadiene, a compound that was present in only one delivery over the 4-year 2003-2006 period.

Potential on-site concentrations of total and respirable dust were also calculated and compared to occupational exposure limits. The calculated maximum 8-hour average total dust concentration was 2.8E-3 mg/m^3, well below the available occupational exposure limits for total dust identified by OSHA and ACGIH (15 mg/m^3 and 10 mg/m^3, respectively). The calculated maximum 8-hour average respirable dust concentration (based on PM10) was 9.6E-4 mg/m^3, also well below the available occupational exposure limits for respirable dust identified by OSHA and ACGIH (5 mg/m^3 and 3 mg/m^3, respectively).

4.4.4.3   *Industrial Hygiene Monitoring*

Industrial hygiene (IH) monitoring is conducted each year for a wide variety of organic compounds and dust in air to ensure that adequate personal protective equipment is being used at the facility. The annual IH surveys monitor workplace breathing zone concentrations of organic compounds and particulate matter among workers employed in a variety of tasks at the facility, for example workers unloading and sampling spent carbon containers, lab technicians and facility assistant managers. The results of annual IH surveys for the past 14 years, from 1993 through 2006, found that the air concentrations of regulated chemicals were either below quantitation limits or typically 100 or more times below occupational permissible exposure limits (PELs). The only exception occurred during the December 1999 IH survey when a spent carbon load containing a high level of benzene (roughly 60,000 ppm in spent carbon) was being unloaded at the outdoor hopper H-2. Three of the five personal samples collected during this survey, all from inside the hopper building, had time-weighted-average (TWA) benzene levels equal to or just above the PEL, ranging from 1.0 to 2.2 parts per million in air (ppm) versus the PEL of 1 ppm. The samples were collected from individuals who were working inside the hopper during the spent carbon unloading and who were wearing personal protective equipment, including respirators, in accordance with the facility's worker protection program. Results for the other 15 organic compounds tested during the December 1999 IH survey were all either below the quantification limit or more than 100 times below their corresponding PELs. Benzene results from all other IH air samples collected during the 1993-2006 period were either below the detection limit or well below the PEL.

4.4.4.4   Conclusions

These results indicate that fugitive air emissions during spent carbon unloading at the outdoor hopper, the activity for which potential impacts associated with fugitive emissions from spent carbon are expected to be highest, would not exceed occupational exposure limits in ambient air within the property boundary. These results are supported by many years of industrial hygiene measurements which have consistently shown air concentrations of regulated chemicals, excluding a few samples collected inside the hopper structure, either below quantitation limits or typically 100 or more times below the occupational PELs.

4.5   Discussion of Uncertainties

All risk assessments involve the use of assumptions, judgment and incomplete data to varying degrees. This results in uncertainty in the final estimates of risk. In accordance with standard risk assessment practice, this section of the analysis presents discussions of key uncertainties affecting the risk assessment.

4.5.1   General Review of Uncertainties

The results of any risk assessment inherently reflect uncertainty because of the many complexities involved in the analysis. This risk assessment, for example, involved the integration of many steps, each of which is characterized by some uncertainty. These steps included the following:

- Calculation of chemical emission rates
- Modeling of potential air concentrations and deposition rates associated with chemical emissions
- Calculation of chemical concentrations in the environment (e.g., soil, beef, produce, and fish) using mathematical models in conjunction with many chemical/physical properties and assumed or site-specific information about the environment in the facility area
- Calculation of potential exposures to humans through multiple pathways using a combination of default and site-specific exposure parameters
- Calculation of potential risks using toxicity information derived in some instances from human data but predominantly derived by extrapolation from experimental data produced in animal studies

There are four types of uncertainty generally associated with a risk assessment, as described in HHRAP and based on Finkel (1990):

- Variable uncertainty
- Model uncertainty
- Decision-rule uncertainty
- Variability

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Variable uncertainty results from uncertainties in the parameter values used in equations in the risk assessment. These uncertainties may stem from measurement, random or systematic errors associated with the numerical values assigned to input parameters. Variable uncertainty may be reducible through additional research or analysis (i.e., better data). Uncertain variables in a risk assessment include chemical-specific input parameters (e.g., biotransfer factors, cancer slope factors), and parameters describing the physical environment (e.g., characteristics of surface water bodies).

Model uncertainty is associated with models used in the risk assessment. The types of models incorporated into risk assessments include animal models used as surrogates for testing the human toxicity of chemicals, dose-response models used to develop chemical toxicity criteria, chemical property models used to calculate chemical-physical properties for the selected compounds, and fate and transport mathematical models used to calculate environmental concentrations of chemicals (e.g., HHRAP equations, ISCST3). Model uncertainty can stem from use of surrogate variables, excluded variables, abnormal conditions, and incorrect model structure.

Decision-rule uncertainty relates to uncertainties stemming from decisions applied in the risk assessment, including methods used to select chemicals for detailed evaluation, the decision to use USEPA default values in the analysis, the decision to use site-specific information to develop input parameters where information was available, and the decision to use USEPA-specified toxicity criteria to evaluate cancer and non-cancer risks.

Variability is related to variations in physical and biological processes, such as the natural differences in how much people weigh or how much they eat. Variability cannot be reduced by doing additional research but it can be addressed by incorporating information on the range of values that might be present in a population. In this risk assessment, many single point values were used for parameters that are known to vary across the population, and most of these were USEPA default values. Although this means that the risk results do not reflect variability in the population, when considered together the single point values, particularly USEPA's defaults, are expected to be more likely to overestimate risks than underestimate risks.

Table 4.5-1 summarizes some of the key elements of uncertainty associated with this analysis and also indicates whether each is expected to underestimate and/or overestimate potential risks. Discussions are also provided below for some additional topics and assumptions relevant to the risk assessment.

The risk assessment results presented earlier in this report reflect the combination of these potential sources of uncertainty. Collectively, however, the assumptions used in this assessment are considered more likely to overestimate risks than underestimate them.

4.5.2   Calculation of Emission Rates

Chemicals that have not been detected in emissions are sometimes included in combustion source risk assessments to ensure that risks are not underestimated. In this assessment, compounds that were not detected were included at the request of USEPA and, as described

in the Workplan, they were evaluated using the common risk assessment practice of assuming they were present at a concentration equivalent to one-half of the reported detection limit. It is, however, uncommon, if not unprecedented, for a combustion source risk assessment to evaluate as many compounds, both detected and not detected, as were considered in this study. The calculation of risk results for over 80 compounds that were not detected adds uncertainty to this study, because these compounds may not actually be present in stack emissions. Overall, the inclusion of so many compounds, including many that were not detected, is considered likely to overestimate risks and unlikely to underestimate risks.

As described in the Workplan, chemical emission rates based on PDT measurements were based on average values across the three test runs. USEPA requested that risks also be considered using emission rates based on the maximum out of the three test runs. This change is only relevant for compounds that were detected in the PDT and for which emission rates were based on PDT results. As noted earlier in this report, the dominant compounds affecting the stack emissions risk assessment results were all evaluated at proposed permit limits, and not based on PDT results (i.e., PCDDs/PCDFs, cadmium, mercury, and chlorine). This alone suggests that the risks would not likely be affected even if maximum emission rates were used instead of averages. Additionally, the emission rates based on PDT results and used in the quantitative risk assessment were compared to the maximum single test run results to determine the potential effect on the calculated risks. This comparison, which was conducted for compounds with emission rates based on PDT measurements, showed that the differences between the average and maximum PDT emission rates ranged from a factor of 1.0 (i.e., no change) to no more than a factor of 3.0. These relatively small differences for compounds that were not risk drivers indicate that the overall risk assessment results would not change if maximum PDT-based emission rates were used rather than averages.

4.5.3   Chemical Concentrations in Spent Carbon

The Workplan indicated that the risk assessment would include a discussion of the representativeness of the spent carbon used during the PDT relative to long-term operating conditions. This issue was examined by developing a profile of the mass-weighted average composition of various organic constituents and metals in the spent carbon received at the facility, based on 2003 through 2006 Toxics Release Inventory (TRI) data. In addition, analytical results from the PDT feed carbon for metals, volatile organics, and semi-volatile organics were averaged across the three test runs and compared to the mass-weighted average carbon profile. The results showed that the concentration of many of the compounds on the PDT feed carbon corresponded well with the mass-weighted average composition based on the TRI data, while other compounds were present on the PDT carbon at concentrations either significantly above or below the mass-weighted average carbon values. For two of the compounds in spent carbon that accounted for the majority of the calculated risks, cadmium and methyl mercury (assessed using elemental mercury in carbon), the concentrations in the PDT feed were higher than the average composition concentrations calculated from the long-term TRI data.

The variation in results from the comparison of the mass-weighted average composition based on the TRI data with the PDT carbon is not unexpected, since the spent carbon fed during the PDT was comprised of the carbon available at the time of testing, and no attempt had been made prior to the test (due primarily to space limitations) to stockpile any particular carbon from specific sources. It was for this very reason that the PDT included the spiking of the feed carbon with principal organic hazardous constituents (POHCs), metals, and a suite of organic surrogate compounds which were believed to represent various classes of compounds and which would likely produce a broad range of combustion by-products and very conservative emissions (i.e., expected to be greater than under typical operating conditions with typical spent carbon).

4.5.4   Examination of Dioxin-Like PCBs

Measurements of specific PCB congeners, compounds believed to have "dioxin-like" properties, were collected during the PDT (Focus 2006). [19] The purpose of this section of the risk assessment is to present an evaluation of the potential impact of the measured dioxin-like PCB congener emissions on the risk assessment results.

The World Health Organization (WHO 1998) has developed toxic equivalency factors (TEFs) for certain dioxin-like PCBs that relate the potential toxicity of each dioxin-like PCB to that of 2,3,7,8-TCDD. For example, the PCB congener 3,4,3',4'-tetrachlorobiphenyl has been assigned a TEF of 0.0001 by WHO, which means that this PCB compound is believed to be 10,000 times less toxic than TCDD. These TEFs, which are also summarized in HHRAP, were used to calculate potential excess lifetime cancer risks for dioxin-like PCBs.

The approach used to perform this evaluation involved several steps. First, emission rates of dioxin-like PCBs based on the PDT were compiled. Second, the potential lifetime average daily dose for each dioxin-like PCB was calculated by multiplying the lifetime average daily dose already calculated for total PCBs by the ratio of the measured PDT emission rate for the dioxin-like PCB divided by the emission rate for total PCBs. The total PCB lifetime average daily dose was based on the receptor and exposure pathway that was found to dominate the risk results for PCDDs/PCDFs (ingestion of fish caught from the Main Drain by an adult). This provided the most conservative indication of the potential impact of dioxin-like PCBs on the risk assessment. The average daily dose for each dioxin-like PCB was then multiplied by its WHO TEF to calculate the TCDD toxic equivalent (TEQ) dose for each dioxin-like PCB. After this, the sum of all the dioxin-like PCB TEQ doses was calculated. Finally, the cancer slope factor for TCDD was multiplied by the total dioxin-like PCB TEQ dose to calculate the associated potential excess lifetime cancer risk. Table 4.5-2 presents the calculations performed for each of these steps.

The resulting excess lifetime cancer risk associated with dioxin-like PCBs was 4.3E-9. This potential risk is lower than the cancer risk already calculated for the fish ingestion pathway for PCDDs/PCDFs (8.7E-9) and negligibly affects the overall results of this risk assessment.

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[19] A PCB congener is a single unique chemical compound in the PCB category. There are 209 PCB congeners, of which 12 are considered by USEPA to be dioxin-like compounds.

There are a variety of uncertainties that are associated with this analysis. For example, the assumption that a dioxin-like PCB compound's potency is directly proportional to the potency of 2,3,7,8-TCDD and that this relationship can be quantified based on a TEF. This analysis is also uncertain because it does not account for the differing physicochemical properties of the PCBs that can affect their environmental fate and transport. In addition, many of the PDT test results for dioxin-like PCBs, and PCB homologue groups, were so low that method blank results were significant in relation to the actual sample results, however, no blank corrections were made. Further, a number of the analytical results for these compounds had to be estimated by the laboratory in a manner that is most likely to give an upper bound result (i.e., flagged as an estimated maximum possible concentration). This means that the PDT test results, and the associated dioxin-like PCB excess lifetime cancer risks, are likely to be overestimated.

4.5.5   Total Organic Emissions

This risk assessment evaluated a very large number of specific chemical compounds, and determined not only that the risk results were below target risk levels, but also that over 97% of the cancer risks were due to two compounds (cadmium and PCDDs/PCDFs evaluated as TEQs) and over 91% of the chronic noncancer hazard quotients were due to two other compounds (chlorine and methyl mercury) when all detected compounds were evaluated. When all compounds except for one that was not detected (benzidine) were evaluated, roughly 80% or more of the cancer risks were due to four compounds (cadmium, PCDDs/PCDFs, arsenic and beryllium) and over 85% of the chronic noncancer hazard quotients were due to two other compounds (chlorine and methyl mercury). All of these risk-driving compounds were evaluated at proposed permit limits and two were not detected in PDT stack emissions (arsenic and beryllium).

The dominance of a few compounds on the risk assessment results suggests that other compounds that may be present in stack emissions but which were not quantitatively evaluated in the risk assessment are unlikely to affect the calculated risk results and would not change the overall conclusions of this risk assessment. In order to evaluate this uncertainty further, this section discusses the potential impacts of total organic emissions on the risk assessment results.

During the PDT, total organic emissions were measured for total volatile organic compounds, total semi-volatile organic compounds and total non-volatile organic compounds (Focus 2006).   These data were used to derive a total organic emissions (TOE) factor to determine the extent to which emissions of organic compounds not specifically evaluated in the risk assessment might affect the overall risk results. The TOE factor is defined as the ratio of the total organic compound emission rate divided by the sum of the emission rates for organic compounds quantitatively evaluated in the risk assessment. Current methods recommended by USEPA were used to derive this factor, though it should

be noted that there are very important uncertainties associated with this practice (USEPA 2005b). In this particular case, a TOE factor of 10 was calculated.[20]

The potential increase in risks associated with the TOE factor was evaluated by examining the excess lifetime cancer risks for the two receptors with the highest cancer risk results (resident receptor R_2 and farmer receptor R_3). The TOE factor was taken into account by assuming that the toxicity of the unidentified organics was the same as the toxicity of all organics that were evaluated, except PCDDs/PCDFs which are in a class by themselves with respect to potential toxicity. The excess lifetime cancer risks for resident receptor R_2 increased by a factor of 1.2 when all detected compounds were considered and a factor of 1.7 when all compounds except benzidine were included (i.e., revised total cancer risks of 8E-8 and 2E-7, respectively). The excess lifetime cancer risks for the farmer receptor R_3 were not affected when all detected compounds were considered and increased by a factor of 1.3 when all compounds except benzidine were included (i.e., an unchanged total cancer risk of 3E-8 and a revised risk of 8E-8, respectively). These results show that total organic emissions that were not evaluated had a negligible effect on the risks already calculated in this report, resulted in risks still well below USEPA target risk levels, and would not change the overall conclusions of this analysis.

4.5.6   Tentatively Identified Compounds and Compounds Without Human Health Toxicity Criteria

Tentatively identified compounds (TICs) in stack emissions were evaluated as part of the PDT. A description of the methods used to identify TICs is provided in the PDT test report (Focus 2006). In general, these methods focused on identifying those TICs present in the largest amounts in the collected stack samples and for which a chemical-specific identification could be made with confidence. In the PDT, 12 compounds were identified as TICs and all of these were selected for consideration in the detailed quantitative risk analysis.

USEPA-approved human health toxicity criteria were, however, not available for the TIC compounds as well as a number of other organics. Of the more than 200 compounds that were selected for detailed evaluation in this risk assessment, a total of 49 did not have chronic toxicity criteria and 17 did not have acute toxicity criteria either in HHRAP or in sources recommended by HHRAP. These compounds are listed in Table 4.5-3, with an indication of whether chronic and/or acute toxicity criteria were lacking.

The potential impact of TICs and other compounds without toxicity criteria on the risk assessment results was addressed by the TOE evaluation presented above. The TOE factor incorporates not only all of the compounds shown in Table 4.5-3 but also other unidentified organics that may potentially be present in stack emissions. The TOE evaluation showed that the overall conclusions of this analysis would not change even if these compounds had been able to be quantitatively evaluated in the risk assessment.

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[20] TOE factor = (TOE emission rate from PDT of 7.63E-3 g/sec) / (sum of emission rates of quantitatively evaluated compounds with chronic toxicity criteria of 7.87E-4 g/sec) = 9.7.

4.5.7   Evaluation of Irrigation Water Use

The IRAP software is not programmed to include inputs from irrigation water in calculating soil concentrations within an area. Soil concentrations were used in this assessment not only to calculate risks from soil ingestion, but also as inputs to the calculation of concentrations in other environmental media (e.g., produce, animals). The effect of this programming limitation was evaluated by comparing the chemical loading to agricultural area soil within the farmer receptor area that was included in IRAP (i.e., residential areas with access to irrigation water and within the modeling domain) to the chemical loading estimated to be due to irrigation water used over the same area. The chemical loading to soil addressed in HHRAP, and programmed into IRAP, reflects inputs due to direct deposition onto the ground surface. The loading was calculated based on a compound's emission rate, the unitized deposition modeling results, the fraction of the compound present in vapor and particulate phases, and the area across which deposition occurs. The loading due to irrigation was calculated based on the compound's concentration in irrigation water and the amount of water applied to the same area.

Irrigation water for the CRIT Reservation is withdrawn from the Colorado River above Headgate Rock Dam in Parker. For the purposes of this comparison, concentrations in irrigation water were assumed to be equivalent to those calculated by the IRAP software for the Colorado River within the modeling domain. The loadings to soil in the agricultural area within the modeling domain due to deposition (evaluated in IRAP) and due to irrigation water use were evaluated for three compounds with different characteristics to represent the range of possible differences in loadings. The three compounds were nickel, an inorganic with a fraction vapor of 0, methylene chloride, a volatile organic compound with a fraction vapor of 1.0, and PCBs (treated as Aroclor 1254), with a fraction vapor of 0.993. The results of the calculations for these three compounds showed that the loadings due to the use of irrigation water on soil were well below those already addressed in IRAP due to direct deposition, ranging from 65 times lower for PCBs to over 850 times lower for methylene chloride and nickel. These results indicate that the risks calculated for farmer receptors would not change if chemical loadings due to irrigation water use were included.

4.5.8   Selection of Meat Exposure Pathways

In this risk assessment, ingestion of several types of animal products was evaluated, consisting of beef, chicken, eggs, and pork. Some people in the facility area may, however, raise and eat goat and lamb (Masters 2007), and some may hunt for animals, including mule deer. Because the IRAP program does not include input parameters necessary to evaluate these pathways, they were not included in the quantitative calculations. As a result, an evaluation was conducted to estimate the extent to which risks might be underestimated by not including these exposure pathways.

This evaluation focused on the compound accounting for the majority of risks from the beef ingestion pathway, which was PCDDs/PCDFs with an excess lifetime cancer risk of roughly 5.6E-9 for the farmer receptor R_3. The total excess lifetime cancer risk for the farmer

receptor R_3 across all evaluated pathways and all detected compounds was 3E-8.[21] PCDD/PCDF concentrations in beef were calculated using biotransfer coefficients that are proportional to the fat content of beef (HHRAP default for beef is 19%). The potential for PCDD/PCDF uptake into goat, lamb and venison was evaluated, relative to beef, by identifying the fat content of each of these animal meats (2.3%, 23% and 2.4%, respectively).[22] The differences in fat content indicate that PCDD/PCDF concentrations could be about eight times lower in venison and goat, and about the same in lamb, compared to beef. If fat on processed lamb is trimmed to a greater extent than beef, then concentrations in lamb could be lower than calculated in beef. Assuming that people eat the same amount of each of these meats as beef, the excess lifetime cancer risk for ingestion of all four meat types was calculated by adjusting the beef ingestion pathway risk. This adjustment conservatively assumed that a farmer would ingest not only locally-raised beef, but also locally-raised lamb and goat, and locally-caught deer. The resulting cancer risk was 1.3E-8,[23] approximately two times higher than the beef risk, which would produce a total cancer risk for farmer receptor R_3 of roughly 4E-8. These results are still well below the target cancer risk of 1E-5, indicating that the overall risk assessment results would not be affected by including these additional meat ingestion pathways.

4.5.9   Evaluation of Subsistence Exposure Pathways

In the Workplan development phase of this project, USEPA (2001a) requested that the risk assessment address exposure due to subsistence hunting, agriculture and gathering of plants for cultural practices. This section discusses the potential impact on risks associated with subsistence agriculture and subsistence hunting. Potential risks associated with use of plants for cultural practices was not addressed in this report because the information request process outlined by CRIT for this project specified that confidential tribal practices would be assessed separately by CRIT.

Potential risks associated with subsistence agriculture, which was assumed to apply to both ingestion of homegrown produce as well as home-raised or locally-raised animal meats, were evaluated by assuming that 100% of the produce and animal meats ingested by a resident would be homegrown or locally-raised. As noted earlier in this report, the local Agricultural Extension Agent, with input from colleagues, estimated that residents in the area may ingest, at most, 20% of their produce and animal products from home-raised or locally-raised sources (Masters 2007).

Potential risks under the hypothetical subsistence agriculture scenario were evaluated for all compounds, both detected and not detected, except for benzidine (i.e., Group 2 compounds, see Section 4.4.1.1). Risks were calculated, by re-running the IRAP software, for the resident receptor and the farmer receptor with the highest excess lifetime cancer risks presented earlier in this report (i.e., 2E-7 for resident receptor R_2 and 6E-8 for the farmer

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[21] Of the total 3E-8 cancer risk, 23% was due to PCDDs/PCDFs of which over 80% was due to beef ingestion. The other dominant compound was cadmium, accounting for 75% of the total, of which over 80% was due to inhalation.

[22] U.S. Department of Agriculture Nutrient Database, Release 19. 2006. http://riley.nal.usda.gov/NDL.

[23] Approximated adjusted excess lifetime cancer risk = beef risk 5.6E-9 + lamb risk 5.6E-9 + goat risk 5.6E-9/8 + venison risk 5.6E-9/8 = 1.3E-8.

receptor R_3). The total excess lifetime cancer risks across all evaluated pathways combined for this subsistence scenario increased by a factor of 1.5, to 3E-7, for the resident receptor R_2, and by a factor of 1.7, to 1E-7, for farmer receptor R_3. These recalculated risks were more than 30 times below the USEPA target cancer risk level of 1E-5, indicating that consideration of a subsistence agriculture scenario would not change the overall results of this risk assessment.

Potential risks for a hypothethical subsistence hunting scenario were evaluated by analogy to the risk results for the beef ingestion pathway for farmer receptor R_3, assuming venison was the subsistence hunted food. As noted above, the risks for farmer receptor R_3, assuming 100% of all produce and animal meats ingested were from local or home sources, was calculated to be 1E-7. The dominant pathway contributing to this result was ingestion of beef (cancer risk of 3.7E-8) and the dominant compounds contributing to the beef risk were PCDDs/PCDFs (cancer risk of 2.8E-8). The analogous cancer risk from PCDDs/PCDFs for 100% ingestion of venison was then calculated to be roughly 3.5E-9, based on the fact that venison has roughly eight times less fat than beef (19% fat in beef / 2.4% fat in venison).[24] Conservatively assuming that all compounds other than PCDDs/PCDFs transfer to venison to the same extent as beef gives a total subsistence venison ingestion cancer risk across all compounds (except benzidine) of roughly 1.2E-8. This result is lower than the risk from 100% beef ingestion and well below USEPA's target risk level, indicating that potential risks from subsistence hunting would not alter the overall findings of this risk assessment.

4.5.10 Evaluation of Facility Effluent on the CRSSJV POTW

The incremental contribution of the facility effluent on chemical concentrations in the CRSSJV outfall and downstream in the Main Drain was evaluated using screening-level mathematical models which introduce uncertainty into this evaluation. Site-specific data were used in the calculations where possible to reduce uncertainty. The available site-specific data included: measurements of chemical concentrations in the facility effluent; measured water flow rates, pH levels and suspended solids levels in the facility effluent and the CRSSJV outfall; and measured water flow rates downstream in the Main Drain.

The analysis focused on a location on the Main Drain downstream of the CRSSJV where detailed water flow measurements and drainage ditch dimension data are collected and publicly available, and where fishing is believed to occur. Detailed local information on fishing behaviors was not available at the time this analysis was conducted and, as a result, it was conservatively assumed that 100% of the fish a person eats (i.e., every fish meal per year for many years) would be obtained solely from the one evaluated location on the Main Drain. This assumption is likely to over-estimate potential risks because people probably fish at a variety of locations, possibly along the Main Drain, possibly in other drains in irrigated areas, and/or in the Colorado River. The location that was evaluated in this analysis was considered likely to reasonably reflect potential risks for a person assumed to fish only from the Main Drain and at the fishing locations identified by USEPA (2001c). At

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[24] Approximated risk = 100% PCDD/PCDF beef ingestion risk of 2.8E-8 * (2.4% fat in venison / 19% fat in beef) = 3.5E-9.

more distant locations than that evaluated, Main Drain water flow rates will be higher (and potential fish tissue concentrations lower) while at closer locations, water flow rates will be lower (and potential fish tissue concentrations higher). Chemical concentrations in fish tissue were calculated using a simplified fish uptake mathematical model and primarily using default fish biotransfer values provided in HHRAP, an approach which may over- or under-estimate fish tissue levels.

5.0   ECOLOGICAL RISK ASSESSMENT

An ecological risk assessment was conducted to determine the potential effects of modeled stack air emissions on ecological receptors within the study area. The overall approach was based on the approved Workplan which was developed from USEPA's Guidelines for Ecological Risk Assessment (USEPA 1998b) and USEPA's Screening Level Ecological Risk Assessment Protocol for Hazardous Waste Combustion Facilities ("Screening Level Protocol") (USEPA 1999).

As described in the Workplan, this ecological risk assessment was designed to present a screening-level assessment focused on the potential effects of stack emissions on selected representative ecological receptors within the facility area considered to be at greatest risk. It was not intended to be an exhaustive evaluation of wildlife species that may be present or to evaluate all possible ecological receptors or exposure pathways.

5.1   Problem Formulation

Problem formulation is the process by which the receptors, endpoints, and pathways which become the focus of the ecological risk assessment are identified. The foundation of problem formulation is an understanding of the predicted relationships between ecological entities and the chemicals to which they may be exposed. From this foundation, the particular receptors and endpoints to be the focus of the assessment are defined.

The problem formulation step of this project was described in the Workplan. In summary, the problem formulation process resulted in the identification of habitat types considered in the risk assessment, as well as the selection of representative ecological receptors for detailed analysis. The habitat types that were considered consisted of creosote bush scrub, agricultural areas, riparian corridors and backwaters, the Colorado River, and the Main Drain. The receptor species or groups selected for evaluation consisted of aquatic life, plants, the badger, Gambel's quail, the great horned owl, the burrowing owl, the southwestern willow flycatcher, the double-crested cormorant, the Yuma clapper rail and mule deer. Table 5.1-1 summarizes the receptor species and pathways for each habitat type that were selected for evaluation in this risk assessment.

For terrestrial receptors, the assessment endpoint was maintenance of long-term health and reproductive capacity of these populations. The measures of effect (measurement endpoints) for these receptors were alteration of reproduction and survival for wildlife and alteration of survival and growth for plants. For aquatic life, the assessment endpoint was maintenance of species abundance and diversity within the study area aquatic community. The measures of effect were alterations of growth, reproduction, or survival in individual species, or changes in community structure, abundance, or diversity in benthic communities. For endangered or threatened species which were selected as receptors (i.e. Yuma clapper rail), the assessment endpoint was reproduction and survival of individual organisms, rather than the population, as specified by USEPA (2003a).

5.2   Risk Analysis Method

Ecological risks were evaluated using a predictive hazard quotient (HQ) approach. In this approach, exposures were calculated for each receptor species or group and then compared to receptor group toxicity reference values (TRVs). This section describes the selection of compounds for the ecological risk assessment, then presents a toxicity assessment, an exposure assessment, an analysis of potential risks, and a discussion of uncertainties.

5.2.1   Selection of Chemicals for Evaluation

Chemicals of potential concern (COPCs) were selected for consideration in the risk assessment in Section 4.1.1 of this report. These compounds were selected based on the results of the PDT and based on their potential to be present in spent carbon.

Starting with the comprehensive COPC list from Section 4.1.1 of more than 225 compounds, available TRVs were identified both from USEPA's 1999 Screening Level Protocol and by referring to the toxicological data sources listed in the Workplan. Compounds for which TRVs were available were quantitatively evaluated in the ecological risk assessment. Compounds without TRVs from the referenced data sources were discussed qualitatively in the uncertainties section.

5.2.2   Toxicity Assessment

A variety of toxicological data sources were consulted to identify TRVs for each selected receptor. TRVs are the estimated dose or exposure level at which no adverse effects are expected to occur. In general, TRVs were obtained from USEPA's Screening Level Protocol or, in the absence of data from this report, from standards, criteria, guidance, or ecological benchmarks from the data sources listed in the Workplan.[25] Consistent with the selected receptor species and groups, available TRVs were compiled for birds, mammals, plants, and aquatic life (surface water and sediment). The TRVs for terrestrial wildlife were based on toxicity studies in which effects on reproduction or survival are measured, since these endpoints are relevant to an assessment of population level effects. For aquatic life, TRVs were based on toxicity studies that examine alterations in growth, reproduction, or survival in individual species, or changes in community structure, abundance, or diversity in benthic species.

As noted in the Workplan, PCDDs/PCDFs were evaluated using a TRV based on 2,3,7,8-TCDD and TEFs for fish and wildlife. These TEFs, which are listed in the Workplan, were applied to express PCDD/PCDF concentrations or doses as 2,3,7,8-TCDD toxic equivalents (TEQs). The TEQs were then summed to calculate the total concentration or dose of 2,3,7,8-TCDD equivalents for each receptor species or group. More information describing the evaluation of PCDD/PCDF mixtures is provided in Appendix M.

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[25] CEPA (2002), AZDEQ (2003), USEPA (1996b, 1999, 2003d, 2005d, 2007b), Sample et al. (1996), Schafer et al. (1983), Schafer and Bowles (1985), EC (2000), Efroymson et al. 1997), Mayer and Ellersieck (1986), NOAA (2006), and MacDonald (2000).

5.2.3   Exposure Assessment

Exposures were calculated for each of the selected receptors in each of the selected habitats described above.   Exposure point concentrations (EPCs) for environmental media (i.e., sediment, surface water, plants and soil) were calculated using the mathematical equations presented in HHRAP, and implemented using the IRAP software. The air dispersion, deposition, and fate and transport modeling conducted to support the human health risk assessment was also used in the ecological risk assessment to calculate the annual average EPC of each chemical in each habitat as a result of stack emissions. The EPCs were evaluated, either in direct comparisons with TRVs for terrestrial plant receptors and aquatic community receptors, or as inputs to food chain calculations for specific mammalian and avian receptors.

Exposures of selected mammalian and avian receptors were expressed as dosages (mg/kg bw) using food chain models conducted according to the methods recommended in the Screening Level Protocol. The food items and environmental media considered in the food chain analysis for each mammalian and avian receptor are shown in Table 5.2-1. Exposure factors for each receptor (e.g., amount and types of food ingested) were then compiled from the published literature for the specific receptors evaluated in this study, as shown in Table 5.2-2. A discussion of the food chain calculation methods is provided in Appendix M.

Chemical concentrations in food items evaluated in the food chain analyses were obtained either from the IRAP software output (i.e., plant and fish tissue concentrations) or calculated from environmental media concentrations using bioaccumulation factors to estimate tissue concentrations in prey items (i.e., invertebrates and small mammals).   The bioaccumulation factors were obtained from values compiled by USEPA in the Screening Level Protocol where available. For compounds not addressed specifically in the Screening Level Protocol, the bioaccumulation factors were derived following the methods outlined in the Screening Level Protocol. One modification to the default bioaccumulation factors in the Screening Level Protocol was made for PCDDs/PCDFs for the Yuma clapper rail. The bioaccumulation factors for invertebrates, the food source for the Yuma clapper rail, that were used in the food chain evaluation for this receptor were developed by USEPA to be generically representative of benthic invertebrates. A detailed assessment of prey of the Yuma Clapper Rail in Arizona and California by the U.S. Fish and Wildlife Service (USFWS 2000) found, however, that the bird's primary prey is crayfish and small fish. Analyses of the stomach contents of 16 Yuma clapper rails collected in the Colorado River area above Laguna Dam[26] found that 94.7% (by volume) of the contents was comprised of crayfish (USFWS 2000). Rather than rely on USEPA's default sediment-to-benthic invertebrate bioconcentration factors (BCFs) for PCDDs/PCDFs, which are based on a 1978 non-specific regression equation (Southworth et al. 1978), recently published literature was reviewed to identify a sediment-to-benthic invertebrate BCF specific to crayfish based on experimental data for the Yuma Clapper Rail food chain analysis (Currie et al. 2000). Appendix M provides additional discussion of the bioaccumulation factors used in the food chain analyses.

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[26] Laguna Dam is located about 13 miles northeast of Yuma, Arizona and about 100 miles south of Parker, Arizona.

5.2.4   Risk Estimation and Description

The potential for ecological risks was quantified using an HQ approach in which exposures were compared to receptor-specific TRVs. An HQ is the ratio of predicted exposure to predicted toxicity. In general, hazard quotients less than 1 indicate that adverse effects from chemical-specific exposures are unlikely, whereas hazard quotients greater than one indicate adverse effects are possible. As directed by USEPA Region 9 (USEPA 2003a) during the Workplan development, this screening-level assessment used an HQ threshold of 0.25, rather than 1.0, to initially characterize the potential for risks.

Potential cumulative toxicity was assessed by summing the HQs for all chemicals, regardless of differences in the mechanism of action of the various compounds, to calculate a hazard index (HI). To be consistent with USEPA Region 9 guidance, the very conservative 0.25 target level was also applied to the HI as an initial step. Most other USEPA regions and states use a target level of 1.0 for evaluating HQ and HI results in ecological risk assessments.

If an HI for all compounds is above 1.0, or above 0.25, this does not mean that adverse ecological effects will occur (for example, because of the safety factors that are incorporated in the TRVs). Rather it indicates that HI values should be recalculated for groups of compounds that act via a similar mechanism of action or the hazard quotient values for those compounds producing an HI above a target level should be examined in more detail. If the HI for compounds with similar mechanisms of action is below 1.0, then adverse health effects are not expected to occur. Even if the HI for compounds with similar mechanisms of action is above 1.0, this does not automatically mean that adverse health effects will occur; rather, this type of result means that there is an increased chance that adverse ecological effects might occur. In this case, further research should be conducted to evaluate the potential for ecological effects.

A summary of all the HI values calculated for receptor species or groups, for all the evaluated habitat areas, is presented in Table 5.2-3. The detailed chemical-specific results are provided in Appendix M. The cumulative HI values were not only below a target of 1.0, but also below the very conservative 0.25 ecological target risk level specified by USEPA Region 9 for this project. Concentrations in surface water and sediment were found to be more than 800 times lower than the 0.25 target hazard index level. Concentrations in plants ranged from just below the 0.25 target level to more than 400 times lower than the 0.25 target level. Exposures to selected bird species were found to be at least five times lower than the 0.25 target level. Finally, exposures to the evaluated mammal species were determined to be at least 5,000 times below the 0.25 target level. These results indicate that adverse ecological effects from exposure to stack emissions are not expected to occur for the evaluated receptors.

Although the results were all below the very conservative 0.25 USEPA Region 9 target level, the data were examined to identify those compounds with the highest HQ results. The highest HQ result was calculated for plants in the creosote bush scrub area, based entirely on one compound which was thallium (HQ=0.18). Thallium was not detected in the PDT and was not detected in any monthly composite spent carbon samples tested from 2003 through

June 2006. It was evaluated in the risk assessment using a stack emission rate derived from its reported detection limit in the PDT. In addition, the TRV for thallium identified in the Screening Level Protocol and used in this analysis incorporates an uncertainty factor of 100. These factors all indicate that the results for thallium are expected to be overestimated. The next highest HQ results were calculated for the double-crested cormorant in the Main Drain exposure area (HQ=0.05) and for the southwestern willow flycatcher in the riparian corridor area (HQ=0.03). These results, while at least five times below a 0.25 target level and 20 times below the more commonly used target level of 1.0, were due to one compound, methyl mercury. As described earlier in the human health risk assessment section of this report, mercury was evaluated in this risk assessment using a permit limit-based emission rate that was about 15 times higher than the measured PDT emission rate. This means that the ecological risk assessment results would be 15 times lower if measured emission rates were used in this analysis.

5.2.5   Discussion of Uncertainties

This section discusses uncertainties associated with the data, calculations, and assumptions specific to the ecological risk assessment. Awareness of important uncertainties involved in the risk assessment is critical to interpreting and understanding the potential risks calculated in this analysis.

5.2.5.1   Selection of Compounds for Detailed Evaluation

Many of the compounds identified for consideration in the ecological risk assessment did not have TRVs available from the data sources consulted (see above), and thus were not quantitatively evaluated. The number of TRVs that were available ranged from about 30 TRVs for birds to roughly 80 TRVs for surface water. This uncertainty could potentially under-estimate ecological risks. On the other hand, the chemicals with TRVs included those compounds generally considered to be of most concern to ecological receptors, such as PCDDs/PCDFs and other compounds with a high bioaccumulation potential, as well as selected inorganic compounds and methyl mercury.

5.2.5.2   Food Chain Models

The food chain model incorporated conservative assumptions in calculating potential exposures which is expected to overestimate potential risks. The screening level risk calculation incorporated the following conservative (i.e., protective) assumptions: a bioavailability from all ingested items of 100%, a body weight based on the low end of the receptor's weight range which results in higher calculated food ingestion rates, an exposure period assumed to occur during the most sensitive receptor life stage, the assumption that each individual species in a community or class-specific guild would be equally exposed, the assumption that 100% of ingested food items and environmental media were potentially contaminated, and the assumption that receptors spend their entire life cycles in the evaluated local habitat areas. The collective impact of these assumptions is expected to be an overestimation of potential exposures and associated risks.

Dietary parameters used in the food chain calculations (e.g., body weight, food intake rate, sediment ingestion rates) were based on literature values. For example, based on the scientific literature, it was assumed that the great horned owl's diet would consist entirely of small mammals, specifically the white-footed mouse. It was also assumed that chemical concentrations modeled in small mammals would be representative of concentrations found in any of the other prey items owls typically consume. It was further assumed that the environmental media concentrations were not high enough to affect viability of the prey populations or viability of vegetation, thus impacting the availability of food. In reality, there will be considerable variability in prey and foraging habits, which could add uncertainty to the ecological risk assessment, and may under- or over-estimate risk.

5.2.5.3   *Exposure Point Concentrations*

The ecological risk calculations relied on maximum annual concentrations associated with stack emissions, thereby conservatively assuming that the each receptor was exposed to the highest annual concentrations over their full life cycle. This assumption may overestimate potential exposures and associated risks.

Plant concentrations were used in the food chain analyses to represent potential concentrations in foods that may be eaten by the herbivores, Gambel's quail and mule deer. The plant concentrations output from the IRAP software based on the USEPA guidance and used in the calculations were for homegrown produce, rather than the specific plant types that may be ingested by these receptors. This may introduce some uncertainty into the exposure point concentrations. For example, differences in plant yields may affect chemical concentrations calculated in plants due to direct deposition, since these concentrations, as calculated by HHRAP methods, are inversely proportional to plant yields.   Thus the lower plant yields characteristic of plants that may be ingested by the quail and mule deer, relative to produce, could possibly result in higher plant concentrations than were used in the food chain analyses. This approach could potentially underestimate food chain exposures and associated risks. The HQ results for Gambel's quail and mule deer, however, were more than 2,000 times below the target level, indicating that this uncertainty will not alter the overall risk assessment results.

Fish tissue concentrations used in the food chain analysis for the cormorant were calculated from the IRAP software for fish at the top of the aquatic food web (i.e., trophic level 4 fish). This approach may overestimate concentrations in fish species ingested by the cormorant since the cormorant will commonly feed on invertebrates and a wide variety of fish from varying trophic levels.

USEPA Region 9 requested that the ecological risk assessment discuss the influence of monsoons on chemical fate and transport. The monsoon season in southern Arizona usually occurs from roughly mid-June through mid-September and is associated with elevated humidity, a reversal of cyclonic wind patterns and severe thunderstorms that are often accompanied by strong winds and short periods of blowing dust.[27] Over the 15-year period

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[27] www.wrh.noaa.gov/psr/general/monsoon/; http://www.public.asu.edu/~aunjs/asuclim_files/azclim.doc; www.ag.arizona.edu/maricopa/garden/html/weather/monsoon.htm;

from 1993-2007, seven thunderstorm and high wind events were recorded by the National Weather Service in Parker and all of these occurred between late June and late August.[28] The chemical fate and transport modeling methods provided by USEPA for combustion source risk assessments, and which were applied in this ecological risk assessment calculate long-term exposure point concentrations to be consistent with the TRVs, and cannot address the short-term impacts associated with brief climate events such as monsoons. This adds uncertainty to the risk assessment results. For example, during a monsoon, stack emissions will be dispersed in the air to a much greater extent than modeled in this study, short periods of intense rainfall could produce higher water flow rates than modeled in this study, and surface soil could become suspended and redistributed during periods of high winds. In general, environmental conditions that enhance mixing such as monsoons are considered more likely to reduce rather than increase potential long-term environmental concentrations due to stack emissions. This uncertainty could only be addressed through very refined site-specific modeling.

5.2.5.4   Toxicity Reference Values

Toxicity reference values for the selected species and communities were based on default values identified by USEPA in the Screening Level Protocol or obtained from standards, criteria, databases or literature noted in the Workplan or recommended by USEPA (1999). In general, TRVs are a major source of uncertainty in an ecological risk assessment. The results of different studies from which TRVs may be obtained often vary by several orders of magnitude, depending on various forms of the chemical, test species, and test endpoints. The sensitivity of receptors in the exposure areas may be different than the sensitivity of species used in tests reported in the literature. Assumptions about the similarity of the chemical speciation between laboratory tests and site conditions must also be made in the absence of speciation analyses. This is a source of uncertainty, since toxicity may vary with the form of the chemical in the environment. Thus, the actual toxicities of chemicals evaluated in this ecological risk assessment could be higher or lower than indicated by the TRVs. On the other hand, many of the TRVs used in this analysis incorporate uncertainty factors which provide an added margin of safety.

5.2.5.5   Dioxin-Like PCBs

The potential impact of emissions of dioxin-like dioxin-like PCBs on the ecological risk results was evaluated using PCB toxic equivalency factors (TEFs) for fish, birds and wildlife developed by the World Health Organization (WHO 1998). The emission rate of each dioxin-like PCB from the PDT was multiplied by the WHO TEFs to calculate the toxic equivalent (TEQ) emission rate for each dioxin-like PCB. These TEQ emission rates were then summed to provide a total TEQ emission rate for all dioxin-like PCBs combined. The resulting total dioxin-like PCB TEQ emission rates using the fish, bird and wildlife TEFs were all determined to be well below the total PCDD/PCDF TEQ emission rate evaluated in the risk assessment, by at least a factor of 35. Since the highest PCDD/PCDF hazard quotient based on the PCDD/PCDF emissions was calculated to be more than 80 times

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[28] http://www4.ncdc.noaa.gov/cgi-win/wwcgi.dll?wwevent~storms

below the conservative 0.25 target level, these findings demonstrate that dioxin-like PCBs would not result in risks to fish, birds or wildlife.

5.2.5.6   *Desert Tortoise*

The desert tortoise receptor was selected for evaluation in the ecological risk assessment, as described in the Workplan, but no TRVs were identified from a search of available toxicity data sources for tortoises or turtles. As a result, potential risks to the desert tortoise are evaluated in this section, by qualitatively discussing factors relevant to the health status of the desert tortoise and the potential for these factors to be adversely affected by SWT facility stack emissions.

As described by the Nevada Fish and Wildlife Service, "Based on more than 40 years of data, we know that tortoises are directly and indirectly impacted by natural as well as human-caused activities. These threats include disease, predation, expanding development, off-highway vehicles, invasion of non-native grasses and weeds, fire, collection, poachers, sheep & cattle grazing, mining, and drought. At this point, there is not one threat that seems to impact tortoises more than another. It is, rather, an accumulation of threats that are taking a toll. Drought, disease, predation, mining, grazing, and off-highway vehicles all impact tortoises." [29]

TRVs are not available for the desert tortoise or any (even remotely related) reptilian species for the compounds considered in this study. Desert tortoises are herbivorous feeding on grasses, herbs, cacti, and some shrubs. Previous research performed by CPF (Chrostowski and Durda 1991) showed that the primary impact of environmental pollution on the desert tortoise was through phytotoxicity that diminished the availability of forage plants. To the extent that this risk assessment shows no impact of stack emissions on plants in general, there is not likely to be an impact on the desert tortoise.

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[29] http://www.fws.gov/nevada/desert_tortoise/dt_threats.html

6.0   QUALITY ASSURANCE PROCEDURES

Risk assessments use data from many different sources in numerous mathematical equations. A multiple-chemical, multiple-pathway combustion source risk assessment, such as this one, generally includes thousands of individual calculations using dozens of input parameters. As a result, a quality assurance (QA) program is an important element in the risk assessment process.

For this project, the QA program included evaluation of input data for accuracy and traceability, documentation of the study process, retention of documents containing data and calculations, and independent QA of calculations by trained scientists who did not conduct the aspects of work they reviewed.

The fate and transport modeling, and exposure and risk assessment calculations for stack and fugitive air emissions, which accounted for the bulk of this study, were performed using the IRAP software. The IRAP software, which was created by Lakes Environmental based on USEPA's HHRAP methodology, relies on quality-assured programmed calculations which incorporate USEPA-specified chemical-specific data and USEPA default input parameters. The program was originally tested and verified in conjunction with USEPA, and the current 2005 version has also been independently verified by Lakes. This software has been widely used in the U.S. (e.g., most USEPA Regions and several states).

Additional QA was conducted for calculations that were independent of the IRAP program (e.g., chemical emission rates, evaluation of wastewater discharge from the facility to the Joint Venture, and QA of inputs entered into the IRAP program).   The QA effort for the air dispersion and deposition modeling included an independent review of the input parameters (e.g., building dimensions, emission source input parameters), selected model options, conversions from English to Metric units, and model output files.

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