FINAL

Remedial Investigation Report
Site-Wide Groundwater Operable Unit (OU-1)
Anaconda Copper Mine Site
Lyon County, Nevada

April 15, 2020

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The draft version of this document (October 20, 2017) was prepared under the supervision of Charles Zimmerman, Brown and Caldwell, with input and direction by EPA, and key technical support by Greg Davis. The team listed above has made significant revisions to the draft version of this document.

TABLE OF CONTENTS

LIST OF FIGURES

LIST OF TABLES

LIST OF APPENDICES

LIST OF ACRONYMS AND ABBREVIATIONS

ACMS	Anaconda Copper Mine Site
AHA	Applied Hydrology Associates
Anaconda	Anaconda Company
ARC	Atlantic Richfield Company
BC	Brown and Caldwell
BCL	Background Concentration Limit
BGQA	Background Groundwater Quality Assessment
BLM	Bureau of Land Management
CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act
CFC	Chlorofluorocarbon
COI	Constituent of Interest
DO	Dissolved Oxygen
DPT	Direct Push Technology
DSR	Data Summary Report
DQO	Data Quality Objective
DWMP	Domestic Well Monitoring Program
EA-IRMS	Elemental Analyzer-Isotopic Ratio Mass Spectrometer
EC	Electrical Conductivity
EPA	U.S. Environmental Protection Agency
ESI	Environmental Standards, Inc.
FEP	Finger Evaporation Ponds
FMS	Fluid Management System
FRIR	Final OU-1 Remedial Investigation Report
FS	Feasibility Study
GC-ECD	Gas Chromatography with Electron Capture Detection
GMP	Groundwater Monitoring Plan
GMR	Groundwater Monitoring Report
GTM	Geothermal Technical Memorandum
HCSM	Hydrogeologic Conceptual Site Model
HDPE	High-Density Polyethylene
HFA	Hydrogeologic Framework Assessment
HLP	Heap Leach Pad
HHRA	Human Health Risk Assessment
HSR	Historical Summary Report
IAOC	Interim Administrative Settlement Agreement and Order on Consent
ICP-MS	Inductively Coupled Plasma Mass Spectroscopy
J	Estimated Concentration
K	Hydraulic Conductivity
K_d	Partition (or Distribution) Coefficient
LEP	Lined Evaporation Pond
LOE	Line of Evidence
MCL	Maximum Contaminant Level
MFR	Mountain-Front Recharge
MIW	Mine-influenced Groundwater
MSA	Multivariate Statistical Analysis
NA	Not Applicable/Not Available
NAC	Nevada Administrative Code
NDEP	Nevada Division of Environmental Protection
NDWR	Nevada Division of Water Resources
NR	Not Recorded
NSA	North Study Area
NS	No Sample
O&M	Operation and Maintenance
ORP	Oxidation-Reduction Potential
OU	Operable Unit
PLS	Pregnant Leach Solution
POD	Point of Diversion
POU	Place of Use
PSTM	Plume Stability Technical Memorandum
PV	Pore Volume
PWS	Pumpback Well System
Q	Quarter
QA/QC	Quality Assurance/Quality Control
QAPP	Quality Assurance Project Plan
RAO	Remedial Action Objective
RER	Replicate Error Ratio
RI	Remedial Investigation
RI/FS	Remedial Investigation and Feasibility Study
RPD	Relative Percent Difference
RSIL	USGS Reston Stable Isotope Laboratory
RSL	Regional Screening Level

SCM	Surface Complexation Model
SERA	Southeast Recharge Area
SOP	Standard Operating Procedure
SOW	Scope of Work
SSPA	S.S. Papadopulos & Associates, Inc.
START	Superfund Technical Assessment and Response Team
STORET	Storage and Retrieval
SWRA	Southwest Recharge Area
TC	Total Carbon
SX/EW	Solvent Extraction/Electrowinning
TIC	Total Inorganic Carbon
TDS	Total Dissolved Solids
TOC	Total Organic Carbon
TU	Tritium Unit
U	Chemical not detected at the indicated sample detection limit
UAO	Unilateral Administrative Order
Ue	Uranium Excess
UEP	Unlined Evaporation Pond
UJ	Chemical not detected at the indicated estimated sample detection limit
USCS	Unified Soil Classification System
USGS	United States Geological Survey
VLT	Vat Leach Tailings
WRCC	Western Regional Climate Center
WRD	Walker River Decree
WRID	Walker River Irrigation District
YPT	Yerington Paiute Tribe

%	percent
‰	per mil
amsl	above mean sea level
bgs	below ground surface
cm^3/kg	cubic centimeters per kilogram
cu ft	cubic feet
cfs	cubic feet per second
°F	degrees Fahrenheit
°C	degrees Celsius or Centigrade
ft	feet
ft/day	feet per day
ft^2/day	square feet per day
ft/yr	feet per year
fmol/kg	femtomole per kilogram
gal	gallons
g/L	grams per liter
gpd	gallons per day
gpm	gallons per minute
L	liter
L/kg	liters per kilogram
µg/L	microgram per liter
µm	micron or micrometer
mg/L	milligram per liter
pCi/g	picocuries per gram
pCi/L	picocuries per liter
pCi	picocurie
pmol/kg	picomoles per kilogram
s.u.	standard units (pH)

MINERALS, CHEMICAL FORMULAS, AND ABBREVIATIONS

$CaSO_4 \cdot 2H_2O$	Gypsum
$Fe_3(OH)_7(s)$ and $Fe_2(OH)_5(s)$	Hydroxy-green rust
$(K_{0.77}Na_{0.03}H_{0.2})Fe_3(SO_4)_2(OH)_6(s)$	Jarosite (ss)
$KFe_3(SO_4)_2(OH)_6(s)$	K-jarosite
$NaFe_3(SO_4)_2(OH)_6(s)$	Na-jarosite
$Fe_8O_8(OH)_{4.8}(SO_4)_{1.6}(s)$	Schwertmannite

(a)	Amorphous
(aq)	Aqueous
(s)	Solid
(ss)	Solid Solution

$\delta^{13}B$	Boron isotopes in water samples
$\delta^{36}Cl$	Chloride isotopes in water samples
$\delta^{34}S_{SO4}$	Sulfur isotopes in dissolved sulfate
$\delta^{34}O_{SO4}$	Oxygen isotopes in dissolved sulfate
$\delta^{15}N_{NO3}$	Nitrogen isotopes in dissolved nitrate
$\delta^{18}O_{NO3}$	Oxygen isotopes in dissolved nitrate

HNO_3	Nitric Acid
H_2SO_4	Sulfuric Acid
SF_6	Sulfur Hexafluoride
SO_2	Sulfur Dioxide

EXECUTIVE SUMMARY

Atlantic Richfield Company (ARC) has prepared this *Final Site-Wide Groundwater Operable Unit (OU-1) Remedial Investigation Report* (FRIR) pursuant to the Interim Administrative Settlement Agreement and Order on Consent (IAOC) between the Nevada Division of Environmental Protection (NDEP) and ARC, dated February 5, 2018 (NDEP, 2018) for the Anaconda Copper Mine Site (ACMS or Site). Consistent with the IAOC requirements, this FRIR 1) summarizes activities conducted to characterize and monitor groundwater in the vicinity of the Site, including on- and off-Site locations; 2) describes the nature and extent of mine-related constituents of interest (COIs) in groundwater; and 3) integrates relevant aspects of other OUs that represent potential sources of chemical loading to groundwater or have the potential to influence groundwater conditions. The FRIR fulfills the study elements and Data Quality Objectives presented in the Revised Groundwater Remedial Investigation (RI) Work Plan (Brown and Caldwell [BC], 2014a), and thereby completes the RI activities for OU-1. Potential human health risks will be addressed comprehensively in a separate OU-1 Human Health Risk Assessment (HHRA) Report[1].

The ACMS is a former copper mine that is located west and northwest of the City of Yerington in Lyon County, Nevada. Historical mining and copper ore beneficiation activities at the ACMS involved the use of sulfuric acid (H_2SO_4) as well as an alkaline flotation circuit and are described in Section 2.0. Mine-related COIs include acidity (i.e., low pH), total dissolved solids (TDS), and major ions including sulfate, metals/metalloids, and uranium (BC, 2014a).

Based on extensive data collection, geochemical analysis, and multiple lines of evidence (LOEs), the nature and extent of mine-influenced groundwater (MIW) in OU-1 has been estimated. MIW in the Shallow Zone has been delineated using two hypotheses resulting in overlapping portions

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[1] Please see Section 8 for discussion of Screening Level Ecological Risk Assessment (SLERA).

of the aquifer where mine-related chemical influences may be present: a less extensive "High Confidence MIW Area" and a somewhat more extensive "Low Confidence MIW Area." A single hypothesis has been used to delineate MIW in the Intermediate through Deep 5 Zones. The information and analyses collected and completed for the RI over many years are sufficient to complete the characterization of groundwater conditions in OU-1, distinguish MIW from water in the Study Area that is chemically influenced by other sources, support risk assessment, and proceed with completion of the Feasibility Study (FS).

Groundwater Investigations and Related Studies

Numerous groundwater investigations, monitoring activities, and related studies conducted by ARC and others provide the geologic, hydrogeologic, and water quality information necessary to complete the OU-1 RI. While substantial information from the mine's operation was also considered and included, this FRIR relies primarily on data obtained after 2005 to characterize groundwater conditions, describe the nature and extent of MIW, and refine the hydrogeologic conceptual site model (HCSM) to support the OU-1 Remedial Investigation and Feasibility Study (RI/FS). Data obtained after 2005 have been selected for these purposes because: 1) data collection was performed pursuant to U.S. Environmental Protection Agency (EPA)-approved quality assurance/quality control (QA/QC) planning documents and OU-1-specific work plans that were developed with input from other stakeholders including the NDEP, Yerington Paiute Tribe (YPT), and the Bureau of Land Management (BLM); 2) the spatial coverage and amount of data increased substantially after 2005; and 3) post-2005 sampling results better represent current conditions at the Site, which is the proper focus of the RI (*see* 40 CFR § 300.430(d)(4)).

The dataset obtained during August 2014 is emphasized in this FRIR for the reasons previously cited and because a variety of data types were concurrently collected and/or evaluated (e.g., groundwater level data, groundwater quality data, and hydrologic tracer data). Consequently, the August 2014 dataset is particularly useful for characterizing spatial aspects of OU-1 groundwater conditions.

Recognizing that groundwater characterization activities would largely involve monitor well installation, a multi-step sequential approach was adopted for field data collection to maximize usable data and optimize the design of a monitor well network intended to serve the data needs for both OU-1 RI characterization and long-term groundwater monitoring. Groundwater RI characterization activities (Section 3.0) resulted in drilling 133 boreholes, logging approximately 33,000 linear feet of core, analyzing groundwater samples collected at up to three depth intervals within the Shallow Zone at 108 locations using direct push technology (DPT) equipment (i.e., a Geoprobe®), analyzing 624 depth-specific (zonal) groundwater samples in all zones of the alluvial aquifer, installing 299 new monitor wells, hydraulic (slug) testing of 296 wells, conducting several long-term pumping tests to estimate hydraulic properties of the alluvial aquifer, development of a groundwater flow model, surface water characterization, hydrologic tracer studies, and bedrock groundwater characterization.

Bedrock Characterization

Bedrock characterization information indicates: 1) a high degree of fracture heterogeneity and vertical hydraulic connection between the bedrock and alluvial groundwater systems and 2) low horizontal hydraulic connectivity and transmissivity of bedrock fractures over horizontal distances that are relevant to the scale of the Site and the surrounding Study Area. In addition to a high degree of three-dimensional (i.e., anisotropic) variability in hydraulic characteristics and hydrologic tracer signatures, the bedrock groundwater system also exhibits three-dimensional spatial variability in chemical concentrations. COI concentrations in bedrock groundwater are 10 to over 1,000 times lower than the values in overlying alluvial groundwater. In addition, areas of elevated mine-related COIs in bedrock groundwater are small in comparison to the alluvial aquifer and found mostly on-Site. Bedrock groundwater flow velocities, average volumetric flow rates and chemical fluxes through the bedrock groundwater system are low, and the bedrock groundwater system is not considered to be an important migration pathway at the Site (EPA, 2015a).

Site Setting

The physical characteristics of the OU-1 study area are described in Section 4.0. The regional hydrogeologic setting is provided in Section 4.8 and the Site and Study Area Local Hydrogeology is described in Section 4.9.

Delineating the Nature and Extent of MIW

Concentrations of COIs measured in alluvial groundwater samples collected in the Study Area are mapped in Section 5.1 to illustrate the nature and extent of contamination. Information presented in other documents, e.g., the Final OU-4a RI Report (Copper Environmental Consulting and Broadbent and Associates, 2019), coupled with the evaluation of Site-wide groundwater chemistry and water level data, document that there are no discernable on-going sources of mine-related COIs currently migrating to groundwater. The distribution of mine-related COIs in groundwater reflect infiltration during historical operations.

Differentiating MIW in the Study Area is complicated by three factors: 1) agricultural practices and naturally-occurring alteration minerals (non-mining sources) contribute some of the same COIs to groundwater in the Study Area, 2) some COIs (e.g., sulfate and uranium) are common to all three sources (MIW, naturally-occurring, and agricultural practices), and 3) commingling of groundwater with three different COI sources. The Background Groundwater Quality Assessment (BGQA, Appendix J-7) used a technical approach that, based on subsequent RI evaluations, identified a large area of groundwater jointly influenced by the three sources but did not subdivide the large area into the smaller area that contains MIW, or delineate other areas primarily influenced by the other two sources.

Development of a LOE Approach to Delineate MIW

Post-BGQA RI evaluations developed a LOE approach to differentiate the MIW component of the BGQA area and identify other subareas of the BGQA that are influenced by the other two sources. The extent of MIW is evaluated using up to six lines of physical, chemical and isotopic evidence (as summarized below and detailed in the identified sections):

- Groundwater flow distances (Section 5.4.1);
- Extent of nitrate from agricultural activities (Section 5.4.2);
- Multivariate statistical analysis (Section 5.4.3);
- Sulfur isotopic information (Section 5.4.4);
- Groundwater age estimates (Section 5.4.5); and
- Collocation of MIW indicators (Section 5.4.6).

Groundwater Flow Distances LOE

The groundwater flow distances LOE uses the physical properties of the alluvium and hydraulic gradients of groundwater to estimate how far groundwater may have migrated from the Unlined Evaporation Pond (UEP) and Lined Evaporation Pond (LEP) since the time of initial releases. Groundwater travel distances are calculated for water in the Shallow Zone through Deep 2 Zones. Deeper zones are not amenable to this approach due to interception of groundwater by large capacity pumping wells.

Groundwater travel distance is calculated by multiplying the average linear groundwater velocity by the time of flow. Groundwater travel distances were calculated over a range of parameter values using a Monte Carlo analysis for the Shallow Zone, as sufficient information was available for this zone to define probability distribution functions for the parameters used in the travel distance calculation. The resulting output from the Monte Carlo simulation provides a description of groundwater travel distance probabilities. For the Intermediate, Deep 1, and Deep 2 Zones, constant parameter values were used for calculating travel distances.

The direction of groundwater flow and hydraulic gradients used in these calculations vary based on the groundwater zone and the time period of calculations. The range of groundwater travel distance estimates are presented in Table 5-2 and Figure 5-15(a-d).

Extent of Nitrate from Agricultural Activities LOE

The extent of nitrate from agricultural activities LOE is based on the fact that nitrate concentrations in groundwater are a tracer of agricultural influences on groundwater quality.

Nitrogen-containing fertilizers are used extensively on the fields north of the Site, a common agricultural practice. Nitrate is a not a mine-related constituent.[2] A large nitrate plume is centered on the agricultural fields north of the Site boundary. Nitrate concentrations exceed 40 milligrams per liter (mg/L) near the center of the irrigated fields, and most shallow wells beneath the fields exceed 10 mg/L.

As discussed in Section 5.4.2, over the past 50 plus years, the estimated amount of irrigation water that has infiltrated to the water table has been more than sufficient to replace all of the water in the Shallow Zone (Appendix M, p. 13). As a result, the current water quality in the Shallow Zone beneath the agricultural fields is related to the quality of the irrigation water applied to the field. It is unlikely that MIW is discernable in such areas where agricultural operations are the dominant influence on current groundwater quality. Although agricultural impacts are evident at lower concentrations, the 5 mg/L nitrate contour is used as an exclusionary criterion to indicate where MIW is not discernable. These areas are indicated on Figures 5-21(b-g).

Multivariate Statistical Analysis LOE

Multivariate statistical analysis (MSA) was used to classify by discriminant analysis all samples from a groundwater monitoring dataset (399 groundwater monitoring samples with 34 analytes per sample) into 18 distinct clusters (Section 5.4.3). Those 18 clusters were then grouped into the three geochemically-distinct primary groundwater types based on interpretation:

- Mine-influenced groundwater,
- Potentially geothermal-influenced groundwater, and
- Regional (includes agricultural/Walker River) groundwater.

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[2] Nitrate measured in on-Site wells and Heap Leach Pads appears to have originated in water supplied by mine water supply wells located in the northern part of ACMS that drew in water from the adjacent agricultural fields.

Greater than 95% of the dataset (380 of 399 samples) was classified with greater than 90% likelihood of proper classification, indicating little uncertainty in source attribution within the dataset.

The water types assigned to each monitor well sample are listed in Table 5-3. This information was used as an independent LOE by mapping the water type for each and then delineating the extent of the water type with a boundary line. Application of this technique for the Shallow Zone is shown in Figure 5-17. The extent of the three water types for all other zones are shown in Figures 5-21(a-g).

Ionic cross plots are included in the MSA LOE and use the distributions of pairs of conservative (non-attenuated) ions to evaluate the presence of dilution and mixing trends of differing water types (Section 5.4.3). The MSA LOE evaluates all the analytes, and the ionic cross plots provide visual clarification of the relationships of some specific analytes, or subsets of the MSA LOE. Mixtures of two end members plot as a linear trend between the end members on the bivariate graphs. If mixing of additional end members with differing chemistries are present, they plot as separate linear trends.

Chloride is used as the abscissa on the ionic cross plots due to the conservative nature of the ion. Ions plotted on the ordinate are dominant mine COIs (sulfate and uranium) as well as a group of COIs with bimodal distributions similar to those of sulfate and uranium (boron, fluoride, lithium and vanadium). The graphs are shown on Figure 5-26a.

The data points are coded by the three water types discussed in Section 5.4.3. The focus of these graphs is to understand dilution trends of the MIW and geothermal ("GEO") water types, and evidence of mixing between these two end members. Dilution trends of the two end members with low ionic strength "background" end members are displayed on each graph.

The MIW and GEO end members plot in different areas of each graph and have different dilution trends. This array of data is caused by the higher chloride content of the GEO end members relative to the MIW dilution trends and "background" end members. Evapoconcentration processes likely also influence the GEO trends (Section 5.4.12). No data are observed in a linear trend directly between the MIW and GEO end members.

Each graph also identifies within a dotted circle the samples that plot along a dilution trend that is not consistent with the water type identified for the indicated sample. For example, there are blue MIW data points that plot along the low end of the GEO dilution trend on the chloride/fluoride and chloride/vanadium cross plots (Figure 5-26a). There are red GEO data points that plot along the upper end of the MIW dilution trend on the chloride/boron cross plot. Additionally, there are MIW and GEO data points that plot between the low ends of the MIW and GEO dilution trends on the chloride/sulfate, chloride/uranium, and chloride/lithium cross plots.

These samples are all from wells located in the LEP and most are in the vicinity of the former Pumpback Well System (PWS; Table 5-4). The hydraulic influences of the PWS when operational may have induced flow of both MIW from the south and the geothermal water type into the PWS wells. Alternatively, these areas may represent an area of mixing of the two water types. In either case, commingled samples of both diluted MIW and geothermal water are found in this area of the former PWS alignment.

Review of molybdenum versus pH data as part of the ionic cross plot evaluation process provided additional insight to the evaluation of MIW and non-MIW constituent migration. Evaluation of the molybdenum and pH data in the Shallow Zone, process chemistry, groundwater chemistry beneath the Evaporation ponds, and published reference on molybdenum water treatment and attenuation processes support the conclusion that elevated concentrations of molybdenum north of ACMS did not originate from the ACMS. Molybdenum is associated with the naturally-occurring geothermal process. The area containing elevated molybdenum is similar to the distribution of other COIs to the north of the Site.

In summary, the ionic cross plot evaluation indicates that wells B/W-32S and B/W-77S do not contain discernable amounts of MIW, further supporting the MSA LOE. The ionic cross plot evaluation indicates that some well samples previously assigned as MIW or geothermal water types may be mixtures of the two water types.

Sulfur Isotopic Information LOE

The sulfur isotope LOE is based on the fact that early copper extraction operations at Yerington (during the 1950s and 1960s) relied on sulfur ores from the Leviathan Mine in California (BC, 2014a), which were isotopically depleted and had a distinct sulfur isotopic signature of -17.6‰ $\delta^{34}S_{SO4}$ (Taylor and Wheeler, 1994). Sulfur sources with different (higher) isotopic signatures were used during later copper extraction operations in the 1970s, 1980s, and 1990s.

The ACMS sulfide ore has a mean $\delta^{34}S_{SO4}$ of -5.45‰, based on the average of five samples reported in the literature (Field, 1966) which ranged from -4.9 to -6.3‰. Lipske (2003) reported a $\delta^{34}S_{SO4}$ value of -4.7‰ for ACMS sulfide ore. Thus, $\delta^{34}S_{SO4}$ values more depleted than ~ -5‰ represent MIW containing sulfur from the Leviathan Mine.

The -5‰ $\delta^{34}S_{SO4}$ contour line is used as the sulfur isotope LOE to delineate the extent of early releases of MIW as shown in Figures 5-19(a-g). As discussed above, later sources of sulfur used at ACMS were less depleted than -5‰ $\delta^{34}S_{SO4}$. Therefore, MIW from later releases could occur in areas beyond the areas shown these figures.

Groundwater Age Estimates LOE

The age of groundwater is the time since the water was recharged to the saturated zone. There are techniques to determine whether groundwater was recharged before Anaconda Company (Anaconda) operations, and hence could not contain MIW. The age of groundwater can be estimated by various methods. A technique based on a combination of tritium and helium was used at the Site to assess where recharge occurred prior to Anaconda operations (Section 5.4.5).

The tritium/helium ($^{3}H/^{3}He$) groundwater age dating method involves measuring both ^{3}H and ^{3}He in the water sample and accounting for the tritium-derived ^{3}He, which is produced at a known rate. The concentration of ^{3}He is set upon exposure to atmospheric gas and "the $^{3}H/^{3}He$ clock does not start until water is below the water table and completely isolated from a gas phase" (Solomon and Cook, 2000, p. 414). The measurement of helium (a decay product of tritium) is useful for estimating the high tritium concentrations that occurred in water after 1952 but is not useful for better determining low concentrations in water recharged prior to 1952. Therefore the tritium only method is used to identify groundwater recharged before 1952.

The $^3H/^3He$ data used to map the extent of MIW were collected in 2012 and 2014. A groundwater age estimate of greater than 60 years derived from the tritium only method indicates that the water was recharged prior to 1954 (conservatively assuming all data were collected in 2014), which essentially precedes Site operations. Copper ore was first delivered to the vat leach plant in November 1953 (CH2M Hill, 2010). This approach is used to delineate the extent of the groundwater age estimates LOE on Figures 5-21(a-g).

Collocation of MIW Indicators LOE

The collocation of MIW indicators LOE is based on the fact that sulfate and uranium are key indicator parameters for MIW (Section 5.4.6). Sulfate and uranium are generally more mobile in Site groundwater than other indicators of MIW that can be less mobile due to changes in pH or rock-water interaction. Although both sulfate and uranium are present due to natural and anthropogenic processes in the Mason Valley, elevated concentrations of both constituents are not always collocated. Elevated concentrations of sulfate and uranium are collocated in some areas of ACMS. The empirical selection of collocated areas where sulfate is greater than 1,000 mg/L and uranium is greater than 100 micrograms per liter (μg/L) is used as a LOE to estimate the extent of MIW in the Shallow Zone.

The distributions of sulfate and uranium in the Shallow Zone are shown in Figures 5-2a and Figure 5-3a. Both contour maps indicate an area of elevated concentrations in the vicinity of the Evaporation Ponds that extends a short distance off-Site to the north/northwest in the vicinity of wells B/W-32S and B/W-77S. The similarity of spatial patterns of other non-pH dependent COIs to the distributions of sulfate and uranium provide support to the collocated sulfate/uranium LOE to map the extent of MIW.

Sulfate and uranium concentrations measured in both the DPT program (Appendix B) and in groundwater monitor wells (3Q 2014) are illustrated on the NW/SE cross section that tracks along the axis of the collocated sulfate/uranium model of MIW migration (Figures 5-25a and 5-25b, respectively). Concentrations are elevated on the southeast end of the cross section beneath ACMS and are lowest on the northwest end of the cross section. As shown on the contour maps of monitor well data and DPT data, a zone with concentrations lower than observed at either the ACMS or wells B/W-32S and B/W-77S is observed between the two areas. Similar patterns are observed in the cross sections illustrating the distributions of lithium and vanadium (Figures 5-25f and 5-25g, respectively).

The distributions of COIs can be interpreted to support a model of MIW migration, and this LOE is summarized in Figure 5-21h as the area in the Shallow Zone where 3Q 2014 groundwater monitoring data indicate sulfate concentrations are greater than 1,000 mg/L and uranium concentrations are greater than 100 µg/L.

Two Hypotheses of Extent of MIW in the Shallow Zone

Hypotheses of the extent of MIW in the Shallow Zone are assembled using mutually supporting LOEs. Review of the LOE information indicates that there are two hypotheses that can be supported by different sets of mutually consistent evidence. The first hypothesis integrates five mutually supportive lines of evidence into a composite evaluation of MIW using MSA (including ionic cross plots), sulfur isotopes, the 50% probability of groundwater travel distance, age dating, nitrate, and excluding the collocation of MIW indicators LOE. The first hypothesis is termed the High Confidence MIW Area Shallow Zone. There is a high probability it contains MIW from ACMS. The second hypothesis uses two mutually supportive lines of evidence to assemble a composite evaluation of MIW (using the conservative 5% probability of groundwater travel distance LOE and the collocation of MIW indicators LOE). The second hypothesis is termed the Low Confidence

MIW Area Shallow Zone. There is a lower probability that it contains MIW and a higher probability that it is influenced by other sources. There is only one hypothesis for the Intermediate through Deep 5 Zones based on the first five lines of evidence listed above[3]. The extent defined by this hypothesis in each zone is termed MIW Area. Groundwater that contains discernable mixtures of MIW are mapped as MIW.

The extents of MIW in the Shallow Zone defined by the two hypotheses are shown in Figures 5-21(a-h). There is little conflict among the five LOEs used to map the High Confidence MIW Area Shallow Zone. The collocation of MIW indicators LOE does not support the extent of the High Confidence MIW Area in the Shallow Zone.

The LOE used to estimate the Low Confidence MIW Area Shallow Zone are mutually supportive. The groundwater travel distance estimate provides a reasonable upper limit (maximum extent) of the possible range of groundwater travel distances estimated by a Monte Carlo probability. The area of 5% probability groundwater travel distance area extends slightly farther than the collocation of MIW indicators LOE. Concentrations of sulfate and uranium generally decrease from the area beneath the ACMS to the area north of wells B/W-32S and B/W-77S. The non-pH influenced COIs have distributions similar to the sulfate and uranium distributions, and thus appear to support the estimate of the Low Confidence MIW Area Shallow Zone.

The Low Confidence MIW Area is not supported by the MSA (including ionic cross plots), sulfur isotope, and age dating LOEs, all of which support a different extent of MIW. The Low Confidence MIW Area is not inconsistent with the sulfur isotope LOE.

The areal extent of MIW is greatest in the Shallow Zone (using either hypothesis) and decreases with depth. MIW is mapped beneath the UEP in all zones. MIW is also mapped beneath the LEP

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[3] Groundwater travel distances are calculated using constant parameter values in the Intermediate through Deep 5 Zones.

in the Shallow (both High and Low Confidence MIW Areas) through Deep 2 Zones, but less so in the deeper zones, in part due to the alluvial aquifer pinching out the west with depth below the LEP. The location of the MIW beneath the UEP and LEP is consistent with the contour maps indicating highest concentrations of sulfate, uranium, and other COIs beneath the UEP and LEP. On-site MIW is also mapped to the east and west of the Evaporation Ponds. On-site COI concentrations in MIW decrease with distance from the UEP and LEP.

The MIW occurs off-Site in the following three areas contiguous to ACMS:

1. A triangular area in the Shallow Zone on the western border of ACMS (High Confidence MIW Area Shallow Zone);
2. An area north of the LEP in the Shallow Zone (this area extends further north in the Low Confidence MIW Area [Figure 5-21h] encompassing the area that includes wells B/W-32S and B/W-77S, sometimes referred to as the "Hot Spot," than in the High Confidence MIW Area [Figure 5-21a]); and
3. An area to the northeast of the intersection of the UEP and LEP that extends progressively further from the Site with depth. In the Deep 3, Deep 4 and Deep 5 Zones, MIW extends beyond the B/W-1 well cluster location.

The water types in the remainder of the Study Area reflect geothermal and regional (agricultural/Walker River) influences as illustrated in Appendix N Figures 8-5(a-h). The geothermal signature is present in groundwater throughout the bedrock and near-bedrock alluvium. The geothermal signature becomes less prominent within the alluvial aquifer towards the surface, although it is present along the western portion of the Study Area. Regional (agricultural/Walker River) groundwater is present within the near-surface alluvium along the eastern portion of the Study Area where irrigation ditches and agriculture fields are prevalent.

Fate and Transport

As required by the IAOC, a Plume Stability Technical Memorandum (PSTM) was finalized and is included in Appendix M and summarized in Section 6. The results of the PSTM evaluations indicate that the areal extent of MIW plume of COIs due to Site operations is changing very slowly. Evaluations of the center-of-mass of sulfate and uranium indicated no movement over the past six years in the centers-of-mass. Under current conditions, groundwater flow in the Shallow, Intermediate, and Deep 1 Zones at the northern Site boundary, with few exceptions, is from the fields north of the Site toward the Site and not from the Site towards the fields, which limits potential migration in these zones. Migration may occur in the deeper aquifer zones in the future, but rate of expansion of the plume of MIW in these zones will be slow. The MIW plume stability is the result of slow groundwater velocities and attenuation by dilution.

Hydrogeologic Conceptual Site Model

The updated Hydrogeologic Conceptual Site Model (HCSM) is presented in Section 7 and includes a diagram of Site operations and groundwater conditions through time. The alluvial aquifer is composed of interbedded sequence of alluvial sediments with clay strata. An approximately 10-foot thick clay layer at a depth of approximately 50 feet ("50-ft Clay"), at an elevation corresponding to the Shallow and Intermediate Zone boundary (4,300 ft above mean sea level [amsl]), has been documented in some boreholes beneath the LEP and to the north and east of the LEP. This clay layer limits vertical groundwater flow from the Shallow to the Intermediate Zones in this area. Clay lithology sections (Figures 4-7[c-g]) show that the 50-ft clay is not continuous beyond this area. As a result, this layer is not a laterally extensive confining unit across the Study Area.

Historical changes in groundwater flow patterns at the Site due to mining include 1) construction and subsequent water infilling of the mine pit, 2) groundwater pumping from mine water supply wells, and 3) to a lesser extent, operation of the PWS. Excavation of the mine pit and associated mine pit dewatering during the Anaconda operational period reversed the general northward groundwater flow in the vicinity of the pit (Figure 7-1). The pit continues to be a groundwater sink maintained by evaporative losses and is projected to have a steady-state long term level that is more than 100 feet lower than pre-mining groundwater levels. A groundwater divide currently exists in the Shallow Zone in the area east of Weed Heights. South of the divide, flow is to the south towards the mine pit. Groundwater inflow to the pit will continue in the future due to continued evaporative losses from the Pit Lake.

Significant amounts of water were pumped by the mine water supply wells, which included wells used for pit dewatering and wells in the northern part of the Site constructed solely for water-supply purposes. The latter wells were screened across much of the alluvial aquifer. Based on historical potentiometric maps and response of the alluvial aquifer to long-term RI pumping tests, the cones of depression created during pumping were likely large enough to create reversals of groundwater flow directions towards the pumping locations when operational. Anaconda pumping of mine water supply wells ceased in 1978.

Infiltration of mine COIs to groundwater and mounding of the water table beneath the Evaporation Ponds is illustrated in Figure 7-1. Some documentation of hydraulic gradients and flow directions is available in the later years of mine operation until the present, but detailed documentation is not available until the RI activities began. During historical operations, OU-4a was a major source of COIs to groundwater (primarily from the UEP). OU-4a is not a discernable on-going source of COIs to groundwater, based on multiple lines of evidence (Copper Environmental Consulting and Broadbent and Associates, 2019). The effectiveness of the PWS, which operated from 1986 to 2009, in capturing groundwater water and influencing flow directions is discussed in Section 3.3.1 and in Appendix J-1.

Conclusions

The FRIR fulfils the study elements and Data Quality Objectives presented in the Revised Groundwater RI Work Plan (Brown and Caldwell [BC], 2014a), and thereby completes the RI activities for OU-1. Potential human health risks will be addressed comprehensively in the OU-1 HHRA Report[4].

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[4] Please see Section 8.0 for discussion of the SLERA.

There are inherent uncertainties and limitations associated with each LOE presented in support of both hypotheses on MIW extent.

The RI presents two hypotheses of the extent of MIW in the Shallow Zone: 1) the High Confidence MIW Area Shallow Zone and 2) the Low Confidence MIW Area Shallow Zone. There is only one hypothesis each for the extent of MIW in the Intermediate through Deep 5 Zones based on five LOEs (MSA, sulfur isotopes, groundwater travel distance, age dating, and nitrate). The extent defined by this hypothesis in each of the Zones is termed MIW Area.

The Low Confidence MIW Area Shallow Zone and the MIW Area in the deeper zones will be used primarily in the risk assessment with additional context provided by the COIs in the High Confidence MIW Area Shallow Zone as discussed in the HHRA Work Plan. All three MIW areas will be considered in the implementation and evaluation of future groundwater monitoring programs. The specific uses of both the High and Low Confidence MIW Areas in the Shallow Zone and the MIW Areas in the deeper zones will be described in the approved HHRA Work Plan and Feasibility Study.

1 INTRODUCTION

Atlantic Richfield Company (ARC) has prepared this *Final Site-Wide Groundwater Operable Unit (OU-1) Remedial Investigation Report* (FRIR) pursuant to the Interim Administrative Settlement Agreement and Order on Consent (IAOC) between the Nevada Division of Environmental Protection (NDEP) and ARC, dated February 5, 2018 (NDEP, 2018) for the Anaconda Copper Mine Site (ACMS or Site). The remedial investigation for OU-1 meets the standard of a Comprehensive Environmental Response, Compensation and Liability Act (CERCLA) Equivalent Remedial Investigation as stipulated in the National Priorities List Deferral Agreement (Deferral Agreement) between the U.S. Environmental Protection Agency (EPA) and NDEP (EPA and NDEP, 2018). This document was prepared consistent with CERCLA guidance regarding the structure and content of a Remedial Investigation (RI) report.

ARC prepared the draft OU-1 RI Report on October 20, 2017 pursuant to Section 7.0 of the Scope of Work (SOW) attached to the Administrative Order for Remedial Investigation and Feasibility Study (RI/FS), CERCLA Docket No. 9-2007-0005 (2007 Order) for the ACMS. The 2007 Order (EPA, 2007a) was issued to ARC by the EPA on January 12, 2007. A revised draft OU-1 Report was submitted to NDEP pursuant to the IAOC on October 4, 2019. This FRIR amends and updates the revised draft OU-1 RI Report in accordance with NDEP comments and direction based on the revised draft OU-1 RI Report.

The term "Site" refers to the area where copper mining activities historically occurred. The Site is located west and northwest of the City of Yerington in Lyon County, Nevada (Figure 1-1). Figure 1-2 depicts the Study Area boundary for OU-1 and the boundaries for the seven other OUs at the Site that were identified in the 2007 Order. The eight OUs identified in the 2007 Order include:

- Site-Wide Groundwater (OU-1);
- Pit Lake (OU-2);
- Process Area (OU-3);
- Evaporation Ponds and Sulfide Tailings (OU-4);

- Waste Rock Areas (OU-5);
- Oxide Tailings (OU-6);
- Wabuska Drain (OU-7); and
- Arimetco Facilities (OU-8).

Subsequently, the EPA approved a subdivision of OU-4 into OU-4a (Evaporation Ponds) and OU-4b (Sulfide Tailings), as well as the transfer of the southern portion of the Calcine Ditch from OU-3 to OU-4a. The EPA-approved OU-4 subdivision and transfer of a portion of the Calcine Ditch to OU-4a was based on a recognition of: 1) the different types of mine-waste materials in the Evaporation Ponds and Sulfide Tailings; 2) the similarity of mine-waste materials in the Calcine Ditch and portions of the Evaporation Ponds; and 3) the differences in the presence and magnitude of constituents of interest (COIs) in groundwater underlying the Evaporation Ponds and Sulfide Tailings. The FRIR retains the OU-based framework in the 2007 Order (updated to include the EPA-approved restructuring). The IAOC (and attached Statements of Work) has grouped the OUs into various Closure Management Units for the remainder of Feasibility Study and Remedial Design/Remedial Action activities.

The term “Study Area” refers to on-Site and off-Site areas in which OU-1 RI-related investigations have been conducted. The Study Area boundary is based on the OU-1 hydrogeologic conceptual site model (HCSM) that was described in the EPA-approved *Site-Wide Groundwater Operable Unit (OU-1) Remedial Investigation Work Plan - Revision 1* (Revised Groundwater RI Work Plan; Brown and Caldwell [BC], 2014a).

1.1 Purpose of Report

Consistent with the IAOC, this FRIR 1) summarizes activities conducted to characterize and monitor groundwater in the vicinity of the Site, including on- and off-Site locations; 2) describes the nature and extent of mine-related COIs in groundwater; and 3) integrates relevant aspects of other OUs that represent potential sources of chemical loading to groundwater or have the potential to influence groundwater conditions. Additionally, this FRIR addresses the fifteen study elements specified in Section 7.0 of the original SOW (EPA, 2007a), which served as the principal bases for RI planning, data collection, and analysis.

Given the complexity of Site-wide groundwater conditions, several phases of investigations have been approved by EPA and conducted by ARC since 2005. During this time period, ARC, NDEP, EPA, and other stakeholders, including the Yerington Paiute Tribe (YPT), and Bureau of Land Management (BLM), periodically held groundwater technical meetings to discuss field data collection activities, technical findings, and remaining data gaps relative to: 1) the 15 study elements specified in Section 7.0 of the SOW attached to the 2007 Order; and 2) the Data Quality Objectives (DQOs) established in the draft and final versions of the remedial investigation work plans for OU-1 (BC and Integral Consulting, Inc., 2007 and BC, 2014a, respectively).

The 15 study elements discussed above are:

> 1) *Specific hydrogeologic characteristics of the bedrock and alluvial groundwater flow systems in the study area including regional and local topography, geology, stratigraphy, structural geology, and depositional history;*
>
> 2) *A description of anthropogenic influences that may affect the hydrogeology in the vicinity of the Site including agricultural irrigation areas, drains, ditches, conveyance altered channels, seals or compacted fill;*
>
> 3) *Identification and characterization of areas and amounts of recharge and discharge, regional and local groundwater flow patterns, and characterization of seasonal variations in the groundwater flow regime;*
>
> 4) *Collection of general meteorological data including, as applicable, daily precipitation and temperature records, annual and monthly precipitation averages, monthly temperature averages, wind speed and direction, evaporation rates, and climatic extremes (including frequency of occurrence);*

5) *An analysis of topographic features that might influence the groundwater flow system including watershed characteristics;*

6) *Assessment of hydraulic relationships between the groundwater and surface water flows, including the Walker River, and a water balance/budget and estimate of steady state hydrologic conditions;*

7) *Installation of exploratory boreholes, groundwater elevation measurements, water quality sampling, and installation and calibration of monitoring equipment;*

8) *Based on field data and lithologic cores, classification and description of the saturated and unsaturated hydrogeologic units that may be part of any migration pathways including, as appropriate, hydraulic conductivity, porosity, effective porosity, pore water, and Darcy velocity; lithology, grain size, sorting, degree of cementation, and an interpretation of the degree of any interconnections between saturated zones;*

9) *Based on field data and lithologic cores, assess structural geologic elements and construct hydrogeologic cross sections and fence diagrams showing any extent (depth, thickness, lateral extent) of hydrogeologic units that may be part of any migration pathways;*

10) *Identify, as appropriate, sand and gravel layers in unconsolidated deposits, zones of fracturing or channeling in consolidated or unconsolidated deposits, zones of higher permeability or lower permeability that might direct and restrict the flow of contaminants, geologic formations or group of formations that are capable of yielding a significant amount of groundwater to wells and springs, and water bearing zones that may serve as a pathway for contaminant migration including perched zones of saturation;*

11) *Based on data obtained from groundwater monitor wells and/or piezometers installed upgradient and downgradient from the potential contaminant sources, define groundwater elevations using water level contours and/or potentiometric maps (legibly displayed on appropriate scale maps), hydrologic cross sections showing vertical gradients, flow systems*

including the vertical and horizontal components of groundwater flow, and seasonal or temporal changes in groundwater elevations or hydraulic gradients;

12) *Definition of aquifer properties (e.g., hydraulic conductivity, transmissivity, and storativity) by a program of aquifer testing to measure the hydraulic connection between areas and throughout the known extent of contamination;*

13) *Definition of contaminant pathways from the Anaconda and Arimetco facilities to all aquifers including boundary conditions, character of water-bearing units, presence of absence of impermeable units or confining layers;*

14) *Groundwater quality data including pH, total dissolved solids (TDS), salinity, specific conductance, concentrations of selected metals and radiochemicals, and a comparison of such data to background concentrations; and*

15) *As related to other Site operable units, collect groundwater quality data at to-be-determined locations underlying the tailings, Evaporation Ponds, portions of the Process Areas, heap leach pads, and other potential sources of contamination.*

Table 1-1 relates the 15 study elements specified in the 2007 SOW to the DQOs presented in the Revised Groundwater RI Work Plan (BC, 2014a).

Table 1-1. Comparison of Study Elements Specified in the SOW to the 2007 Order to DQOs Presented in the Revised Groundwater RI Work Plan

DQO	DQO Title	SOW Study Element														
		1	2	3	4	5	6	7	8	9	10	11	12	13	14	15
1	Discriminate Background and Mine-Influenced Groundwater		X									X			X	
2	Identify Potential Chemical Loading Sources to Groundwater		X									X		X		X
3	Determine Geochemical Mobilization/Attenuation Processes											X				
4	Characterize Chemical Distribution and Migration Pathways	X						X	X	X	X	X			X	X
5	Determine Aquifer Properties								X				X			
6	Determine Groundwater Flow and Chemical Transport Rates	X	X		X	X	X	X		X	X	X				
7	Assess Anthropogenic Influences on Groundwater and Surface Water/Groundwater Interactions	X	X	X			X									
8	Determine Pumpback Well System Efficiency	X														
9	Assess Human Health and Ecological Effects							X	X	X	X	X		X	X	

Numerous investigations and monitoring activities conducted by ARC and others provide substantial geologic, hydrogeologic, and water quality information relevant to OU-1. The results of each phase of work were used to identify data gaps and work plans for subsequent phases of work. Therefore, the HCSM was progressively modified during the RI to reflect the increased understanding of site conditions. The HCSM (Section 7.0) represents the current culmination of understanding developed during the entire RI. The analytical results used to define the nature and extent of contamination and fate and transport of COIs relies primarily on data obtained after 2005 and up through 2019 to address the study elements specified in Section 7.0 of the SOW, characterize groundwater conditions, and refine the HCSM to support the OU-1 RI/FS. Data obtained after 2005 are used primarily for these purposes because: 1) data collection was performed pursuant to EPA-approved quality assurance/quality control (QA/QC) planning documents and OU-1 specific work plans that were developed with input from other stakeholders including the YPT, NDEP, and BLM; 2) the spatial coverage and amount of data increased substantially after 2005; and 3) post-2005 sampling results represent current conditions at the Site, which is the proper focus of the RI.

The August 2014 dataset is emphasized in this FRIR because a variety of data types were concurrently collected (e.g., groundwater level data, groundwater quality data, and hydrologic tracer data) and/or evaluated (e.g., geochemical speciation modeling to help assess COI transport). Consequently, the August 2014 dataset is particularly informative for characterizing spatial aspects of OU-1 groundwater conditions. Temporal aspects of OU-1 groundwater conditions are based on post-2005 data, which indicate that the August 2014 dataset is generally representative of post-2005 groundwater conditions.

Long-term monitoring of Site-wide groundwater conditions is conducted pursuant to the *Site-Wide Groundwater Monitoring Plan - Revision 2* (GMP; BC, 2012a), which was prepared by ARC pursuant to Section 6.0 of the 2007 SOW. The development of the monitoring program and a description of the monitor well network, including modifications implemented under the IAOC, are provided in this FRIR.

Potential human health risks will be addressed comprehensively in a separate OU-1 Human Health Risk Assessment (HHRA) report. The FRIR provides the basis for ARC to identify remedial action objectives (RAOs) and screen/evaluate remedial alternatives for OU-1, which will occur during the feasibility study (FS).

1.2 Site and Study Area Description

The Site and Study Area are located in the Mason Valley in Lyon County, Nevada. The Site boundary includes portions of Township 13 North, Range 25 East, Sections 4, 5, 8, 9, 16, 17, 20, and 21 (Mount Diablo Baseline and Meridian) on the Mason Valley and Yerington United States Geological Survey (USGS) 7.5-minute quadrangles. The IAOC defines the Site as encompassing approximately 3,468 acres (5.4 square miles) of land used for copper mining and processing activities, although other reports describe the Site as ranging from approximately 3,100 to 3,600 acres in size . Including the Site, the Study Area covers approximately 19,300 acres (30.2 square miles).

The Mason Valley Basin (Basin no. 108, as defined by the Nevada Division of Water Resources [NDWR]) is located within the larger Walker River Hydrographic Basin (no. 9). Mason Valley covers about 510 square miles, and the valley floor occurs at an elevation between 4,300 and 4,700 feet above mean sea level (amsl). The principal agricultural activities in the valley include farming (hay, grain, and onions) and cattle ranching (Lopes and Allander, 2009a, 2009b; Carroll et al., 2010). Irrigation water is provided by surface water diversions from the Walker River and from pumped groundwater. The Walker River flows northerly and northeasterly between the Site and the City of Yerington. The river is within a quarter mile of the southern portion of the Site (Figure 1-1).

1.3 Groundwater Zone Designations

Groundwater zone designations based on elevation are used in this FRIR to identify and group monitor wells with similar screen interval elevations in the alluvial aquifer, consistent with previous groundwater-related reports, as follows:

- Shallow (S): screen intervals that straddle the water table, or are within 50 feet of the water table when a shallower well does not exist, typically >4,300 feet amsl
- Intermediate (I): 4,250 to 4,300 feet amsl

- Deep (D): <4,250 feet amsl; given the thickness of alluvium, the Deep Zone is further subdivided as follows:
    - Deep 1 (D1): 4,200 to 4,250 feet amsl
    - Deep 2 (D2): 4,120 to 4,200 feet amsl
    - Deep 3 (D3): 4,000 to 4,120 feet amsl
    - Deep 4 (D4): 3,900 to 4,000 feet amsl
    - Deep 5 (D5): <3,900 feet amsl

Monitor wells with screen intervals in bedrock, regardless of elevation, are designated as bedrock ("B") wells. The groundwater zone designation is included as a suffix to the monitor well identification number (e.g., the "S" suffix in monitor well identification B/W-1S indicates that the screen for this well is positioned in the Shallow Zone).

1.4 Report Organization

Section 2.0 of this FRIR summarizes the Site operation history. Section 3.0 details the investigations related to OU-1 and relevant Site-wide studies and evaluations. Section 4.0 describes the physical characteristics of the Study Area. Section 5.0 describes the nature and extent of contamination, which is based on an evaluation procedure to distinguish mine-influenced groundwater (MIW) from other water types that contain some of the same COIs (agriculturally-influenced water and geothermally-influenced water). Section 5.0 also discusses the primary sources of releases of mine-related COIs to groundwater. Section 6.0 describes the fate and transport of contaminants in Study Area groundwater and includes the presentation of temporal trends in COI concentrations in wells containing MIW. Section 7.0 summarizes the updated HCSM for OU-1. Section 8.0 introduces the risk evaluation process. Report conclusions are presented in Section 9.0. Section 10.0 lists the references cited in this FRIR. The organization of this report is consistent with EPA guidance (EPA, 1988) and recommendations provided during groundwater technical meetings in 2015 and 2016, and in subsequent discussions with NDEP.

The FRIR contains the following appendices that provide detailed backup information for the summary information presented in the main text. Appendix A provides historical mining-related information including the *Final Historical Summary Report* (HSR; CH2M Hill, 2010) and historical Anaconda water supply and use information. Appendix B provides the *Shallow Zone Data Summary Report - Revision 1* (BC, 2010a), which refined the distributions of select COIs in Shallow Zone groundwater north of the Site and helped guide subsequent monitor well installation efforts. Appendix C presents information on the groundwater monitor wells including lithologic logs, well construction information, and depth-specific (zonal) groundwater quality data obtained during borehole drilling and well installation.

Appendix D presents water level and groundwater quality information. Appendix E provides soil sampling data. Appendix F presents hydraulic conductivity information and analyses. Appendix G presents regional and local surface water data. Appendix H presents hydrologic tracer data and supplemental information. Appendix I presents the *Bedrock Groundwater Assessment Technical Memorandum* (BC, 2015a). Appendix J provides various groundwater studies, evaluations, and reports that were conducted to fulfill certain study elements specified in Section 7.0 of the SOW attached to the 2007 Order (EPA, 2007a) and have been previously submitted to the EPA. These include an evaluation of the effectiveness of the Pumpback Well System (PWS), a Pit Lake water balance, public information pertaining to agricultural water use, the groundwater flow model report, the *Geochemical Characterization Data Summary Report - Revision 2* (BC, 2016a), Site-specific chemical distribution coefficients, and the *Background Groundwater Quality Assessment - Revision 3* (BC, 2016b). Appendix K presents groundwater chemical distribution maps of select COIs, including pH, sulfate, dissolved uranium, alkalinity, nitrate, and dissolved arsenic. Appendix L, which was included in the Revised Draft RIR, has been deleted from this FRIR. Appendix L contained groundwater temperature maps that were a precursor to and superseded by the more extensive geothermal system evaluation presented in Appendix N.

Appendix M is the Final Plume Stability Technical Memorandum (PSTM) and Appendix N is the Final Geothermal Technical Memorandum (GTM). Appendix O contains responses to NDEP comments on the revised draft RIR.

2 SITE HISTORY

This section on Site History contains a summary of key historical mining practices, releases, and features relevant to the historical and/or current aspects of the HCSM for OU-1. Historical mining-related features are shown on Figure 1-2. Historical ore processing facilities located within the Process Areas (OU-3) are shown on Figure 2-1. This summary of the operational history of the Site paraphrases and/or is derived from the HSR (CH2M Hill, 2010), which is provided in Appendix A-1. Topics covered in detail in the HSR include: 1) Site chronology; 2) processing operations utilized by the various owners and operators; 3) historical mine Site water usage and quality information; 4) uses and releases of chemicals; and 5) current Site status since 2000 focusing on removal actions conducted by EPA.

2.1 Mining and Processing Operations

Copper in the Yerington district was initially discovered in the late 1860s, with large-scale exploration of the porphyry copper system occurring in the early 1900s when the area was organized into a mining district by Nevada-Empire Copper Mining and Smelting Company. Mining and ore processing operations at the Site were conducted by various owners from 1953 to November 1999.

2.1.1 Anaconda Operations

The Anaconda Company (Anaconda) became involved in the Site when it entered into a lease agreement and acquired the claims in 1941. Anaconda purchased the property in 1951 and the mine began producing copper in November 1953, producing approximately 1.7 billion pounds of copper during its operations. Anaconda divested itself of the Site on June 30, 1978. Anaconda mining operations generated approximately 360 million tons of ore, 15 million tons of overburden and waste rock (400 acres), 3,000 acres of tailings, and 1,377 acres of disposal ponds.

Mined materials included oxide ore, sulfide ore, low-grade dump-leach ore, low-grade sulfide ore, and alluvium and waste rock overburden. Several processes were required to extract copper from the ore, as discussed further below. Briefly, all oxide and sulfide ore were crushed prior to leaching or processing in the various plant facilities. Copper was extracted from oxide ore using a sulfuric acid leach solution. The vast majority of leaching was conducted in vat leach tanks. A leach dump was operated over a much shorter period (discussed below). Pregnant copper solutions from the tanks and dump were stored in large solution tanks. Copper extracted from the oxide ore was recovered from the acidic leach solutions in the precipitation plant by precipitating (i.e., cementing) the copper onto iron scraps. A concentration/flotation process was used to extract copper from the sulfide minerals.

<u>Dump Oxide Ore Processing</u>

Crushed oxide ore was bedded into vat leach tanks capable of holding 12,000 dry tons of ore and 800,000 gallons of sulfuric acid leach solution. Spent ore, known as oxide tailings or vat leach tailings (VLT), was excavated from the vat leach tanks and disposed in the Oxide Tailings. The vats typically operated on a 96-hour or 120-hour leaching cycle, with an additional 32- to 40-hour wash period and 24 hours required to excavate and refill. The entire cycle required approximately eight days. Thus, eight leach vats were installed and used to process ore.

Following the leaching process, the ore underwent three wash cycles. Acidic leach solutions were recirculated and pumped at a rate of 2,000 gallons per minute (gpm). Copper-enriched wash solutions were put into three of the four open solution storage tanks located between the vat leach tanks and precipitation plant. The three solution tanks used for storing pregnant copper solutions had a total storage capacity of 1.4 million gallons. The additional storage tank, referred to as the wash water sump, stored up to 845,000 gallons of wash water from the leaching circuit, which included slurry from the scrubber in the sulfuric acid plant. Copper was recovered from the leach solution in the precipitation plant, which consisted of the iron launders, solution sumps, an adjacent launder pump station, scrap iron storage, and trommel screens.

The iron launders consisted of 20 parallel launders that were filled with scrap iron used to precipitate (i.e., cement) copper from the sulfuric acid leach solution pumped out of the leach vats and temporarily stored in the solution tanks. The waste product from the precipitation plant was an iron-sulfate solution that was conveyed in unlined ditches (such as the Calcine Ditch) to Lined and Unlined Evaporation Ponds in the northern portion of the Site (as further discussed in Section 2.2). Pregnant copper solution from dump leaching activities (discussed below) was also sent to the precipitation plant but was kept separate from the vat leach solutions. Historical information on flows and chemical concentrations of solutions in various stages of the cementation circuit are provided in Table 2-1. Following cementation, the copper cement product was washed and dried to reduce moisture content prior to shipment off-Site for final smelting (Skillings, 1972).

Table 2-1. Average Assay Values of Solutions at Various Stages in the Cementation Circuit

	Flow (gpm)	Cu (g/L)	H_2SO_4 (g/L)	Fe (g/L)	Fe^{3+} (g/L)
Primary and Scavenger Launders					
New Solution	700	20.0	5.8	7.2	5.4
Recirculated Solution	900	3.5	2.4	23.6	0.5 a
Total Feed (new solution plus recirculated solution)	1,600	10.7	3.8	16.4	2.6
Discharge	1,600	3.8	2.5	23.2	Trace
Secondary Launders					
Recirculated Solution (feed)	900	3.5	2.4	23.6	0.5
Discharge	900	1.0	2.1	26.4	b
Stripping/Settling Launders					
Feed	700	1.0	2.1	26.4	- -
Discharge	700	0.5	2.0	28.1	- -

Notes:

[a] The recirculated solution in the primary launders is the same strength as the recirculated solution in the secondary launders.

[b] The discharge solution in the secondary launders is the same strength as the feed solution to the stripping bank (U.S. Bureau of Mines, 1958).

Cu = copper; H2SO4 = sulfuric acid; Fe = iron; Fe3+ = ferric iron; gpm = gallons per minute, g/L = grams per liter

Dump Leaching

In February 1965, Anaconda began dump leaching low-grade oxide ore in the W-3 Waste Rock Area (Figure 1-2). Copper-enriched solutions were stored in the Dump Leach Surge Pond (Figure 2-1) prior to conveyance to the iron launders for copper recovery. Copper extraction by heap leaching was in in its infancy at the time and recovery from the W-3 Waste Rock Area was inefficient because there were large quantities of ore that never came into contact with the acid-bearing leach solutions. Due to poor copper recovery, Anaconda ceased dump leaching in 1968.

Sulfide Ore Processing

To process sulfide ore, a froth flotation system was constructed and began operating on September 25, 1961. Flotation separation was accomplished by mixing very finely ground ore (pulp) with water and a chemical "collector" (typically xanthates and aerofloats) to make sulfide minerals hydrophobic, and then sparging air and a surfactant chemical "frother" (typically pine oil) through the mixture to create froth. The collector attaches to the sulfide minerals making it hydrophobic and susceptible to attachment to the stabilized air bubbles in the froth mixture, which were skimmed off as copper concentrate. The concentrate was further beneficiated in a scavenger flotation circuit, dewatered and dried, and subsequently hauled by truck to the Wabuska railroad spur and transported to the Washoe Smelter in Anaconda, Montana (Skillings, 1972). Excess pulp after the flotation separation was disposed in the Sulfide Tailings area as a slurry mixture. Operation of the concentrator required approximately 3,000 gpm of water.

Acid Plant

Sulfuric acid was produced at the Site in the fluosolids and acid plant from 1952 to 1978. Raw sulfur ore (predominantly native sulfur and sulfide minerals) was hauled by truck to the Site from the Leviathan Mine located in Alpine County, California until 1962. The fluosolids system roasted the sulfur ore to generate sulfur dioxide (SO_2) gas, which was converted to sulfuric acid (H_2SO_4) in the contact acid plant. In 1962, Anaconda ceased mining sulfur ore from the Leviathan Mine. Between 1962 and 1978, previously stockpiled sulfur ore was blended with liquid sulfur, which was purchased from several suppliers and hauled to the Site where it was used as feed to the Acid Plant to generate sulfuric acid.

The final product was a 93 percent (%) sulfuric acid solution that was used in the vat leach tanks and dump leach of oxide ores. Byproducts, such as selenium, were generated during production of sulfuric acid (CH2M Hill, 2010). Operation of the sulfuric acid plant was discontinued in 1978 and the plant was dismantled by Arimetco, Inc. (Arimetco) in 1992. For its leaching operations, Arimetco purchased liquid sulfuric acid from off-Site vendors and stored it in tanks at the Site. The Acid Plant and surrounding area has subsequently been buried under the Arimetco Phase III South Heap Leach Pad (HLP).

2.1.2 Post-Anaconda Operations

Subsequent operators and lessees used some of the buildings within the Process Areas for operational support, storage, and various light industrial activities; however, the Anaconda-constructed processing components remained inactive after 1978.

In 1982, Copper Tek Corp. operated the mine under the ownership of Don Tibbals and leased the Site for reprocessing tailings and low-grade copper ore using heap leaching and solvent extraction/electrowinning (SX/EW) processes in the area to the south of the Process Areas. In 1988, Don Tibbals sold his interests (except for the Weed Heights community and certain other parcels) to Arimetco. Prior to the sale, Arimetco (operating under the name Arizona Metals Company) had leased a large portion of the mine Site from Don Tibbals. By 1989, Arimetco had also acquired 90% of Copper Tek.

2.2 Arimetco Operations

From 1989 to November 1999, Arimetco conducted the following SX/EW operations on the Site:

- Phase I/II HLP: operated from 1990 to 1996, plus five months in 1997;
- Phase III South HLP: operated from August 1992 to early 1997, plus a few months in 1998;
- Phase III 4X HLP: operated from August 1995 to 1999;
- Phase IV Slot HLP: operated from March 1996 to November 1998; and
- Phase IV VLT HLP: operated from August 1998 to November 1998.

The HLPs (Figure 1-2) were constructed over high-density polyethylene (HDPE) liners with leak detection systems. The HLPs were leached with a sulfuric acid solution and the acidic, copper-laden drain-down fluids were collected in HDPE-lined perimeter ditches, routed to HDPE-lined collection ponds, and conveyed to the SX/EW Plant. The copper-laden acid solution was then stripped of copper in a solvent extraction circuit using a mixture of kerosene and an organic hydroxyamine-based chelating agent (tradename ACORGA) in three process vats (total of approximately 200,000 gallons).

In the SX circuit, the copper in the dump leach liquor was concentrated by the organic in exchange for hydrogen ions producing a strong acid that became the electrolyte for the EW circuit. In the EW circuit, the copper was electroplated to stainless-steel sheets to produce 99.999% fine copper and in the process, additional sulfuric acid was generated. It was this strong acid in the EW circuit that was used to exchange copper from the loaded organic chelating agent.

Arimetco recirculated the acid solution from the EW vats back into the HLPs, which currently continue to drain fluids to the present. The electrolyte circulated between the EW plant and the tail end of the SX plant (called raffinate). The kerosene and organic reagent were also recirculated within the SX/EW circuit, being loaded and stripped repeatedly.

In January 2000, Arimetco, on the verge of bankruptcy and unable to make payroll, abandoned operations at the Site. From 2000 to 2004, NDEP managed HLP drain-down fluids by re-circulation and evaporation. In 2005, ARC was required by EPA to assume responsibility for fluid management operations at the Site. Since 2006, EPA has conducted various RI/FS and closure-related activities associated with the Arimetco Facilities (OU-8).

Inactive Arimetco HLPs that continue to produce drain-down fluids include the Phase I/II HLP, two Phase III HLPs, the Phase IV Slot HLP, and the Phase IV VLT HLP (BC, 2014b). HLP drain-down fluids are currently stored and conveyed in a network of ponds, ditches, and 25,000 feet of pipe, collectively referred to as the fluid management system (FMS). The HLPs and associated FMS components are briefly described below. Additional FMS details are provided in annual operation and maintenance (O&M) reports for the FMS (e.g., BC, 2014b).

Phase I/II Heap Leach Pads

The Phase I/II HLP covers an area of approximately 14 acres (Phase II was extended to the west and north of Phase I). A solution ditch with eleven leak detection points was constructed around the HLP. A variable two- to ten-foot-thick layer of VLT was placed on a single 40-mil (0.04-inch-thick) HDPE liner. The solution ditch that surrounds the Phase I/II HLPs drained to the Phase I Pond.

Phase III Heap Leach Pads

The 46-acre Phase III South HLP and the 50-acre Phase III 4X HLP were constructed by Arimetco to leach low-grade oxide ores. A single 40-mil HDPE liner was installed by Arimetco to recover drain-down solution, and the drainage ditch was designed with a leak detection system over a second, 40-mil HDPE liner. The solution ditches surrounding the Phase III South HLP and the Phase III 4X HLP drained to the Phase III Bathtub Pond and to the Mega Pond, respectively.

Phase IV Slot Heap Leach Pad

The approximate 86-acre Phase IV Slot HLP was constructed by Arimetco on a pad excavated into the W-3 waste rock dump and an asphalt-lined area and was expanded northward between 1993 and 1996 on a 40-mil HDPE liner over a secondary liner of compacted clay. This HLP is surrounded by a berm and double HDPE-lined collection ditch with leak detection between the membranes and seven leak detection monitoring points. Drain-down solutions flow to one of two pregnant leach solution (PLS) ponds.

Until late 2003, drain-down solutions were pumped by NDEP from the PLS ponds to the surface of the HLP for evaporation. In 2006, EPA relined the northern Phase IV Slot PLS Pond, and solutions from this pond were routinely conveyed to the FMS Evaporation Pond (also known as the EPA 4-Acre Pond) constructed by EPA in 2007.

Phase IV VLT Heap Leach Pad

The 54-acre Phase IV VLT HLP was constructed by Arimetco on the southern portion of the former Finger Evaporation Ponds and on native alluvial soils, north of the Oxide Tailings OU, and consists of oxide tailings and run-of-mine and crushed ore from the MacArthur Mine. The Phase IV VLT HLP was constructed on a 40-mil HDPE liner overlying a secondary liner of compacted clay. The solution drainage ditch includes a leak detection system over a 40-mil HDPE liner designed with five leak detection points and drains to the northeast corner of the HLP to a single PLS pond (5.04 million-gallon capacity).

Drain-down solutions from the Phase IV VLT HLP flow by gravity to the VLT Pond and, as needed to improve evaporation efficiency of the FMS, are pumped to one of two new FMS Evaporation Ponds (B and C) described below. EPA completed a VLT pond liner replacement project in October 2012 (BC, 2014b).

2.3 Evaporation Ponds

From the Process Areas, spent process solutions resulting from the beneficiation of copper oxide and sulfide ores were conveyed in unlined trenches to the Lined and Unlined Evaporation Ponds, and ponds in the northern portion of the Site. The Evaporation Ponds in the northern portion of the Site are identified on Figure 1-2 as the Unlined Evaporation Pond (UEP), the Lined Evaporation Pond (LEP), and the Finger Ponds. The Sulfide Tailings were also used to dispose spent oxide ore process solutions prior to the mining of sulfide ores. A brief description and period of known usage of these ponds follows:

- UEP (1954-1978): consists of a large northern section (98 acres) and a much smaller southern section (4.1 acres) constructed on alluvial soils without a liner surrounded by berms constructed of VLT. The volume of pond sediments contained in the UEP is approximately 270,230 cubic yards based on average thicknesses of approximately 1.5 and five feet in the northern and southern sections, respectively.
- LEP (1972-1978): consists of three sections (North, Middle, and South), which were lined with a relatively thin (0.5 to one-inch-thick) liner consisting of a mixture of asphalt tar and crushed gravel. The asphalt liner was placed over one to 2.5 feet of VLT materials. The LEP has a total combined area of approximately 101 acres. The thickness of the pond sediments averages three to six inches, with a maximum measured thickness of approximately 12 inches within the central, topographically lower portion of the LEP.

The volume of pond sediments contained in the LEP is approximately 65,800 cubic yards. Prior reports cite use of the LEP beginning in 1974, but it is present in a 1972 aerial photo.

- Finger Ponds: consist of four narrow "Finger Ponds" and one larger "Thumb Pond." The four Finger Ponds (17.8 acres) were lined similarly to the LEP without the VLT sub-base. The estimated volume of sediments within the Finger Ponds is 5,838 cubic yards based on an average thickness of four inches. The unlined Thumb Pond has elevated embankments along its north and east margins. The exposed portion of the Thumb Pond (i.e., the portion not covered by the Arimetco Phase IV VLT HLP in 1995) covers about 69 acres and was capped in 2010 with VLT materials. The volume of pond sediments contained within the Thumb Pond is 95,000 cubic yards based on an average thickness of 3.5 feet.

In 1955, the flow rate to the Evaporation Ponds averaged approximately 2,000,000 gallons per day (gpd) or 1,385 gpm and water quality characteristics of the fluid showed a free acid concentration of 1.0 grams per liter (g/L), total soluble salts concentration of 171 g/L, and total iron concentration of 37.5 g/L (Nesbitt, 1955; Dalton, 1998). The flow rate to the Evaporation Ponds varied considerably over the period of operations, however. A quantitative water balance of inflow and evaporation is not available. Therefore, a comprehensive and quantitative estimate of the total flux of COIs to the subsurface from the Evaporation Ponds is not possible. The application of a rudimentary calculation to qualitatively estimate the mass of COIs released to the subsurface was explored but found to involve uncertainties as well. The COI maps presented in Section 5.0 provide the proper and reliable information needed to assess the current nature and extent of COIs.

During the operational period, infiltration of process solutions at these locations due to increased hydraulic heads associated with impounded fluids likely raised groundwater elevations and created mounding effects that influenced groundwater gradients, flow directions and velocities, and groundwater quality. More information on the Evaporation Pond operational period is presented in the Final OU-4a RI Report (Copper Environmental Consulting and Broadbent and Associates, 2019). The term "Evaporation Ponds" used in the following sections refers to the LEP and UEP.

2.4 Historical Mine Site Groundwater Pumping, Distribution, and Use

Details of Anaconda's historical groundwater pumping, distribution, and water management at the Site are provided in Appendix A-2 and summarized below.

Mine Site Water

Gill (1951) conducted a groundwater investigation to support open pit mining and reported that the groundwater table around the proposed open pit was approximately 4,350 to 4,380 feet amsl, with variable water levels a result of bedrock compartmentalization of groundwater. Gill (1951) also reported that most groundwater in the proposed open pit was recharged by the Walker River. Dewatering of the pit in advance of mining operations in the early 1950s resulted in a depressed water table.

Groundwater produced from the pit area wells and other supply wells was primarily used in the beneficiation of copper oxide and sulfide ores in the Process Areas. Pit dewatering ended in 1978. The resulting Pit Lake functions as a hydraulic sink that captures alluvial and bedrock groundwater, as discussed in Section 3.3.2.

Historical Groundwater Pumping and Water Management

The known locations of historical wells used for mining-related activities are shown on Figure 2-2. These wells include those that have been abandoned per the State of Nevada Administrative Codes (NAC) 534.420, 534.4365 or 534.4371 and others that have not been abandoned but are not currently in active use.

Groundwater pumped by Anaconda was used for four general purposes: 1) to depress the water level in the open pit to achieve safe working conditions for mining operations; 2) copper ore beneficiation activities in the Process Areas; 3) residential use at Weed Heights, commonly referred to as the "Townsite" in archived documents (Anaconda Mining Company, 1953 and 1966, 1955, and 1957); and 4) ancillary operations at the Site (e.g., fire protection, dust control, drilling, blasting, and supply to shops).

Groundwater use was less during the period from 1952 through 1963 when only oxide ores were leached than in the period from 1963 through 1978 when the copper sulfide ore milling circuit was added to the existing copper oxide ore leaching operations, which resulted in an increased demand for groundwater. References in archived documents to the “Plant” generally refer to oxide leaching facilities prior to 1963, and combined copper oxide and copper sulfide ore beneficiation operations after 1963. Groundwater supplies were obtained from four geographic areas: open pit area wells; evaporation area wells; well WW-10 in the Process Areas; and off-Site area wells. Excess pumped water from pit dewatering activities, was discharged to the Walker River.

Discharge of water to the Walker River peaked in 1953 at 2,373 acre-feet per year, which is equivalent to approximately 1,471 gpm, and generally declined through 1963. At this point, pumped water to the combined Townsite and Plant rose steadily until 1974 at which time it reached 11,388 acre-feet per year (7,058 gpm). There was a general decline in total water pumped during the last four years of Anaconda operations. The annual average pumping rate at the Site ranged from 1,658 gpm in 1978 (the final year of operations) to 7,119 gpm in 1974 (the peak year of water production).

Although the monthly water reports did not specify on-Site water use, some details of water distribution to operational areas are available for 1964 and the first half of 1978 (Table 2-2). In 1964, the Plant received 2,055 acre-feet (45% of total) and the sulfide milling circuit (Concentrator) received 1,511 acre-feet (33% of total).

Table 2-2. Water Distributed to Operational Areas During 1964 and 1978

Operational Area or Use	1964 (acre-feet)	1964 (% of Total)	Jan-Jun 1978 (acre-feet)	Jan-Jun 1978 (% of Total)
Townsite	455	9.9%	58	4.3%
Roads	51	1.1%	31	2.3%
Leach Vats	2,055	44.7%	271	20.2%
Precipitation Plant	---	---	15	1.1%
Sulfide Concentrator	1,511	32.9%	---	---
Acid Plant	481	10.5%	538	40.1%
Water Discharged to Walker River	46	1.0%	0	---
W-3 Waste Rock Dump	---	---	259	19.3%
Dust System	---	---	169	12.6%
Total	4,600	100%	1,340	100%

Open pit area wells were installed during the period 1952-1954 (WW-1 through WW-7) and in 1969 (WW-36) to dewater the pit, supply water to the Townsite, and supply water for copper oxide ore beneficiation. Well logs are available for WW-1 through WW-4 and WW-36 in Appendix A-2. These well logs show large perforated casing intervals (174-284 ft) primarily within the bedrock. The perforated casing in WW-2 includes approximately 39 ft of gravel overlying the bedrock. During July 1955, the combined flow from six of the open pit area wells was estimated to be 2,454 gpm, and the total demand was 2,553 gpm. The use of recycled process water during this period made up for the approximate 100 gpm difference.

Evaporation area wells were installed during the period 1959-1961 (WW-8, WW-9, and WW-11) and in 1965 (WW-12C) to provide the required make-up water (i.e., approximately 1,000 gpm) for the Sulfide Concentrator, which began operation in 1963. Well logs for WW-8, WW-9, and WW-11 are available in Appendix A-2. Well logs indicate perforated casing intervals were large and straddle the alluvium-bedrock boundary. Alluvium at WW-8 was logged at depths of zero to 220 ft, with casing perforated from 100 to 270 ft. WW-9 had an alluvial thickness of 330 ft, with casing perforated from 100 to 415 ft. WW-11 had an alluvial thickness of 310 ft, with casing perforated from 240 to 358 ft. Based on perforated casing positions relative to alluvium-bedrock contact depths these wells drew water from Deep 3 through Deep 5 Zones. However, accurate elevations of perforated casing intervals are not available. Based on 1964 monthly water reports, evaporation area wells had a combined pumping rate that ranged from 442 to 1,390 gpm with an annual average rate of 690 gpm.

Well WW-10 was installed in the Process Areas in 1960 to provide additional water for the copper oxide and copper sulfide ore beneficiation operations. The well log for WW-10 is available in Appendix A-2. The alluvial thickness (depth to bedrock) for this well was logged as 200 ft. Casing was perforated within the alluvium and bedrock from depths of 105 to 505 ft.

Per the well log on file with the NDWR, the well was drilled to a depth of 610 feet and penetrated 200 feet of alluvial materials before reaching bedrock. At that time, the depth to groundwater was 100 feet below ground surface (bgs). The well casing was perforated from 105 to 505 feet bgs, resulting in about 95 feet of alluvial materials and 305 feet of bedrock that could yield groundwater. When tested for two hours at a rate of 595 gpm, WW-10 exhibited a drawdown of 81 feet (close to the alluvium-bedrock contact).

There are few continuous historical pumping records for WW-10. Table 2-3 summarizes 1964 monthly water reports for well WW-10. The monthly pumping rate was calculated by dividing the monthly volume by the number of days in each month and the number of minutes in each day. The pumping rate ranged from 102 gpm in January 1964 to 254 gpm in October 1964, with an average annual rate of approximately 169 gpm for the 10 months with pumping data.

Table 2-3. 1964 Monthly Pumping Volumes and Rates for Mine-Water Supply Well WW-10

Month, 1964	Volume Pumped (cu ft)	Volume Pumped (gal)	Average Pumping Rate (gpm)
January	606,470	4,537,000	102
February	NA	NA	NA
March	NA	NA	NA
April	1,135,410	8,494,000	190
May	1,230,851	9,208,000	206
June	1,329,595	9,946,700	223
July	1,134,621	8,488,100	190
August	1,362,839	10,195,400	228
September	1,227,109	9,180,000	206
October	1,514,771	11,332,000	254
November	1,220,291	9,129,000	205
December	1,139,420	8,524,000	191
Annual Values	11,901,377	89,034,200	169

NA = not available; cu ft = cubic feet; gpm = gallons per minute; gal = gallons

Groundwater quality samples were obtained from WW-10 from August 1991 to June 2006. From August 1991 through 1994, at least three samples were collected during each year. From 1995 through June 2006, samples were collected quarterly. Not all parameters were monitored in each sampling event. Results for sulfate, uranium, and arsenic are discussed below.

Concentrations of sulfate in the 49 samples collected from August 1991 to June 2006 ranged from 457 to 2,485 milligrams per liter (mg/L). Concentrations of uranium in the 10 samples collected from September 2003 to June 2006 ranged from 190 to 310 micrograms per liter (µg/L). From August 1991 to June 2006, 95% (i.e. 35 of the 37) reported arsenic results were less than or approximately equal to the laboratory analytical reporting limits. Laboratory reporting limits for arsenic were commonly higher (i.e., 25 µg/L) during the early part of the monitoring history and lower (as low as 4 µg/L) during the latter part of the monitoring history. In September and October 1991, arsenic concentrations were reported as 1,040 and 3,475 µg/L, respectively. The arsenic results for these two sampling events are inconsistent with and three orders of magnitude greater than the results for the other 35 sampling events, indicating a potential units discrepancy between mg/L and µg/L. Sulfate, uranium, and arsenic concentrations at WW-10 were reviewed over the intervals described above and no visual increasing trends were identified.

2.5 Pumpback Well System

The PWS and associated monitor wells were constructed under an Administrative Order on Consent issued by NDEP on October 1, 1985. ARC operated the PWS located along the northern margin of the Site to collect shallow groundwater beginning in March 1986 (Piedmont Engineering Inc., 2001). The initial PWS consisted of five extraction wells (PW-1 through PW-5; see Figure 3-3[5]), which were constructed in 1985, and are located near the northeast corner of the UEP, east of the LEP and near the Wabuska Drain. A clay-lined 23-acre evaporation pond was also constructed for containment of extracted groundwater.

In 1998, six additional extraction wells (PW-6 through PW-11; see Figure 3-3) were installed and operated as part of the PWS. These wells are located on an east-west trending line approximately 200 feet south of the northern extent of the LEP. The distance from PW-6 to PW-11 is approximately 2,400 feet. The total depth of the wells in the PWS range from 48 to 56 feet, and the wells typically have a 20-foot screen in the Shallow Zone. Other improvements to the PWS included partitioning the 23-acre evaporation pond into three cells and installing an HDPE liner on top of the clay liners in the middle and south cells to limit infiltration through desiccation cracks in the clay liners. No HDPE liner was placed on the north cell.

In the initial phase of operation of the PWS, from March 1986 through 1998, the system consisted of only three operating wells, PW-2, PW-3, and PW-4, the three wells located between the

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[5] In addition to showing monitor well locations and other select Site features, Figure 3-3 also shows the North Study Area (NSA). The NSA is defined and discussed further in Section 5.1.1.

northeast corner of the UEP and the Wabuska Drain. Well PW-4, the closest of these wells to the LEP, is located about 900 feet east of the LEP and 2,000 feet south of the northern extent of the LEP. From 1991 through 1996 these wells were only operated six months of the year, the remainder of the time, the wells operated all year except in 1998 when PW-4 was not operated. In the second phase of operation of the PWS, from 1999 through March 2009, the system consisted of 11 operating wells, PW-1 through PW-11, except in 1999 when PW-1 was not operated (Norwest, 2009). EPA approved the shutdown of the PWS on March 25, 2009 to allow for a characterization of groundwater conditions at the northern Site margin. Subsequently, the pumpback wells were used as part of an aquifer test to characterize hydraulic properties of the Shallow Zone (ARC, 2010). The test data were used to: 1) delineate the hydraulic capture zones of the PWS; and 2) assess the historical effectiveness of the PWS in limiting the off-Site migration of shallow groundwater. The PWS evaluation is discussed in Section 3.3.1. Since completion of hydraulic testing, the PWS has been shut down and wells PW-1 to PW-11 have been monitored pursuant to the GMP (BC, 2012a).

2.6 Wabuska Drain

The Wabuska Drain is a 13.8-mile long unlined ditch that extends from the Site to the Walker River. The grade of the Wabuska Drain between the Site and the southern margin of the YPT Reservation is approximately 0.15% over 4.1 miles. The grade increases to about 0.16% along the 1.1-mile length within the YPT Reservation. From the northern margin of the YPT Reservation to its intersection with the Walker River, the average grade is approximately 0.04% (BC, 2015b). Adjacent surrounding agricultural fields slope gently toward the Wabuska Drain or connect to the drain through lateral ditches that feed into the drain by gravity flow. The drain was constructed in the late 1930s, when the regional groundwater table was higher, to intercept shallow groundwater to stabilize areas north of the Site adjacent to the tracks of the former Nevada Copper Belt Railroad and several farms. The Wabuska Drain alignment near the Site has changed over time (BC, 2015b; CH2M Hill, 2010), as shown on Figure 2-3. Portions of these former alignments are now buried beneath the Evaporation Ponds and the Hunewill Ranch agricultural fields (immediately to the north of the mine-Site boundary).

Currently the drain functions as one of many irrigation return-flow ditches in the northern Mason Valley. These drains collect irrigation tail water and run-off from agricultural fields and convey water to downgradient agricultural areas for further irrigation uses and/or discharge to the Walker River (CH2M Hill, 2010).

Historically, the Wabuska Drain alignments near the Site intercepted shallow groundwater (CH2M Hill, 2010). However, the various drain alignments near the Site no longer intercept shallow groundwater due to basin-wide groundwater level declines (Section 4.9). In the northern part of the Wabuska Drain, inputs also include intercepted shallow groundwater and deeper water associated with alluvial groundwater discharge and geothermal springs that coincide with a series of northwest trending faults referred to as the Wabuska lineament (Stewart, 1988). Other potential past and/or current inputs include discharges from, or groundwater influenced by, the Thompson smelter and various geothermal power production activities. Details regarding the Wabuska Drain are provided in the HSR (CH2M Hill, 2010; Appendix A-1). Results of the ongoing RI for the Wabuska Drain will be reported separately.

3 GROUNDWATER INVESTIGATIONS AND RELATED STUDIES

Numerous investigations and monitoring activities conducted prior to 2005 provide substantial geologic, hydrogeologic, and water quality information pertaining to OU-1. These pre-2005 activities were primarily associated with few monitor wells having screens positioned across the water table located around the northern Site margin. Sampling methods and the quality of laboratory analytical results prior to 2005 were not well documented.

Pre-2005 investigations and reports for the Site and surrounding area are summarized in Section 3.1. Post-2005 investigations and reports performed and/or prepared by ARC are summarized in Section 3.2. Pre-2005 data are used to address historical aspects of the HCSM because historical conditions (e.g., groundwater elevations and flow directions) were different from current conditions due to mine operations, agricultural activities, groundwater and surface water usage, and weather conditions (BC, 2014a). Historical aspects of the HCSM are generally more qualitative relative to aspects of the HCSM developed using data collected after 2005 because the historical data are typically not spatially robust and data quality is often not well documented.

Since 2005, numerous investigations and monitoring activities have been conducted by ARC with EPA and stakeholder involvement, and the sampling methods and quality of the laboratory analytical results since 2005 have been well documented. A draft Quality Assurance Project Plan (QAPP) that included standard operating procedures (SOPs) for sampling and field data collection methods was prepared in 2003 to establish and implement strict QA/QC procedures and, subsequently, was periodically revised to result in the *Quality Assurance Project Plan - Revision 5* (Environmental Standards, Inc. [ESI], and BC, 2009). The current *Quality Assurance Project Plan - Revision 5.1* is dated September 5, 2018 (ARC, 2018). Other QA planning documents that were prepared pursuant to the 2007 Order included the: 1) *Data Management Plan for the Yerington Mine Site* (BC, 2007a); 2) GMP (BC, 2012a); and 3) EPA-approved work plans specific to OU-1.

3.1 Pre-2005 Investigations

Investigations and reports relevant to the OU-1 Study Area that were conducted prior to 2005 are provided below, generally listed in chronological order:

1. Gill, D.K., 1951. Groundwater at the Yerington Mine, Lyon County, Nevada, a consultant report prepared for Anaconda that describes the results of aquifer testing and provides projections of groundwater inflows and dewatering rates for the open pit.
2. Huxel, C.J., Jr. and E.E. Harris, 1969. Water Resources and Development in Mason Valley, Lyon and Mineral Counties, Nevada, 1948-1965, NDWR Bulletin No. 38 prepared in cooperation with the USGS. This is a comprehensive hydrologic study of the Mason Valley area including water budgets and effects of agriculture on surface water and groundwater quality and quantity.
3. Seitz, H., A.S. Van Denburgh, and R.J. La Camera, 1982. Ground Water Quality Downgradient from Copper Ore Milling Wastes at Weed Heights, Lyon County, Nevada, USGS Open File Report 80-1217. This study presents hydrologic and geochemical data on the effects of mining on groundwater quality from several monitor wells, most of which are no longer operational.
4. Applied Hydrology Associates (AHA), 1983. Evaluation of Water Quality and Solids Leaching Data, a consultant report prepared for Anaconda Minerals Company. This report includes surface water and solids leaching data in addition to groundwater sampling data that are compared to the data reported by Seitz et al. (1982).
5. Anaconda Minerals Company, 1984. Water Quality Investigation and Mitigation Plan, Yerington Mine Site, Yerington, Nevada, a report prepared for NDEP that summarized additional field data and groundwater conditions north of the Site.
6. Proffett, J.M., Jr., and J.H. Dilles, 1984. Geologic Map of the Yerington District, Nevada, Nevada Bureau of Mines and Geology, Map 77.
7. Nork, W., 1989. MacArthur Project Hydrogeologic Investigation, Lyon County, Nevada, a consultant report prepared for MacArthur Mining and Processing Co. that describes the general hydrogeologic conditions associated with a proposed project to develop an open pit mine located to the northwest of the Site.
8. Dalton, D., 1998. *Arimetco Yerington Mine and Process Facility Site Assessment of Groundwater Quality*, a consultant report prepared for Arimetco for submittal to NDEP in response to NDEP's Finding of Alleged Violation and Order of February 1997.

9. Lewis, B., 2000. *Geophysical Survey Results of the Yerington Mine, Mason Valley, Nevada*, a BLM report on electro-magnetic and resistivity surveys north of the Site.
10. Superfund Technical Assessment and Response Team (START), 2000 and 2001. *Expanded Site Inspection: Yerington Mine* and *Anaconda, Yerington Mine Site Emergency Response Assessment Final Report,* reports prepared for the EPA that describe Site conditions including groundwater quality.
11. Piedmont Engineering, Inc., 2001. *Yerington Shallow Aquifer Data Evaluation Report*, consultant prepared for ARC. Interpretations of data presented in this report related to the nature and extent of mine-influenced groundwater.
12. AHA and Norwest Applied Hydrology, 2000 through 2007. *Annual Monitoring and Operation Summary: Pumpback Well System, Yerington Nevada*, annual consultant reports prepared for ARC. These reports provide groundwater elevation and water quality data for the pumpback system and associated monitor wells. The reports also include pumping rates and time-concentration plots for select chemicals.
13. BC, 2002. *Installation of Two Monitor Wells at the Yerington Mine Site, Lyon County, Nevada.* This letter report described the drilling and well construction activities of two monitor wells, which was an interim action required by NDEP, EPA, and BLM.

3.2 Post-2005 Investigations

A generalized chronology of the phased, groundwater-related field investigations conducted since 2005 is provided in Figure 3-1. The following subsections describe the post-2005 investigations and related evaluations by media. Groundwater characterization activities largely involved monitor well installation. Monitor wells installation procedures are described in Section 3.2.1. After installation, initial sampling and testing for OU-1 characterization purposes, these wells were subsequently incorporated into the Site-Wide Groundwater Monitoring Program, which is described in Section 3.2.3.

3.2.1 Analytical Parameters and Constituents of Interest

Groundwater investigations prior to 2005 obtained data for a variety of different analytical parameters and the evaluations focused on different COIs to fulfill various objectives. Beginning in approximately 2002, communications between ARC, EPA, NDEP, BLM, YPT and other stakeholders established a preliminary list of analytical parameters and potential COIs during a series of groundwater technical meetings. Subsequently, analytical parameters and potential COIs evolved over time based on regulatory input, knowledge of available ore beneficiation processes, available site data, the evolving HCSM, and specific DQOs.

Ultimately EPA recommended an expanded analytical parameter list based on the rationale that an extensive list of analytes should be considered potential COIs until sufficient spatial and temporal data had been obtained to develop an HCSM suitable for RI decision-making, determine background concentration limits, and delineate the extent of contamination. These recommendations resulted in the analytical parameter list presented in the EPA-approved Hydrogeologic Framework Assessment (BC, 2005), as well as numerous subsequent groundwater investigation work plans and groundwater monitoring plans that were approved by EPA and stakeholders.

As RI characterization activities progressed, EPA recommended that ARC use spatial distributions and temporal trends of select COIs (specifically pH, sulfate, uranium, arsenic, alkalinity, nitrate) to help display the results of the RI characterization and groundwater monitoring activities. The rationale for displaying spatial and temporal trends for these six parameters (which has been presented in past groundwater-related work plans and reports) is summarized in Section 5.0.

Evaluations conducted since submittal of the draft OU-1 RIR (BC, 2017) have utilized a broader set of analytical parameters to characterize Site-wide groundwater conditions and, in particular, to better assess background groundwater quality. These more recent evaluations, and rationale for selection and use of additional COIs, are discussed in Sections 3.3.6 and 5.0.

3.2.2 Monitor Well Installations

Recognizing that groundwater characterization activities would largely involve monitor well installation, ARC and EPA adopted the following sequential approach to field data collection to maximize usable data and optimize the design of a monitor well network intended to serve the data needs for both OU-1 RI characterization and long-term Site-wide groundwater monitoring.

- Borehole drilling using a roto-sonic core drilling rig and lithologic logging of continuous cores to identify coarse-grained or potentially transmissive intervals where chemicals could potentially migrate.
- Depth-specific (zonal) groundwater sample collection in the alluvial aquifer at nominal 20-foot intervals from the top of the water table to the target depth of each borehole using low-flow, minimal drawdown purging and sampling procedures approved by EPA.
- Collection of field measurements from zonal samples including pH, specific conductivity, temperature, dissolved oxygen (DO), oxidation-reduction potential (ORP), sulfate, alkalinity, and total and ferrous iron, using routinely-calibrated field meters and accuracy-checked water quality field test kits.
- Laboratory analyses of zonal samples for total and dissolved uranium, total and dissolved arsenic, total organic carbon (TOC), and uranium isotopes (^{234}U, ^{235}U, and ^{238}U).
- Based on the zonal groundwater sample results, construction of new monitor wells in various groundwater zones using methods and materials specified in EPA-approved SOPs and work plans, with EPA approval of well screen lengths and positions.
- Surveying of well location coordinates and reference point elevations followed by measurement of groundwater elevations in all new and existing monitor wells.
- Hydraulic (slug) testing of monitor wells and analysis of hydraulic test data.
- Deployment of pressure transducers and data loggers in select monitor wells with EPA approval to collect groundwater elevation data at four-hour intervals and assess temporal water level fluctuations.
- Collection of hydrologic tracers from monitor wells.
- Incorporation of the new monitor wells into the GMP (BC, 2012a), discussed below in Section 3.2.3.

In total, the groundwater RI characterization activities described above resulted in drilling 133 boreholes, logging approximately 33,000 linear feet of core, collecting and analyzing 624 zonal groundwater samples, installing 299 new monitor wells, and hydraulic (slug) testing 296 wells. Appendix C provides monitor well information including: 1) borehole lithologic information and well construction diagrams; 2) well construction and location information for the active monitor wells; 3) construction information for abandoned and inactive wells not used for routine groundwater monitoring; and 4) zonal groundwater sample results.

The various investigations involving monitor well installations are integrated into the discussion of the Site-wide groundwater monitoring program in the following section. Appendix D provides OU-1/Site-wide groundwater monitoring information including water level and chemical data, water-level hydrographs, charts illustrating temporal changes in vertical gradients at monitor well clusters, and charts illustrating temporal changes in chemical concentrations at monitor wells.

3.2.3 Shallow Zone Groundwater Investigation

The 2009 Shallow Zone investigation is detailed in the *Shallow Zone Data Summary Report - Revision 1* (BC, 2010a; Appendix B) and summarized below. The Shallow Zone investigation was designed to improve the understanding of hydrogeologic and geochemical conditions in the Shallow Zone of the alluvial aquifer to the north of the Site by refining the distributions of sulfate, uranium, uranium isotopes, dissolved metals, TOC and alkalinity in Shallow Zone groundwater. This information was used to evaluate potential sources of chemicals in groundwater in this portion of the Study Area and identify portions of the Intermediate or Deep Zones in the alluvial aquifer that would warrant the installation of monitor wells.

Shallow Zone characterization activities during 2009 included:

- Direct push technology (DPT) with Geoprobe® equipment was used to obtain continuous, high-resolution electrical conductivity (EC) measurements of subsurface materials at 93 locations. EC profiling was initially conducted at four locations (OU1-DPT-18, OU1-DPT-40, OU1-DPT-16, and OU1-DPT-24) where lithologic logging and zonal sampling had been previously conducted during borehole drilling and well installation at the B/W-2, B/W-3, B/W-18, and USGS-13S/W32DC-D well clusters, respectively, and one location (OU1-DPT-13) where geophysical logging had been conducted in 1983 (W5AB-2). This comparison was intended to help correlate EC data with clays and/or elevated solute concentrations in groundwater. In addition, EC data were also used to make decisions regarding the number and depths of sampling intervals at each individual DPT location.

- Field parameters were measured in groundwater samples collected from the sampled intervals within the Shallow Zone including pH, specific conductivity, temperature, sulfate, and total alkalinity (alkalinity) using routinely-calibrated field meters and accuracy-checked water quality field test kits.
- DPT equipment was used to collect groundwater samples at each location from as many as three intervals within the Shallow Zone, and samples were submitted to the analytical laboratory for chemical analyses including total and dissolved uranium, TOC, 28 metals, uranium isotopes, and sulfur and oxygen isotopes in dissolved sulfate at EPA-selected locations.
- DPT locations were surveyed by a registered Nevada surveyor.
- Upon completion of EC profiling and/or groundwater sample collection, all boreholes were abandoned in compliance with Nevada regulatory requirements.

In 2010, DPT equipment and identical sample collection methods were used at 10 additional locations to obtain groundwater samples for laboratory analysis (BC, 2013a). EC profiling was not conducted during the 2010 investigation. Both the 2009 and 2010 DPT locations are shown on Figure 3-2. Chemical distributions in the Shallow Zone of the alluvial aquifer are shown on figures provided in Appendix B and are described as follows:

- The highest concentrations of major ions and metals (e.g., aluminum, copper, iron, manganese, nickel, vanadium, and zinc) and uranium in the Shallow Zone were typically detected beneath the central portion of the UEP and the south-central and north-central portions of the LEP. Low pH values occur beneath the LEP and UEP (Appendix B, Figure 4-12). Alkalinity was depressed or non-detectable beneath the UEP (Appendix B, Figure 4-15). Elevated alkalinity (e.g., >500 mg/L) occurred down-gradient of the Weed Heights sewage lagoons.
- The high chemical concentrations beneath the Evaporation Ponds decrease laterally by varying orders of magnitude because of past and current physical and chemical attenuation processes. West of the LEP, concentrations of sulfate, other mobile chemicals, and metals decrease rapidly with distance from the Site (Appendix B, Figure 4-14). To the east beneath the agricultural fields, chemical concentrations are generally lower than other locations beneath or near the Site. Elevated concentrations of select constituents in groundwater samples were observed from sample locations on the agricultural fields and included: alkalinity, manganese, molybdenum, nickel, potassium, selenium, sodium, uranium, vanadium, and zinc.

- The distribution of dissolved (i.e., filtered) uranium in the Shallow Zone is generally consistent with the distribution of many other Site chemicals in that: 1) the highest concentrations occur beneath the central portion of the UEP, and the south-central and north-central portions of the LEP; 2) elevated concentrations extend beyond the Site boundary along a northwest alignment from the Evaporation Ponds; and 3) elevated chemical concentrations beneath the Evaporation Ponds decrease laterally by varying orders of magnitude (Appendix B, Figure 4-32). However, there are important differences that suggest that alkalinity and calcium influence the mobility/attenuation of uranium. Uranium concentrations rapidly decrease laterally to the west, similar to other chemicals, but do not decrease as rapidly to the east beneath the irrigated agricultural fields. Elevated concentrations of uranium in DPT locations including DPT-41, DPT-42, DPT-46, DPT-49, DPT-61, and DPT-72 are roughly coincident with: 1) the areas of locally high concentrations of alkalinity (over 300 mg/L) in Shallow Zone groundwater at the northwest and northern edge of the agricultural fields; and 2) high calcium concentrations to the northwest of the agricultural fields and general widespread distribution of calcium throughout the agricultural fields.
- The distribution of arsenic in Shallow Zone groundwater differs from the distributions of the parameters and chemicals described above. The highest concentrations of arsenic were detected in Shallow Zone groundwater at DPT-23 (up to 620 µg/L), DPT-28 (up to 580 µg/L), and other nearby locations, which are located approximately 3,000 to 3,500 feet north of the Evaporation Ponds (Appendix B, Figure 4-16). Beneath the Evaporation Ponds, arsenic concentrations were much lower and typically ranged from approximately 10 to 160 µg/L. In this area, the lowest arsenic concentrations in Shallow Zone groundwater occur along a northwest trend from the Weed Heights sewage lagoons. To the west of the LEP, arsenic occurs in Shallow Zone groundwater at concentrations at or slightly above 50 µg/L.

3.2.4 Site-Wide Groundwater Monitoring Program

Groundwater monitoring in the Study Area has evolved over time in response to regulatory requirements and incorporation of new wells installed during the OU-1 RI. Currently, long-term Site-wide groundwater monitoring is conducted in accordance with the GMP (BC, 2012a) and EPA- and NDEP- approved modifications listed in Table 3-1. Provided below is a summary of the: 1) development of the monitoring program; 2) current active monitor well network; 3) water level monitoring activities; and 4) groundwater quality monitoring activities.

Monitoring Program Development

Table 3-1 provides a chronological summary of groundwater characterization and monitoring activities at the Site and the evolution of the monitor well network over time.

Table 3-1. Chronology of Groundwater Characterization and Monitoring Activities

Date	Activity
1976-1982	The USGS conducted groundwater investigations north of the Site boundary, which culminated in a report entitled: *Ground-water quality down-gradient from copper-ore milling wastes at Weed Heights, Lyon County, Nevada* (Seitz et al., 1982).
1982-1985	1982 – An NDEP Order required groundwater investigations near the Sulfide Tailings and Evaporation Ponds and initial groundwater monitoring. 1985 – An NDEP Order required construction of the PWS and performance of associated O&M and groundwater monitoring activities.
1997	An NDEP Order was issued that required Arimetco to conduct groundwater investigations and monitoring of the Site, requesting both a complete hydrogeological assessment for the Yerington Mine (including existing and projected Pit Lake conditions), and a facility assessment to identify all areas where constituent concentrations in groundwater exceed the drinking water standards or background.
1999	Implementation of a Geoprobe® investigation of Shallow Zone alluvial aquifer conditions north of the Site, which consisted of collecting 29 samples from 18 locations (AHA, 2000).
2002	Two groundwater monitor wells, MW-2002-1 (subsequently re-named B/W-2S) and MW-2002-2, were installed (BC, 2002) under an interim action directed by NDEP.
2004	Several groundwater characterization boreholes were drilled to collect groundwater grab samples, and three groundwater monitor wells were installed in the Process Areas, pursuant to the *Final Draft Process Areas Work Plan* (BC, 2003).
2005	Implementation of the *First-Step Hydrogeologic Framework Assessment* (HFA; BC, 2005) under the Unilateral Administrative Order (UAO) issued by EPA for Initial Response Activities, EPA Docket No. 9-2005-0011, including the installation of the first phase of B/W wells.
2007-2008	Implementation of the Second-Step HFA (BC, 2007b) included a second phase of well installations and the preparation of OU-specific work plans that describe additional on-Site monitor well installations under the 2007 Order. ARC submitted the *Draft Site-Wide Groundwater Monitoring Plan* (BC, 2007c).
2008	Monitor well identification numbers modified to include a suffix designating the groundwater zone in which the well screen is positioned, including the Shallow, Intermediate, Deep, and bedrock zones.
2008	EPA approved the elimination of well MW-1S from the monitoring network due to an obstruction.
2008	Collection of groundwater grab samples and groundwater levels beneath the Anaconda Evaporation Ponds pursuant to the *Anaconda Evaporation Ponds Removal Action Characterization Work Plan* (BC, 2008a).
2009	-Shutdown of the pumpback wells on March 25, 2009 with EPA approval. -Implementation of activities pursuant to the *Pumpback Well System Characterization Work Plan* (ARC, 2008) including: 1) installation of nine groundwater monitor wells north of the PWS and the LEP; and 2) pumpback well aquifer testing pursuant to the *PWS Characterization Work Plan Addendum - Revision 2* (ARC, 2010), which was performed in 2010. - Implementation of activities pursuant to the *Shallow Alluvial Aquifer Characterization Work Plan for Northern Portion of the Yerington Mine Site* (BC, 2008b). -EPA approved the QAPP - Revision 5 dated May 20, 2009 (ESI and BC, 2009), which contains SOPs for groundwater monitoring. -Implementation of a separate Domestic Well Monitoring Plan (BC, 2010b) as a revision to the QAPP per EPA direction. The Domestic Well Monitoring Plan and related Bottled Water Program are discussed further in Section 3.4.

Table 3-1. Chronology of Groundwater Characterization and Monitoring Activities

Date	Activity
	-ARC submitted the *Site-Wide Groundwater Monitoring Plan - Revision 1* (BC, 2009a).
2010-2011	Installation of 123 wells per the *2010 Groundwater Monitor Well Work Plan - Revision 2* (BC, 2010c) and the *Agricultural Fields Characterization Work Plan - Revision 2* (BC, 2010d).
2011	-Installation of 58 wells per the *On-Site Monitor Well Installation Work Plan - Revision 1* (BC, 2011a). -Aquifer testing of an agricultural well (WDW019) north of the Site, including a 96-well observation network, pursuant to the *Aquifer Test Work Plan - Revision 1* (BC, 2011b).
2012	Submittal of the *Site-Wide Groundwater Monitoring Plan - Revision 2* (BC, 2012a). Addition of new well YPT-MW-15I to the monitoring program in November 2012.
2013	-Installation of 58 monitor wells per the *Additional Monitor Well Work Plan - Revision 1* (Additional Monitor Well Work Plan; BC, 2013b). -Addition of five EPA Arimetco wells, nine YPT wells, and the one MacArthur mine well (MMW-2) to the monitoring program after the 4Q 2011 event. -EPA (2013a) approved of eliminating well USEPA2S from the active monitor well network. EPA (2013b) approves of abandoning well USEPA2S and eliminating the following five metals from the analyte list for wells having at least four quarters of data: lead, silver, thallium, tin, and titanium. EPA (2013b) also approved of reducing the sampling frequency of 118 wells from quarterly to semi-annually, with sampling to be conducted in the first and third quarters of subsequent years.
2014	-EPA (2013c) approved the abandonment of well B/W-14S and the well was abandoned in January 2014. -Well USEPA2S was abandoned in April 2014 and was moved from the inactive to abandoned well list. -ARC submitted the *Technical Memorandum: Site-Wide Groundwater Monitoring Optimization* (ARC, 2014) proposing several modifications to the GMP (BC, 2012a). -EPA approved of reducing the frequency of manual water level monitoring in wells installed before 2013 from monthly to quarterly, to coincide with the quarterly sampling events (EPA, 2014a). EPA also approved reducing the frequency of collecting groundwater samples from wells installed before 2013 for nitrate analysis from quarterly to semi-annually, with sampling to be conducted in the first and third quarters of subsequent years (EPA, 2014a).
2015	Installation of six additional monitor wells (B/W-63 cluster) in the third quarter of 2015 (3Q 2015), pursuant to the Additional Monitor Well Work Plan (BC, 2013b). Initial water level monitoring and sampling of these six wells in 4Q 2015.
2019	NDEP approved elimination of Thorium-228 and Thorium-230 monitoring in all wells. NDEP approved elimination of Radium-226 and Radium-228 monitoring in all wells except B/W-8, B/W-32, B/W-57, YPT-MW-12, and YPT-MW-15.

The network development detailed in Table 3-1 included the addition and elimination of monitor locations as summarized in Table 3-2.

Table 3-2. Inventory of Monitor Well and Piezometer Locations (2007 - 2019)

Date (Through)	Total (1)	Monitor Wells and/or Piezometers	Pumpback Wells
2007	87	76	11
2008	101	90	11
1Q 2009	110	99	11
2Q 2009	114(2)	103(2)	11
3Q 2009	114(2)	103(2)	11
4Q 2009	114(2)	103(2)	11
1Q 2010	114(2)	103(2)	11
2Q 2010	114(2)	103(2)	11
3Q 2010	114(2)	103(2)	11
4Q 2010	114(2)	103(2)	11
1Q 2011	237(3)	226(3)	11
2Q 2011	237(3)	226(3)	11
3Q 2011	237(3)	226(3)	11
4Q 2011	242(4)	231(4)	11
1Q 2012	309(5)	298(5)	11
2Q 2012	309(5)	298(5)	11
3Q 2012	309(5)	298(5)	11
4Q 2012	310(6)	299(6)	11
1Q 2013	309(7)	298(7)	11
2Q 2013	309(7)	298(7)	11
3Q 2013	309(7)	298(7)	11
4Q 2013	309(7)	298(7)	11
1Q 2014	308(8)	297(8)	11
2Q 2014	325(9)	314(9)	11
3Q 2014	354(10)	343(10)	11
4Q 2014	354(10)	343(10)	11
1Q 2015	354(10)	343(10)	11
2Q 2015	354(10)	343(10)	11
3Q 2015 -4Q 2019	360(11)	349(11)	11

Notes:

1. Total does not include domestic/supply wells that were part of the network until 2010. Total does include the eleven pumpback wells.
2. Includes four Lyon County wells.
3. Includes four Lyon County wells and 123 wells installed in 2010/2011.
4. Includes four Lyon County wells, 123 wells installed in 2010/2011, and five EPA Arimetco wells (sampled in 3Q 2011 and added to the monitoring program after the 4Q 2011 event).
5. Includes four Lyon County wells, 123 wells installed in 2010/2011, 58 wells installed in 2011/2012, five EPA Arimetco wells, eight YPT wells (excluding YPT-MW-7), and one MacArthur mine well (MMW-2). Prior to 1Q 2012, these wells were or may have been sampled; however, sample collection methods were either inconsistent with EPA-approved sample collection methods or were not documented.
6. Includes four Lyon County wells, 123 wells installed in 2010/2011, 58 wells installed in 2011/2012, five EPA Arimetco wells, nine YPT wells (including new well YPT-MW-15I in November 2012), and one MacArthur mine well (MMW-2).
7. With EPA approval, well USEPA2S moved to inactive well list in March 2013, subsequently proposed for abandonment in August 2013, and abandoned in April 2014.
8. With EPA approval, well B/W-14S was abandoned in January 2014.

9. Includes 17 wells installed in 2013/2014 and first sampled in 2Q 2014.
10. Includes 29 wells installed in 2013/2014 and first sampled in 3Q 2014.
11. Includes six wells (B/W-63 cluster) installed in 3Q 2015 and first sampled in 4Q 2015.

Active Monitor Well Network

The active monitor well network included 360 wells at 170 locations: 133 wells in the Shallow Zone, including 11 PWS wells formerly used for groundwater extraction (currently in shutdown mode); 55 wells in the Intermediate Zone; 105 wells in the Deep Zone; and 67 bedrock wells (Table 3-3 and Figure 3-3). Of the 360 monitor wells, seven are used only for water level measurements, and the remaining 353 are monitored for both water levels and water quality.

A generalized cross-section that depicts alluvial monitor well screen intervals and groundwater zone designations for active wells within the monitoring network is presented in Figure 3-4(a-c). A generalized cross-section that depicts bedrock monitor well screen intervals is presented in Figure 3-5(a-b). All monitor wells in the network were surveyed by a Nevada-registered surveyor. Project datum is Nevada State Plane West Zone coordinate system (NAD27).

Table 3-3. Well Construction Details for the Active Groundwater Monitoring Network

Well Name (1)	Completion Date	Groundwater Zone	Well Type	Well Screen Interval		
				feet amsl		
Pumpback Wells						
PW-1S	10/1/85	Shallow	Sampling	4335.02	-	4312.52
PW-2S	10/1/85	Shallow	Sampling	4335.73	-	4315.23
PW-3S	10/1/85	Shallow	Sampling	4333.88	-	4313.38
PW-4S	10/1/85	Shallow	Sampling	4331.48	-	4311.98
PW-5S	10/1/85	Shallow	Sampling	4334.23	-	4313.73
PW-6S	10/21/98	Shallow	Sampling	4340.11	-	4323.11
PW-7S	10/22/98	Shallow	Sampling	4339.32	-	4319.82
PW-8S	10/22/98	Shallow	Sampling	4336.63	-	4316.63
PW-9S	10/23/98	Shallow	Sampling	4337.38	-	4317.38
PW-10S	10/23/98	Shallow	Sampling	4338.46	-	4318.46
PW-11S	10/24/98	Shallow	Sampling	4339.68	-	4319.68
Shallow Zone Monitor Wells						
B-2S	5/18/89	Shallow	Water Level	NR	-	NR
B-3S	5/18/89	Shallow	Water Level	NR	-	NR
B/W-1S	1/23/08	Shallow	Sampling	4334.71	-	4314.71
B/W-2S	6/13/02	Shallow	Sampling	4330.95	-	4320.95
B/W-3S	9/26/07	Shallow	Sampling	4332.50	-	4312.50
B/W-4S	1/21/08	Shallow	Sampling	4316.74	-	4296.74
B/W-5RS	11/16/07	Shallow	Sampling	4326.12	-	4306.12
B/W-6S	1/30/08	Shallow	Sampling	4326.78	-	4306.78
B/W-8S	10/9/07	Shallow	Sampling	4325.95	-	4305.95
B/W-9S	11/7/07	Shallow	Sampling	4331.77	-	4311.77
B/W-10S	1/23/08	Shallow	Sampling	4321.56	-	4301.56
B/W-11S	11/4/07	Shallow	Sampling	4330.42	-	4310.42
B/W-13S	7/13/05	Shallow	Sampling	4364.14	-	4344.14
B/W-15S	7/22/05	Shallow	Sampling	4348.48	-	4328.48
B/W-16S	10/7/07	Shallow	Sampling	4328.68	-	4308.68
B/W-18S	2/19/08	Shallow	Sampling	4333.87	-	4308.87
B/W-19S	1/9/08	Shallow	Sampling	4331.43	-	4311.43
B/W-20S	7/13/07	Shallow	Sampling	4377.44	-	4357.44
B/W-21S	7/24/07	Shallow	Sampling	4338.99	-	4318.99
B/W-22S	7/18/07	Shallow	Sampling	4309.55	-	4289.55
B/W-25S	1/31/08	Shallow	Sampling	4322.63	-	4302.63
B/W-27S	2/7/08	Shallow	Sampling	4338.98	-	4318.98
B/W-28S	1/15/08	Shallow	Sampling	4331.67	-	4311.67
B/W-29S	1/6/08	Shallow	Sampling	4314.97	-	4294.97

Table 3-3. Well Construction Details for the Active Groundwater Monitoring Network

Well Name (1)	Completion Date	Groundwater Zone	Well Type	Well Screen Interval
				feet amsl
B/W-30S	10/25/10	Shallow	Sampling	4325.10 - 4305.10
B/W-31S1	12/8/10	Shallow	Sampling	4330.77 - 4315.77
B/W-31S2	12/8/10	Shallow	Sampling	4304.95 - 4294.95
B/W-32S	1/11/11	Shallow	Sampling	4328.60 - 4308.60
B/W-33S	8/4/10	Shallow	Sampling	4328.23 - 4308.23
B/W-34S	12/5/10	Shallow	Sampling	4337.68 - 4317.68
B/W-36S	8/11/10	Shallow	Sampling	4329.76 - 4319.76
B/W-37S	6/6/10	Shallow	Sampling	4331.62 - 4311.62
B/W-38RS	10/11/10	Shallow	Sampling	4320.17 - 4300.17
B/W-40S	1/10/11	Shallow	Sampling	4318.41 - 4298.41
B/W-41S	2/8/11	Shallow	Sampling	4324.54 - 4304.54
B/W-42S	11/9/10	Shallow	Sampling	4326.05 - 4306.05
B/W-43S	12/17/10	Shallow	Sampling	4323.75 - 4303.75
B/W-44S	9/24/10	Shallow	Sampling	4324.88 - 4304.88
B/W-45S	1/17/11	Shallow	Sampling	4331.84 - 4311.84
B/W-46S	11/8/10	Shallow	Sampling	4327.09 - 4307.09
B/W-50S (2)	2/11/14	Shallow	Sampling	4337.83 - 4317.83
B/W-51S	8/25/10	Shallow	Sampling	4303.87 - 4293.87
B/W-52S	8/18/10	Shallow	Sampling	4329.90 - 4309.90
B/W-53S1	1/20/11	Shallow	Sampling	4310.26 - 4290.26
B/W-53S2	1/19/11	Shallow	Sampling	4265.87 - 4255.87
B/W-54S	8/20/10	Shallow	Sampling	4298.38 - 4288.38
B/W-55S	10/20/10	Shallow	Sampling	4327.27 - 4307.27
B/W-56S	3/13/12	Shallow	Sampling	4334.12 - 4314.12
B/W-57S	3/15/12	Shallow	Sampling	4325.36 - 4305.36
B/W-58S	3/14/12	Shallow	Sampling	4294.04 - 4284.04
B/W-59S (2)	11/20/13	Shallow	Sampling	4338.55 - 4318.55
B/W-60S	1/8/11	Shallow	Sampling	4342.73 - 4322.73
B/W-61S	8/27/10	Shallow	Sampling	4342.05 - 4322.05
B/W-62S	11/22/10	Shallow	Sampling	4333.94 - 4313.94
B/W-63S (3)	8/9/15	Shallow	Sampling	4325.73 - 4305.73
B/W-64S	12/6/10	Shallow	Sampling	4348.03 - 4328.03
B/W-65S	9/29/10	Shallow	Sampling	4325.29 - 4305.29
B/W-66S	12/5/10	Shallow	Sampling	4313.88 - 4293.88
B/W-67S	1/23/11	Shallow	Sampling	4329.26 - 4309.26
B/W-68S (2)	4/30/14	Shallow	Sampling	4325.57 - 4305.57
B/W-69S (2)	4/15/14	Shallow	Sampling	4319.18 - 4299.18

Table 3-3. Well Construction Details for the Active Groundwater Monitoring Network

Well Name (1)	Completion Date	Groundwater Zone	Well Type	Well Screen Interval
				feet amsl
B/W-70S	10/20/11	Shallow	Sampling	4338.80 - 4318.80
B/W-71S	10/12/11	Shallow	Sampling	4342.25 - 4322.25
B/W-73S	9/10/11	Shallow	Sampling	4357.74 - 4337.74
B/W-74S	10/26/11	Shallow	Sampling	4342.98 - 4322.98
B/W-75S	12/20/11	Shallow	Sampling	4346.69 - 4326.69
B/W-76S	12/15/11	Shallow	Sampling	4335.33 - 4315.33
B/W-77S (2)	4/24/14	Shallow	Sampling	4320.30 - 4300.30
B/W-78S (2)	4/23/14	Shallow	Sampling	4329.30 - 4309.30
B/W-79S (2)	4/25/14	Shallow	Sampling	4335.29 - 4315.29
B/W-81S (2)	3/10/14	Shallow	Sampling	4308.10 - 4288.10
B/W-82RS (2)	11/3/13	Shallow	Sampling	4310.40 - 4290.40
B/W-83S (2)	2/5/14	Shallow	Sampling	4326.66 - 4306.66
D4BC-1S	10/1/85	Shallow	Sampling	4333.98 - 4313.98
D5AC-1S	5/6/84	Shallow	Sampling	4332.48 - 4327.48
FMS-05S (4)	10/20/13	Shallow	Sampling	4335.34 - 4315.34
FMS-06S (4)	11/6/13	Shallow	Sampling	4336.55 - 4316.55
FMS-07S (4)	11/14/13	Shallow	Sampling	4337.75 - 4317.75
HLP-03S (4)	11/16/13	Shallow	Sampling	4341.79 - 4321.79
HLP-04S (4)	10/8/13	Shallow	Sampling	4340.55 - 4320.55
HLP-08S (4)	10/21/13	Shallow	Sampling	4331.83 - 4311.83
LC-MW-1S (5)	1/18/06	Shallow	Sampling	NR - 4303.80(6)
LC-MW-2S (5)	1/18/06	Shallow	Sampling	NR - 4313.90(6)
LC-MW-3S (5)	1/18/06	Shallow	Sampling	NR - 4323.70(6)
LC-MW-5S (5)	NR	Shallow	Sampling	NR - 4323.10(6)
LEP-MW-1S	2/26/09	Shallow	Sampling	4330.92 - 4320.92
LEP-MW-2S	2/27/09	Shallow	Sampling	4331.46 - 4321.46
LEP-MW-3S	2/28/09	Shallow	Sampling	4333.75 - 4323.75
LEP-MW-5S	3/2/09	Shallow	Sampling	4336.35 - 4326.35
LEP-MW-6S	3/2/09	Shallow	Sampling	4327.51 - 4317.51
LEP-MW-7S	3/3/09	Shallow	Sampling	4342.81 - 4332.81
MW2002-2S	6/14/02	Shallow	Sampling	4323.78 - 4313.78
MW-2S	12/13/92	Shallow	Sampling	4326.61 - 4311.61
MW-4S	12/10/92	Shallow	Sampling	4325.68 - 4310.68
MW-5S	10/20/95	Shallow	Sampling	4330.79 - 4315.79
MW-SXN	7/26/09	Shallow	Sampling	4355.39 - 4335.39
MW-SXS	8/28/09	Shallow	Sampling	4354.32 - 4334.32
PA-MW-1S	1/20/05	Shallow	Sampling	4347.32 - 4327.32

Table 3-3. Well Construction Details for the Active Groundwater Monitoring Network

Well Name [(1)]	Completion Date	Groundwater Zone	Well Type	Well Screen Interval
				feet amsl
PA-MW-2S	1/21/05	Shallow	Sampling	4347.37 - 4327.37
PA-MW-3S1	1/19/05	Shallow	Sampling	4348.13 - 4328.13
PA-MW-3S2	11/19/11	Shallow	Sampling	4309.85 - 4299.85
PA-MW-4S	10/18/11	Shallow	Sampling	4348.09 - 4328.09
PA-MW-5S1	11/17/11	Shallow	Sampling	4344.01 - 4324.01
PA-MW-5S2	11/14/11	Shallow	Sampling	4311.16 - 4301.16
PA-MW-7S	10/25/11	Shallow	Sampling	4317.46 - 4297.46
PLMW-2S	8/3/11	Shallow	Sampling	4369.05 - 4349.05
PLMW-4S	10/31/11	Shallow	Sampling	4319.72 - 4289.72
PW10-P1	9/27/05	Shallow	Water Level	4339.10 - 4319.10
USGS-13S	6/10/76	Shallow	Sampling	4342.06 - 4332.06
USGS-2BS	6/8/76	Shallow	Sampling	4326.34 - 4324.44
UW-1S	10/1/85	Shallow	Sampling	4333.32 - 4313.32
W5AA-2S	10/26/83	Shallow	Water Level	4333.65 - 4313.65
W5AA-3S	10/24/98	Shallow	Sampling	4342.86 - 4332.86
W5AB-2S	10/1/83	Shallow	Sampling	4337.68 - 4322.68
W5AD-1S	5/2/82	Shallow	Water Level	4330.91 - 4325.91
W5BB-S	10/23/83	Shallow	Sampling	4337.12 - 4307.12
W5DB-S	10/9/10	Shallow	Sampling	4345.06 - 4325.06
WRP-1S	6/19/07	Shallow	Water Level	4382.53 - 4372.53
WRP-2S	6/19/07	Shallow	Water Level	4382.29 - 4372.29
YPT-MW-6S	1/11/02	Shallow	Sampling	4320.21 - 4315.21
YPT-MW-8S	1/9/02	Shallow	Sampling	4322.26 - 4317.26
YPT-MW-11S	1/11/02	Shallow	Sampling	4317.43 - 4312.43
Intermediate Zone Monitor Wells				
B/W-2I	10/17/07	Intermediate	Sampling	4279.78 - 4259.78
B/W-3I	9/27/07	Intermediate	Sampling	4266.40 - 4246.40
B/W-4I	1/21/08	Intermediate	Sampling	4276.50 - 4256.50
B/W-5RI	11/16/07	Intermediate	Sampling	4278.65 - 4258.65
B/W-6I	9/26/05	Intermediate	Sampling	4259.84 - 4249.84
B/W-7I	8/14/05	Intermediate	Sampling	4304.69 - 4284.69
B/W-8I	8/21/05	Intermediate	Sampling	4284.16 - 4264.16
B/W-9I	11/7/07	Intermediate	Sampling	4281.19 - 4261.19
B/W-19I	1/11/08	Intermediate	Sampling	4281.40 - 4261.40
B/W-27I	8/17/10	Intermediate	Sampling	4274.77 - 4254.77
B/W-28I	1/17/08	Intermediate	Sampling	4277.23 - 4257.23
B/W-29I	12/19/07	Intermediate	Sampling	4288.07 - 4278.07

Table 3-3. Well Construction Details for the Active Groundwater Monitoring Network

Well Name (1)	Completion Date	Groundwater Zone	Well Type	Well Screen Interval
				feet amsl
B/W-30I	10/25/10	Intermediate	Sampling	4267.63 - 4247.63
B/W-31I	12/7/10	Intermediate	Sampling	4287.82 - 4246.82
B/W-32I	1/10/11	Intermediate	Sampling	4286.67 - 4266.67
B/W-33I	8/3/10	Intermediate	Sampling	4265.02 - 4255.02
B/W-34I	12/5/10	Intermediate	Sampling	4303.85 - 4283.85
B/W-37I	8/10/10	Intermediate	Sampling	4296.85 - 4276.85
B/W-38RI	10/9/10	Intermediate	Sampling	4287.91 - 4267.91
B/W-41I	2/7/11	Intermediate	Sampling	4278.31 - 4268.31
B/W-42I	11/8/10	Intermediate	Sampling	4266.21 - 4246.21
B/W-46I	11/7/10	Intermediate	Sampling	4276.66 - 4256.66
B/W-51I	9/9/10	Intermediate	Sampling	4264.06 - 4244.06
B/W-52I	8/20/10	Intermediate	Sampling	4296.06 - 4276.06
B/W-54I	8/21/10	Intermediate	Sampling	4277.24 - 4267.24
B/W-57I	3/14/12	Intermediate	Sampling	4270.47 - 4250.47
B/W-63I (3)	8/9/15	Intermediate	Sampling	4285.58 - 4265.58
B/W-65I	9/29/10	Intermediate	Sampling	4285.30 - 4265.30
B/W-66I	12/5/10	Intermediate	Sampling	4268.85 - 4248.85
B/W-67I	1/22/11	Intermediate	Sampling	4289.41 - 4269.41
B/W-70I	10/22/11	Intermediate	Sampling	4288.59 - 4268.59
B/W-71I	10/11/11	Intermediate	Sampling	4281.11 - 4261.11
B/W-74I1	10/20/11	Intermediate	Sampling	4307.69 - 4297.69
B/W-74I2	10/21/11	Intermediate	Sampling	4277.50 - 4257.50
B/W-76I	12/13/11	Intermediate	Sampling	4276.82 - 4256.82
B/W-82RI (2)	11/2/13	Intermediate	Sampling	4280.35 - 4270.35
HLP-03I (4)	11/5/13	Intermediate	Sampling	4300.00 - 4280.00
HLP-08I (4)	10/20/13	Intermediate	Sampling	4296.56 - 4276.56
LEP-MW-4I	3/1/09	Intermediate	Sampling	4266.95 - 4256.95
LEP-MW-8I	3/4/09	Intermediate	Sampling	4271.83 - 4261.83
LEP-MW-9I	3/6/09	Intermediate	Sampling	4258.17 - 4248.17
MW-4I	8/30/10	Intermediate	Sampling	4285.18 - 4265.18
MW-5I	1/23/11	Intermediate	Sampling	4269.38 - 4249.38
PA-MW-2I	9/8/11	Intermediate	Sampling	4296.59 - 4276.59
PA-MW-3I	11/18/11	Intermediate	Sampling	4281.86 - 4271.86
PA-MW-4I	10/17/11	Intermediate	Sampling	4273.89 - 4253.89
W4CB-1I	10/27/83	Intermediate	Sampling	4280.31 - 4265.31
W4CB-2I	10/28/83	Intermediate	Sampling	4307.74 - 4295.74
W5AA-1I	10/26/83	Intermediate	Sampling	4293.56 - 4278.56

Table 3-3. Well Construction Details for the Active Groundwater Monitoring Network

Well Name (1)	Completion Date	Groundwater Zone	Well Type	Well Screen Interval
				feet amsl
W5AB-3I	9/19/97	Intermediate	Sampling	4308.70 - 4284.20
W5DB-I	10/10/10	Intermediate	Sampling	4287.77 - 4267.77
YPT-MW-9I	1/8/02	Intermediate	Sampling	4282.60 - 4272.60
YPT-MW-12I	1/10/02	Intermediate	Sampling	4280.36 - 4270.36
YPT-MW-13I	7/20/04	Intermediate	Sampling	4287.78 - 4262.78
YPT-MW-15I	10/5/12	Intermediate	Sampling	4275.21 - 4270.21
Deep Zone Monitor Wells				
B/W-1D1	11/5/07	Deep	Sampling	4229.76 - 4209.76
B/W-1D2	10/22/07	Deep	Sampling	4139.92 - 4119.92
B/W-1D3	11/5/05	Deep	Sampling	4028.63 - 4018.63
B/W-1D5	1/7/11	Deep	Sampling	3877.18 - 3867.18
B/W-2D1	9/10/05	Deep	Sampling	4224.01 - 4204.01
B/W-2D3	1/23/11	Deep	Sampling	4049.28 - 4029.28
B/W-2D4	1/21/11	Deep	Sampling	3938.99 - 3918.99
B/W-3D1	8/31/05	Deep	Sampling	4221.87 - 4201.87
B/W-4D1	8/26/05	Deep	Sampling	4228.07 - 4208.07
B/W-5RD1	11/16/07	Deep	Sampling	4241.21 - 4221.21
B/W-9D2	9/14/05	Deep	Sampling	4206.72 - 4186.72
B/W-10D1	8/5/05	Deep	Sampling	4241.10 - 4221.10
B/W-11D2	9/28/05	Deep	Sampling	4197.64 - 4177.64
B/W-18D1	2/19/08	Deep	Sampling	4232.79 - 4212.79
B/W-18D2	12/15/07	Deep	Sampling	4194.17 - 4174.17
B/W-19D1	6/14/07	Deep	Sampling	4216.51 - 4196.51
B/W-25D1	2/1/08	Deep	Sampling	4249.71 - 4229.71
B/W-25D2	1/19/08	Deep	Sampling	4133.82 - 4113.82
B/W-27D2	2/6/08	Deep	Sampling	4124.99 - 4104.99
B/W-27D3	1/6/11	Deep	Sampling	4022.95 - 4002.95
B/W-27D4(2)	2/21/14	Deep	Sampling	3944.83 - 3924.83
B/W-27D5(2)	2/11/14	Deep	Sampling	3879.66 - 3859.66
B/W-28D1	6/28/07	Deep	Sampling	4221.83 - 4201.83
B/W-29D1	12/16/07	Deep	Sampling	4225.24 - 4215.24
B/W-29D3	9/25/07	Deep	Sampling	4050.12 - 4030.12
B/W-30D1	10/26/10	Deep	Sampling	4228.86 - 4208.86
B/W-31D2	11/7/10	Deep	Sampling	4199.84 - 4179.84
B/W-32D2	1/9/11	Deep	Sampling	4147.42 - 4127.42
B/W-32D5	10/24/10	Deep	Sampling	3886.73 - 3866.73
B/W-33D1	7/29/10	Deep	Sampling	4239.39 - 4229.39

Table 3-3. Well Construction Details for the Active Groundwater Monitoring Network

Well Name (1)	Completion Date	Groundwater Zone	Well Type	Well Screen Interval
				feet amsl
B/W-34D1	12/4/10	Deep	Sampling	4257.96 - 4237.96
B/W-37D1	6/5/10	Deep	Sampling	4218.80 - 4198.80
B/W-38RD1	10/10/10	Deep	Sampling	4210.93 - 4190.93
B/W-40D1	1/20/11	Deep	Sampling	4222.20 - 4202.20
B/W-40D3	11/3/10	Deep	Sampling	4057.58 - 4037.58
B/W-41D2	2/7/11	Deep	Sampling	4198.22 - 4178.22
B/W-41D4	2/5/11	Deep	Sampling	4004.14 - 3984.14
B/W-42D1	10/25/10	Deep	Sampling	4210.91 - 4190.91
B/W-44D1	9/23/10	Deep	Sampling	4229.65 - 4209.65
B/W-44D2	9/22/10	Deep	Sampling	4152.72 - 4132.72
B/W-45D1	1/18/11	Deep	Sampling	4252.78 - 4232.78
B/W-45D2	11/20/10	Deep	Sampling	4209.84 - 4189.84
B/W-46D1	11/6/10	Deep	Sampling	4219.76 - 4199.76
B/W-50D1(2)	2/10/14	Deep	Sampling	4206.81 - 4186.81
B/W-50D2(2)	2/8/14	Deep	Sampling	4125.75 - 4105.75
B/W-50D3(2)	2/5/14	Deep	Sampling	4024.73 - 4014.73
B/W-52D2	8/17/10	Deep	Sampling	4177.59 - 4157.59
B/W-55D1	10/20/10	Deep	Sampling	4251.44 - 4241.44
B/W-55D2	10/13/10	Deep	Sampling	4171.59 - 4151.59
B/W-57D1	3/14/12	Deep	Sampling	4212.37 - 4192.37
B/W-57D4	3/13/12	Deep	Sampling	3940.67 - 3920.67
B/W-58D1	3/16/12	Deep	Sampling	4234.41 - 4214.41
B/W-58D3	3/25/12	Deep	Sampling	4054.51 - 4044.51
B/W-59D3(2)	11/19/13	Deep	Sampling	4126.65 - 4106.65
B/W-60D1	12/17/10	Deep	Sampling	4247.69 - 4227.69
B/W-60D3	12/16/10	Deep	Sampling	4036.75 - 4016.75
B/W-60D5	12/7/10	Deep	Sampling	3881.82 - 3861.82
B/W-61D1	8/23/10	Deep	Sampling	4247.00 - 4227.00
B/W-61D3	8/29/10	Deep	Sampling	4036.94 - 4016.94
B/W-62D1	11/21/10	Deep	Sampling	4243.89 - 4223.89
B/W-62D2	11/20/10	Deep	Sampling	4173.88 - 4153.88
B/W-62D4	11/19/10	Deep	Sampling	3953.94 - 3933.94
B/W-62D5	1/6/11	Deep	Sampling	3833.92 - 3813.92
B/W-63D1(3)	8/8/15	Deep	Sampling	4240.50 - 4220.50
B/W-63D2(3)	8/7/15	Deep	Sampling	4170.83 - 4150.83
B/W-63D3(3)	8/5/15	Deep	Sampling	4015.78 - 3995.78
B/W-63D5(3)	7/29/15	Deep	Sampling	3900.65 - 3880.65

Table 3-3. Well Construction Details for the Active Groundwater Monitoring Network

Well Name (1)	Completion Date	Groundwater Zone	Well Type	Well Screen Interval
				feet amsl
B/W-64D1	12/5/10	Deep	Sampling	4260.09 - 4240.09
B/W-64D2	12/3/10	Deep	Sampling	4175.77 - 4155.77
B/W-65D1	9/27/10	Deep	Sampling	4213.36 - 4193.36
B/W-65D5	9/23/10	Deep	Sampling	3750.51 - 3740.51
B/W-66D1	12/4/10	Deep	Sampling	4208.81 - 4188.81
B/W-66D5	12/2/10	Deep	Sampling	3761.03 - 3751.03
B/W-67D1	1/21/11	Deep	Sampling	4245.24 - 4225.24
B/W-67D3	1/13/11	Deep	Sampling	4125.04 - 4105.04
B/W-68D1(2)	4/29/14	Deep	Sampling	4240.74 - 4220.74
B/W-68D4(2)	4/28/14	Deep	Sampling	3964.32 - 3954.32
B/W-69D1(2)	4/14/14	Deep	Sampling	4259.33 - 4239.33
B/W-69D2(2)	4/13/14	Deep	Sampling	4194.30 - 4174.30
B/W-69D5(2)	4/9/14	Deep	Sampling	3782.33 - 3772.33
B/W-70D2	10/25/11	Deep	Sampling	4143.64 - 4123.64
B/W-71D1	10/5/11	Deep	Sampling	4222.09 - 4202.09
B/W-71D3	10/3/11	Deep	Sampling	4094.16 - 4074.16
B/W-74D1	11/20/11	Deep	Sampling	4247.72 - 4227.72
B/W-76D1	10/4/11	Deep	Sampling	4251.74 - 4231.74
B/W-81D1(2)	3/9/14	Deep	Sampling	4243.06 - 4223.06
B/W-81D2(2)	3/10/14	Deep	Sampling	4153.13 - 4133.13
B/W-83D1(2)	2/4/14	Deep	Sampling	4216.67 - 4196.67
B/W-83D3(2)	1/29/14	Deep	Sampling	4066.59 - 4046.59
HLP-08D1(4)	10/19/13	Deep	Sampling	4249.87 - 4229.87
HLP-08D2(4)	10/15/13	Deep	Sampling	4174.99 - 4154.99
LEP-MW-2D1	10/25/10	Deep	Sampling	4229.98 - 4209.98
LEP-MW-2D3	10/22/10	Deep	Sampling	4100.11 - 4080.11
MW-5D2	1/12/11	Deep	Sampling	4194.22 - 4174.22
MW-5D3	1/18/11	Deep	Sampling	4119.72 - 4099.72
MW2002-2D1	7/12/07	Deep	Sampling	4249.75 - 4239.75
PA-MW-4D2	10/15/11	Deep	Sampling	4192.92 - 4172.92
W32DC-D1	10/25/83	Deep	Sampling	4240.41 - 4197.41
W4CB-2D1	9/15/10	Deep	Sampling	4240.56 - 4220.56
W4CB-2D3	9/14/10	Deep	Sampling	4065.76 - 4045.76
W4CB-2D4	11/8/10	Deep	Sampling	3965.54 - 3955.54
W5DB-D1	10/19/98	Deep	Sampling	4239.49 - 4211.49
W5DB-D3	11/17/10	Deep	Sampling	4091.93 - 4071.93
W5DB-D4	10/25/10	Deep	Sampling	4009.93 - 3989.93

Table 3-3. Well Construction Details for the Active Groundwater Monitoring Network

Well Name [(1)]	Completion Date	Groundwater Zone	Well Type	Well Screen Interval
				feet amsl
YPT-MW-14D1	7/21/04	Deep	Sampling	4255.83 - 4235.83
Bedrock Monitor Wells				
B/W-1B	5/19/10	Bedrock	Sampling	3700.10 - 3690.10
B/W-2B	1/12/11	Bedrock	Sampling	3839.17 - 3819.17
B/W-6B	1/25/11	Bedrock	Sampling	4172.04 - 4152.04
B/W-11B	11/3/07	Bedrock	Sampling	4132.88 - 4122.88
B/W-12RB	12/6/11	Bedrock	Sampling	4382.05 - 4302.05
B/W-17B	10/30/11	Bedrock	Sampling	4385.06 - 4365.06
B/W-22B	5/8/10	Bedrock	Sampling	4261.26 - 4241.26
B/W-23B	8/8/07	Bedrock	Sampling	4340.26 - 4330.26
B/W-26RB	11/3/11	Bedrock	Sampling	4367.92 - 4347.92
B/W-27B (2)	11/12/13	Bedrock	Sampling	3800.16 - 3780.16
B/W-33B	7/26/10	Bedrock	Sampling	4167.48 - 4157.48
B/W-34B	12/1/10	Bedrock	Sampling	4203.76 - 4183.76
B/W-36B	6/17/10	Bedrock	Sampling	4271.60 - 4261.60
B/W-37B	5/27/10	Bedrock	Sampling	4166.84 - 4146.84
B/W-38RB	10/8/10	Bedrock	Sampling	4166.90 - 4146.90
B/W-39B	10/7/10	Bedrock	Sampling	4309.10 - 4299.10
B/W-44B	9/16/10	Bedrock	Sampling	4124.48 - 4104.48
B/W-51B	6/25/10	Bedrock	Sampling	4198.79 - 4188.79
B/W-53B	12/17/10	Bedrock	Sampling	4240.77 - 4220.77
B/W-54B	7/10/10	Bedrock	Sampling	4261.30 - 4251.30
B/W-58B	2/4/12	Bedrock	Sampling	4014.50 - 3994.50
B/W-61B	7/9/10	Bedrock	Sampling	3684.05 - 3664.05
B/W-62B	9/25/10	Bedrock	Sampling	3690.87 - 3670.87
B/W-64B	12/2/10	Bedrock	Sampling	4089.75 - 4069.75
B/W-70B	8/30/11	Bedrock	Sampling	4060.86 - 4040.86
B/W-71B	9/1/11	Bedrock	Sampling	3931.06 - 3911.06
B/W-73B	9/7/11	Bedrock	Sampling	4307.60 - 4287.60
B/W-74B	9/21/11	Bedrock	Sampling	4207.18 - 4187.18
B/W-75B	1/7/12	Bedrock	Sampling	4266.82 - 4246.82
B/W-82RB (2)	11/1/13	Bedrock	Sampling	4235.38 - 4215.38
B/W-83B (2)	1/24/14	Bedrock	Sampling	3943.51 - 3913.51
HLP-01B (4)	9/20/13	Bedrock	Sampling	4333.97 - 4313.97
HLP-02B (4)	9/22/13	Bedrock	Sampling	4406.47 - 4386.27
HLP-03B (4)	10/18/13	Bedrock	Sampling	4236.98 - 4206.98
HLP-05B (4)	10/5/13	Bedrock	Sampling	4346.26 - 4306.26

Table 3-3. Well Construction Details for the Active Groundwater Monitoring Network

Well Name (1)	Completion Date	Groundwater Zone	Well Type	Well Screen Interval
				feet amsl
HLP-06B (4)	10/1/13	Bedrock	Sampling	4338.55 - 4318.55
HLP-07B (4)	9/24/13	Bedrock	Sampling	4345.04 - 4325.04
HLP-08B (4)	10/8/14	Bedrock	Sampling	4117.44 - 4097.44
LEP-MW-2B	10/13/10	Bedrock	Sampling	4040.47 - 4020.47
MMW-2	12/6/92	Bedrock	Sampling	4246.34 - 4186.34
MW-4B	8/28/10	Bedrock	Sampling	4251.41 - 4231.41
MW-5B	1/7/11	Bedrock	Sampling	3984.29 - 3964.29
MW-H12	8/6/09	Bedrock	Sampling	4353.58 - 4323.58
MW-H4SN	8/11/09	Bedrock	Sampling	4371.56 - 4341.56
MW-H4SS	8/13/09	Bedrock	Sampling	4360.63 - 4330.63
PA-MW-1B	8/6/11	Bedrock	Sampling	4290.87 - 4270.87
PA-MW-2B	9/3/11	Bedrock	Sampling	4210.44 - 4190.44
PA-MW-3B	10/11/11	Bedrock	Sampling	4246.82 - 4226.82
PA-MW-4B	9/14/11	Bedrock	Sampling	4157.96 - 4137.96
PA-MW-5B	8/20/11	Bedrock	Sampling	4281.60 - 4261.60
PA-MW-7B	9/30/11	Bedrock	Sampling	4174.49 - 4154.49
PLMW-1B	9/29/11	Bedrock	Sampling	4218.23 - 4168.23
PLMW-2B	8/2/11	Bedrock	Sampling	4313.20 - 4293.20
PLMW-3RB	11/12/11	Bedrock	Sampling	4237.72 - 4197.72
PLMW-4B	10/20/11	Bedrock	Sampling	4094.72 - 4064.72
PLMW-5B	9/18/11	Bedrock	Sampling	4243.58 - 4203.58
W4CB-2B	7/9/10	Bedrock	Sampling	3844.55 - 3824.55
W5DB-B	9/26/10	Bedrock	Sampling	3781.04 - 3761.04
WRA3-1B	10/1/11	Bedrock	Sampling	4369.32 - 4339.32
WRA3-2B	10/19/11	Bedrock	Sampling	4322.60 - 4302.60
WRA3-3B	12/5/11	Bedrock	Sampling	4330.39 - 4310.39
WW-1B	NR	Bedrock	Sampling	4364.42 - 4344.42
WW-2B	NR	Bedrock	Sampling	4342.48 - 4322.48
WW-36B	4/15/69	Bedrock	Sampling	4305.78 - 4105.78
WW-40B	NR	Bedrock	Sampling	NR - NR
WW-59B	11/20/72	Bedrock	Sampling	4280.04 - 3888.04
YPT-MW-10B	1/7/02	Bedrock	Sampling	4107.46 - 4097.46

Notes:

1. The names of 117 wells for which the sampling frequency has been reduced from quarterly to semi-annually are bold and italicized. Well names for older wells reflect revisions based on their alluvial aquifer zone designations.
2. New well added to the monitor well network and first sampled in 3Q 2014.
3. New well added to the monitor well network and first sampled in 4Q 2015.
4. New well added to the monitor well network and first sampled in 4Q 2014.
5. Lyon County well.

6. The bottom of screen elevations for the Lyon County wells are based on a 2009 survey conducted for ARC and the total depth of each well measured in the field by BC. The measured well depths are not consistent with the information on the well logs provided by Lyon County (see Appendix C-1 for the well logs).
7. NR = not recorded on well construction logs. amsl = above mean sea level

Water Level Monitoring

Routine water level monitoring was initiated in 2006 to evaluate seasonal and temporal changes in groundwater flow directions and hydraulic gradients and aquifer responses to irrigation practices. Water level elevation monitoring was historically conducted monthly, and subsequently reduced to quarterly in 2014 for wells installed before 2013, in accordance with the *Technical Memorandum: Site-Wide Groundwater Monitoring Optimization* (ARC, 2014).

As outlined in the GMP (BC, 2012a), water levels are measured within a three-day (or shorter) period, for representative aquifer conditions throughout the monitoring network. Water level elevations are also measured electronically using pressure transducers/data loggers, at four-hour intervals at select monitor wells, and at one-hour intervals at the Pit Lake. Water level data from transducers are typically downloaded in conjunction with monthly water level measurements. Appendix D provides groundwater level data, hydrographs, and vertical gradient information.

Groundwater Quality Monitoring

Monitor wells comprising the active monitoring network are sampled on a quarterly or semi-annual frequency pursuant to the GMP (BC, 2012a) using EPA-approved low-flow, minimal drawdown purging and sampling procedures, where applicable. Groundwater samples are analyzed for the constituents listed in Table 3-4 pursuant to the data requirements presented in the QAPP (ESI and BC, 2009).

Table 3-4. Analyte List for Active Monitor Well and Surface Water Sampling

Parameter or Analyte	Total/ Dissolved (1)	Method (2)	Reporting Limit (2)	Units
Physical Parameters and Major Anions/Cations				
Alkalinity, Bicarbonate (as $CaCO_3$)	Total	SM 2320B	2.0	mg/L
Alkalinity, Carbonate (as $CaCO_3$)	Total	SM 2320B	2.0	mg/L
Alkalinity, Total (as $CaCO_3$)	Total	SM 2320B	2.0	mg/L
Chloride	Total	EPA 300.0	0.5	mg/L
Fluoride	Total	EPA 300.0	0.5	mg/L
Nitrate, as N	Total	EPA 300.0	0.1	mg/L
Nitrate/Nitrite (NO_3 + NO_2 as N)	Total	EPA 300.0	0.1	mg/L
Nitrite, as N	Total	EPA 300.0	0.1	mg/L
Sulfate	Total	EPA 300.0	0.5	mg/L
pH (Lab)	Total	SM 4500B	0.1	s.u.
Total Dissolved Solids (TDS) (3)	Total (Lab Filtered) (3)	SM 2540C	10	mg/L
Total Organic Carbon (TOC)	Total	SM 5310B	1.0	mg/L
Metals				
Aluminum	Total + Dissolved	EPA 200.7	0.05	mg/L
Antimony	Total + Dissolved	EPA 200.8	2.0	µg/L
Arsenic	Total + Dissolved	EPA 200.8	1.0	µg/L
Barium	Total + Dissolved	EPA 200.8	1.0	µg/L
Beryllium	Total + Dissolved	EPA 200.8	0.5	µg/L
Boron	Total + Dissolved	EPA 200.7	50	µg/L
Cadmium	Total + Dissolved	EPA 200.8	1.0	µg/L
Calcium	Total + Dissolved	EPA 200.7	0.1	mg/L
Chromium	Total + Dissolved	EPA 200.8	2.0	µg/L
Cobalt	Total + Dissolved	EPA 200.8	1.0	µg/L
Copper	Total + Dissolved	EPA 200.8	1.0	µg/L
Iron	Total + Dissolved	EPA 200.7	0.04	mg/L
Lead	Total + Dissolved	EPA 200.8	1.0	µg/L
Lithium	Total + Dissolved	EPA 200.7	50	µg/L
Magnesium	Total + Dissolved	EPA 200.7	0.02	mg/L
Manganese	Total + Dissolved	EPA 200.8	1.0	µg/L
Mercury	Total + Dissolved	EPA 245.1	0.2	µg/L

Table 3-4. Analyte List for Active Monitor Well and Surface Water Sampling

Parameter or Analyte	Total/ Dissolved (1)	Method (2)	Reporting Limit (2)	Units
Molybdenum	Total + Dissolved	EPA 200.8	2.0	µg/L
Nickel	Total + Dissolved	EPA 200.8	2.0	µg/L
Phosphorus	Total + Dissolved	EPA 200.7	0.04	mg/L
Potassium	Total + Dissolved	EPA 200.7	0.5	mg/L
Selenium	Total + Dissolved	EPA 200.8	0.6	µg/L
Silica	Total + Dissolved	EPA 200.7	0.05	mg/L
Silver	Total + Dissolved	EPA 200.8	1.0	µg/L
Sodium	Total + Dissolved	EPA 200.7	0.5	mg/L
Strontium	Total + Dissolved	EPA 200.7	0.02	mg/L
Thallium	Total + Dissolved	EPA 200.8	1.0	µg/L
Tin	Total + Dissolved	EPA 200.7	100	µg/L
Titanium	Total + Dissolved	EPA 200.7	0.005	mg/L
Uranium, Total	Total + Dissolved	EPA 200.8	1.0	µg/L
Vanadium	Total + Dissolved	EPA 200.8	2.0	µg/L
Zinc	Total + Dissolved	EPA 200.8	10	µg/L
Radiochemicals (5)				
Gross Alpha	Dissolved	EPA 900.0	1.0	pCi/L
Gross Beta	Dissolved	EPA 900.0	1.0	pCi/L
Radium-226	Dissolved	EPA 903.0	1.0	pCi/L
Radium-228	Dissolved	EPA 904.0	1.0	pCi/L
Thorium-228	Dissolved	HASL 300	1.0	pCi/L
Thorium-230	Dissolved	HASL 300	1.0	pCi/L

Notes:

1. Dissolved constituents are field-filtered with a new disposable 0.45-micron (µm) filter. Dissolved (filtered) metals collected quarterly. Total (unfiltered) metals collected in two non-consecutive quarters once a well is installed and/or initially included in the GMP (BC, 2012a).
2. Except for lithium and selenium, EPA laboratory analytical methods and reporting limits are consistent with those provided in Revision 5 of the QAPP (ESI and BC, 2009); alternative analytical methods identified in the QAPP may also be used. For lithium, the lab was unable to get reproducible results using EPA Method 200.8 (as indicated in QAPP); therefore, the lab has used EPA Method 200.7 for lithium, which has a higher reporting limit than indicated in the QAPP. For selenium, the reporting limit of 0.6 µg/L is lower than that indicated in the QAPP (2 µg/L).
3. The samples for TDS are filtered in the analytical laboratory with a new disposable 0.45 µm filter.
4. s.u. = pH standard units; mg/L = milligrams per liter; µg/L = micrograms per liter; pCi/L = picocuries per liter.
5. See Table 3-1 for reduction in radiochemical analysis in 2019.

Appendix D provides groundwater quality data associated with the Site-wide groundwater monitoring program and charts illustrating temporal changes in chemical concentrations.

Dissolved versus Total Metals

Beginning with the 3Q 2010 sampling event, samples from monitor wells at the Site have been periodically collected in two different quarters from each well and analyzed for both dissolved (0.45 µm-filtered) and total (unfiltered) metals to determine whether the two sampling methods produce comparable results. The results of the comparative statistical analysis of the available dissolved and total metals datasets have been periodically reported in previous quarterly and annual groundwater monitoring reports (GMRs). The most recent and final sampling for dissolved and total metals occurred during the 3Q 2014 and 1Q 2015 events for 29 off-Site wells that were installed in 2013/2014 and first sampled in 3Q 2014. The statistical comparison of the complete dissolved and total metals datasets is presented in the 2015 Annual GMR (BC, 2016c).

Based on the statistical analyses, differences between dissolved and total metals concentrations in groundwater samples collected from monitor wells are non-existent or are too small to be meaningful. It is concluded that groundwater sampling both with and without filtering of samples produce equivalent datasets.

3.2.5 Soil Sampling and Testing

Pursuant to the *Agricultural Fields Characterization Work Plan - Revision 2* (BC, 2010d), soil samples were collected from select borehole cores in the Shallow, Intermediate, and Deep Zones. The types of samples collected, and a brief description of the sampling objectives, are provided below.

- Soil samples were collected for analysis of grain size distribution to generate laboratory-determined Unified Soil Classification System (USCS) soil descriptions for comparison to USCS descriptions made in the field at the time of drilling.
- Non-redox preserved soil samples were collected for bulk chemical analyses to characterize chemical concentrations in soils.
- Redox-preserved soil samples were collected using EPA-specified procedures that preserve the subsurface oxidation state of the sediments and archived at the Site for

potential testing pursuant to the *Aquifer Solids Testing Work Plan - Revision 1* (BC, 2010e).

Soil sampling information is provided in Appendix E and discussed briefly below.

Grain Size Analysis

A total of 37 samples of aquifer materials were obtained from 16 borehole locations for grain size distribution analyses by sieving of material larger than 75 µm (i.e., retained on a No. 200 sieve). Laboratory reports with the grain size distribution data are provided in Appendix E-1. Grain size distribution results were used to generate laboratory-derived USCS lithologic classifications for comparison to the field-derived USCS lithologic classifications based on visual inspection of core during drilling. As shown in Table 3-5, field description and laboratory analyses of the dominant soil grain size are generally consistent.

Table 3-5. Sample Locations for Grain Size Analysis

Borehole Name	Sample Interval (feet bgs)	USCS Classification (Field)	Percent Fines (Field)	USCS Classification (Laboratory)	Percent Fines (Laboratory)
B/W-2	378-384	SW	5	SM	12.7
B/W-2	442-454	SC	35	SC	26.4
B/W-32	21-27	SM	15	SM	18.7
B/W-32	411-414	SW-SM	10	SM	13.5
B/W-36	57-71	SM	30	SM	13.5
B/W-37	111-117	CL	80	CL	82.0
B/W-38R	140-143	GW	5	SW-SM	7.8
B/W-38R	208-212	SC	35	SC	25.7
B/W-38R	249-253	SM	30	SM	18.6
B/W-40	220.5-226	SM	15	SM	19.2
B/W-40	451-456	SP	5	SM	20.9
B/W-42	159-165	SM	15	SP-SM	5.9
B/W-51	64-69	SC	35	SC	15.0
B/W-54	46-52	CL	65	CL	63.1
B/W-54	52-61	SW	10	SW-SM	8.3
B/W-55	42-51	CH	95	CL	74.4
B/W-55	118-125	SM	20	CL	51.2
B/W-55	135-145	SW	5	SP	4.9
B/W-55	175-185	SP	5	SW	2.9
B/W-60	32-36	SM	35	SM	23.3
B/W-60	132.5-136	ML	60	CL	70.1
B/W-60	450-456	SM	25	SM	22.6
B/W-61	299-306	SP	10	SP-SM	7.7
B/W-64	27-31	SP	10	SP-SM	12.0
B/W-64	43-47	CL	75	CL	93.0
B/W-64	67-77	SW	10	SP-SM	12.0
B/W-64	177-185	SW	10	SW	5.0
B/W-66	59-64	CH	70	CL	66.1
B/W-66	65.5-68	SC	40	SC	23.0
B/W-66	89-93	SC	20	SC	48.7
B/W-67	27-32	SP	5	SM	28.5
B/W-67	38-55	CL	60	CL	64.0
B/W-67	142-146	SC	40	SC	31.8
LEP-MW-2	61-66	CL	50	SC	22.1
LEP-MW-2	212-217	CH	95	SC	34.4
LEP-MW-2	266-273	SW-SM	10	SM	12.8
LEP-MW-2	341-346	CL	50	SC	36.2

Notes:
SW = Well Graded Sand or Well Graded Sand with Gravel (where gravel is more than 15%).
SP = Poorly Graded Sand or Poorly Graded Sand with Gravel (where gravel is more than 15%).
SW-SM = Well Graded Sand with Silt or Well Graded Sand with Silt and Gravel (where gravel is more than 15%).
SP-SM = Poorly Graded Sand with Silt or Poorly Graded Sand with Silt and Gravel (where gravel is more than 15%).
SC = Clayey Sand or Clayey Sand with Gravel (where gravel is more than 15%).
SM = Silty Sand or Silty Sand with Gravel (where gravel is more than 15%).

CL = Lean Clay, Lean Clay with Sand, Sandy Lean Clay or Sandy Lean Clay with Gravel (where gravel is more than 15%).
CH = Fat Clay, Fat Clay with Sand, Sandy Fat Clay or Sandy Fat Clay with Gravel (where gravel is more than 15%).
GW = Well Graded Gravel with Sand.

Solids Sampling for Bulk Chemistry

Samples of archived core were collected from select depths in boreholes B/W-1, B/W-31, B/W-32, B/W-42, B/W-46, B/W-61, B/W-62, B/W-65, B/W-66, B/W-67, and MW-5 and submitted to the laboratory for bulk chemical analysis of the parameters listed in Table 3-6. Specifically, samples were collected at: 1) multiple intervals in both the unsaturated and saturated portions of the shallow zone; 2) locations where both high and low readings were obtained during EC profiling; and 3) locations immediately above and below the interface between the saturated and unsaturated zones. Sample collection methods conformed to SOP-11 of the QAPP. Concentrations of metal/metalloids (hereinafter referred to as metals) and radiochemicals in the solid soil samples were determined by microwave- assisted digestion using EPA Method 3051A (HNO_3). The locations, sample depths and laboratory results are summarized in Appendix E-2.

Vertical profiling of chemical concentrations in soils beneath the agricultural fields was performed to help understand potential chemical loading to groundwater unrelated to mining. Other evaluations involving characterization of groundwater quality upgradient and downgradient of agricultural fields, and the sulfur isotope signatures associated with gypsum, an agricultural fertilizer/soil amendment, proved more useful for evaluating groundwater impacts associated with agricultural activities (see Section 5.2.2).

Table 3-6. Analyte List for Soil Samples

Parameter or Analyte	Method (1)	Reporting Limit (1)	Units (2)
Soil pH	EPA 9045C	0.1	s.u.
Total and Acid Soluble Sulfur	Method 9030B	0.4	mg/kg
Chloride	EPA 300.0	5	mg/kg
Nitrate (as N)	EPA 300.0	1.1	mg/kg
Sulfate	EPA 300.0 (4)	5.0	mg/kg
TOC, TC, TIC (3)	EPA LG601 (2)	1.0	mg/kg
Aluminum	EPA 6010B	10	mg/kg
Antimony	EPA 6020	1.0	mg/kg
Arsenic	EPA 6020	0.5	mg/kg
Barium	EPA 6020	0.5	mg/kg
Beryllium	EPA 6020	0.3	mg/kg
Boron	EPA 6010B	5.0	mg/kg
Cadmium	EPA 6020	0.5	mg/kg
Calcium	EPA 6010B	15	mg/kg
Chromium	EPA 6020	1.0	mg/kg
Cobalt	EPA 6020	0.5	mg/kg
Copper	EPA 6020	1.0	mg/kg
Iron	EPA 6010B	5.0	mg/kg
Lead	EPA 6020	0.5	mg/kg
Magnesium	EPA 6010B	10	mg/kg
Manganese	EPA 6020	0.5	mg/kg
Molybdenum	EPA 6020	1.0	mg/kg
Nickel	EPA 6020	1.0	mg/kg
Potassium	EPA 6010B	50	mg/kg
Selenium	EPA 6020	1.0	mg/kg
Sodium	EPA 6010B	50	mg/kg
Uranium, Total	EPA 6020	0.10	mg/kg
Vanadium	EPA 6020	1.0	mg/kg
Zinc	EPA 6020	10	mg/kg
Uranium-234, 235, 238	HASL 300 (U-02-RC)	1.0	pCi/g

Notes:

1. EPA laboratory analytical methods and reporting limits are consistent with those provided in QAPP (ESI and BC, 2009); alternative analytical methods identified in the QAPP may also be used.
2. s.u. = standard units; mg/kg = milligrams per kilogram; pCi/g = picocuries per gram.
3. Total Organic Carbon (TOC), Total Carbon (TC), and Total Inorganic Carbon (TIC).
4. EPA Method LG601 (Dry Combustion, Infrared Detection) as described in EPA, 2005.

Redox-Preserved Soil Sampling and Archiving

During the 2007 Second-Step HFA (BC, 2008c) and 2010 field investigation (BC, 2013a), soil samples were collected using EPA-specified procedures that preserved the subsurface oxidation state of the sediments. Redox-preserved soil samples were collected at select borehole locations near the Evaporation Ponds (B/W-11, B/W-18, LEP-MW-9I, MW-5, and W4CB-2), the agricultural fields adjacent to the Site (B/W-61, B/W-65, and B/W-66), and at B/W-32 (i.e., at DPT-28, which was identified during the Shallow Zone investigation in 2009).

The redox-preserved soil samples were archived at the Site for potential laboratory testing (BC, 2010e) to evaluate geochemical processes that affect the release and/or attenuation of chemicals from/onto aquifer solids (in particular, chemical partitioning to various mineral fractions) and the mobility and transport of chemicals in groundwater at the Site. The disposition of archived, redox-preserved soil samples is described in Section 3.3.5, which addresses chemical transport evaluations.

3.2.6 Aquifer Hydraulic Properties Testing

The 2007 SOW (EPA, 2007a) required "Definition of aquifer properties (e.g., hydraulic conductivity, transmissivity, and storativity) by a program of aquifer testing to measure the hydraulic connection between areas and throughout the known extent of contamination." In addition, characterization of aquifer hydraulic properties was identified as DQO #5 in the Revised Groundwater RI Work Plan (BC, 2014a).

Hydraulic properties of the alluvial aquifer, as well as bedrock, are provided in Appendix F and have been estimated based on a variety of small- to large-scale test methods including:

- Slug testing of groundwater monitor wells (Appendix F-1);
- Analysis of steady-state drawdown data obtained during routine quarterly low-flow sampling of monitor wells (Appendix F-2);
- Constant-rate pumping tests of the eleven wells comprising the PWS during 2010 (Appendix F-3);
- Slug testing of piezometers installed near the PWS that were used as observation wells during constant-rate pumping tests of the 11 wells comprising the PWS during 2010 (Appendix F-4); and
- A constant-rate pumping test of agricultural well WDW019 using an observation network of 93 monitor wells, of which 61 exhibited pumping-related responses (Appendix F-5).

Small-scale test methods, such as slug testing, provide data that are useful for identifying spatial patterns related to geology, guiding characterization, and as a preliminary estimate of hydraulic conductivity. Because slug test data are available throughout the Study Area, this dataset is used to evaluate spatial patterns in hydraulic conductivity within the Study Area. Data from the pumping test at WDW019, the other large-scale pumping tests, and subsequent groundwater model development using a parameter estimation technique (Doherty, 2009), have been used to develop representative field-scale estimates of hydraulic conductivity and, therefore, groundwater velocity.

3.2.7 Surface Water Characterization

The hydrology of the study area is dominated by groundwater recharge from surface water associated with agricultural irrigation (Carroll et al., 2010; Huxel and Harris, 1969; Myers, 2001). Therefore, understanding the flows in the Walker River and diversions for nearby agricultural activities is important for understanding and contextualizing the data collected in the Study Area. Both regional and local (i.e. Study Area) characterization activities were conducted.

To characterize regional surface water hydrology, daily streamflows for the Walker River are obtained at several gaging locations throughout the Mason Valley, both upstream and downstream of the Site. The data collected from the gaging stations are maintained by the USGS, often in cooperation with state and local agencies, and are available at the USGS website (http://waterdata.usgs.gov/nv/nwis/sw). Surface water quality in the Walker River is also routinely monitored by various federal, state, and miscellaneous agencies/entities. Much of this data is assembled and made publicly available through the EPA's STOrage and RETrieval (STORET) Data Warehouse. This dataset supplements the data collected by ARC.

To evaluate the quality of surface water used to irrigate the Hunewill Ranch agricultural fields next to the Site, samples were collected and analyzed from the West Campbell Ditch (SW-WCD-01) and the Walker River (SW-WR-01), pursuant to the *Agricultural Fields Characterization Work Plan - Revision 2* (BC, 2010d). Sampling locations are shown on Figure 3-6. West Campbell Ditch receives its water directly from the Walker River. The Walker River monitoring point (SW-WR-01) is located less than 1,000 feet upstream of the diversion point for West Campbell Ditch. The monitoring location in West Campbell Ditch (SW-WCD-01) is located about three miles farther downstream from the diversion point. Potential temporal trends in surface water quality during non-irrigation and irrigation periods were addressed by collecting samples monthly at these locations for 12 months during 2010/2011.

Field parameters (temperature, pH, specific conductivity, DO, sulfate, and turbidity) were measured at the time of sample collection, and samples were submitted for the analysis (total concentrations) of the parameters listed in Table 3-4. Surface water samples were collected using the direct-grab method described in SOP-18 in the QAPP (ESI and BC, 2009). This surface water sample collection method is consistent with the method used by NDEP to collect samples at other surface water monitoring stations in the Mason Valley. Laboratory analyses were conducted in accordance with the QAPP. Surface water data are presented in Appendix G and summarized in Section 4.7.

3.2.8 Hydrologic Tracer Studies

Hydrologic tracer investigations were initiated to help characterize Study Area groundwater conditions, refine the HCSM, and identify background groundwater quality types (BC, 2008c, 2012b, 2014a). A variety of hydrologic tracers were initially identified as having the potential to provide information on the origin, age, sources of dissolved constituents, and migration pathways of groundwater and surface water in the Study Area.

To evaluate the feasibility of using hydrologic tracers to support these objectives, samples were collected prior to 2010 from a select number of groundwater monitor wells and surface water features. Based on the apparent efficacy of using hydrologic tracers to assess Study Area groundwater conditions, additional EPA-approved hydrologic tracer sampling events were conducted. Over time, the hydrologic tracer sampling events evolved with changes primarily related to increases in the number of groundwater monitor wells that were sampled, opportunistic collection of standing rainwater samples, and elimination of select tracers considered less useful for characterizing groundwater conditions. Table 3-7 provides a chronology of the various OU-1 hydrologic tracer sampling events.

Table 3-7. Chronology of Hydrologic Tracer Sampling Events

Date	Sampling Locations[1]	Hydrologic Tracers[2]	Information Source(s)
July/August 2008	Hydrologic tracer samples were collected from 47 of the 94 (50%) active groundwater monitor wells at the time that routine groundwater monitoring was conducted.	Uranium isotopes, sulfate isotopes, tritium, tritium/ helium, $\delta^{18}O/\delta^{2}H$ in water, and nitrate isotopes.	*Second-Step Hydrogeologic Framework Assessment Data Summary Report* (BC, 2008c).
February 2011	Samples were collected from the Walker River and West Campbell Ditch, and from 127 of the 223 (57%) active groundwater monitor wells at the time of sampling.	Uranium isotopes, sulfate isotopes, tritium, tritium/ helium, $\delta^{18}O/\delta^{2}H$ in water, nitrate isotopes, CFCs, $\delta^{13}B$, and $\delta^{36}Cl$.	*Draft Site-Wide Groundwater Operable Unit (OU-1) Remedial Investigation Work Plan* (BC, 2012b). Revised Groundwater RI Work Plan (BC, 2014a).
May 2012	Samples were collected from: 1) three surface water locations (Pit Lake, Walker River and West Campbell Ditch); and 2) 279 of the 287 (97%) active groundwater monitor wells at the time of sampling.	Uranium isotopes, sulfate isotopes, tritium, tritium/ helium, CFCs, and SF_6.	Conclusions about the usefulness of specific hydrologic tracers collected in 2011 were noted in correspondence between ARC and EPA (2012a). With EPA approval (2012b), samples collected during May 2012 were not analyzed for stable isotopes in water or nitrate isotopes. May 2012 results were provided in the *Background Groundwater Quality Assessment - Revision 2* (BC, 2015c).
July 2013	Collection of 14 standing rainwater samples following a large precipitation event.	Uranium isotopes and sulfate isotopes.	July 2013 results provided in the *Background Groundwater Quality Assessment - Revision 2* (BC, 2015c).
August 2014	Groundwater samples were collected from all (100%) of the new wells installed pursuant to the Additional Monitor Well Work Plan (BC, 2013b) except well HLP-02B because it was dry. Also sampled were the four wells at the B/W-65 cluster, which were inaccessible in May 2012, and monitor well YPT-MW-15I, which was installed in October 2012. Five wells were resampled to evaluate the results reported in May 2012.	Uranium isotopes, sulfate isotopes, tritium, and tritium /helium.	August 2014 results provided in the *Background Groundwater Quality Assessment - Revision 2* (BC, 2015c).

Notes:

1. Hydrologic tracer samples from monitor wells were collected in conjunction with routine groundwater monitoring events associated with the GMP (BC, 2012a).
2. Uranium isotopes include ^{234}U, ^{235}U, and ^{238}U; Sulfate isotopes = $\delta^{34}S/\delta^{18}O$ in dissolved sulfate; Nitrate isotopes = $\delta^{15}N/\delta^{18}O$ in dissolved nitrate; $\delta^{13}B$ = boron isotopes in the water samples; $\delta^{36}Cl$ = chloride isotopes in the water samples; CFCs = chlorofluorocarbons; SF_6 = Sulfur Hexafluoride.

Hydrologic tracer samples were collected from monitor wells in conjunction with routine Site-wide groundwater sampling events using low-flow, minimal drawdown sample collection procedures specified in the GMP (BC, 2012a), as well as tracer-specific sampling protocols specified in SOP-17 of the QAPP (ESI and BC, 2009). Surface water hydrologic tracer samples were collected using the direct-grab method described in SOP-18 of the QAPP. This surface water sample collection method is consistent with the method used by the NDEP to collect samples at other surface water monitoring stations in the Mason Valley. Table 3-8 presents the parameters, analytical methods, reporting limits, and accuracy and precision goals for the hydrologic tracer analyses.

Table 3-8. Analyte List for Hydrologic Tracer Samples

Parameter	Analytical Method	Analytical Precision (1)	Reporting Limit (2)	Matrix Spike Accuracy	Lab Control Sample Accuracy
Tritium (^{3}H)	EA-IRMS (^{3}He-ingrowth)	± 0.1 TU (3)	NA	NA	NA
Tritium/Helium (^{3}H/^{3}He)	Noble Gas MS	± 1%	NA	NA	NA
^{34}S in Sulfate	EA-IRMS (Combination to SO_2) USGS RSIL Lab Code 1951	± 0.5‰	NA	NA	NA
^{18}O in Sulfate	EA-IRMS (Combination to CO_2) USGS RSIL Lab Code 1951 (4)	± 0.5‰	NA	NA	NA
^{234}U, ^{235}U, ^{238}U	HASL-300 (U-02-RC) (5)	RPD<20% or RER<2	1 pCi/L	70-130%	75-125%
Total Uranium	EPA 200.8 ICP-MS	20%	0.1 mg/L	70-130%	80-120%
Chlorofluorocarbons (CFCs)	GC-ECD	0-2%	0.001 x 10^{-12} pmol/kg	NA	NA
Sulfur Hexafluoride (SF_6)	GC-ECD	1-3% (6)	0.01 x 10^{-15} fmol/kg	NA	NA

Notes:

1. Precision is the average standard deviation (1-sigma) in per mil units (‰). Precision limit applicable for matrix spike/matrix spike duplicate, laboratory duplicate, laboratory control sample/ laboratory control sample duplicate, or reference standard analyses.
2. The method detection limits presented are laboratory-derived limits.
3. TU = tritium unit; NA = not applicable; RPD = relative percent difference; RER = replicate error ratio; EA-IRMS = elemental analyzer-isotopic ratio mass spectrometer; ICP-MS = inductively coupled plasma mass spectroscopy; GC-ECD = Gas Chromatography with Electron Capture Detection; % = percent
4. USGS Reston Stable Isotope Laboratory (RSIL) Lab Code 1951 (USGS, 2006).
5. Method U-02-RC: see Isotopic Uranium in Biological and Environmental Materials for water samples as documented in HASL-300
6. (Rev. 1, February 2000) available at URL address: http://www.eml.st.dhs.gov/publications/procman/.
7. Wanninkhof et al. (1991); Law et al. (1994).
8. mg/L = milligrams per liter; µg/L = micrograms per liter; pmol/kg = picomoles per kilogram; fmol/kg = femtomole per kilogram; pCi/L = picocuries per liter.

Laboratory analytical results for hydrologic tracers achieved the completeness, accuracy, and precision goals specified in relevant planning documents including the QAPP (ESI and BC, 2009) and SOP-17. Hydrologic tracer information that relates directly to the groundwater recharge aspects of the HCSM and the background groundwater assessment was obtained in May 2012 and August 2014. These data are discussed in Section 5.0. Appendix H provides supplemental information about hydrologic tracer sampling and analysis including:

- A detailed discussion of the locations where hydrologic tracer samples were collected during May 2012, July 2013, and August 2014;
- Analytical results of hydrologic tracer samples of standing rainwater impounded on mine waste features after a large rain event (average of 1.55 inches on-Site) on July 4, 2013;
- A detailed description of the sample collection procedures, analytical methods, laboratory precision goals for each hydrologic tracer, and QA/QC sample results;
- An evaluation and discussion of the usefulness of CFC and SF_6 data for estimating groundwater ages in the Study Area;
- A discussion of the principles and application of uranium isotopes to groundwater interpretation;
- A discussion of the principles of groundwater age estimation using data for tritium and tritium/helium in groundwater;
- A discussion of the additional sources of sulfate isotope data potentially relevant to groundwater conditions in the Study Area; and
- Electronic copies of the analytical results provided by the laboratory and laboratory-calculated apparent groundwater ages.

3.2.9 Bedrock Groundwater Characterization

Since 2005, phased field investigations associated with OU-1 have included characterization of both the alluvial and bedrock groundwater systems. The Revised Groundwater RI Work Plan (BC, 2014a) presented available groundwater information through May 2012 and described an updated HCSM for the bedrock and alluvial groundwater systems. The Revised Groundwater RI Work Plan also outlined the approach for completing the bedrock groundwater study elements specified in the 2007 SOW. Bedrock characterization activities that were approved by EPA are described below in Table 3-9.

Table 3-9. Chronology of Bedrock Groundwater Characterization Activities

2004-2007	Bedrock characterization (including borehole drilling, lithologic logging, well installation, hydraulic testing, water level monitoring, and groundwater quality monitoring) conducted pursuant to the First-Step HFA Work Plan (BC, 2005) and Second-Step HFA Work Plan (BC, 2007b).
2007	Characterization of bedrock groundwater conditions is required under the Administrative Order for Remedial Investigation and Feasibility Study (RI/FS), CERCLA Docket No. 9-2007-0005 dated January 12, 2007.
2007-2011	Bedrock characterization (including borehole drilling, lithologic logging, well installation, hydraulic testing, water level monitoring, and groundwater quality monitoring) conducted pursuant to the *2010 Groundwater Monitor Well Work Plan - Revision 2* (BC, 2010c), *Agricultural Fields Characterization Work Plan - Revision 2* (BC, 2010d), *On-Site Monitor Well Installation Work Plan - Revision 1* (BC, 2011a), and the *Addendum to the On-Site Monitor Well Installation Work Plan* (ARC, 2011).
August 17, 2011	Groundwater technical meeting with EPA, ARC and stakeholders to discuss progress of the phased approach to groundwater RI activities, which resulted in concurrence to conduct initial bedrock characterization activities to support a more comprehensive assessment of bedrock groundwater conditions as part of the RI characterization.
September 29, 2011	EPA (2011a) provided comments on the *2010 Annual Site-Wide Groundwater Monitoring Report* dated April 15, 2011, and the First and Second Quarter (1Q and 2Q) 2011 Groundwater Monitoring Reports dated July 1, 2011 and August 26, 2011, respectively, that pertained to bedrock characterization.
January 5, 2012	ARC submitted the *Proposed Initial Bedrock Characterization Activities* (ARC, 2012a).
April 12, 2012	EPA (2012c) provided comments on the *Proposed Initial Bedrock Characterization Activities* (ARC, 2012a).
June 18, 2012	ARC submitted preliminary responses to EPA comments on the *Proposed Initial Bedrock Characterization Activities.*
August 28, 2012	Groundwater technical meeting with EPA, ARC and other stakeholders to resolve comments on the *Proposed Initial Bedrock Characterization Activities.*
October 11, 2012	Submittal of ARC final responses to EPA comments on the *Proposed Initial Bedrock Characterization Activities,* and submittal of the *Proposed Initial Bedrock Characterization Activities - Revision 1* (ARC, 2012b).
October 22, 2012	EPA (2012d) approval of the *Proposed Initial Bedrock Characterization Activities - Revision 1,* included as Attachment D to the Revised Groundwater RI Work Plan (BC, 2014a).
March 2013	ARC began implementation of the EPA-approved *Proposed Initial Bedrock Characterization Work Plan - Revision 1.*
November 20, 2013	ARC submitted the *Initial Bedrock Characterization Data Summery Report* (BC, 2013c).
February 7, 2014	ARC submitted the *Site-Wide Groundwater Remedial Investigation Work Plan - Revision 1* (BC, 2014a), which presented available groundwater information through May 2012, and described an updated HCSM for the bedrock and alluvial groundwater systems.
January 28, 2015	ARC submitted the *Bedrock Groundwater Assessment Technical Memorandum* (BC, 2015a).
June 11, 2015	ARC submitted the *Additional Monitor Well Installation Data Summary Report* (BC, 2015d) detailing installation and testing of bedrock and alluvial wells installed in 2013 and 2014.
July 31, 2016	EPA (2016a) approved the *Bedrock Groundwater Assessment Technical Memorandum* (BC, 2015a).

After installation and testing of new bedrock monitor wells in late 2013 and 2014, and a technical meeting in May 2015 to discuss the full set of bedrock information, EPA (2015a) concluded that sufficient data had been collected to conclude that bedrock is not an important migration pathway at the Site, and requested preparation of a technical memorandum to update the bedrock HCSM. The *Bedrock Groundwater Assessment Technical Memorandum* (BC, 2015a) is provided in Appendix I and bedrock information is summarized in Section 4.9.7.

3.3 Site-Wide Groundwater Studies and Evaluations

Studies and evaluations relying on OU-1 RI data that were conducted to fulfill certain study elements specified in Section 7.0 of the SOW attached to the 2007 Order (EPA, 2007a) are described below. Reports describing the approach, analysis, and results of these groundwater related studies and evaluations are provided in Appendix J.

3.3.1 Pumpback Well System Effectiveness

The effectiveness of the PWS in limiting the off-Site migration of MIW was evaluated in accordance with the *Pumpback Well System Characterization Work Plan Addendum - Revision 2* (ARC, 2010). The 11 pumpback wells ceased pumping on March 25, 2009 and were subsequently hydraulically tested to generate information to support a capture zone analysis using an analytical element model. These activities provided a preliminary assessment sufficient to conclude that the PWS was only partially effective in capturing Shallow Zone mine-influenced groundwater during its operational life. The PWS effectiveness evaluation is described in the *Summary of PWS Aquifer Testing* (BC, 2010f), which is included as Appendix J-1.

Pumping rates and the quality of groundwater pumped from the PWS for the period 1997 through 1997 through 2008 are documented in the 2008 Annual Report for the PWS (Norwest, 2009) and in the project water-quality data base. Incomplete information is available for the period 1986 through 1996 on pumping rates and water quality, though annual water-quality data are available for PW-4 for this period. Based on the available information, annual pumping rates and sulfate concentrations for the wells in the PWS were compiled or estimated for the period 1986 through 2009. The annual pumping rates for each of the wells are listed on Table 3-10 and the annual average sulfate concentrations are listed on Table 3-11. The total mass of sulfate removed at each of the wells annually was calculated based on the pumping rates and sulfate concentrations and are listed on Table 3-12.

Table 3-10. Average Annual Pumping Rates of PWS Wells (gpm)

Year	Well											
	PW-1	**PW-2**	**PW-3**	**PW-4**	**PW-5**	**PW-6**	**PW-7**	**PW-8**	**PW-9**	**PW-10**	**PW-11**	**Total**
1985												
1986		5.5	3.5	3.4								**12.4**
1987		7.3	4.7	4.6								**16.6**
1988		7.3	4.7	4.6								**16.6**
1989		7.3	4.7	4.6								**16.6**
1990		7.3	4.7	4.6								**16.6**
1991		3.6	2.3	2.3								**8.3**
1992		3.6	2.3	2.3								**8.3**
1993		3.6	2.3	2.3								**8.3**
1994		3.6	2.3	2.3								**8.3**
1995		3.6	2.3	2.3								**8.3**
1996		3.6	2.3	2.3								**8.3**
1997		7.3	4.7	4.6								**16.6**
1998		6.0	5.3									**11.4**
1999		8.5	7.5	7.4	3.1	2.4	1.9	2.6	3.2	1.6	4.8	**43.2**
2000	9.4	8.3	7.4	7.3	3.1	2.4	1.9	2.5	3.2	1.6	4.7	**51.8**
2001	10.1	6.6	5.3	5.1	5.8	2.1	3.0	3.0	3.5	1.0	4.4	**49.8**
2002	11.6	9.2	4.7	6.6	6.8	2.8	2.6	1.0	2.7	0.6	3.1	**51.7**
2003	11.0	8.5	5.4	5.0	5.2	1.8	2.3	0.5	2.8	0.6	1.9	**44.9**
2004	11.0	7.1	4.8	4.6	5.3	1.3	1.3	0.4	2.6	0.7	1.4	**40.7**
2005	13.0	8.5	5.4	5.8	5.4	1.7	1.7	0.3	1.4	1.3	1.3	**45.5**
2006	16.7	10.3	6.8	5.5	5.7	2.4	3.7	0.5	2.1	1.2	1.5	**56.5**
2007	16.5	10.5	5.3	4.9	5.3	2.8	4.5	0.4	1.8	1.0	1.0	**54.0**
2008	14.9	8.8	4.4	3.5	5.6	2.8	2.8	0.3	1.5	0.6	0.6	**45.7**
2009	2.5	1.5	0.7	0.6	0.9	0.5	0.5	0.1	0.3	0.1	0.1	**7.6**
Average	**12.7**	**6.8**	**4.5**	**4.4**	**5.1**	**2.3**	**2.6**	**1.1**	**2.5**	**1.0**	**2.5**	

Note: Shaded pumping rates represent estimated values.

Table 3-11. Average Yearly Sulfate Concentrations in PWS Wells (mg/L)

Year	Well										
	PW-1	PW-2	PW-3	PW-4	PW-5	PW-6	PW-7	PW-8	PW-9	PW-10	PW-11
1985	5500	14400	27500	38500	11500						
1986		13872	26153	23667							
1987		13343	24807	33967							
1988		12815	23460	20600							
1989		12287	22113	20000							
1990		11758	20767	16100							
1991		11230	19420	9550							
1992		10702	18073	6150							
1993		10173	16727	5250							
1994		9645	15380	5100							
1995		9117	14033	8130							
1996		8588	12687	4945							
1997	3940	8060	11340	4465							
1998	3625	7200	10245	3985	5700	1960	4090	1390	4180	8950	7120
1999	3365	6775	9150	5323	2900	2220	4535	3625	4630	10500	6615
2000	2750	5720	8025	5425	4460	2000	4325	2850	4680	9975	5775
2001	3025	5600	8250	4750	3450	1900	4375	3500	4800	10250	5700
2002	3050	5875	7600	4975	2980	1950	4200	3300	4750	10125	5820
2003	3240	6100	6740	4025	3150	1925	5360	3175	5100	9850	5483
2004	4000	6820	8050	4550	3675	2175	6050	2800	5417	9840	6525
2005	3550	5840	7580	4500	3175	1925	5200	2340	5100	9700	6125
2006	3050	4967	5833	4133	2550	1650	4425	2700	4875	10025	5700
2007	2925	4850	5980	4050	3340	1475	3975	3150	4950	9800	6025
2008	2800	5460	6133	3625	2175	1675	4380	2775	5000	9225	6250
2009	2075	5767	5133	2186	1880	1258	4433	4300	4150	7825	5100
2010	2100	3740	4300	2650	870	1250	4600	4575	4000	7525	4860
2011	2283	3400	4350	2820	1148	1300	5340	5200	3675	7375	4680
2012	2840	3125	4617	2520	1125	1078	5040	5575	4125	7380	4575
2013	2050	3467	4367	2625	1100	880	4333	6267	3933	7267	4300
2014	1950	3250	4150	2520	1350	680	3600	6400	3900	6750	5000
2015	1783	3200	3600	2500	1150	815	3700	7500	3550	6550	4550
2016	1875	2850	3767	2600	1150	1050	3050	8300	3500	6650	4400
2017	2017	3425	4233	2950	1200	1400	3450	9650	3900	7000	4600
2018	1933	3267	3850	2540	915	1100	4233	8750	3950	6867	4650

Note: Shaded numbers indicated values estimated by linear interpolation.

Table 3-12. Annual Sulfate Mass Removal from PWS Wells (metric tons)

Year	Well										
	PW-1	PW-2	PW-3	PW-4	PW-5	PW-6	PW-7	PW-8	PW-9	PW-10	PW-11	Total
1985												
1986		151	183	162								**496**
1987		193	231	310								**735**
1988		186	219	188								**593**
1989		178	206	183								**567**
1990		170	194	147								**511**
1991		81	91	44								**216**
1992		78	84	28								**190**
1993		74	78	24								**176**
1994		70	72	23								**165**
1995		66	65	37								**169**
1996		62	59	23								**144**
1997		117	106	41								**263**
1998		86	109									**195**
1999		114	136	79	18	11	17	19	30	34	64	**521**
2000	51	94	117	78	27	9	16	14	29	31	54	**524**
2001	61	74	87	49	39	8	27	21	33	20	49	**467**
2002	70	107	70	65	40	11	22	6	26	12	36	**466**
2003	71	102	72	40	32	7	25	3	28	11	21	**413**
2004	87	97	77	42	39	6	16	2	28	13	19	**426**
2005	92	99	81	51	34	6	17	1	14	24	15	**435**
2006	101	102	79	45	29	8	33	3	20	24	17	**461**
2007	96	101	63	39	35	8	36	3	18	19	12	**430**
2008	83	95	53	25	24	9	24	2	15	11	7	**349**
2009	10	17	7	3	3	1	4	0	2	2	1	**51**
Totals	**723**	**2515**	**2540**	**1726**	**322**	**85**	**237**	**74**	**243**	**201**	**296**	**8960**

Pumping Rates and Groundwater Flow

The average annual total pumping rates from the PWS are estimated to have ranged from 8.3 to 16 gpm in the period 1986 through 1998 when only wells PW-2, PW-3, and PW-4 were operated. In the latter period, 1999 through March 2009, total average annual pumping rates averaged between 41 and 57 gpm. The combined average pumping rate of the six wells along the northern boundary of the LEP, PW-6 through PW-11, was 12 gpm between 1999 and March 2009, with average pumping rates at individual wells ranging from 1.0 to 2.6 gpm. The wells with the highest average pumping rates were PW-1 through PW-5, which all had average pumping rates of greater than 4 gpm. PW-1 had the highest average pumping rate, 12.7 gpm. A total of 340 million gallons were extracted from the wells in the PWS.

The total volume of groundwater extracted along the northern perimeter of the LEP was 64 million gallons over the 10-year period wells PW-6 through PW-11 were operated. This volume of groundwater represents the amount of groundwater in storage in the Shallow Zone in a band approximately 400 feet wide along the northern extent of the LEP. The operation of the PWS can be represented as removing the groundwater near the northern extent of the LEP in a band 200 feet wide on other side of the line of PWS wells (note that 200 feet is the approximate distance from the PWS wells to the northern extent of the LEP).

The operation of wells PW-6 through PW-11 created drawdown cones around each of the wells, and the magnitude of these drawdown cones was well documented in the detailed aquifer testing that was conducted as each of the wells. The testing consisted of four-day aquifer tests with piezometers located approximately five, 10, and 20 feet from each of the PWS wells (BC, 2010). The magnitude of the drawdowns, except at the pumping wells, was small consistent with the low yields of the wells.

The effect of the pumping along the northern extent of the LEP on regional groundwater flow was evaluated by calculating the hydraulic gradient toward the north immediately north of the LEP and the hydraulic gradient to the west along the northern extent of the LEP prior to operation of the PWS wells PW-6 through PW-11, during operation of these wells between 1999 and 2009, and following operation of these wells. Water level data from monitor wells W5AB-2S,[6] W5BB-S, and USGS-13 were used for these evaluations (Figure 3-3). Wells W5AB-2S and W5BB-S are

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[6] Note that monitor well USGS-7 was located along Luzier Lane in close proximity to W5AB-2S (Seitz et al., 1982). This well was constructed in 1976 and data from this well is used for gradient calculations prior to construction of well W5AB-2S in 1983.

located along Luzier Lane, just north of the LEP, approximately 2,000 feet apart. USGS-13 is located approximately 1,450 feet due north of W5AB-2S. Well USGS-13 was constructed in 1976 and wells W5AB-2S and W5BB-S were constructed in 1983. The calculated hydraulic gradients toward the north and the west in the area north of the LEP are shown on Figure 3-7.

The hydraulic gradient toward the north, based on wells W5AB-2S/USGS-7 and USGS-13, fluctuated about zero for several years before operation of wells PW-6 through PW-11, during operation of these wells, and following operation of these wells. This indicates that operation of the PWS system along the northern extent of the LEP had at most a small effect on groundwater flow north of the LEP. As shown on Figure 3-7, from about 1992 through 2012, the hydraulic gradient fluctuated about zero, indicating that there was a negligible component of flow toward the north (a positive gradient indicated flow to north and a negative gradient indicates flow to south). Prior to 1986, there was a consistent hydraulic gradient to the north averaging a little greater than 0.001 feet/foot, and then after, the hydraulic gradient declined rapidly from 1986 to 1992. This decline in the hydraulic gradient corresponds with a significant increase in irrigated lands and irrigation adjacent to the Site in 1987 (refer to Figure 4-4 in Draft OU-7 RI, Wood, 2019).

The hydraulic gradient toward the west, as shown on Figure 3-7, fluctuated between 0.0015 and 0.004 between 1997 and 2012. This indicates that during this period groundwater flow immediately north of the LEP was consistently toward the west, and that operation of the PWS wells along the northern extent of the LEP did not change the overall direction of groundwater flow. During this period, given that the northward component of flow was approximately zero, groundwater flow was predominately toward the west.

Amount of Sulfate Removed from Groundwater

The total estimated amount of sulfate removed from groundwater by operation of the PWS was 8,960 metric tons (Table 3-12). Seventy-six percent of the sulfate mass was removed from the three wells at the northeast corner of the UEP (PW-2, PW-3, and PW-4). The total amount of sulfate removed from the wells along the northern extent of the LEP (PW-6 through PW-11) was only 1,135 metric tons.

The water-quality in wells PW-2, PW-3, and PW-4 changed significantly as the result of operation of the PWS. For example, the average sulfate concentration in PW-3 declined from 27,500 mg/L in 1985 prior to operation of the PWS to 3,267 mg/L in 2018. The large change in concentrations is the result of capture by these wells of irrigation waters applied on the adjacent agricultural fields. The average sulfate concentration in the six wells along the northern extent of the LEP was 4,615 mg/L in 1998 prior to operation of the PWS and 4,925 mg/L in 2018. An evaluation of groundwater chemistry collected in 2014 from areas including the vicinity of the PWS is provided in Section 5.4.3.

3.3.2 Pit Lake Water Levels

The Pit Lake (OU-2), which is currently refilling with groundwater from bedrock and alluvial flow systems (BC, 2014a), has been studied to better understand its influence on Site-wide groundwater conditions. Pit Lake studies related to OU-1 include routine monitoring of the Pit Lake water level elevation beginning in September 2007 and a water balance evaluation (Appendix J-2) to predict the future “steady-state” elevation of the Pit Lake.

Groundwater inflow, based on the lake water balance study, is estimated to be slightly greater than the current rate of evaporation. Thus, the lake level is slowly rising with time. The Pit Lake water balance and projection of the pit refilling curve (Figure 3-8) indicate that the lake is expected to reach a steady-state level, where water inflow and evaporation are balanced, prior to 2030. The steady-state Pit Lake elevation is estimated to be in the range of 4,249 to 4,253 feet amsl, with more recent data indicating that the steady-state elevation may fall within the lower end of this range. The steady-state Pit Lake elevation is approximately: 1) 100 feet lower than the pre-mining groundwater elevation range of 4,350 to 4,375 feet amsl reported by Gill (1951); 2) 135 feet lower than current groundwater levels in the alluvial aquifer adjacent to the Walker River just east of the Pit Lake; 3) 80 feet lower than the current groundwater levels beneath the

Evaporation Ponds; and 4) 150 and 340 feet below the east and west pit rim elevations, respectively.

The steady-state Pit Lake level is projected to be lower than the pre-mining water level as the result of the significant evaporation that occurs from the Pit Lake surface. Consequently, the lake is and will continue to be a groundwater sink that creates a cone of depression (extending as far north as the Process Areas) with local groundwater flow toward the pit. Because the Pit Lake does not and will not in the future discharge into the Site-wide groundwater system, the Pit Lake is not a source of COIs to Site-wide groundwater.

3.3.3 Groundwater Pumping and Surface Water Points of Diversion

Groundwater conditions in the Study Area are influenced by groundwater pumping and surface water diversion associated primarily with irrigation and, to a lesser extent, stock watering and mining/milling (BC, 2014a; S.S. Papadopulos & Associates, Inc. [SSPA], 2014).

Publicly-available groundwater pumping and surface water diversion information applicable to the Study Area is provided in the *Revised Public Information for the Northern Portion of the Background Groundwater Study Area* (BC, 2013d) included in Appendix J-3. That document also includes well ownership, location, and construction; underground and surface water rights and points of diversion (PODs); well pumping records from 1993 to 2010; subsurface lithology and, as applicable, depth to bedrock; and groundwater elevations from the NDWR and the USGS.

PODs from an underground source (i.e., groundwater) for the wells with water rights within and adjacent to the Study Area are shown on Figure 3-9, along with diversion rates and annual duties. All agricultural wells within and near the Study Area are screened in the alluvial aquifer. Although well construction varies greatly, agricultural wells used to extract groundwater for crop irrigation are either screened beginning at or near the water table to the total depth of installation and/or have been installed with a permeable filter pack from above the water table surface to the total depth of installation. Annual pumping inventories (i.e., actual total amounts pumped each year) for wells in the Mason Valley from 1994 to 2003 and from 2004 to 2010 have been reported by

Gallagher (2004) and Gallagher (2013), respectively. Since 2011, the Nevada State Engineer has published an annual inventory of pumping in the Mason Valley (Nevada Department of Conservation and Natural Resources, 2011, 2012, 2013, 2014, 2015, 2016, 2017).

Within the Study Area, there are 20 wells used for irrigation, four wells used for stock watering, one used for mining/milling, and one used for commercial purposes (Gallagher, 2013). The 20 irrigation wells are currently permitted to irrigate a total of 5,509 acres using an annual duty of 15,788 acre-feet with a combined diversion rate of 46.36 cubic feet per second (cfs). Of the 36 active water rights, 26 allow for pumping to occur on a year-round basis, nine of the rights can only be pumped during the irrigation season, and one right can only be used in the winter.

The place of use (POU) of 37 surface water rights within and adjacent to the Study Area that are identified in the amended Walker River Decree (WRD), Case in Equity, C-125, filed April 24, 1940 (WRD C-125; WRD, 1940) are shown on Figure 3-10. This figure also shows the POUs of surface water rights approved by NDWR as either new appropriations or applications to change WRD rights. Additional information about the distribution and routing of surface water is included in the discussion of surface water hydrology in Section 4.7.

The POUs of flood waters permitted by NDWR Application 5528, Certificate 8859 are shown on Figure 3-11 by quarter-section. Application 5528 was filed by the Walker River Irrigation District (WRID) to divert flood waters from the Walker River for irrigation from May 1 to July 31 of each year. Application 5528 was certificated for 491.2 cfs, not to exceed 89,612 acre-feet per season (the total duty of water cannot exceed 4.0 acre-feet per acre per season from any and/or all sources). The lands irrigated under this Certificate during any one season cannot exceed 30,000 acres.

3.3.4 Groundwater Model Development

The 2007 SOW (EPA, 2007a) specified that the OU-1 RI "extrapolate the future contaminant transport using a comprehensive groundwater flow and fate-and-transport model." In addition, determining groundwater flow and chemical transport rates was identified as DQO #6 in the Revised Groundwater RI Work Pan (BC, 2014a).

The technical and programmatic framework to address quantitative numerical modeling of groundwater flow and chemical transport was developed during iterative technical discussions with the EPA, ARC, and other stakeholders and documented in the Revised Groundwater RI Work Plan (BC, 2014a). Table 3-13 summarizes the chronology of investigations, evaluations, communications, and documents related to groundwater flow modeling.

Table 3-13. Chronology of Groundwater Modeling Activities

2004-2007	Characterization of groundwater conditions in the Study Area pursuant to the First-Step HFA Work Plan (BC, 2005) and Second-Step HFA Work Plan (BC, 2007b).
2007	Characterization of bedrock groundwater conditions is required under the Administrative Order for Remedial Investigation and Feasibility Study (RI/FS), CERCLA Docket No. 9-2007-0005 dated January 12, 2007.
2007-2011	Characterization of groundwater conditions in the Study Area pursuant to the various work plans and related correspondence (BC, 2008c, 2010c, 2010d, 2011a; ARC, 2011).
May 16, 2011	Conference call with EPA, ARC, and other stakeholders to discuss groundwater flow modeling.
August 17, 2011	Groundwater technical meeting with EPA, ARC, and stakeholders to discuss the status of RI activities, which resulted in concurrence to conduct groundwater modeling to support a quantitative evaluation of groundwater flow and chemical transport.
June 4, 2012	Conference call with EPA, ARC, and other stakeholders to discuss groundwater flow modeling activities, which resulted in a request by EPA that ARC submit a document describing key groundwater modeling deliverables and milestones, and a draft table of contents for a groundwater modeling work plan.
June 25, 2012	ARC submitted the *Groundwater Flow Modeling Deliverables* (ARC, 2012c), which included: 1) a preliminary summary of key groundwater modeling deliverables and milestones; and 2) a draft table of contents for a groundwater flow model work plan.
July 11, 2012	EPA (2012e) provided comments on the *Groundwater Flow Modeling Deliverables* (ARC, 2012c).
July 17, 2012	Groundwater technical meeting with EPA, ARC, and stakeholders to discuss findings of the 2011 Monitor Well Installation investigation, and related RI activities, which resulted in an EPA request that ARC submit a document summarizing groundwater modeling objectives.
August 14, 2012	ARC submitted the *Objectives for Groundwater Modeling in the RI/FS Process, Yerington Mine Site* (SSPA, 2012a).
August 29, 2012	Groundwater technical meeting with EPA, ARC, and other stakeholders to discuss the groundwater modeling objectives.
October 15, 2012	ARC submittal of the *Objectives for Groundwater Modeling in the RI/FS Process - Revision 1, Yerington Mine Site* (SSPA, 2012b).
October 26, 2012	EPA (2012f) approval of the *Objectives for Groundwater Modeling in the RI/FS Process - Revision 1, Yerington Mine Site* (SSPA, 2012b).
December 28, 2012	ARC submittal of the *Draft Groundwater Flow Model Work Plan for the Yerington Mine Site* (SSPA, 2012c).
March 29, 2013	EPA (2013d) provided comments on the *Draft Groundwater Flow Model Work Plan for the Yerington Mine Site* (SSPA, 2012c).
May 21, 2013	ARC submittal of the *Groundwater Flow Model Work Plan – Revision 1* (SSPA, 2013) included as Attachment E to the Revised Groundwater RI Work Plan, along with responses to EPA comments on the *Draft Groundwater Flow Model Work Plan* (SSPA, 2012c).
March 18, 2014	ARC submittal of the *Groundwater Flow Model Yerington Mine Site* (SSPA, 2014). This report synthesized available hydrologic and geochemical information into a quantitative representation of the current and historical HCSM. The report also contained: 1) documentation of the study goals; 2) a discussion of the modeling strategy and assumptions; 3) details about model construction, calibration and validation; 4) a summary of model predictions; and 5) an analysis of the uncertainty associated with the model predictions.
October 28, 2014	EPA provided comments on the *Groundwater Flow Model Yerington Mine Site* (SSPA, 2014).
February 3, 2015	ARC submits the *Flow Model Supplemental Materials* (SSPA, 2015) in response to EPA comments.
May 18, 2015	EPA (2015b) provided comments on the *Flow Model "Supplemental" Materials* (SSPA, 2015) and approved the *Groundwater Flow Model Yerington Mine Site* (SSPA, 2014).

Based on review of the *Flow Model Supplemental Materials* (SSPA, 2015), EPA (2015b) constrained the modeling objective and approved the groundwater flow model, noting that: "The primary goal foreseen for the Yerington groundwater model is to provide a management tool that can be used to evaluate possible remediation options. As such, its greatest value will be in allowing short-term comparisons of remedial designs and possible effectiveness of different remediation scenarios using a common tool and less so in predicting long-term migration of contaminants. It appears that this tool is adequate for that purpose."

The groundwater flow model is provided in Appendix J-4. The flow model domain, which encompasses an area of approximately 86 square miles, consists of that portion of the Mason Valley west of the Walker River and north of Mason that is underlain by saturated alluvium (Regional Domain). Nested within the model domain is the Study Area which encompasses an area of approximately 23 square miles that is bounded to the north by Campbell Lane, to the west by the Singatse Range, to the east by a north-south trending line located one mile east of Highway 95, and to the southeast by the Walker River (Local Domain). The Local Domain is nested within the Regional Domain so that appropriate boundary conditions along the northern and eastern boundaries of the overall model domain can be calculated. In addition, the model domain is subdivided to reflect two different sources of data, which differ in data quality.

The vertical extent of the model domain extends from the ground surface to the alluvial/bedrock contact and into the portions of the bedrock groundwater system in hydrologic communication with the alluvial aquifer. The model domain extends laterally to include monitor well locations for identifying background groundwater quality and groundwater impacted by mining and other anthropogenic activities.

Since 2005, hydrogeologic data within the Local Domain have been and continue to be collected as part of the RI process, pursuant to EPA-approved planning documents and work plans. Thus, these data are high quality and there is a high degree of confidence in the data. Hydrogeologic data from outside the Local Domain but within the Regional Domain are from multiple sources and are not verified. Much of these data are from the USGS (e.g., water-level data) and the NDWR (e.g., well logs).

Temporal (e.g., seasonal, annual) variations in groundwater flow patterns and chemical concentrations continue to be assessed due to variability in hydrologic stresses on the groundwater system. Monitor wells installed for groundwater characterization purposes continue to be routinely monitored to address temporal aspects of the study within the Local Domain. Within the Regional Domain, available water-level and surface water flow data from the USGS and NDWR were used to assess temporal variations in groundwater conditions.

3.3.5 Chemical Transport Evaluations

The technical and programmatic framework for characterizing groundwater geochemical conditions and assessing geochemical processes that affect the release and subsequent mobility or attenuation of COIs during groundwater transport in the Study Area was presented as DQO #3 in Revised Groundwater RI Work Plan (BC, 2014a).

Table 3-14 summarizes the chronology of investigations, evaluations, communications, and documents related to chemical transport evaluation.

Table 3-14. Chronology of Activities to Determine Geochemical Mobilization/Attenuation Processes

2008	The Second-Step HFA Work Plan (BC, 2007b) included collecting and archiving redox-preserved samples of saturated and unsaturated alluvium. EPA technical staff observed the redox-sample collection and archiving methods and provided input on locations and depth intervals for collecting an initial set of samples. These initial samples were collected for use in a "methods development" phase of work intended to develop Site-specific testing procedures.
June 21, 2010	ARC submitted the *Draft Aquifer Solids Testing Work Plan* (BC, 2010g).
September 13, 2010	EPA (2010a) provided comments on the *Draft Aquifer Solids Testing Work Plan* (BC, 2010g).
September 21, 2010	ARC submitted the *Aquifer Solids Testing Work Plan - Revision 1* (BC, 2010e), which was revised in response to EPA comments.
September 30, 2010	EPA (2010b) approved the *Aquifer Solids Testing Work Plan - Revision 1* (BC, 2010e).
2011	ARC developed Site-specific procedures and methods to physically separate redox-preserved samples into solid and liquid fractions for subsequent characterization of total metals concentrations, mineralogy, and pore water chemistry.
February 28, 2012	EPA technical staff visited the testing laboratory (Hazen Research, Inc. in Golden, Colorado) and observed the Site-specific testing procedures.
August 17, 2012	ARC submittal of *SOP-23: Aquifer Solids Testing SOP - Redox-Preserved Sample Preparation and Testing* (BC, 2012c).
September 26, 2012	EPA (2012g) transmitted comments on *SOP-23: Aquifer Solids Testing SOP- Redox-Preserved Sample Preparation and Testing* (BC, 2012c).
October 15, 2012	ARC submittal of *SOP-23: Aquifer Solids Testing SOP - Redox-Preserved Sample Preparation and Testing - Revision 1* (BC, 2012d).
October 22, 2012	EPA (2012d) approval of SOP-23 Revision 1, pending minor changes. These minor changes were incorporated into SOP-23 Revision 2, included as Appendix H-1 to the Revised Groundwater RI Work Plan (BC, 2014a).
February 7, 2014	ARC submitted the Revised Groundwater RI Work Plan (BC, 2014a), which included: 1) DQO #3 pertaining to geochemical attenuation/mobilization; 2) the thermodynamic data for Site-specific geochemical modeling; and 3) Site-specific distribution coefficients (a simple, lumped-parameter variable that describes either the relative affinity of the aquifer matrix for a particular ion or the mobility of the ion in a groundwater flow system) based on chemical concentrations in co-located aquifer sediment and groundwater samples.
October 9, 2014	EPA (2014b) approved the Revised Groundwater RI Work Plan including the thermodynamic data presented in Appendix H-4 for Site-specific geochemical modeling and directed ARC to prepare a *Groundwater Geochemical Characterization Data Summary Report*.
December 30, 2014	ARC submitted the *Groundwater Geochemical Characterization Data Summary Report* (BC, 2014c). As noted in ARC's transmittal letter, the document partially fulfilled the requirements for the geochemical characterization and ARC recommended additional refinements to the thermodynamic database for geochemical modeling.
April 27, 2015	ARC transmitted recommendations to EPA for refining the thermodynamic database to be used for geochemical modeling (via e-mail).
May 4, 2015	EPA approved ARC's recommendations on refining the thermodynamic database to be used for geochemical modeling (also via e-mail).
September 23, 2015	EPA, ARC, and other stakeholders agreed during a conference call that geochemical characterization to be performed for the OU-1 RI should focus on geochemical modeling of: 1) controls on the fate and transport of key COIs anticipated to drive decision-making in the FS; and 2) other chemicals that may affect their mobility and transport in groundwater.
December 11, 2015	ARC submitted the *Groundwater Geochemical Characterization Data Summary Report – Revision 1* (BC, 2015e).
July 2016	EPA conditionally approved the document on July 31, 2016 (EPA, 2016b) subject to minor editorial changes and revision of statements referencing COI concentrations and spatial extent relative to background chemical concentrations presented in the *Background Groundwater*

Table 3-14. Chronology of Activities to Determine Geochemical Mobilization/Attenuation Processes

	Quality Assessment - Revision 2 (BC, 2015c). EPA recommended that a revised version of the report be included as an appendix to the OU-1 RI Report.

The *Groundwater Geochemical Characterization Data Summary Report - Revision 2* (BC, 2016a) is provided in Appendix J-5. The chemical speciation model and approach to calculating Site-specific distribution coefficients is summarized below.

Chemical Speciation Model Development

The specific objectives of the *Groundwater Geochemical Characterization Data Summary Report - Revision 2* are as follows:

- Describe the occurrence and distributions of select chemicals in Study Area groundwater based on the comprehensive set of monitor well data obtained during August 2014; and
- Using the EPA-approved thermodynamic database developed for the Site and geochemical modeling, evaluate the aqueous geochemical speciation of select COIs and potential formation of solid mineral phases in Study Area groundwater to assess chemical mobility/attenuation.

The primary geochemical data inputs used to identify the geochemical processes controlling chemical transport consist of: 1) groundwater chemical data from monitor wells installed in the groundwater zones in the alluvial aquifer and bedrock; 2) field parameter measurements that characterize the pH and redox status of the groundwater system (because these affect the aqueous speciation of inorganic chemicals and formation of mineral phases); and 3) thermodynamic data describing chemical reactions for each of the important aqueous species, minerals comprising the aquifer solids, gases, and adsorbed species. The geochemical assessment primarily relied on groundwater information associated with the August 2014 groundwater monitoring event. Approximately 2% of the August 2014 dataset had speciated charge imbalances outside the acceptable range of ±10%, and groundwater data obtained in October 2014 were substituted for August 2014 data.

Geochemical modeling using the Site-specific thermodynamic database with PHREEQC version 3.1.5 was conducted to determine the chemical speciation of aqueous constituents and the saturation indices of solid mineral phases in equilibrium with the groundwater samples. The geochemical modeling did not involve adsorption to aquifer soil/sediments or organics in aquifer materials. Details regarding the development of the Site-specific thermodynamic database are provided in Appendix J-5 and key modifications are discussed briefly below.

The WATEQ4F database was used as the starting point for database development because its major-element data are consistent with the Nordstrom et al. (1990) data compilation, which is a reliable and internally consistent dataset. Subsequently, the WATEQ4F database was modified by replacing aqueous speciation and solid-phase solubility data for uranium, phosphate, vanadium, sulfide, arsenic, and copper with new data that have been critically reviewed by federal agencies (e.g., compilations prepared by the Nuclear Energy Agency were the principal sources of the uranium data in the ARC database) or in the peer-reviewed literature (e.g., Dong and Brooks, 2006). Uranyl species $Ca_2UO_2(CO_3)_3^0$, $CaUO_2(CO_3)_3^{-2}$, $Mg_2UO_2(CO_3)_3^0$ and $MgUO_2(CO_3)_3^{-2}$ are of particular importance in evaluating the mobility of uranium; therefore, the thermodynamic data for these constituents were added to the Site-specific database. Thermodynamic solubility data for schwertmannite (an oxyhydroxide sulfate mineral) reported by Bigham et al. (1996) and confirmed by Sánchez-España et al. (2011) were included in the database. In addition, thermodynamic solubility data reported by Bourrié et al. (1999) for three hydroxy-green rusts were included in the database.

PHREEQC is a geochemical software model distributed by the USGS. The model assumes equilibrium mass transfer and does not account for the kinetics of mineral precipitation and dissolution reactions using applicable reaction rate laws (Parkhurst and Appelo, 1999; EPA, 2007b). Chemical speciation modeling describes the distribution of chemical mass between aqueous and solid mineral phases, and hence, predicts the geochemical conditions under which various constituents might be sequestered by mineral precipitation or remain mobile in the groundwater flow system. Both chemical speciation and mineral precipitation are pertinent data for evaluating the mobility of constituents in the groundwater system.

Information generated from the geochemical assessment is incorporated into the discussion of contaminant fate and transport in Section 6.0. This information may be used to guide the development of quantitative approaches to representing chemical transport in the numerical groundwater flow model (SSPA, 2014) to evaluate various remedial alternatives during the FS. As noted by the EPA (2016b), decisions will be made during the FS regarding the most appropriate reactive transport modelling approach and whether it will be necessary and/or beneficial to integrate the models or how that can be accomplished to efficiently meet the technical needs of the OU-1 RI/FS without introducing unnecessary complexity to the modeling efforts.

Distribution Coefficients

The partition (distribution) coefficient (K_d) is a simple, lumped-parameter variable that is used to assess contaminant transport by describing either the relative affinity of the aquifer matrix for a particular ion or the mobility of the ion in a groundwater flow system (EPA, 2007b; Freeze and Cherry, 1979).

The initial approach to developing Site-specific distribution coefficients based on chemical concentrations in co-located groundwater and aquifer sediment samples was presented in the Revised Groundwater RI Work Plan (BC, 2014a) and is reproduced in this OU-1 RI Report as Appendix J-6. The approach to developing the distribution coefficients is summarized below.

Distribution coefficients were calculated for a variety of chemicals in Site groundwater including aluminum, arsenic, barium, boron, calcium, chloride, chromium, cobalt, copper, fluoride, iron, lead, lithium, magnesium, manganese, molybdenum, nickel, nitrate, potassium, selenium, sodium, sulfate, uranium, vanadium, and zinc. Distribution coefficients were not calculated for parameters that were only infrequently detected in groundwater or are not likely to be the subject of FS transport modeling including alkalinity, antimony, beryllium, cadmium, mercury, phosphorous, silica, silver, strontium, thallium, tin, and titanium.

Site-specific distribution coefficients were calculated using data from a single set of soil samples that were collected during the borehole drilling for monitor well installation and two distinct sets of water quality data. Soil samples were analyzed for a variety of bulk chemical concentrations (analyses were performed on liquid extracts from treatment of the solid samples by microwave-assisted digestion using EPA Method 3051A).

The first set of water quality data used in K_d calculations was the zonal water quality data that was collected at the time of borehole drilling. As discussed in Section 3.2.1, zonal groundwater samples were analyzed only for sulfate, uranium and arsenic. These data were collected over small depth intervals, typically ranging from three to five feet. Co-located zonal groundwater and soil samples were collected at multiple depth intervals in 13 locations throughout the Site that included B/W-1, B/W-2, B/W-3, B/W-4, B/W-11, B/W-31, B/W-32, B/W-42, B/W-46, B/W-61, B/W-62, B/W-65, and B/W-66.

The second set of water quality data used in K_d calculations included groundwater quality data obtained during quarterly sampling events from 15 monitor wells typically having a screen interval length of 20 feet. Monitor well samples were analyzed for the broader set of constituents listed in Table 3-4. The soil sample data used in the calculations were selected such that the sample intervals were within the screened interval of the well. The wells considered in this portion of the analysis were B/W-2D1, B/W-3I, B/W-4I, B/W-4D1, B/W-11D2, B/W-31S1, B/W-31S2, B/W-32S, B/W-42S, B/W-46S, B/W-61S, B/W-62S, B/W-65S, B/W-66S, and B/W-67S. The quarterly groundwater quality results collected closest to the date of the zonal soil sample collection for each particular well were used to calculate K_d values to minimize potential effects from variability in groundwater concentrations over time.

The distribution coefficient is calculated as the ratio of the concentration of a chemical adsorbed onto the solid phase (commonly expressed as milligrams [mg] of chemical per kilogram [kg] of solid) to the dissolved concentration of the chemical in the water (mg of chemical per liter [L] of solution) at equilibrium (Freeze and Cherry, 1979). Based on the formulation below, K_d values are expressed in units of liters per kilogram (L/kg).

$$\mathbf{K_d} = \frac{\mathbf{C}_{\mathbf{adsorbed}}}{\mathbf{C}_{\mathbf{water}}}$$

where: $C_{adsorbed}$ = adsorbed chemical concentration (mg/kg)

C_{water} = dissolved chemical concentration in water (mg/L)

Initially, distribution coefficients were calculated using water chemical data and chemical concentrations in the aquifer sediment sample that were determined using EPA digestion Method 3051A. Because of the relatively aggressive digestion method, the aquifer sediment data represent the bulk (i.e., total) chemical concentration in the solid rather than the adsorbed chemical concentration. As recommended by EPA (1999), trace metals that are present in crystalline lattice sites of minerals present in soils do not participate in adsorption/desorption reactions and should not be included in the K_d calculation. Consequently, the original K_d values were revised for this OU-1 RI Report. To better estimate K_d values, the adsorbed amount of a chemical was estimated by subtracting the average chemical concentration in Sub-area A-1 soils (BC, 2009b) from the bulk (i.e., total) chemical concentration in the individual aquifer sediment sample, as follows:

$$\mathbf{K_d} = \frac{\mathbf{C}_{\mathbf{soil}} - \mathbf{C}_{\mathbf{bkgd}}}{\mathbf{C}_{\mathbf{water}}}$$

where: $C_{adsorbed} = C_{soil} - C_{bkgd}$

C_{soil} = bulk chemical concentration in the solid (mg/kg)

C_{bkgd} = average background chemical concentration in the solid (mg/kg)

C_{water} = dissolved chemical concentration in water (mg/L)

The revised Site-specific distribution coefficient values are presented in the contaminant fate and transport discussion in Section 6.0.

As noted in Table 3-14, the scope of work considered necessary to fulfill the DQO pertaining to characterizing geochemical mobilization/attenuation processes evolved as more RI characterization data was obtained. Originally, K_d values were developed to provide general insights to the variability of the attenuation characteristics of the alluvial aquifer, and to help guide preparation of a detailed work plan for obtaining sufficient data to characterize attenuation characteristics for geochemically-reactive COIs along two flow pathways from the Site to potential receptors. The attenuation characteristics of the alluvial aquifer were to be used in a combined fluid-flow and reactive chemical transport model to predict future chemical distributions during the FS. During a teleconference call on September 23, 2015, EPA, NDEP, ARC, and other stakeholders agreed that the scope of characterization for this DQO should focus on geochemical modeling of: 1) controls on the fate and transport of key COIs anticipated to drive decision-making in the FS (i.e., mobile COIs such as sulfate and uranium); and 2) other chemicals that may affect their mobility and transport in groundwater. The decision to focus on mobile COIs, and thus eliminate the aquifer solids testing program the geochemically reactive COIs (such as metals), was based on two considerations: 1) the recognition that transport modeling of future COI distributions in groundwater would likely yield more accurate predictions for mobile COIs than geochemically reactive COIs; and 2) the recognition that evaluation of empirical data (specifically time-trend analysis of metals concentrations in groundwater monitor wells) would yield a better understanding of metals transport rates than predictive chemical transport modeling. During that teleconference call, EPA recommended that the K_d values be presented in the Draft OU-1 RIR for the sake of completeness, and ARC has retained the discussion in the Final OU-1 RIR for the sake of consistency.

3.3.6 Background Groundwater Quality Assessment

Identifying background water types and defining the extent of mine-influenced groundwater was specified in the 2007 SOW (EPA, 2007a) and identified as DQO #1 in the Revised Groundwater RI Work Plan (BC, 2014a). The background groundwater quality assessment (BGQA) has been integrated into groundwater characterization activities performed in the Study Area after 2007.

Table 3-15 summarizes the chronology of the BGQA and other background-related orders, investigations, and documents.

Table 3-15. Chronology of Activities Related to Establishing Background Groundwater Quality

2004-2008	Monitor wells B/W-13S, B/W-14S, B/W-15S, B/W-20S, and B/W-21S were installed adjacent to Walker River and hydraulically upgradient of the Site, pursuant to the First-Step HFA Work Plan (BC, 2005) and the Second-Step HFA Work Plan (BC, 2007b).
2007	Characterization of background groundwater quality is required under the Administrative Order for Remedial Investigation and Feasibility Study (RI/FS), CERCLA Docket No. 9-2007-0005 dated January 12, 2007.
2007-2011	Background characterization conducted pursuant to the *2010 Groundwater Monitor Well Installation Work Plan - Revision 2* (BC, 2010c), *Agricultural Fields Characterization Work Plan - Revision 2* (BC, 2010d), *On-Site Monitor Installation Work Plan - Revision 1* (BC, 2011a), and the *Addendum to the On-Site Monitor Well Installation Work Plan* (ARC, 2011).
August 17, 2011	Groundwater technical meeting with EPA, ARC, and stakeholders to discuss progress of the phased approach to groundwater RI activities, which resulted in concurrence to accelerate background groundwater characterization activities and an EPA request for ARC to prepare and submit a BGQA. Also discussed were installation of additional well clusters (B/W-12R, B/W-17, and B/W-22R) in areas south and southwest of the Site, pursuant to the *On-Site Monitor Well Installation Work Plan* (BC, 2011a), to support background groundwater characterization.
September 7, 2011	ARC submitted the *Addendum to the On-Site Monitor Well Installation Work Plan* (ARC, 2011) proposing additional well clusters at B/W-12R, B/W-17, and B/W-26R to support background groundwater characterization.
September 28, 2011	ARC submitted the *Draft Background Groundwater Quality Assessment* (BC, 2011c), which recommended the installation of three monitor well clusters (B/W-56, B/W-57, and B/W-58) located in the northern portion of the Study Area.
September 30, 2011	The *Addendum to the On-Site Monitor Well Installation Work Plan* (ARC, 2011) was approved by EPA (2011b).
December 7, 2011	Via e-mail communication, ARC requests and receives EPA approval to install well clusters B/W-56, B/W-57, and B/W-58 proposed in the Draft BGQA during implementation of the *On-Site Monitor Well Installation Work Plan - Revision 1.*
February 7, 2012	EPA (2012a) provided comments on the Draft BGQA.
March 19, 2012	ARC (2012d) submitted a request to implement a comprehensive hydrologic tracer sampling event that was larger in scope than the sampling event proposed in the *On-Site Monitor Well Installation Work Plan - Revision 1* (BC, 2011a) and to eliminate select hydrologic tracers. Hydrologic tracers are considered one line of evidence that may be useful for determining background groundwater quality.
April 18, 2012	ARC (2012e) submitted responses to EPA comments on the Draft BGQA.
April 27, 2012	EPA approved the comprehensive hydrologic tracer sampling event and request to eliminate select tracers (EPA, 2012b).
May 2012	Comprehensive hydrologic tracer sampling event conducted concurrent with the 2Q 2012 quarterly groundwater monitoring event.
August 28, 2012	Groundwater technical meeting with EPA, ARC, and other stakeholders to clarify and resolve comments on the Draft BGQA.
November 19, 2012	Submittal of final ARC responses to EPA comments on the *Draft BGQA* and the *Background Groundwater Quality Assessment - Revision 1* as Attachment A to the *Draft Site-Wide Groundwater Operable Unit (OU-1) Remedial Investigation Work Plan* (BC, 2012b).

Table 3-15. Chronology of Activities Related to Establishing Background Groundwater Quality

June 26, 2013	Submittal of the *Draft Additional Monitor Well Work Plan* (BC, 2013e) as Attachment B to the *Draft Site-Wide Groundwater Operable Unit (OU-1) Remedial Investigation Work Plan* (BC, 2012b), to address data gaps identified by ARC and EPA, including groundwater conditions in the north and northeastern portion of the Study Area. ARC recommended sampling of all wells proposed for hydrologic tracers to supplement data from the May 2012 hydrologic tracer sampling event.
July 29, 2013	EPA (2013e) provided comments on the *Draft Additional Monitor Well Work Plan*.
October 8, 2013	ARC submitted the Additional Monitor Well Work Plan (BC, 2013b). This work plan was also included as Attachment B to the Revised Groundwater RI Work Plan (BC, 2014a). EPA (2014b) approved the Revised Groundwater RI Work Plan including Attachment B on October 9, 2014.
September 2013 to July 2014	Installation, development, and hydraulic testing of new wells installed pursuant the Additional Monitor Well Work Plan (BC, 2013b).
August 2014	Hydrologic tracer sampling of new wells installed pursuant to the Additional Monitor Well Work Plan (BC, 2013b).
May 2015	Receipt of hydrologic tracer laboratory analytical results for new wells installed pursuant to the Additional Monitor Well Work Plan (BC, 2013b).
June 11, 2015	ARC submitted the *Additional Monitor Well Installation Data Summary Report* (BC, 2015d).
July 2, 2015	ARC submitted the *BGQA –Revision 2* (BC, 2015c), which described: 1) the technical approach, scope, rationale and methods to establish background groundwater quality; and 2) multiple supporting lines of evidence for defining the extent of mine-influenced groundwater and identifying other anthropogenic groundwater impacts.
February 11, 2016	EPA (2016c) provided comments on the *Background Groundwater Quality Assessment - Revision 2*.
June 14, 2016	ARC (2016b) provided responses to EPA Comments on the *Background Groundwater Quality Assessment - Revision 2*.
June 29, 2016	EPA, ARC, and other project stakeholders held a groundwater technical meeting to discuss the background assessment.
September 27, 2016	EPA (2016d) letter to ARC providing final direction on the background groundwater quality assessment including an attachment (EPA, 2016e) dated September 2, 2016 and titled EPA Memorandum, Subject: Yerington Mine Site, Yerington Nevada (16-R09-003) Responses to ARC Responses to Comments on the Background Groundwater Quality Assessment - Revision 2.
November 11, 2016	ARC submitted the *Background Groundwater Quality Assessment - Revision 3* (BC, 2016b).
February 16, 2017	EPA (2017) approved the *Background Groundwater Quality Assessment - Revision 3 (included herein as Appendix J-7)*.
October, 2018	ARC submitted the Draft Geothermal Technical Memorandum (Copper Environmental Consulting, 2018)
April 1, 2019	NDEP provided comments and direction to ARC on finalization of the Geothermal Technical Memorandum (NDEP, 2019)
October, 2019	ARC submitted the Final Geothermal Technical Memorandum (included herein as Appendix N).

Subsequent to EPA approval of the BGQA in 2017, additional evaluations were conducted by ARC to addresses the acknowledged limitations and conservative assumptions that were identified by ARC, EPA,NDEP, and other stakeholders during development of the background assessment approach. In particular, it was recognized that: 1) multiple sources including mining, natural processes, and agricultural and irrigation practices contributed COIs to Study Area groundwater (BC 2016a, 2017); and 2) the EPA directed ARC to delineate the extent of mine-influenced groundwater in the BGQA using an approach that did not account for all anthropogenic or naturally-occurring background sources within the Study Area (EPA 2016a, 2016b).

A more detailed discussion of evaluations conducted from 2017 to the present to better assess background groundwater quality is provided in Section 5.0. These more recent evaluations form the basis for establishing background groundwater quality conditions in this FRIR.

3.4 Former Domestic Well Monitoring and Bottled Water Programs

Water quality monitoring of domestic, commercial, and irrigation wells (collectively referred to as domestic wells) located near the Site has evolved over time. Domestic well monitoring began in late 1983. Up through early 2009, domestic well monitoring activities were performed pursuant to:

- Paragraphs 15(e) and 15(f) of the Unilateral Administrative Order for Initial Response Activities, Docket No. 9-2005-0011 (2005 Order);
- Section 6.0 of the 2007 SOW; and
- The Administrative Order on Consent and Settlement Agreement for Removal Actions and Past Response Costs, Docket No. 09-2009-0010 (2009 Order).

In March 2009, EPA requested that ARC expand the domestic well monitoring program because of the EPA-approved shutdown of the PWS to evaluate OU-1 hydrogeologic conditions. The expanded domestic well monitoring program has been conducted pursuant to the *Domestic Well Monitoring Plan - Revision 3* (DWMP; BC, 2010b), which was prepared as an addendum to the Site-Wide QAPP (ESI and BC, 2009). Results of domestic well monitoring have been used to: 1) characterize the quality of groundwater used for drinking water or other domestic water supply purposes; and 2) determine eligibility for receiving bottled water as part of an interim response action. In 2019, the NDEP eliminated monitoring of radium-226 and radium-228 in all domestic wells except for DW-58, DW-61, DW-65, DW-205, DW-208, DW-210, and YPT Well 4.

The Bottled Water Program was initiated in March 2004. Domestic well owners were deemed eligible to receive bottled water if uranium concentrations measured during domestic well monitoring exceeded 25 µg/L.

The number of wells/properties included in the DWMP and Bottled Water Program was substantially reduced in 2016 (ARC, 2016a; EPA, 2016f). As part of the settlement entered in the class action lawsuit *Roeder et al. v. Atlantic Richfield Company et al.*, D. Nev., Case No. 3-11-cv-00105-RCJ-WGC (Roeder Settlement Agreement), ARC provided funding to the City of Yerington to extend municipal water service to then-existing residences located within that part of the settlement class area that was also within the city's projected future service area. Domestic well owners who connected to the City of Yerington's municipal water system could elect to either abandon their well or apply for a state permit to authorize withdrawals of groundwater for outdoor use only (landscape watering). Each property owner who received a connection to the City Water System executed and recorded an environmental covenant either prohibiting future domestic use of groundwater altogether or limiting it to outdoor purposes.

Construction of the expanded water system began in the Fall 2014 and the construction of new mains and service connections was completed in June 2016. The first phase of well abandonments and system testing was completed as of August 1, 2016. The water system is functional, and domestic wells for all participating property owners have been abandoned or disconnected from the residences within the expansion area. A relatively small number of domestic wells (see Figure 3-12) were not disconnected or converted to outdoor use only in 2016. In 2017 ARC remained in communication with the owners of these wells and connected seven additional properties to the City of Yerington's municipal water system. Since 2017, no other wells have been connected to City Water System. The NDEP recently posted the bottled water cessation criteria and indicated that if access to municipal water is available, but the homeowner refuses to connect, they will no longer be eligible for bottled water service. No domestic well owners have been added to the Bottled Water Program since the Roeder Settlement Agreement.

4 PHYSICAL CHARACTERISTICS OF THE STUDY AREA

This section describes the physical characteristics of the Study Area including demographics, land use, climate, topography, geology and soils, hydrology and groundwater, ecological setting, and vegetation.

4.1 Demographics and Study Area Land Use

Lyon County, Nevada covers approximately 1,993 square miles, and its population in 2013 was 51,585 (U.S. Census Bureau, 2015). Communities near the Site include Yerington (population 3,486), Weed Heights (population 500), and the YPT (approximate population 575). The regional population and industrial centers near the Site include Fernley (47 miles north), Fallon (59 miles northwest), Hawthorne (57 miles southeast), and Reno (85 miles northwest). Yerington's economic base is primarily agriculture.

Land use has included mine operations, ranching, agriculture, urban development, establishment of the YPT colony, BLM range land, and residential development. Mason Valley has long been the largest agricultural area in the Walker River Basin and the most productive area in Nevada. In 2000, a total of 88,600 acres of irrigated land was mapped in the Walker River Basin. Total irrigated land included 39,100 acres (44%) in Mason Valley (Lopes and Allander, 2009a).

4.2 Climate

Nevada is located on the leeward side (rain shadow) of the Sierra Nevada mountain range, which results in a dry climate. The climate in Lyon County is warm and arid. Snow melt is the primary natural source of streamflow and groundwater recharge in the Walker River Basin (Lopes and Allander, 2009b).

The average annual precipitation in Yerington is approximately 5.1 inches, and average snowfall is 6.7 inches (Western Regional Climate Center [WRCC], 2015). The annual average precipitation rate is low relative to the regional pan evaporation rate of about 69 inches per year. The average monthly temperature for the period of record (March 1, 1894 through January 20, 2015) ranges from a maximum of 92.4 degrees Fahrenheit (°F) in July to a minimum of 17.8°F in January.

Tables 4-1 and 4-2 summarize monthly climate data for the City of Yerington weather station for the period from 1894 through 2015 (WRCC, 2015). Table 4-1 summarizes monthly minimum and maximum temperatures, and monthly precipitation. Table 4-2 provides monthly average, maximum, and minimum precipitation values, and the one-day maximum rainfall event.

Table 4-1. Average Monthly Climate Data for the Yerington Weather Station #269229 (1894 - 2015)

	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Annual
Average Maximum. Temperature (°F)[1]	46.2	52.5	59.7	67.0	75.1	83.8	92.4	91.0	83.1	70.8	56.8	47.1	68.8
Average Minimum Temperature (°F)[1]	17.8	22.6	27.0	32.4	40.2	46.8	52.7	50.4	42.3	33.3	23.5	17.9	33.9
Average Total Precipitation[2]	0.57	0.53	0.42	0.41	0.63	0.46	0.26	0.25	0.24	0.35	0.42	0.52	5.06
Average Snow Fall[2]	1.9	1.2	1.0	0.4	0.1	0.0	0.0	0.0	0.0	0.1	0.8	1.2	6.7

Notes:
1. °F = Degrees Fahrenheit
2. Precipitation values in inches

Table 4-2. Precipitation Data for the Yerington Weather Station #269229 (1894 - 2012)

Month	Mean	Maximum	Year	Minimum	Year	1-Day Maximum (Year)
January	0.57	3.67	1916	0.00	1915	1.40 (1943)
February	0.53	2.62	1962	0.00	1953	1.28 (1962)
March	0.42	1.83	1991	0.00	1914	0.98 (1941)
April	0.41	1.80	1990	0.00	1916	1.30 (1990)
May	0.63	3.04	1995	0.00	1916	1.90 (1939)
June	0.46	2.01	1997	0.00	1895	1.02 (1997)
July	0.26	2.00	2003	0.00	1916	1.75 (1984)
August	0.25	2.37	1983	0.00	1895	1.46 (1983)
September	0.24	2.15	1955	0.00	1920	2.02 (1955)
October	0.35	3.02	1993	0.00	1895	1.83 (1993)
November	0.42	2.39	1965	0.00	1894	1.04 (1974)
December	0.52	3.51	1955	0.00	1917	2.00 (1955)
Annual	5.06	10.58	1983	1.61	1947	2.02 (1955)

Notes:
1. Precipitation values presented in inches.
2. Most minimum values (11 of 12 months) of 0.00 inches were recorded prior to 1920.

Wind speed and direction at the Site vary on the local scale due, in part, to the heterogeneous natural topography (i.e., micro-climates) and modified topography from surface mining operations. Meteorological data collected since 2002 indicate that wind direction is variable at the Site with no quadrant representing over 50% of the total measurements. However, when wind speeds are above 15 miles per hour, a predominant wind direction from southwest to northeast has been documented (BC, 2008c).

4.3 Topography

The Site is in Mason Valley, which is a north-south trending structural valley (graben) within the Basin and Range physiographic province filled with up to 1,000 feet of unconsolidated sediments. Mason Valley occupies a structural graben (i.e., down-dropped faulted basin) immediately east of the Singatse Range, Desert Mountains to the north, and the Wassuk Range to the east. Elevations in the Wassuk and Singatse Ranges reach 9,000 and 6,700 feet amsl, respectively (Huxel and Harris, 1969). Alluvial fans occur along the boundaries of Mason Valley and the

Wassuk and Singatse Ranges and are characterized as moderate (1-6%) to steep (up to 25%) slopes between the valley floor and mountain ranges.

The valley ranges in width from about nine miles in the south to nearly 15 miles in the north and is about 40 miles long. The valley floor slopes gradually to the north and ranges from approximately 4,600 feet amsl in elevation at the south end to 4,290 feet amsl at the north end. The center of the Process Areas is at an elevation of approximately 4,450 feet amsl and is situated above the valley floor on an alluvial fan between the Singatse Range and Ground Hog Hills.

Walker River flows south to north through Mason Valley and topography in the valley is consistent with cross-valley migration of the Walker River. Valley floor topography is generally flat and slopes to the north, with isolated bedrock outcrops such as Mason Butte that create areas of higher elevation. Remnant river channels on the valley floor are present that create localized topographical differences on the order up to +/- 10 ft elevation change. At many places along the margins of Mason Valley, in particular where the western edge of the valley floor meets alluvial fans on the Singatse Range, there are topographical depressions. A topographic depression occurs just northwest of the Evaporation Ponds (near B/W-32). Another topographic depression occurs at the north end of the valley where the Wabuska Hot Springs discharge at the surface. These topographic depressions are five to 10 ft lower than adjacent areas on the valley floor.

The topographic depression near B/W-32 is approximately five to seven ft lower than the Hunewill Ranch fields to the east. This depression occurs immediately adjacent to alluvial deposits from the Singatse Range but is not associated with a single large drainage from the range or a well-defined alluvial fan. The alluvial fan near Sunset Hills is situated just to the north of this topographical depression.

Rainfall runoff occasionally ponds in some of these topographic depressions, but recharge to groundwater from tributary surface flows in ephemeral drainages account for less than one half of one percent of the total recharge in the groundwater model and recharge from precipitation falling directly on the valley floor is negligible (SSPA 2014).

4.4 Ecological Setting

The Study Area is part of the Great Basin sagebrush-steppe ecosystem (Lopes and Allander 2009a). The Singatse Range to the west and the Mason Valley is dominated by a scrub brush community, except along the Walker River where a riparian community occurs. These communities support resident and migrating birds, mammals, reptiles, amphibians, and invertebrates. The Walker River flows within 0.25 miles of the southeastern end of the Site. Although riparian systems comprise an extremely small fraction of the Great Basin region, they are critical centers of biodiversity; more than 75% of the species in the region are strongly associated with riparian vegetation. The Study Area ecosystem has been impacted by anthropogenic activity, including mining, cattle ranching, and agriculture. Site activities have resulted in the large piles of tailings and waste rock, which could be used as vantage points for predators surveying the surrounding area, and steep-sloped piles may potentially be used by nesting birds (e.g., swallows).

4.5 Vegetation

The terrestrial ecosystem in the Study Area not disturbed by anthropogenic activities supports an arid sagebrush-steppe vegetative community that is dominated by sagebrush and other low-lying woody vegetation, interspersed with a variety of forbs and grasses. The scrub brush community in the Study Area is predominately sparse greasewood, sagebrush, and rabbitbrush (Lopes and Allander, 2009a). Livestock and wildlife preference for grasses contributes to the domination of vegetation in this system by sagebrush and other shrubs (Ricketts et al., 1999).

The riparian community along the Walker River supports a variety of trees, shrubs, and grasses (Lopes and Allander, 2009b). Vegetation can be dense with large trees such as Freemont cottonwood, Russian olive, and invasive Tamarisk (Salt Cedar). Saltbush may be abundant where riverbank soil is saline. As previously stated, many areas on the Site have been disturbed to varying degrees by historical mining activities, but still retain areas of sandy soil interspersed with vegetation typical of the sagebrush-steppe vegetative mix of shrubs, forbs, and grasses. Mason Valley has long been the most agricultural part of the Walker River Basin and remains one of the most productive agricultural areas in Nevada (Lopes and Allander, 2009a). During the growing season, agricultural fields to the north may include onions, alfalfa, winter wheat, and sorghum.

4.6 Regional and Site Geology

Mason Valley is a structural graben that has been filled with unconsolidated alluvial deposits derived by erosion of the emerging mountain horst blocks, and from materials transported into the valley by the East and West Walker Rivers (Huxel and Harris, 1969). The alluvial apron and the valley floor are the two major landforms comprising the lowland area. The mountain blocks, and bedrock beneath the basins, are primarily composed of granitic, metamorphic, and volcanic rocks of Precambrian to Tertiary age and, to a lesser extent, of consolidated to semi-consolidated sedimentary rocks of Paleozoic to Cenozoic age (Heath, 1984; Proffett and Dilles, 1984; Proffett, 1977). Faults along the eastern margin of the Singatse Range are gently to steeply dipping normal faults that generally trend north-northeast (Figure 4-1) and dip to the east (Proffett and Dilles, 1984; Proffett, 1977). Faulting caused moderate to steeply westward tilting of the bedrock.

Unconsolidated deposits underlying the valley floor are collectively termed valley-fill deposits and, where saturated, constitute the valley-fill alluvial aquifer. Huxel and Harris (1969) reported that the valley-fill deposits include four stratigraphic units: 1) younger alluvium, including lacustrine deposits associated with Pleistocene Lake Lahontan (Reheis, 1999); 2) younger alluvial fan deposits resulting from the uplift of mountain blocks; 3) older alluvium; and 4) older alluvial fan deposits.

Older and younger alluvial fan deposits are generally coarse-grained, poorly-sorted, and have relatively few inter-bedded clay lenses (Huxel and Harris, 1969; Plume, 1996; Mifflin, 1988). The grain size of the valley-fill deposits generally decreases toward the center of Mason Valley (Huxel and Harris, 1969; Plume, 1996), and transitional facies have been identified in the Study Area (BC, 2008c). Basin-scale variability in valley-fill deposits leads to variation in hydraulic properties of the alluvial aquifer, which is discussed in Section 4.9.6.

Bedrock and alluvial deposits in Mason Valley, and their associated hydrologic characteristics, are summarized in Table 4-3, which is reproduced from Huxel and Harris (1969). Lake Lahontan lacustrine deposits of Pleistocene age have been eroded or reworked by the Walker River as it meandered across Mason Valley. Lake Lahontan strandline units, consisting of beach, bar, and beach-ridge deposits, were formed for the most part on alluvial aprons between altitudes of 4,340 and 4,375 feet amsl (Huxel and Harris, 1969). The occurrence of Lake Lahontan within Mason Valley had a geologically short life, and probably was less than 60 feet deep during much of its existence (Morrison, 1964).

Table 4-3. Mason Valley Geologic Units: Lithologic and Hydrologic Characteristics (from Huxel and Harris, 1969)

Geologic Age		Geologic Unit		Thickness (feet)	Lithology	Hydrologic Characteristics
Quaternary	Pleistocene to Holocene	Valley Fill	Younger Alluvium	0-100±	Loose, well-sorted sand, gravel, cobbles, and boulders, with layers of silt or sandy clay. Comprises channel, floodplain, and terrace deposits laid down by the Walker River and its major tributaries, plus strandline and bottom deposits of Pleistocene Lake Lahontan. Bottom deposits consist of silt, fine sand, and clay.	Channel and floodplain deposits are highly permeable and are good aquifers. Significant infiltration of surface waters, which recharges the alluvial-fill aquifer, occurs through the coarse deposits in the Holocene channels of the Walker River.
			Younger Fan Deposits	0-100±	Poorly-sorted gravelly clay, sandy clay, and fine sand with occasional stringers and lenses of sand and gravel. Locally, derived from erosion of older rocks and deposits in Mason Valley; generally equivalent to younger alluvium.	In general, younger and older fan deposits are of low permeability. However, stock watering and mining wells penetrating buried sand and gravel deposits yield small to moderate amounts of water. Properly constructed, large-diameter wells may yield up to several hundred gpm.
	Pleistocene		Older Fan Deposits	0-700±	Sandy- to gravelly-clay with abundant cobbles and boulders and occasional lenses of semi-consolidated to cemented sand and gravel. Locally-derived from erosion of consolidated rocks of the surrounding mountains. Equivalent in part to older alluvium.	
			Older Alluvium	0-500±	Similar in lithology to younger alluvium described above. Deposited by ancestral Walker River; underlies valley floor at depths greater than 100 feet. Not exposed at land surface.	Constitutes largest and most productive aquifer in the area, with tested transmissibility as high as 270,000 gallons per day/foot. Wells yield up to 3,000 gpm.
Tertiary	Miocene and Pliocene	Consolidated Rocks	Sedimentary Rocks	--	Sandstone, mudstone, shale, marl, diatomite, and limestone. Includes interbedded tuffaceous rocks, lava flows, and breccia.	Consolidated rocks generally have low permeability. However, where they are fractured or jointed, they yield small to moderate amounts of water to wells.
	Oligocene to Pliocene		Volcanic Rocks	--	Rhyolite flows and tuff, andesite and dacite lava flows, breccia, and agglomerate. Includes interbedded sedimentary rocks and, locally, thin basalt flows with interbeds and scoriaceous basalt breccia.	
Cretaceous			Granitic Rocks	--	Granodiorite, quartz monzonite, and granite porphyry.	

Table 4-3. Mason Valley Geologic Units: Lithologic and Hydrologic Characteristics (from Huxel and Harris, 1969)

Geologic Age		Geologic Unit		Thickness (feet)	Lithology	Hydrologic Characteristics
Permian to Jurassic			Metamorphic Rocks	--	Metamorphosed andesite, basalt, and rhyolite flows, tuff and breccia, metamorphosed limestone, lime shale, dolomite, and gypsum and volcanically-derived sedimentary rocks.	

Bedrock in the Study Area forms a U-shaped graben structure that reaches its lowest point beneath the north end of the Hunewill Ranch, at an elevation of approximately 3,600 feet amsl (700 feet bgs). The elevations of the alluvium-bedrock contact, shown in plan view on Figure 4-2, clearly depict this graben structure in the Study Area. From its lowest elevation, bedrock rises in elevation south toward the Site. The U-shaped graben ends at the open pit and the alluvial-bedrock contact is exposed on the pit walls. The bedrock exposed in the open pit is the host rock for the Yerington copper porphyry deposit. East and west of the Site, bedrock rises to mapped outcroppings associated with the Singatse Range (west) and Singatse Spur (east); this term refers to two adjacent bedrock outcrops located east of the Site called the Ground Hog Hills and McLeod Hills as shown on Figure 4-2. At the north end of the Study Area, bedrock outcrops occur in the Sunset Hills area. In the northeast portions of the Study Area (i.e., toward the Mason Butte bedrock outcrop), bedrock rises in elevation. Range-bounding faults in the Study Area include steeply-dipping and shallower-dipping normal faults (Proffett and Dilles, 1984).

The unconsolidated alluvial deposits in the Study Area were derived primarily from erosion of the uplifted mountain block of the Singatse Range, with minor deposition of fluvial sediments in the Walker River floodplain. In addition, lacustrine deposits derived from ancestral Lake Lahontan occur north of the Site (Reheis, 1999). Uplift and erosion of the Singatse Range formed the east-dipping alluvial fan deposits, which include distal facies that extend into the transitional environment. Concurrent with the development of the alluvial fan, flat-lying fluvial sediments (e.g., sands and gravels) were deposited in the Walker River floodplain. Flat-lying clay-rich deposits have been preserved in the transitional setting, and these deposits are interpreted to have formed within the ancestral Lake Lahontan depositional environment.

Regional metal mineralization and hydrothermal alteration occurs in portions of Mason Valley, and the Singatse Range in particular, in areas of porphyry and skarn copper deposits. The Yerington copper porphyry district is located within the productive Walker Lane mineralized belt in western Nevada (Tetra Tech, 2010). The Walker Lane is a northwest-trending zone of active crustal movement (i.e., right-slip transcurrent faulting) that extends for more than 500 miles from Las Vegas, Nevada to beyond Honey Lake, California (Bell and Slemmons, 1979).

In addition to the Yerington and MacArthur open pit mines along the eastern margin of the Singatse Range, other areas of mineralization include the Bluestone and Ann Mason mines, and the Bear deposit. Areas of known mineralization and ore deposits in the Mason Valley are shown on Figure 4-3. The Bear deposit is located beneath the Sulfide Tailings and Hunewill Ranch, in a structurally-uplifted segment of the Singatse Range. The Pumpkin Hollow copper skarn deposit, located across Mason Valley from the Site, occurs along the margin of the intrusive rocks that host the Yerington porphyry copper deposit.

4.7 Surface Water Hydrology

The Mason Valley Basin (Basin no. 108, as defined by the NDWR) is located within the larger Walker River Hydrographic Basin (no. 9). The Walker River Hydrographic Basin extends from the Sierra Nevada Mountain Range above Bridgeport, California and Topaz Lake to Walker Lake located north of Hawthorne, Nevada. Most streamflows in the basin originate as snowmelt in the Sierra Nevada, with headwaters at elevations of more than 12,000 feet amsl (Lopes and Allander, 2009a, 2009b).

The Walker River originates in two distinct headwater areas in the Sierra Nevada that source the East and West Walker Rivers (Figure 3-6). The East Walker River is sourced above Bridgeport, California. Streamflows are regulated before flowing into the Mason Valley. The West Walker River is sourced above Topaz Lake, a reservoir located along the California-Nevada border, and passes through the town of Wellington, Nevada on its way to the Mason Valley. The confluence of the East and West Walker Rivers occurs in Mason Valley at a location approximately seven miles upstream (three miles directly south) of the Site. The main stem of the Walker River flows north past the Site at the City of Yerington, traverses the geothermal discharge area near the town of Wabuska, exits the north end of Mason Valley at Walker Gap approximately 4.5 miles east of the town of Wabuska, and then turns south and empties into Walker Lake (a terminal lake with no outlet).

4.7.1 Surface Water Flows

Mason Valley is the largest irrigated agricultural area within the Walker River Basin including irrigated areas along the West and East Forks, and the main stem, of the Walker River. Key documents providing information on streamflows and water budgets in the Mason Valley include Huxel and Harris (1969), Lopes and Allander (2009b), and Carroll et al. (2010).

Each of these three documents present information on streamflows and water budgets for different periods of time. Appendix G-1 presents surface water flow information for 1948 to 2001, a period longer than addressed in these three documents. Streamflow and water budget information from these three documents and Appendix G-1 are summarized in Table 4-4 and discussed below.

Table 4-4. Summary of Mason Valley Streamflow and Water Budget Information

	Huxel and Harris (1969)	Lopes and Allander (2009b)	Carroll et al. (2010)	Flow Data Appendix G-1
Period of Record	1948 - 1965	1971 - 2000	1996 - 2006	1948 - 2011
	18-year Average	30-year Average	11-year Average	48-year Average (1)
Stream Inflows (acre-feet) (2)	216,000	269,000	277,832	207,900
Stream Diversions (acre-feet)	140,000	117,000	139,643	NA
Stream Outflows (acre-feet) (3)	107,200	138,000	129,471	80,400
Total Stream Loss (acre-feet) (4)	109,300	131,000	148,361	127,500
Stream Loss as Percent of Inflow	50%	49%	62%	NA
Irrigated Area (acres)	30,000	38,964	38,721	NA
Surface Water Diversion Rate (ft/yr) (5)	3.6	3.4	3.8	NA
Groundwater Pumpage (acre-feet/yr)	4,000	40,000	77,423	NA
Groundwater Application Rate (ft/yr) (6)	0.1	1.0	2.0	NA
Crop Consumption Rate (ft/yr)	1.0	1.6 (7)	2.9 - 3.1	NA

Notes:

1. Excludes 1979 - 1994 because flow data were not collected during winter months (October through March).
2. Sum of streamflow at Hudson (East Walker River) and Strosnider Ditch (West Walker River) gages (USGS gages 10300000 and 10293500, respectively).
3. Streamflow at Wabuska gage (USGS gage 10301500).
4. Total Stream Loss = Stream Inflows - Stream Outflows.
5. Surface Water Diversion Rate = Total Stream Loss/Irrigated Area.
6. Groundwater Application Rate = Groundwater Pumpage/Irrigated Area.
7. Value of 1.6 ft/yr from Myers (2001) cited by Lopes and Allander (2009b).
8. ft/yr = feet per year; % = percent; NA = not available

Lopes and Allander (2009b) provide a surface water budget for Mason Valley based on data collected from 1971 to 2000. Combined average annual inflows to Mason Valley were estimated to be 269,000 acre-feet per year. The average annual outflow from Mason Valley was estimated to be 138,000 acre-feet per year. The average net annual diverted surface water in Mason Valley was estimated to be 117,000 acre-feet per year. Stream infiltration and riparian evapotranspiration was estimated to be 14,000 acre-feet per year (Lopes and Allander, 2009b).

An analysis of Walker River streamflow data from 1948 to 2011 (Appendix G-1) indicates a median annual flow at the confluence of the East and West Walker Rivers of 207,900 acre-feet.

The minimum recorded flow was 65,900 acre-feet per year, and the maximum recorded flow was 596,500 acre-feet per year. Outflows from the Mason Valley are recorded near Wabuska, north of the Study Area. The median annual outflow was 80,400 acre-feet. The minimum recorded outflow was 15,800 acre-feet per year. The maximum recorded out flow was 417,900 acre-feet per year. In all months of all years, combined flows at the confluence of the East and West Walker Rivers were greater than outflows from the Mason Valley, with greater differences observed in summer months compared to winter months. Differences between inflows and outflows are accounted for by seepage losses, evapotranspiration and diversions for crop irrigation.

The disposition and routing of surface water within the Mason Valley is complex. Detailed information is provided in the report *Groundwater Flow Model Yerington Mine Site* (SSPA, 2014).

4.7.2 Recharge to the Alluvial Aquifer

Percolation of surface water is the primary source of groundwater recharge to the alluvial aquifer in the Mason Valley, with mountain-front recharge (MFR) contributing significantly less (Carroll et al., 2010; Huxel and Harris, 1969; Myers, 2001; SSPA, 2014). The amount of recharge derived by infiltration from stream channels, ditches, and percolation from flooded agricultural fields varies from year to year, depending upon the volume of Walker River flow entering the basin, the amount of surface water diverted from the river for irrigation, and the amount of available groundwater storage.

Huxel and Harris (1969) estimated that the annual recharge from the sources listed above ranged from 30,000 to 100,000 acre-feet, with an average of about 70,000 acre-feet, for the period from 1948 to 1965. These estimates were calculated as inflows minus the sum of surface-water outflows and consumptive use by crops and pastures and assumed that all streamflows not consumptively used for irrigation or flowing out of the valley recharged the valley-fill alluvial aquifer. Carroll et al. (2010) estimated that recharge from the sources listed above ranged from 60,400 to 99,400 acre-feet per year for the time period 1996 to 2006 and noted the consistency between their more recent estimates and those provided by Huxel and Harris (1969).

The groundwater flow model water budget (SSPA, 2014) indicates that the alluvial aquifer is primarily recharged by downward percolation from irrigated fields (49%), leakage from irrigation ditches such as the West and East Campbell Ditches (29%), infiltration from the channel of the Walker River (20%), and infiltration through bedrock of precipitation that falls on surrounding mountain ranges and minor tributary surface flows in ephemeral drainages (2%). Recharge from precipitation falling directly on the valley floor is negligible based on work by Huxel and Harris (1969) and Lopes and Allander (2009a, 2009b), as well as data from stable isotope (i.e., oxygen/deuterium) analysis of precipitation and groundwater (BC, 2014a; EPA, 2012b).

4.7.3 Surface Water Quality

Surface water quality is discussed below with a focus on data at sample locations SW-WR-01 (Walker River) and SW-WCD-01 (West Campbell Ditch), and chemicals considered primary indicators of mine-influenced groundwater (i.e., sulfate and uranium). A detailed analysis of the surface water quality data is presented in Appendix G-2. Chemical concentrations in Walker River and West Campbell Ditch samples are similar with low TDS (110 to 300 mg/L; average: 194 mg/L) and relatively low sulfate (7.7 to 54 mg/L; average: 29 mg/L) and dissolved uranium (3.7 to 19 µg/L; average: 9.3 µg/L). Surface water pH is slightly alkaline (7.72 to 8.36 s.u.; average: 8.05 s.u.). Temporal trends indicate more elevated major ion (e.g., sulfate, calcium, and chloride) values and trace metal (e.g., dissolved arsenic and dissolved uranium) values in samples at both locations during the winter months relative to the summer months. Overall, the STORET surface water quality data is similar to the surface water quality data collected by ARC.

Shallow Zone groundwater near the Walker River and West Campbell Ditch exhibits similar chemical characteristics to surface water quality, which is the primary source of groundwater in Mason Valley (Carroll et al., 2010; Huxel and Harris, 1969; Myers, 2001; SSPA, 2014). General ion chemistry in groundwater was found to be statistically similar to surface water for five of eight major ions. Calcium, chloride, and sulfate were found to be higher in groundwater than surface water. Dissolved metals in groundwater were found to be statistically similar to surface water for some parameters (14 of 27) but different for others (13 of 27). However, differences between surface water and Shallow Zone groundwater in major ions and dissolved metals reflect changes in geochemical conditions in groundwater arising from the effects of residence time, presence (or absence) of dissolved atmospheric gasses, mineralization, and influences from land-surface features that alter groundwater quality as it recharges from surface water sources.

4.8 Mason Valley Regional Hydrogeology

Groundwater conditions in Mason Valley are based on 1) general characteristics of groundwater flow for the Basin and Range province; 2) investigations specific to the Mason Valley and/or the Walker River Basin; and 3) groundwater data available from the USGS and/or NDWR. The general conceptual model for groundwater flow in the Basin and Range province (Heath, 1984; Maurer et al., 2004) is movement of groundwater in unconsolidated sediments deposited within the basins that occur between uplifted mountain blocks comprised of consolidated bedrock.

The groundwater flow system in Mason Valley consists of 1) a heterogeneous valley-fill alluvial aquifer system comprised of laterally-discontinuous confining units of clay or other low-permeability sediments, and unconfined (i.e., water table), semi-confined, and confined aquifers; and 2) a relatively impermeable bedrock flow system underlying and bounding the alluvial aquifer with low permeability and groundwater flow, primarily focused along faults and fractures (Maurer et al., 2004; Thomas, 1995; Tetra Tech, 2010; Huxel and Harris, 1969).

Groundwater in the alluvial aquifer within Mason Valley generally flows from south to north toward the topographically lowest part of the valley at the northern end of the valley (Figure 4-4). Similar water-level patterns are depicted in Huxel and Harris (1969, Plate 2), Lopes and Allander (2009a, Plate 1), and Tetra Tech (2010, Figure 24-3). Locally, groundwater flow directions are affected by 1) bedrock outcrops within and along the margins of the valley; 2) drawdown from pumped wells; and 3) irrigation activities on cultivated fields. The Walker River is generally a losing stream except in the far northeastern portion of the valley where it is generally a gaining stream. Water level elevations shown on Figure 4-4 are based on USGS monitor well data from October through December 2010, a period when agricultural pumping is low or non-existent. Table 4-5 summarizes the USGS wells and water level data used to develop the water table (i.e., alluvial aquifer potentiometric surface) map shown on Figure 4-4.

Table 4-5. 2010 USGS Monitor Well Data for Mason Valley

USGS Site Number	Site Name	Latitude	Longitude	Surface Elev. (feet amsl)	Vertical Datum	Measure-ment Date	Depth to Water (feet)	Water Elev. NGVD29 (feet amsl)	Well Depth (feet bgs)	Boring Depth (feet bgs)	Number of Measur-ements
391655119330901	103 N16 E22 06ACD1 HIWAY 50	39.28200000	119.5524167	4352.1	NAVD88	10/13/10	55.4	4293.29	96	96	154
391729119294501	103 N17 E22 34DBDD1 EUREKA	39.29147220	119.4957500	4283.8	NAVD88	10/14/10	10.06	4270.35	35	35	101
391711119303301	103 N16 E22 04AAAD1 RA-4	39.28647220	119.5091667	4288.5	NAVD88	10/14/10	4.8	4280.30	14	14	69
391625119324801	103 N16 E22 07AAAA1 R-3	39.27352778	119.5468056	4303.9	NAVD88	10/14/10	9.07	4291.42	35	35	66
391605119331901	103 N16 E22 07ACCB1 R-2	39.26797220	119.5551389	4308.1	NAVD88	10/14/10	8.27	4296.41	30	30	64
384942119100801	108 N11 E25 10DBCD1	38.82802778	119.1703610	4565	NGVD29	11/23/10	98.14	4466.86	597	597	42
390715119095901	108 N15 E25 34ACDD1	39.12075170	119.1673767	4292	NGVD29	11/22/10	13.48	4278.52	370	370	41
390006119043901	108 N13 E26 09DBCC1	39.00158530	119.0784852	4396	NGVD29	11/24/10	60.37	4335.63	166	166	39
390011119060201	108 N13 E26 08CACA1	39.00297394	119.1015412	4367	NGVD29	11/24/10	21.88	4345.12	130	130	37
390127119030001	108 N13 E26 02BBCC1	39.02408569	119.0509851	4406	NGVD29	11/24/10	87.92	4318.08	203	203	36
390203119055101	108 N14 E26 32BDDD1	39.03408520	119.0984860	4352	NGVD29	11/30/10	13.56	4338.44	104	104	32
385903119073001	108 N13 E25 13DDDD1	38.98408457	119.1259859	4380	NGVD29	11/24/10	16.92	4363.08	280	280	32
390531119115901	108 N14 E25 08ADDC1	39.09186235	119.2007101	4320	NGVD29	11/22/10	30.73	4289.27	523	523	32
385720119085001	108 N13 E25 26DDCC1	38.95547285	119.1482085	4409	NGVD29	11/24/10	26.14	4382.86	160	NA	31
385255119090501	108 N12 E25 23DCC 1	38.88186075	119.1523750	4462	NGVD29	11/23/10	15.84	4446.16	325	325	31
385456119091901	108 N12 E25 11CACD1	38.91547224	119.1562641	4439	NGVD29	11/23/10	21.5	4417.50	245	245	31
390137119065402	108 N14 E26 31DCCC2	39.02686280	119.1159861	4357	NGVD29	11/30/10	13.44	4343.56	400	400	30
390558119094701	108 N14 E25 03DDDC1	39.09936270	119.1640431	4323	NGVD29	11/22/10	20.54	4302.46	85	258	30
390611119110301	108 N14 E25 04DACC1	39.10297367	119.1851545	4321	NGVD29	11/22/10	20.75	4300.25	451	451	30
385447119075901	108 N12 E25 12CDAA1	38.91297248	119.1340414	4476	NGVD29	11/23/10	59.94	4416.06	102	102	28
390004119103001	108 N13 E25 10CDB 1	39.00102868	119.1759868	4380	NGVD29	11/30/10	9.19	4370.81	328	328	27
390026119090401	108 N13 E25 11ACBD1	39.00714008	119.1520976	4370	NGVD29	11/24/10	13	4357.00	435	435	25
385717119080901	108 N13 E25 25CDDA2	38.95463960	119.1368194	4419	NGVD29	11/24/10	33.88	4385.12	106	106	21
385109119085601	108 N12 E25 35DCDD2	38.85241595	119.1498750	4505	NGVD29	11/22/10	35.38	4469.62	NA	NA	20
385003119085201	108 N11 E25 11AACC1	38.83408240	119.1487638	4565	NGVD29	11/23/10	97.14	4467.86	256	256	19
390057119080001	108 N13 E25 01DBCC1	39.0157514	119.1343196	4365	NGVD29	11/19/10	19.96	4345.04	570	570	19

385047119080401	108 N11 E25 01ACCB1	38.84630495	119.1354302	4547	NGVD29	11/23/10	75.13	4471.87	526	526	18
385018119091101	108 N11 E25 02CDDD1	38.83813889	119.1538889	4544	NGVD29	11/23/10	73.59	4470.41	554	560	17
385718119101301	108 N13 E25 27DCCD2	38.95491700	119.1712645	4409	NGVD29	11/23/10	19	4390.00	440	440	17
390152119104401	108 N14 E25 34CBCA2	39.03102880	119.1798760	4362	NGVD29	11/30/10	25.56	4336.44	415	430	14
391741119150601	102 N17 E24 35DAAB1 OLD BUCKLANDS STATION	39.29472220	119.2516667	4203	NGVD29	10/12/10	16.23	4186.77	93	93	14
391610119115801	102 N16 E25 05DCCA1 USBLM	39.27464167	119.2004333	4219	NGVD29	10/12/10	70.24	4148.76	127	NA	12
391757119151801	102 N17 E24 35ACAA1 OLD WEEKS SIDING	39.29916667	119.3050000	4206	NGVD29	10/12/10	18.28	4187.72	23	23	11
392522119101901	102 N18 E25 15CBCA1 STUCCO	39.42288889	119.1718889	4213	NAVD88	10/12/10	57.72	4152.13	200	200	11
392546119121201	102 N18 E25 17BDAA TRAILER GRAVEYARD	39.42947220	119.2034167	4201	NAVD88	10/12/10	14.92	4182.94	170	170	11
392222119075101	103 N17 E25 01BAB1 E OF LAHONTAN	39.37283330	119.1307222	4202	NAVD88	10/12/10	61.18	4137.68	72	72	9
390416119112401	108 N14 E25 16DCCB1 CMPBLL SHALLOW	39.07097220	119.1900833	4336	NGVD29	11/22/10	20.14	4315.86	25	25	8
385249119221401	107 N12 E23 26ABAD1 85471	38.8803611	119.3706667	4729	NGVD29	11/18/10	13.76	4715.24	340	340	8
391727119190701	103 N17 E24 32CDBB1 BULL CANYON	39.29088889	119.3184722	4250.5	NAVD88	11/15/10	27.8	4219.363	41	41	8
391706119322601	103 N16 E22 05BACD1	39.28488889	119.5405556	4345	NAVD88	10/23/10	59.9	4281.691	182	182	8
391709119314001	103 N16 E22 04BBCB1 CARDELLI	39.28569444	119.5278333	4304.5	NAVD88	11/15/10	15.85	4285.244	39	39	2

Notes:
All groundwater elevation data downloaded from USGS website http://nwis.waterdata.usgs.gov/nv/nwis/gwlevels, accessed on August 15, 2011.
amsl = above mean sea level; bgs = below ground surface; NA = not available.

Hydraulic properties of the basin-fill sediments of Mason Valley vary both laterally and vertically because of variable depositional facies and environments observed in the valley. The transmissivity of the basin fill deposits was stated by Huxel and Harris (1969) to generally range from 6,700 square feet per day (ft^2/day) to 27,000 ft^2/day. Based on an average basin-fill thickness of 500 feet, this is equivalent to average hydraulic conductivities in the range of 13 feet per day (ft/day) to 53 ft/day. Based on unspecified geotechnical investigations, Nork (1989) reported hydraulic conductivity values ranging from 0.35 to 0.50 ft/day for older alluvial fan deposits. Consolidated rocks beneath the unconsolidated basin-fill sediments and/or comprising the adjacent mountain ranges have low hydraulic conductivities but may transmit water where fractures are open and interconnected (Lopes and Allander, 2009a). Nork (1989) reported hydraulic conductivity values ranging from 0.23 to 1.5 ft/day for weathered intrusive rocks in the Mason Valley area.

Groundwater in Mason Valley is primarily recharged by downward percolation of surface water diverted from the Walker River to irrigation ditches and irrigated fields, downward percolation of groundwater applied to irrigated fields, and infiltration of surface water from the channel of the Walker River. To a lesser degree, valley-fill sediments are also recharged by MFR, which includes a variety of hydrologic processes such as partitioning of precipitation and snowmelt into deep infiltration through bedrock (i.e., along faults and fractures), surface runoff, focused flow and subflow along mountain stream channels and alluvial fans, and diffuse movement of groundwater through the underlying mountain block (Wilson and Guan, 2004; Huxel and Harris, 1969; Myers, 2001). Huxel and Harris (1969) considered recharge from direct precipitation on the valley floor to be negligible. Recharge from irrigation water and seasonal pumping of irrigation wells affects the vertical flow of groundwater in the alluvial aquifer (i.e., a seasonal increase in the downward vertical gradient in the alluvial aquifer).

Discharge from the Walker River Basin occurs as evapotranspiration from irrigated crops and natural vegetation (e.g., phreatophytes and wetland vegetation) as described by Heath (1984) and Carroll et al. (2010), and as direct evaporation from shallow groundwater (Huxel and Harris, 1969; Lopes and Allander, 2009a). Huxel and Harris (1969, Plate 2) identified an area of artesian conditions (17 flowing wells) in the northern portion of Mason Valley where the alluvial aquifer thins and pinches out and reported that groundwater in this area exhibited elevated specific conductance values (i.e., dissolved solids concentrations) due to evapoconcentration and possible effects of geothermal discharge associated with the Wabuska Lineament.

Lopes and Allander (2009a, 2009b) report that: 1) in 2008, no flowing wells were observed in the Wabuska area due to groundwater pumping; 2) water depths in this area were less than five feet, and efflorescent salts formed where groundwater evaporated from the shallow water table; and 3) pumping in Mason Valley since the early 1960s had caused groundwater levels to decline as much as 60 feet. The long-term decline in water levels is reflected in Site hydrographs for select monitor wells that cover the time period from 1985 to 2015, as shown on Figure 4-5. In addition to factors described above, the sharp decline in the 1980s in water levels in well UW-1S, located near the northern end of the Process Areas, is in part attributed to cessation of mining activities in 1978. Discharge of groundwater through bedrock from the Mason Valley Basin to other groundwater basins may occur but is not an important component of the water budget (Thomas, 1995; Tetra Tech, 2010).

4.9 Site and Study Area Local Hydrogeology

The Site is in a dead-end alluvium filled depression open only on the north to the main alluvial aquifer in the Mason Valley. The Site is surrounded on three sides by bedrock ridges: the Ground Hog Hills to the east of the Site, the Singatse Range to the west of the Site, and a bedrock ridge in the south. Recharge to the alluvial groundwater system in this dead-end alluvium filled depression is restricted to infiltration from the Weeds Heights sewage treatment ponds, which is estimated to be about 10 gpm, and flow in the bedrock from the Singatse Range into the alluvial valley, which is estimated to be no more than 3 gpm (SSPA, 2014). Thus, the total recharge to the alluvial system at the Site is only about 13 gpm, which is the total flow available for migration northward from the Site, absent external factors. This is a very small amount of water and, as a result, groundwater velocities at the Site are low.

Immediately to the north of the mine-Site boundary are approximately 718 acres of irrigated fields (Hunewill Ranch). Some of these fields have been irrigated for a long time. Approximately 300 irrigated acres are visible on an air photo from 1954, approximately 600 acres of irrigated fields are visible on a 1967 air photo, and approximately 718 acres of irrigated fields are visible on a 1990 air photo. The initial irrigation well for these fields was constructed in 1961 and two additional irrigation wells were constructed in 1966. One of these wells (WDW019, NDWR log #26694) continued to be used through 2009. Currently, all irrigation water for these fields comes from the Walker River and two irrigation wells located near the river approximately one mile to the east of the Site. Recharge on these agricultural fields from infiltration of irrigation water, on an average annual basis, is estimated to be about 375 gpm (SSPA, 2014). As this recharge rate is significantly higher than that occurring on the Site to the south of the fields, groundwater flow patterns near the Site in the alluvial aquifer system are controlled by this recharge.

As a result of recharge on the agricultural fields immediately north of the Site, groundwater flow in the shallower zones in the northern part of the Site is toward the west and northwest, away from the agricultural fields adjacent to the Site's northeastern boundary. In the deeper alluvial aquifer zones, groundwater flow is generally to the north and northeast.

A generalized hydrogeologic south-north cross-section with a vertical exaggeration of 20V:1H is presented in Figure 4-7a (location shown on Figure 4-6). This cross-section (A-A') transects the Study Area and extends from well B/W-13S, which is the southernmost well in the Study Area, to a monitor well cluster B/W-82R, which is located at the north end of the Study Area. The lithology shown on Figure 4-7a is consistent with the vertical and horizontal complexity expected from basin fill associated with alluvial fan, fluvial, and lacustrine depositional environments (Table 4-3).

The south-north cross-section depicts: 1) the alluvial materials in the valley-fill alluvial aquifer within the Study Area; 2) the alluvial aquifer zone designations; 3) the occurrence of bedrock outcrops at the open pit and the north end of the Study Area; 4) the water table elevation in the alluvial aquifer in August 2015; 5) the depth of the open pit and the Pit Lake level in August 2015; and 6) the cone-of-depression associated with the open pit. The water table in the Shallow Zone of the alluvial aquifer generally slopes toward the north on this cross-section, though the elevation of the water is relatively flat in much of the Site.

The presence of an approximately 10 ft thick clay layer occurring at an elevation corresponding to the Shallow and Intermediate Zone boundary (4,300 ft amsl) has been documented in some boreholes in the vicinity of the LEP. This clay layer has been termed the “50-ft Clay” because it is situated around 50 ft below the LEP. Existing borehole logs beneath and around the Evaporation Ponds were evaluated for the potential presence of the “50-ft Clay” layer to determine the potential for this layer to create a continuous and laterally extensive confining hydro-stratigraphic layer.

Figure 4-7b provides a map showing boreholes evaluated for the “50-ft Clay” and the location of five section lines of clay lithology in Figures 4-7(c-g). For this evaluation, potential “50-ft Clay” locations were conservatively identified by the following criteria: 1) >5 ft thickness; and 2) has any occurrence in the 20-ft elevation range around the Shallow-Intermediate Zone boundary (4,290 - 4,310 ft). Only boreholes that penetrate below an elevation of approximately 4,250 ft and able to provide adequate characterization at an elevation of 4,300 ft were evaluated.

The clay lithology sections (Figures 4-7[c-g]) show that there is no correlation of clay lithology layers for the large horizontal distances between boreholes. Regardless, where there are two adjacent boreholes with clay layers meeting the potential “50-ft Clay” criteria above, a dashed line indicates the potential for continuity between boreholes. There is not continuity of the “50-ft Clay” layer over horizontal distances in excess of 1,000 feet.

The approximate lateral extent of a potentially continuous “50-ft Clay” layer, per the criteria above, is shown on Figure 4-7b. The extent of the potentially continuous layer extends northward from the eastern and northern borders of the LEP. The extent of potentially continuous “50-ft Clay” does not include a large portion of the LEP or UEP. Because of this evaluation of the “50-ft Clay”, and the presence of MIW in the Intermediate Zone and deeper beneath the LEP and UEP (see Section 5.4), this layer is not a laterally extensive confining unit capable of disconnecting the Shallow and Intermediate Zone groundwater beneath the Evaporation Ponds. Where the “50-foot Clay” does exist beneath and north of the LEP, it limits the migration of water from the Shallow Zone to the Intermediate Zone.

4.9.1 Depth to Groundwater

Contour maps of the depth to groundwater (i.e., depth to water table) below the ground surface in February 2015 and August 2015 are presented on Figures 4-8 and 4-9, respectively. February 2015 and August 2015 represent the non-irrigation and irrigation seasons, respectively. The depth to groundwater is typically less than 20 feet beneath irrigated areas such as the Hunewill Ranch and between monitor wells B/W-59S and B/W-68S. In areas beneath the Site, beneath the Sunset Hills neighborhood, and in the northern portions of the Study Area, the depth to groundwater is greater than 20 feet. To the west of the Site and beneath the Process Areas, the depth to groundwater exceeds 100 feet. Between February 2015 and August 2015, the depth to groundwater beneath the irrigated fields immediately north of the Site as well as beneath the Evaporation Ponds changed by less than one foot with water levels in some wells increasing and in other wells decreasing. Depth to groundwater further north and northeast in the Study Area generally increased by several feet during this period as a result of heavy agricultural pumping during the 2015 irrigation season. Very little surface water was available in 2015 for irrigation as a result of minimal snowpack in the Sierras and as a result groundwater pumping was increased to make up for the surface water deficit. The surface water available for irrigation in 2014 and 2015 was the lowest since groundwater irrigation began in the early 1960's. Groundwater levels beneath the Evaporation Ponds and the adjacent irrigated fields changed little in 2015 relative to levels further to the north and northeast, in part, because groundwater applied to the fields adjacent to the ponds was obtained from wells located about one mile east of the fields near the Walker River.

4.9.2 Saturated Alluvial Thickness

The thickness of saturated alluvium in the Study Area in August 2015 is shown on Figure 4-10, which is similar in shape to the alluvium-bedrock contact map shown on Figure 4-2 (the similarity results from a relatively flat water table beneath the Study Area). Saturated alluvium is bounded to the west by the Singatse Range, to the northwest by the bedrock outcrops in the Sunset Hills area, to the northeast by the Mason Butte bedrock outcrop, to the east by the Singatse Spur, and to the south by the local bedrock high exposed within the open pit and, locally, by that portion of the Singatse Range located south of the Site. As shown on Figure 4-10, saturated alluvium is thickest (more than 700 feet) beneath the northern portion of the Hunewill Ranch.

4.9.3 Alluvial Groundwater

This section discusses groundwater flow conditions in the alluvial aquifer under historical and current (RI) conditions. The potentiometric surfaces of the zones in the aquifer have been rigorously mapped during the RI and document seasonal changes in flow directions and gradients. Due to the substantially fewer number of monitoring points, potentiometric data for the aquifer prior to the RI are not as extensive as during the RI. Therefore, the following discussion of historical flow conditions is necessarily a qualitative discussion, versus the subsequent quantitative discussion of RI conditions.

In the pre-mining period before circa 1950, little agricultural groundwater use was reported. Photographs indicate irrigation in some areas using surface water delivered by canals (approximately 300 acres, Appendix M, p. 9). Based on available data, groundwater flow direction in the alluvial aquifer was generally northward during this period (Huxel and Harris, 1969).

As discussed in Appendix M, during the period 1959 through 1978, when active mining was occurring, groundwater pumping occurred from five deep wells located beneath the Evaporation Ponds and an additional three wells located on the adjacent agricultural fields. The average pumping rate for all of these wells during this period is estimated to have been about 3,000 gpm (SSPA, 2014). This pumping created zones of capture in the deeper alluvial zones with radial groundwater flow towards the pumping wells. Groundwater studies in the late 1970s by the USGS reported the following conclusions (Seitz et al., 1982, p. 44):

> *"Contaminated shallow ground water has been identified with certainty only at test wells 2A and 2B, which are near the easternmost evaporation ponds … At site 4, which is near the tailings-water recycling ditch, groundwater 27.5 – 29.4 feet below land surface (well 4B) chemically resembles the tailings fluid in several respects. Shallow groundwater at wells 4A, 9, and 11 also resembles the tailings fluid, but to a much lesser degree. Ground water at site 7, adjacent to the northernmost evaporation pond, showed evidence of slight but increasing contamination by the acid brine. Water from the remaining nine shallow wells – all of which are more than 0.2 mile downgradient from the ponds – showed little if any resemblance to the waste fluids. This evidence suggests that, as of 1978, contaminated shallow groundwater was localized in occurrence.*

The Seitz et al. (1982) study also indicated that groundwater flow immediately north of the Evaporation Ponds was toward the north in the Shallow Zone in the late 1970s (water level measurements were collected in the period 1976 to 1980).

In the period 1977 to 2000, agricultural groundwater use increased, especially at wells located in the Hunewill Ranch fields north of the Site (SSPA, 2014, Figures C-3 and C-4). Application of this water to an expanded area of agriculture created flow conditions radially outward (west and northwest) from the irrigated areas. These flow conditions are simulated in the groundwater model (SSPA, 2014, Figure 11a). The expanding RI groundwater program began to measure and document groundwater flow conditions during the latter part of this time period.

Agricultural pumping at wells located in the Hunewill Ranch fields north of the Site ceased in 2009. Groundwater irrigation pumping continued in more distant locations to the northeast and established the current groundwater flow patterns.

The PWS operated along the northern boundary of the LEP in the period 1999 to 2009, and near the northeast corner of the UEP from 1986 to 2009. Operation was discontinued due to small capture zones relative to the east to west groundwater flow direction (i.e., the performance of the system was low due to the east-west alignment of wells which is parallel to shallow groundwater flow in this area) as documented in Appendix J-1. The effectiveness of the PWS is further discussed in Section 3.3.1.

Groundwater flow patterns are well documented by the RI program due to substantial increases in the number of additional monitor wells, spatial coverage, and amount of data collected (as discussed previously in Section 3.2.3). Potentiometric surface maps for the various alluvial aquifer zones in 3Q 2018 are provided in Figures 4-11(a-g). The 1Q 2018 potentiometric surface contours presented in the 2018 Annual Groundwater Monitoring Report (Copper Environmental Consulting and Broadbent and Associates, 2018a) for the Shallow, Intermediate, Deep 2, and Deep 5 Zones are added to Figures 4-11a, 4-11b, 4-11d, and 4-11g, respectively, to illustrate seasonal trends in groundwater flow conditions. The potentiometric levels vary little seasonally except in the vicinity of irrigated fields and the Walker River and irrigation canals and in the deeper zones. Application of irrigation water typically steepens the radially-directed hydraulic gradient in the shallow intervals during the summer as evidenced by the downgradient displacement of each contour between winter and summer in these areas (e.g., Figures 4-11a and 4-11b for the Shallow and Intermediate Zones, respectively). Recharge from the West Campbell Ditch is illustrated by the downstream “V” in the 4335 ft contour in Figure 4-11a. In the deeper zones, the rebound of the potentiometric surface during the non-irrigation season results in higher potentiometric levels in the winter than in the summer (Figure 4-12).

Groundwater flow in the northern portion of the Site under OU-4a is primarily to the west, with a slight southerly component. The consistency of the western flow direction in the Shallow, Intermediate and Deep 1 Zones beneath the UEP and LEP during the RI period of monitoring is documented in the rose diagrams[7] shown on Figures 4-11(h-j).

Groundwater flow in the Shallow Zone north of the Site is influenced by irrigation on the Hunewill Ranch fields. The flow direction in the Shallow and Intermediate Zones is directed radially outward to the southwest, west, and northwest from the areas of irrigation. Immediately north of Hunewill Ranch, groundwater flow in these zones is to the northwest.

Groundwater flow in the Deep 1 and 2 Zones in the north of the Site is generally toward the north. Groundwater flow in the Deep 3 through Deep 5 Zones is overall to the north-northeast across the Study Area.

As seen on the Shallow Zone potentiometric surface map (Figure 4-11a), the Pit Lake is currently a hydraulic sink that is refilling with groundwater predominantly derived from alluvial groundwater recharged locally from the Walker River and, to a lesser extent, bedrock groundwater (Hershey, 2002). As noted in Section 3.3.2, the lake is and will continue to be a groundwater sink due to the large amount of evaporation that occurs from the lake surface.

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[7] Rose diagrams illustrate the relative frequencies of groundwater flow directions over a period of time.

Saturated alluvium is in contact with the Pit Lake on the western margin of the open pit (Figure 4-11a). However, groundwater gradients in this area are toward the Pit Lake, and alluvial groundwater recharges the Pit Lake rather than the Pit Lake recharging the alluvial aquifer. On the eastern margin of the open pit, groundwater derived from seepage from the Walker River flows into the pit. Beneath the Process Areas, the Pit Lake cone-of-depression creates a groundwater divide in the Shallow Zone (Figure 4-11a). Because of local bedrock elevations, only the Shallow, Intermediate, and a small portion of the Deep 1 Zones exist in this area (Figures 4-11 (a-c), respectively).

North of the Site, recharge from the Walker River and its surface water diversions, as well as irrigation practices in the Study Area, are the primary influences on groundwater flow directions. In the Shallow, Intermediate, and Deep 1 Zones, groundwater flows away from, or parallel to, the Walker River and West Campbell Ditch (i.e., north/northwest), indicating recharge of the alluvial aquifer from these features. In addition, groundwater mounding beneath irrigated areas is observed in these zones. In the Deep 2 through Deep 5 Zones of the alluvial aquifer, the groundwater flows to the northeast.

Irrigation practices on the Hunewill Ranch, located immediately north of the Site, locally affect groundwater flow conditions. Historically, irrigation practices on the Hunewill Ranch included seasonal diversion of surface water from the Walker River via the West Campbell Ditch and pumped groundwater, as necessary, from up to three wells. These three wells were located within 2,500 feet of the B/W-1 monitor well cluster and included Well Log No. 82983 (also referred to as WDW019), Well Log No. 26694, and Well Log No. 78925. Well WDW019 and other underground water rights points of diversion are shown on Figure 3-9. Groundwater pumping from the Hunewill Ranch wells to support agricultural irrigation ceased in September 2009. In 2011 and subsequent years, crops on the Hunewill Ranch were irrigated with surface water diverted from the Walker River and groundwater pumped from a nearby parcel located near the

Walker River to the east. Additional information about these wells is provided in the *Aquifer Test Data Summary Report - Revision 1* (BC, 2012e).

Historical irrigation practices for the Hunewill Ranch prior to 2010, which included seasonal groundwater pumping and surface application to the fields, created a cone-of-depression around WDW019 that was most pronounced in the Deep 1 through Deep 3 Zones but was also manifested in the Shallow and Intermediate Zones (BC, 2014a). The cone-of-depression in the combined Deep 1 through Deep 3 Zones extended beneath the Sulfide Tailings area and the Evaporation Ponds, and as far north as the Sunset Hills neighborhood. Figure 4-12 provides monthly water level hydrographs of Shallow and Deep monitor wells in the B/W-1 and B/W-27 well clusters (located beneath and adjacent to the Hunewill Ranch, respectively) that illustrate the hydraulic head drawdown during and after operation of the Hunewill Ranch pumping wells. Although agricultural pumping in the Study Area continues to affect hydraulic head in wells B/W-1D3 and B/W-27D2, hydraulic head drawdown in these wells was up to three times greater during operation of the Hunewill Ranch pumping wells than it has been in recent years.

Figure 4-13 presents monthly vertical groundwater gradients in the alluvial aquifer at select locations. Vertical gradients were calculated as the difference in water levels between the shallowest and deepest alluvial monitor wells in a cluster divided by the distance between the midpoints of the screened intervals of the wells. If a well was screened across the water table, then the water table elevation was used in place of the midpoint of the screened interval for that well. Monthly vertical gradients were calculated using water level measurements from 2013, which was the last full calendar year during which water levels in all active monitor wells were measured monthly. For wells installed after 2013 pursuant to the Additional Monitor Well Work Plan (BC, 2013b), monthly vertical gradients were calculated using water level measurements from September 2014 through August 2015. Where available, monthly vertical gradients were also calculated using water level measurements from 2009 to illustrate groundwater conditions during operation of the Hunewill Ranch pumping wells.

Alluvial vertical gradients beneath the Process Areas are generally upward (PA-MW-4 well cluster), reflecting potential discharge of bedrock groundwater to alluvium as a potential source of groundwater to this portion of the Site (i.e., MFR). Beneath the Evaporation Ponds (B/W-11 well cluster) and Hunewill Ranch (B/W-1 well cluster), alluvial vertical gradients are downward, with stronger vertical gradients corresponding to months when irrigation, and thus groundwater recharge and pumping, occurs. In addition, alluvial vertical gradients were even more strongly downward in these areas in 2009 when the Hunewill Ranch pumping wells were operating. In other irrigated areas (i.e., B/W-68 and B/W-81 well clusters), alluvial vertical gradients are also consistently downward, again with stronger vertical gradients in months when irrigation occurs.

Immediately northwest of the Site at the B/W-41 well cluster, vertical gradients are upward in the winter months, reflecting potential discharge of bedrock groundwater to alluvium (i.e., MFR), and downward in the summer months, reflecting the influences of agricultural pumping. Downward vertical gradients at the B/W-41 well cluster are approximately two orders of magnitude smaller than the downward vertical gradients in irrigated areas, reflecting the greater distance of B/W-41 from irrigated areas. The B/W-28 well cluster (located in the Sunset Hills neighborhood) also exhibits seasonal changes in the direction of vertical gradients - upward vertical gradients occur in the winter (reflecting the non-irrigation season) and downward vertical gradients occur in the summer (reflecting the influences of agricultural pumping). In 2009, the effects of operation of the Hunewill Ranch pumping wells, in addition to other agricultural pumping in the Study Area, caused strongly downward vertical gradients that resulted in water levels in monitor well B/W-28S to decline below the bottom of the screened interval in August and September.

4.9.4 Groundwater Recharge

Recharge to the alluvial aquifer in the Mason Valley primarily occurs from surface water diverted from the Walker River within unlined irrigation ditches, infiltration of surface water and groundwater applied to irrigated fields, and infiltration of surface water from the channel of the Walker River. Recharge to the alluvial aquifer also occurs along the range front via a variety of hydrologic processes.

As indicated above, infiltration of surface water is the primary source of groundwater recharge to the alluvial aquifer in the Mason Valley, with MFR contributing significantly less (Carroll et al., 2010; Huxel and Harris, 1969; Myers, 2001; SSPA, 2014). The annual amount of recharge derived by infiltration from stream channels, ditches, and agricultural fields is a function of Walker River flows, the volumes of surface water and groundwater used for irrigation, and water table depths within Mason Valley.

Hydrologic tracer data for tritium/helium ($^{3}H/^{3}He$) groundwater age estimates (Figure 4-14[a-g]) and uranium isotopes (Figure 4-15[a-e]) are consistent with the two principal recharge components of the HCSM: 1) seepage from the Walker River and irrigation ditches, and infiltration from irrigated fields on the east margins of the Study Area; and 2) MFR on the west side of the Study Area adjacent to the Singatse Range.

The use of groundwater uranium isotopes (and their relationship to tritium/helium groundwater age estimates) is briefly summarized as follows. In groundwater systems, ^{234}U is more environmentally mobile than ^{238}U due to physical recoil of the atom following alpha decay of ^{238}U, and the subsequent displacement of the ^{234}U atom to weaker binding sites within the crystalline lattice of the mineral in which it is contained. Thus, the two isotopes are released (weathered) at different rates, and the $^{234}U/^{238}U$ ratio is generally greater than unity in natural waters. Changes in the isotopic ratios (and uranium excess [Ue] values derived from the ratios) are assumed to be solely associated with transport/contact time between groundwater and aquifer solids. Consequently, high Ue values are associated with “long” periods of contact between groundwater and aquifer solids (i.e., “old” water) whereas low Ue values are associated with “short” periods of contact between groundwater and aquifer solids (i.e., “young” water). However, $^{234}U/^{238}U$ ratios in mine-influenced groundwater are also close to unity (resulting in low Ue values) because acidic process solutions leach both isotopes from ore material with equal effectiveness (Iles et al., 1995).

On the east side of the Study Area, Shallow Zone alluvial groundwater directly recharged by surface water commonly exhibits younger, more modern age estimates and low Ue values. Low Ue values are also observed in mine-influenced Shallow Zone groundwater beneath the Evaporation Ponds. In contrast, the older groundwater age estimates and highest Ue values commonly occur on the west side of the Study Area adjacent to the Singatse Range, in Deep alluvial groundwater, and in bedrock groundwater. An exception to the typical vertical distribution of higher Ue values and older groundwater age estimates is evident at the B/W-1 well cluster where younger groundwater age estimates and lower Ue values occur locally in the Deep groundwater zones.

This local pattern of Ue values and groundwater age estimates around the B/W-1 well cluster is consistent with the HSCM, which recognizes groundwater mixing due to agricultural pumping, especially former pumping at well WDW019 (which is located adjacent to the B/W-1 well cluster). Pumping at WDW019 has resulted in the migration of mine-influenced groundwater present beneath the Evaporation Ponds into the Intermediate and Deep Zones of the alluvial aquifer beneath the Hunewill Ranch.

Samples from B/W-32S and B/W-77S yield tritium groundwater age estimates of > 60 years and Ue values of 414 and 438, respectively. There are higher Ue values to the west of B/W-32S and B/W-77S that have higher Ue values (e.g., greater than 600) and are considered “old water” as described above.

However, the tritium age dating methodology provides quantitative estimates of groundwater age whereas the uranium isotope disequilibrium method provides non-quantitative information about groundwater age based on the activity ratio of ^{234}U and ^{238}U and excess daughter ^{234}U (referred to as uranium excess, Ue) which represents the deviation from secular equilibrium of the $^{234}U/^{238}U$ activity ratio, as described above. Ue values do not quantify the groundwater age because there is no absolute scale for relating Ue values to groundwater ages, and because changes in the nature of the uranium host phases, water rock interactions, and groundwater redox conditions alter the Ue value (Porcelli and Swarzenki, 2003) without affecting the groundwater age. Consequently, no direct correlation between tritium groundwater ages and Ue values is possible. Nonetheless, Ue values are useful for providing constraints on groundwater conditions (Cook and Herczeg, 2000; Kendall and McDonnell, 1998). Therefore, it is not inconsistent for a water sample to be older than 60 years as dated with tritium and to be classified as “young” by the uranium disequilibrium method.

4.9.5 Alluvial Aquifer Hydraulic Property Data

Hydraulic conductivity values in alluvium from slug tests (Appendix F-1) range from 0.0399 to 157 ft/day, with a median value of 6.6 ft/day. The 25th and 75th percentiles are 3.1 ft/day and 20.7 ft/day, respectively. Hydraulic conductivity values in alluvium calculated from an analysis of low-flow sampling data (Appendix F-2) range from 0.08 to 240 ft/day, with a median value of 18.4 ft/day. The 25th and 75th percentiles are 6.8 ft/day and 33.6 ft/day, respectively. Statistical analyses indicate that: 1) hydraulic conductivity values in each alluvial zone exhibit similar ranges, median values, and distributional shapes; and 2) hydraulic conductivity values throughout the alluvial aquifer are statistically equivalent in their distributions from zone to zone. Hydraulic conductivity estimates from slug tests (Appendix F-1) for the individual groundwater zone are provided in Table 4-6.

Table 4-6. Hydraulic Conductivity by Groundwater Zone (from Slug Test Data)

Zone	25th Percentile K (ft/day)	Median K (ft/day)	75th Percentile K (ft/day)	Standard Deviation (ft/day)	Count
Shallow	3.1	5.8	11.6	25.7	98
Intermediate	4.2	10.6	33.8	26.1	48
Deep 1	5.0	8.4	30.6	27.5	45
Deep 2	2.8	5.5	21.9	17.9	23
Deep 3	2.4	4.3	8.8	14.9	18
Deep 4	4.6	6.7	36.6	29.6	8
Deep 5	1.8	12.4	16.9	10.3	9
Bedrock	0.2	1.7	11.7	66.0	57
All Alluvial Zones	3.1	6.6	20.7	24.8	249
All Zones	2.3	5.9	18.3	36.5	306

Notes: K = hydraulic conductivity.

Constant-rate testing of the eleven pumpback wells in 2010, which are screened in the Shallow Zone (Appendix F-3), resulted in hydraulic conductivity estimates ranging from 0.9 to 47 ft/day (median of 9.4 ft/day). Slug testing of piezometers (33 in total, all screened in the Shallow Zone) installed near the PWS, that were used as observation wells during constant-rate pumping tests of the 11 pumpback wells during 2010 (Appendix F-4), provided hydraulic conductivity estimates ranging from 1.1 to 83 ft/day (median of 4.1 ft/day). Testing of five pumpback wells (PW-6, PW-7, PW-9, PW-10, and PW-11) in 2000 yielded hydraulic conductivity estimates that ranged from 6.4 to 33 ft/day, with a geometric mean of 16 ft/day, based on an assumed aquifer thickness of 50 feet (AHA, 2000).

A constant-rate test of well WDW019, using an observation network of 93 monitor wells, resulted in estimated values for hydraulic conductivity and specific storage at 61 observation wells that exhibited pumping-related responses (Appendix F-5). Hydraulic conductivity values in alluvium derived from constant-rate pumping tests of WDW019 ranged from 4.9 to 1,200 ft/day, with a median value of 77 ft/day. The 25th and 75th percentiles are 38 ft/day and 195 ft/day, respectively. Five hydraulic conductivity values in bedrock ranged from 13.0 to 92 ft/day.

Specific storage values in alluvium from constant-rate pumping tests of WDW019 range from 1.45×10^{-8} to 1.46×10^{-3} $(\text{feet})^{-1}$, with a median value of 1.14×10^{-5} $(\text{feet})^{-1}$. The 25th and 75th percentiles are 5.16×10^{-6} $(\text{feet})^{-1}$ and 2.73×10^{-5} $(\text{feet})^{-1}$, respectively.

4.9.6 Spatial Variation in Hydraulic Conductivity

The spatial distributions of slug-test hydraulic conductivities in each alluvial groundwater zone are shown on Figures 4-16(a-g), and bedrock slug-test hydraulic conductivities are shown on Figure 4-17. Slug-test hydraulic conductivities represent the largest hydraulic conductivity dataset for evaluating spatial variation within the Study Area. To date, 295 measurements of hydraulic conductivity have been obtained using slug-test methods (Appendix F-1).

Although analysis of drawdown measurements during low-flow sampling of monitor wells has yielded more measurements of hydraulic conductivity than slug testing (318 measurements versus 295 measurements through August 2015), the approach has limitations that that do not capture the highest and lowest values of hydraulic conductivity in the Study Area. Due to the low flow rates used, measurable drawdown (i.e., drawdown exceeding 0.01 feet) does not occur during the sampling of many monitor wells that have sufficiently high hydraulic conductivities. In practice, hydraulic conductivities of greater than 100 ft/day cannot be determined with this method. Conversely, most monitor wells that have hydraulic conductivities less than 0.1 ft/day never achieve steady-state drawdown conditions during low-flow sampling (i.e., the water level in the monitor well continues to fall during the entire sampling period).

Spatial variation in slug-test hydraulic conductivities reflects the heterogeneous lithology of the alluvium underlying the Study Area. Alluvial slug-test hydraulic conductivities beneath the Site (i.e., beneath the Process Areas, Sulfide Tailings, and the Evaporation Ponds) are generally in the range of 1 to 10 ft/day, with some infrequent exceptions. To the west of the Site, along the margin of the Singatse Range, alluvial slug-test hydraulic conductivities extend into the range of 10 to 100 ft/day, with some locations still in the range of 1 to 10 ft/day. This trend continues north towards the Sunset Hills. Beneath the Hunewill Ranch, alluvial aquifer slug-test hydraulic conductivity values generally range from 1 to 50 ft/day, with noted high conductivity exceptions at monitor wells B/W-60S, B/W-61S, and B/W-60D1.

Alluvial slug-test hydraulic conductivities near the Walker River (east of the Pit Lake) are generally higher than alluvial slug-test hydraulic conductivities beneath the Site, as evidenced by monitor wells B/W-13S, B/W-14S, B/W-15S, B/W-21S, and PLMW-2S. In this area, alluvial slug-test hydraulic conductivities are in the range of 10 to 50 ft/day. North of the Site and east of the West Campbell Ditch, alluvial slug-test hydraulic conductivities are also high, generally ranging from 10 to 100 ft/day or higher.

The spatial variability of alluvial slug-test hydraulic conductivity values reflects the varied depositional environments in Mason Valley, which are discussed in Section 4.8. The areas beneath the Site represent older fan deposits, which typically display lower permeability compared to the valley-fill sediments (i.e., younger and older alluvium). North of the Site, a transitional environment with higher permeability exists between the older fan deposits and the valley-fill sediments. Areas near the Walker River and generally to the east of West Campbell Ditch appear to represent valley-fill sediments, which exhibit the highest overall permeability. To the west of the Site, a narrow north-south trending band of higher permeability aquifer materials occurs along the flank of the Singatse Range, which represents coarser-grained alluvial fan materials eroded from the Singatse Range.

4.9.7 Bedrock Groundwater

This subsection summarizes information presented in the Bedrock Groundwater Assessment Technical Memorandum (Appendix I). The conceptual model of OU-1 bedrock groundwater flow conditions is based on regional and Site-specific information, including: 1) the lithologic and structural geology information presented in Proffett and Dilles (1984) and Proffett (1977); 2) a general understanding of bedrock groundwater flow in the Great Basin portion of the Basin-and-Range Physiographic Province; 3) hydrogeologic information obtained from drilling, lithologic logging, testing, and monitoring of 67 bedrock groundwater monitor wells located in the OU-1 Study Area (typically installed with 20-30 foot long screen intervals positioned in the upper 50 feet of bedrock); 4) evaluation of hydraulic head data in the bedrock and alluvial groundwater systems; 5) hydrologic tracer data for stable isotopes of oxygen and hydrogen in water ($^{18}O/^{2}H$), tritium/helium ($^{3}H/^{3}He$) groundwater age estimates, and uranium isotopes; and 6) bedrock monitor well chemical data.

As noted previously, the Site and surrounding Study Area are in a U-shaped graben structure on the western margin of Mason Valley. Within the Study Area, the depth to bedrock is highly variable and ranges from 0 to 750 feet bgs. Bedrock within the Mason Valley and Study Area is comprised of consolidated granitic, metamorphic, and volcanic rocks. The bedrock groundwater system consists of a fractured rock aquifer where water moves predominantly through fracture porosity, and matrix permeability is considered negligible. The fractures occupy only a small fraction of the bedrock.

Large-scale geologic structures (i.e., faults) result in structural compartmentalization of the bedrock groundwater system in the Study Area, with low groundwater flow across and along faults that are commonly characterized as containing fine-grained, low-permeability fault gouge and brittle or plastic clay. The fractured rock aquifer exhibits high, three-dimensional (i.e., anisotropic) spatial variability in hydraulic conductivity (and hence groundwater flow rate). Hydraulic conductivity (K) values in bedrock monitor wells have been derived from slug tests performed after the wells were constructed, and from an analysis of low-flow sampling of bedrock monitor wells during groundwater monitoring activities (BC, 2015a). Both methods yield comparable results. In addition, both methods provide estimates of hydraulic properties local to the test well and, consequently, are primarily used to assess the spatial distribution of bedrock hydraulic properties in the Study Area. Mapping of the hydraulic conductivity values indicate a high degree of spatial variability with significant changes (often greater than three orders of magnitude) over distances that are small relative to the size of the Study Area.

In bedrock, estimated conductivities range from approximately 0.002 to 334 feet/day, with the higher values measured in wells located near faults and the open pit (Figure 4-17). The 25th and 75th percentiles were 0.15 and 11.7 feet/day, respectively. The median bedrock slug-test K value was 1.7 feet/day. The lowest bedrock K values are up to two orders of magnitude lower than the lowest K values in alluvium in the Study Area. The minimum measured low-flow sampling K value in bedrock was 0.013 feet/day, and the maximum value was 67 feet/day. The 25th and 75th percentiles were 0.485 feet/day and 16.6 feet/day, respectively. The median bedrock low-flow sampling K value was 4.3 feet/day. Consistent with the slug-test dataset, the lowest K values in bedrock are up to two orders of magnitude lower than the lowest K values in alluvium in the Study Area.

Similar water level responses in paired bedrock/alluvial monitor wells at any given location throughout the Study Area result from: 1) the interconnectivity between the bedrock and overlying alluvial groundwater flow systems (except locally around the B/W-1 well cluster where an aquitard separates the two flow systems); and 2) the transmission of stresses through the skeletal structure of the aquifer solids. Seasonal fluctuations in bedrock groundwater levels (and vertical gradients between the bedrock and alluvial groundwater systems) due to agricultural pumping from the alluvial aquifer are observed beneath the Hunewill Ranch and Evaporation Ponds, and in the Sunset Hills area. Both the bedrock and alluvial groundwater systems exhibit similar long-term water level declines.

Vertical interconnection between bedrock and overlying alluvial groundwater is indicated by water level data (and the presence of locally-elevated concentrations of sulfate and uranium in bedrock groundwater that are sourced from overlying mine-influenced alluvial groundwater). Spatially, vertical gradients between bedrock and alluvium generally reflect the recharge components of the alluvial groundwater HCSM (BC, 2014a), with downward vertical gradients east and north of the Site driven by recharge of surface water and irrigation water on crop fields, and upward vertical gradients in western portions of the Study Area driven by MFR.

The largest vertical gradients between bedrock and alluvium occur: 1) within the Pit Lake cone of depression; and 2) beneath the Hunewill Ranch fields and Evaporation Ponds. In all other portions of the Study Area, vertical gradients between bedrock and alluvium are relatively small. Seasonal crop irrigation effects are observed near the Hunewill Ranch fields, Evaporation Ponds, and Sunset Hills, as represented by locations with both upward and downward vertical gradients.

Stable isotopes ($^{18}O/^{2}H$) in bedrock groundwater are generally more depleted with respect to Walker River surface water and alluvial groundwater (BC, 2014a). The most depleted stable water isotope signatures in the bedrock groundwater are associated with the oldest apparent groundwater ages, as determined by $^{3}H/^{3}He$ age dating (Figure 4-18). The youngest groundwater ages are associated with the least depleted bedrock groundwater samples, which also overlap the region of cool season Walker River flows. Walker River samples display an evaporative fractionation signature, with less fractionated values occurring during periods of snowmelt runoff and more fractionated values occurring during periods of lower flows during the summer.

The greater degree of $^{18}O/^{2}H$ depletion of bedrock groundwater compared to the alluvial aquifer and Walker River surface water suggests different recharge processes. The depleted stable isotope signature indicates that bedrock groundwater is: 1) sourced from snowmelt recharged directly in the Singatse Range, which does not undergo the same evaporative fractionation as Walker River water, and/or fossil water recharged during the Pleistocene (a cooler and more humid climate than the current climate); and 2) older and of a different origin than surface water and alluvial groundwater. Bedrock groundwater ages are older than 1954, and essentially pre-date Site mining activities, occur throughout most the Study Area. Younger bedrock groundwater within and downgradient of the Singatse Spur, proximal to the Walker River, exhibits isotopic similarities between younger bedrock groundwater and cool season Walker River water.

The widespread occurrence of older groundwater ages and the lesser occurrence of younger groundwater ages suggests slow movement of bedrock groundwater. The spatial distribution of Ue values generally comports with the spatial distribution of $^3H/^3He$ groundwater ages. This pattern of widespread higher Ue values (i.e., “old” water) with small areas of lower Ue values (i.e., “young” water), irrespective of hydraulic conductivity, again suggests slow movement of bedrock groundwater.

In addition to a high degree of anisotropy in hydraulic characteristics and hydrologic tracer signatures, the bedrock groundwater system also exhibits three-dimensional (i.e., anisotropic) spatial variability in chemical concentrations. As noted in BC (2014a) and discussed further in Section 5.0, concentrations of mine-related chemicals in groundwater are most elevated in the Shallow Zone of the alluvial aquifer beneath the LEP, UEP, Finger Ponds, Phase IV VLT HLP, and Phase IV VLT Pond. COI concentrations decrease with vertical depth and horizontal distance from the Site sources, resulting in values in bedrock groundwater that are one to several orders of magnitude lower than the values in overlying alluvial groundwater.

In addition, areas of elevated COIs in bedrock groundwater are small in comparison to the alluvial aquifer and found mostly on-Site. The small areas of elevated COI concentrations in bedrock groundwater indicate that bedrock fractures have low hydraulic connectivity and transmissivity over horizontal distances relevant to the scale of the Study Area.

As summarized in the Bedrock Groundwater Assessment Technical Memorandum (Appendix I, Executive Summary, p. 3), the bedrock characterization information indicates: 1) a high degree of vertical hydraulic connection between the bedrock and alluvial groundwater systems; and 2) low horizontal hydraulic connectivity and transmissivity of bedrock fractures, especially over horizontal distances that are relevant to the scale of the Site and the surrounding Study Area. Despite small areas of relatively high hydraulic conductivity, bedrock groundwater flow velocities, average volumetric flow rates and chemical fluxes through the bedrock groundwater system are low.

5 NATURE AND EXTENT OF GROUNDWATER CONTAMINATION

This section describes the nature and extent of groundwater contamination in OU-1. Section 5.1 provides contour maps that characterize the nature and extent of COIs in groundwater. These maps contribute to the overall MIW evaluation discussed in later subsections. COI distributions are described in general terms without attempt to attribute elevated concentrations to a specific source, although insights to sources can be preliminarily ascertained by the proximity of elevated concentrations to certain Study Area features and past and current activities. Consistent with past reports, the distributions of six COIs (pH, sulfate, uranium, alkalinity, nitrate, and arsenic) are used to help understand groundwater conditions in the various zones comprising both the alluvial groundwater aquifer and the bedrock groundwater system. Recent evaluations (described further below) have identified and evaluated supplemental COIs (copper, iron, molybdenum, and zinc) that are relevant to source attribution for the alluvial aquifer. The distributions of the six COIs listed above used in past reports have been found to be sufficient for RI-decision making about impacts to the bedrock groundwater system. Chemical distributions in alluvial aquifer zones are described in Section 5.1.1 and chemical distributions in the bedrock groundwater system are described in Section 5.1.2.

Section 5.2 discusses the key sources of COIs to Study Area groundwater, which include: 1) mine activities; 2) agriculture activities; and 3) geologic materials. Consistent with Section 7.0 of the 2007 SOW, this FRIR discusses the relevant source-related information for the specific mine waste facilities (i.e., other Site OUs) that represent sources of chemical loading to groundwater (Section 5.2.1). Other major non-mining sources of COIs to groundwater are discussed in Sections 5.2.2 (Agriculture) and 5.2.3 (Geologic Sources).

Section 5.3 summarizes the BGQA, the background approach formerly used to delineate an area of elevated COIs in the larger OU-1 groundwater Study Area. As noted previously in Section 3.3.6, the BGQA did not account for all anthropogenic or naturally-occurring background sources within the Study Area (EPA, 2016a and 2016b). Section 5.3.2 describes the post-BGQA RI evaluations that utilize a multiple lines of evidence approach to differentiate the MIW component of the BGQA area and identify other subareas of the BGQA that are influenced by the anthropogenic or naturally-occurring background sources. In Section 5.4, MIW in the Shallow Zone has been delineated using two "hypotheses," resulting in overlapping portions of the aquifer where mine-related chemical influences may be present: a less extensive "High Confidence MIW Area Shallow Zone" and a somewhat more extensive "Low Confidence MIW Area Shallow Zone." MIW areas are also delineated in the Intermediate through Deep 5 Zones.

5.1 Contoured Chemical Distributions

The following sections discuss the distributions of select parameters and COIs in Study Area groundwater. The contour maps characterize the nature and extent of COIs and contribute to the MIW evaluation process.

5.1.1 Alluvial Aquifer

The distributions of six COIs (pH, sulfate, uranium, alkalinity, nitrate, and arsenic) in groundwater during August 2014 (the comprehensive RI data collection event for groundwater chemistry and isotopic data) are illustrated on cross sections (cross section locations with 20V:1H vertical exaggeration are shown on Figure 4-6) and maps for the Shallow and Deep 3 groundwater zones (Figures 5-1 through 5-6). The distributions of the six chemicals in all the groundwater zones are shown on maps in Appendix K. A northwest/southeast trending cross section, D-D' with 5 x vertical exaggeration for the Shallow Zone is shown on Figure 4-6 and Figures 5-25(a-g).

The six COIs were selected from the groundwater analyte list for use in prior RI groundwater reports characterizing the nature and extent of contamination from mining and non-mining sources for the following reasons. Review of historical operational information and groundwater quality data collected adjacent to mine facilities indicate that elevated acidity (i.e., low pH), sulfate, and uranium are indicator parameters for mine-influenced groundwater at the Site (Section 2.1.1). Copper was extracted from oxide ore using a sulfuric acid leach solution which was the primary source of low pH and sulfate that infiltrated to groundwater from mine operations. Groundwater monitoring data indicate that uranium is also elevated in groundwater samples collected adjacent to mine facilities as well as other non-mining sources as described in Sections 5.2.2 and 5.2.3. Sulfate and uranium are more mobile in groundwater relative to other mine-related parameters, such as iron (Section 6.2). However, there are other sources of sulfate and uranium in the Study Area (Sections 5.2.2 and 5.2.3). Alkalinity in groundwater is important in understanding the nature and extent of uranium because complexation of dissolved uranium with bicarbonate and calcium enhances the solubility and mobility of uranium in groundwater (EPA, 2010c; BC, 2014a).

Sulfate and nitrate are ingredients of agricultural amendments and fertilizers and are indicators of agricultural-influenced groundwater (BC, 2014a). Although arsenic is detected at locally high concentrations in both on-Site and off-Site locations, it occurs naturally in the Study Area (and throughout Nevada) and exhibits complex geochemical transport behavior, which limits its usefulness as an indicator of MIW in the Study Area (EPA, 2016e).

As discussed in Section 5.2.3 and Appendix N (Final GTM), geothermal alteration minerals also enrich groundwater in sulfate and uranium. Therefore, all three potential sources of COIs to groundwater in the Study Area (mine, agriculture, and geothermal) are enriched in sulfate and uranium. A multiple line of evidence (LOE) approach was developed to identify the areas of groundwater containing MIW (Section 5.4).

pH

The distributions of pH values in the Shallow and Deep 3 groundwater zones in August 2014 are shown on Figures 5-1a and 5-1b, respectively. The lowest pH values are observed in Shallow Zone groundwater beneath the Evaporation Ponds and downgradient of the Phase IV VLT HLP and VLT Pond. pH values generally increase laterally and vertically away from these structures by orders of magnitude.

Sulfate and Uranium

The distributions of sulfate and uranium in groundwater in August 2014 are illustrated on Figures 5-2 and 5-3, respectively. The most elevated concentrations of sulfate and uranium are observed in Shallow Zone groundwater beneath the Evaporation Ponds and downgradient of the Phase IV VLT HLP and VLT Pond, and concentrations generally decrease laterally and vertically away from these features by orders of magnitude.

There are elevated concentrations of sulfate (Figure 5-2a) and uranium (Figure 5-3a) in the Shallow Zone north of the Site in the vicinity of well B/W-32S. Sulfate (Figure 5-2b) and uranium (Figure 5-3b) distributions exhibit a northeast longitudinal orientation in Deep Zone groundwater from the northern portion of the Site to beneath the Hunewill Ranch toward former agricultural wells used seasonally to extract groundwater for crop irrigation.

In addition, elevated concentrations of sulfate and uranium (as well as alkalinity and nitrate discussed below) occur in Shallow, Intermediate, and Deep 1 Zones beneath agricultural fields in the North Study Area (NSA). The NSA refers to the portion of the groundwater Study Area located northeast of West Campbell Ditch and north of the Sunset Hills (Figure 3-3) and is discussed further in Section 5.2.2. Given the low concentrations of sulfate and uranium detected in all groundwater zones at well cluster B/W-27 (Figures 5-2d and 5-3d, respectively), elevated concentrations of these chemicals in groundwater beneath the agricultural fields located in the NSA are not physically connected along current and plausible historical groundwater flow paths back to known sources of MIW beneath the Site. A more detailed discussion of data specific to the NSA is provided in Section 5.4.

Alkalinity

The distribution of alkalinity in groundwater in August 2014 is illustrated on Figures 5-4(a-e). The lowest values of alkalinity (less than 100 mg/L) occur in the Shallow Zone beneath the UEP, and elevated values occur in the northern portion of the Site. Alkalinity in the Shallow Zone is non-detected (< 4 mg/L) at pH less than 5 and increases when pH is above 6.

Dissolution of carbonate minerals by low pH MIW can be a source of alkalinity as illustrated in the following example of acid in contact with calcite:

$$CaCO_3 + H^+ \rightleftharpoons Ca^{+2} + HCO_3^-$$

Alkalinity in Deep Zone groundwater exhibits a northeast longitudinal orientation from the Site toward former (e.g., WDW019) and existing agricultural wells used seasonally to extract groundwater for crop irrigation (Figure 5-4b). Alkalinity values above 200 mg/L occur beneath the Hunewill Ranch and other agricultural fields located in the northeastern portion of the Study Area.

Nitrate

The distribution of nitrate in groundwater in August 2014 is illustrated on Figures 5-5(a-e). The most elevated nitrate concentrations occur in groundwater beneath the Hunewill Ranch and other agricultural fields located in the northeast portion of the Study Area (Figures 5-5a, 5-5d, and 5-5e). Beneath the Hunewill Ranch, nitrate concentrations in Shallow Zone groundwater were as high as 42 mg/L in August 2014. Former groundwater extraction from high-capacity wells (especially WDW019) for irrigation of crops on the Hunewill Ranch has mobilized nitrate from the Shallow Zone into the Intermediate and Deep Zones of the alluvial aquifer system (Figures 5-5b and 5-5d). Vertical mobilization of nitrate is not as evident in the NSA compared to the Hunewill Ranch (Figure 5-5e).

Regions of low nitrate concentration within the alluvial aquifer include along the Walker River east of the Pit, along the West Campbell Ditch, near the Singatse Range, beneath the majority of the ACMS, and immediately north/northwest of the ACMS (Figures 5-5[a-e]). These regions represent areas where groundwater recharge was not influenced by agricultural activities or residential areas (i.e. septic systems). The majority of the supply water used during mining was extracted from the alluvium and bedrock with low nitrate concentrations until agricultural activities began to increase immediately north of the mine which may have increased nitrate in the mine water supply.

In summary, regions with high nitrate concentrations are clearly influenced by either agricultural activities or residential water use.

Arsenic

The distribution of arsenic in alluvial groundwater in August 2014 (Figures 5-6[a-e]) differs from the distributions observed for sulfate, nitrate, and alkalinity. The most elevated arsenic values occur in Shallow Zone groundwater at off-Site wells B/W-77S (560 µg/L) and B/W-32S (420 µg/L), which are located approximately 2,500 feet north of the Evaporation Ponds (Figure 5-6a). Arsenic values decrease laterally and vertically from these two wells by at least an order of magnitude. In contrast to the elevated arsenic values at these two off-Site wells, the most elevated arsenic values in groundwater beneath the Site are approximately 3.5 times lower.

The most elevated arsenic concentrations in on-Site groundwater range from approximately 120 to 160 µg/L in Shallow Zone wells MW-5S, FMS-06S, and MW-2S (Figure 5-6a). These three wells are located near the Thumb Pond and the Phase IV VLT HLP and VLT Pond. Historical process solutions discharged to the Thumb Pond were reported to contain 42 mg/L of arsenic (Seitz et al., 1982). Arsenic values in the range of 50 to 80 µg/L occur in the Shallow, Intermediate, and Deep Zones west and northwest of the Site.

In addition, data from zonal groundwater samples and monitor wells installed in the NSA indicate elevated concentrations of arsenic in Deep Zone groundwater that recharges the NSA. Groundwater arsenic concentrations in this portion of the Study Area increase as a function of depth in the alluvial aquifer (at a relatively uniform pH) as indicated best by the zonal groundwater sample data from wells B/W-56 and B/W-69 (Figure 5-7) where arsenic concentrations are as high as 83 µg/L. At other NSA monitoring locations (such as B/W-59, B/W-57, and B/W-68), chemical profiling did not extend as deep as the B/W-56 and B/W-69 locations. Nonetheless, arsenic concentrations at these three locations begin increasing as a function of depth below 4,200 feet amsl. Had sampling been conducted to greater depths at these locations, elevated arsenic concentrations would be expected since no known plausible hydrologic/geochemical explanation would limit arsenic enrichments strictly to those areas around B/W-56 and B/W-69.

Because profile locations B/W-56 and B/W-69 represent upgradient groundwater inflows to the NSA, chemical concentrations at these locations represent naturally-occurring concentrations and are not related to advancement of MIW from the south. Furthermore, the elevated arsenic concentrations at this depth are not associated with agricultural activity, which is shown in Section 5.2.2 to affect the Shallow, Intermediate and Deep 1 zones in the NSA. Instead, these elevated arsenic concentrations are associated with regional groundwater conditions in the Mason Valley, likely associated with deep bedrock discharge to the alluvial basin aquifer.

Supplemental COI Contour Maps

Contour maps of copper, iron, molybdenum, and zinc measured during the 3Q 2014 groundwater monitoring event in the Shallow Zone and the Deep 3 Zone are shown in Figures 5-24(a-h), respectively.

The distribution of copper in the Shallow Zone (Figure 5-24a) is similar to the distribution of pH less than 5 shown on Figure 5-1a. The highest concentration of copper was measured in a sample from well MW-5S, which is on the downgradient boundary of the Evaporation Ponds. Copper concentrations in the Deep 3 Zone (Figure 5-24b) are not elevated; the maximum concentration is 3.6 µg/L and most results are non-detect. The pH of groundwater in the Deep 3 Zone is generally greater than 6.

Iron concentrations in the Shallow Zone (Figure 5-24c) are elevated in areas where pH is less than 6. The maximum iron concentration is 8,000 mg/L, which was measured in a water sample with pH of 3.8. Iron concentrations in the Deep 3 Zone are less than 0.03 mg/L, and many are non-detect (Figure 5-24d). The pH values in the Deep 3 Zone are greater than 6.

Molybdenum concentrations in the Shallow Zone (Figure 5-24e) are highest in the vicinity of wells B/W-32S and B/W-77S (11,000 µg/L). Molybdenum concentrations in the northern part of the LEP range up to 830 µg/L where pH is greater than 6 but are low to non-detect in the areas where pH is less than 6. Well B/W-37S in the eastern part of the Site produced a sample containing 1,700 µg/L molybdenum with a pH of 7.6. The distribution of molybdenum in the Shallow Zone is discussed in detail in Section 5.4.3.

The distributions of zinc in the Shallow and Deep 3 Zones (Figures 5-24g and 5-24h, respectively) are similar to copper, as the mobility of zinc is also pH-dependent. The maximum concentration is 13,000 µg/L measured beneath the Evaporation Ponds in acidic pH conditions.

5.1.2 Bedrock Groundwater

Most of the bedrock wells have 20- to 30-foot long screens that are positioned in the upper 50 feet of the bedrock to straddle fractures that yield groundwater for monitoring purposes. The pH measurements and concentrations of sulfate, uranium, alkalinity, nitrate, and arsenic in the bedrock groundwater system in August 2014 are shown on Figures 5-8(a-f). Most pH measurements in bedrock groundwater are near-neutral (i.e., approximately 7.0).

Sulfate and uranium were detected in bedrock groundwater during August 2014 at concentrations up to 1,600 mg/L and 950J (estimated concentration) µg/L, respectively. For both chemicals, the most elevated concentrations were detected in bedrock groundwater within the Site. The most elevated sulfate concentrations in bedrock groundwater occurred at well B/W-36B. Off-Site, sulfate was detected in bedrock groundwater at a concentration above 500 mg/L in well B/W-58B. The most elevated uranium concentrations in bedrock groundwater occurred at on-Site wells MW-5B and B/W-74B. Similarly, the most elevated alkalinity concentrations in bedrock groundwater occurred at on-Site wells MW-5B and B/W-74B (as well as MW-H4SN).

Nitrate concentrations were most elevated in bedrock groundwater at wells B/W-73B and HLP-06B, which are located to the west and southwest of the Site and where the water table is in the bedrock.

Elevated arsenic concentrations (≥20 µg/L) in bedrock groundwater occurred in three distinct areas. Elevated arsenic concentrations were detected in: 1) four wells (PA-MW-2B, PA-MW-4B, HLP-03B, and HLP-06B) located around the Phase III 4X HLP and nearby Calcine Ditch; 2) five wells (MW-4B, B/W-34B, B/W-6B, LEP-MW-2B, and B/W-33B) located to the immediate west of the Evaporation Ponds in an area potentially influenced by MFR; and 3) in two wells (B/W-54B and YPT-MW-10B) located north of Sunset Hills.

After installation and testing of bedrock monitor wells in late 2013 and 2014, and a technical meeting in May 2015 to discuss the full set of bedrock information, EPA (2015a) concluded that sufficient data had been collected to conclude that bedrock is not an important migration pathway at the Site, and requested preparation of a technical memorandum to update the bedrock HCSM. The Bedrock Groundwater Assessment Technical Memorandum (BC, 2015a) is provided in Appendix I.

5.2 Sources of Impacted Groundwater

OU-1 RI characterization activities indicate that sources of COIs to Study Area groundwater have included:

- Mine waste facilities, which have been grouped into the various Site OUs for individual RI/FS investigations;
- Agriculture activities that contribute COIs or affect the release of naturally-occurring COIs in geologic materials (BC, 2014a, 2016b); and
- A modeled fossil geothermal system and associated mineralization of the aquifer materials (Appendix N, BC, 2009a, 2014a, 2016b).

5.2.1 Mine Waste Facilities

The major past sources of COIs to groundwater include: 1) OU-4a, the Evaporation Ponds (BC, 2014a); 2) OU-8, the Arimetco Facilities (CH2M Hill, 2010, 2011a, 2011b); and 3) OU-3, the Process Areas (BC, 2014a). Concentrations of COIs in groundwater beneath OU-4b (Sulfide Tailings) are generally at least 10 times lower than COI concentrations in groundwater beneath OU-4a. Furthermore, OU-4b is located hydraulically upgradient or cross-gradient of OU-4a in the Shallow and Intermediate Zones, and groundwater beneath OU-4b flows toward the higher COI concentrations under OU-4a.

Evaporation Ponds (OU-4a)

OU-1 RI information that is particularly relevant to OU-4a includes the following: 1) the most elevated concentrations of mine-related chemicals occur in groundwater in this area; 2) concentrations of mine-related chemicals in groundwater beneath OU-4a are 100 or more times greater than chemical concentrations in groundwater beneath other OUs.

The Final OU-4a RI Report (Copper Environmental Consulting and Broadbent and Associates, 2019) concludes that:

- During historical operations, OU-4a was the major source of COIs to groundwater (primarily from the UEP).
- OU-4a, including the UEP, LEP, Finger Ponds (FEPs 1-4), Thumb Pond (FEP 5), Calcine Ditch, and other facilities, is not discernable as an ongoing source of COIs to groundwater based on the evaluation of multiple lines of evidence including vadose zone modeling, monitoring, and geochemical data, including an assessment of uncertainty.

Movement of COIs through the vadose zone to groundwater was different during historical operations compared to the post-operational condition (post mine operation). During the historical operations period, spent process-related solutions were applied to the impoundments which created a source of infiltrating pore water with COIs to the groundwater. During the post-operational period, infiltrating pore water was only through direct precipitation (including snowmelt) on the Evaporation Ponds and Calcine Ditch and is the only potential source of water for infiltration. The average annual precipitation rate is approximately five inches/year. The annual pan evaporation rate is about 13 times greater than the precipitation rate resulting in a strongly net evaporative condition in the vadose zone. Evaporation from the vadose zone reduces the soil moisture in shallow soil and sediment to less than field capacity, which is the point at which gravity drainage ceases.

Thus, under average annual conditions, deep flux of pore water to groundwater does not occur. Multiple lines of evidence were used to evaluate actual ongoing COI transport from the OU-4a facilities through the vadose zone to groundwater in the post-operational condition. These lines of evidence are:

- Vadose zone modeling;
- Soil moisture trends and pore water samples collected from various depths in the lysimeter monitoring program; and
- Laboratory measurements of moisture content, porosity (and therefore percent saturation), and saturated and unsaturated hydraulic conductivity of sediment and soil samples.

The evidence regarding the current post-operational COI transport pathway beneath each OU-4a area is discussed below. As discussed in Copper Environmental Consulting and Broadbent and Associates (2019) and summarized below, the monitoring period included the opportunity to evaluate infiltration during several large precipitation events. The vadose zone monitoring period included the evaluation of soil moisture conditions and infiltration during and following a large precipitation event between January 4 and 13, 2017 (2.34 inches of precipitation at AM-6 on-Site) and a subsequent November 2017 event (0.52 to 0.76 inches of precipitation). Off-site, the 10-day total of 4.35 inches measured during the January event at Yerington was 86% of annual mean precipitation (5.06 inches) and was between a 500- and 1,000-year storm event. No deep percolation from these large events was observed. In many cases, the soil moisture conditions during the monitoring period were relatively invariant demonstrating that infiltration from these large precipitation events did not percolate to the sensor depth.

UEP

Vadose zone modeling results for the UEP depict an upward net annual moisture flux in the soil profile for the modeled time period. The dominantly upward flux is caused by upward hydraulic gradients driven by evaporation.

The depth to groundwater at the UEP vadose zone monitoring location is approximately 20 feet bgs, and the vadose zone monitoring equipment is located at 1.5, 5, and 15 feet bgs. The UEP soil moisture monitoring data indicate that the shallow 1.5-foot depth sensor responded to the precipitation events, but sensors placed at 15 feet did not respond to either precipitation event. These data indicate that even significant precipitation events during the monitoring period did not result in deep infiltration of a wetting front at the vadose zone monitoring locations in the UEP.

The vadose zone pore water concentrations of COIs measured at the UEP lysimeter are greatest in clay present at the 5-foot depth, and concentrations in silty sand at the 15-foot depth are similar to concentrations in groundwater. The similarity of the chemistry of pore water at this depth and groundwater reflects the upward flux of groundwater in the capillary fringe.

LEP

The LEP has an asphalt liner. The effect of this is evident in that the LEP contains more ponded surface water after precipitation events than the lithologically similar UEP. Therefore, while the liner is an important hydrologic component in the soil profile affecting infiltration to the vadose zone, this component was not included in the model, therefore limiting the usefulness of the LEP modeling.

The depth to groundwater at the LEP vadose zone monitoring location is approximately 23 feet bgs, and the vadose zone monitoring equipment is located at 3, 10, and 20 feet bgs. The soil moisture data collected in the LEP indicate that ponding at the surface due to precipitation, which was observed by the 3-foot deep soil moisture sensor, did not result in measurable changes in soil moisture at depth (10- and 20-foot depths). These data indicate that even significant precipitation events do not result in deep infiltration of a wetting front in the LEP.

The evaluation of COI concentrations observed in lysimeter pore water samples indicate that the 3-foot depth samples reflect the chemistry of LEP surface water quality. Below this shallow lysimeter, the chemistry of pore water samples collected from the 10-foot depth is relatively invariant, often near background concentration limits (BCLs), and the chemistry of samples collected from the 20-foot depth is similar to that of the underlying groundwater.

The overall evaluation of pore water measurements at the 10- and 20-foot depths in the LEP lysimeter array indicates that movement of mine-influenced water through the vadose zone was not extensive based on the distribution of COIs and the soil hydrogeologic properties. Therefore, future migration through the vadose zone at this location is not likely under the current, post-operational conditions that are even less favorable to downward migration.

Finger Ponds (FEPs 1-4)

Vadose zone monitoring was not performed for FEPs 1-4. However, boring data indicate that COI concentrations in alluvium beneath the Finger Ponds are much lower than beneath the Thumb Pond. FEPs 1-4 contain an asphalt liner and were operated for only four years. Based on the model for the Thumb Pond and modeled water content near the water table for FEPs 1-4, it is expected that an upward moisture flux exists at the Finger Ponds under current conditions.

Thumb Pond

The simulated average annual moisture flux beneath the Thumb Pond is upward for all years in the vadose zone model. The depth to groundwater at the Thumb Pond monitoring location is approximately 45 feet bgs, and the vadose zone monitoring equipment is located at 10, 20, 30, and 40 feet bgs. The vadose zone soil moisture monitoring data from all sensor depths indicate no advancing wetting fronts caused by the precipitation events at any of the monitored depths.

The concentrations of COIs in lysimeter samples appear static and provide no indication of downward COI transport. At the deeper locations, concentrations of COIs are generally similar to groundwater concentrations.

Calcine Ditch

The depth to groundwater at the Calcine Ditch vadose zone monitoring location is approximately 55 feet bgs, and the vadose zone monitoring equipment is located at 2, 6, 20, 30, and 47 feet bgs. The soil moisture monitoring data indicates that the surficial VLT and near-surface sulfide tailings sensors responded to the precipitation events, but the three deeper sensors did not respond to the precipitation events. These data indicate that even significant precipitation events such as the January 2017 storm do not result in deep infiltration of a wetting front beneath the Calcine Ditch.

Concentrations of COIs in most of the lysimeter pore water samples are static and provide no indication of downward COI transport. The similarity of water quality measured in the deepest lysimeter with local groundwater quality reflects an upward flux of groundwater in the capillary fringe.

Other Facilities

The PWS Evaporation Pond is a facility containing solids from the evaporation of groundwater pumped during a past interim action. Due to the presence of clay liner in the three ponds, and HDPE on top of the clay liner in two of the three ponds, transport of COIs to groundwater from the PWS pond may not have been significant. The ancillary peripheral areas are small, and transport of COIs from these facilities to groundwater under post-operational conditions is unlikely.

In summary, OU-4a, including the Thumb Pond, FEPs 1-4, Calcine Ditch, and other Ancillary Facilities, is not an on-going or likely future source of any discernible flux of COIs to groundwater. This conclusion is based on multiple lines of evidence and the evaluation of current moisture conditions, depth to water, and geochemical distribution of COIs in pore water, soil, and groundwater. It is possible in the future that the UEP and to a lesser extent, the LEP, could potentially contribute limited mass of COIs through the vadose zone to groundwater if rare combinations of storm events and antecedent soil moisture conditions resulted in downward flux in some portions of these ponds, but the results of site characterization work indicate such limited contribution is not likely to occur.

Arimetco Facilities (OU-8)

OU-8 components located throughout the Site include five HLPs (Phase I/II, Phase III South, Phase III 4X, Phase IV Slot, and Phase IV VLT), the FMS (which stores and conveys drain-down solution via a network of ponds, ditches, and 25,000 feet of pipe), and the SX/EW Plant.

RI activities characterized the nature and extent of radiochemicals, metals, and physical properties of the OU-8 HLPs and their associated ponds and ditches. Sources of contamination include:

- Leachable metals (aluminum, copper, iron, and manganese and, to a lesser extent, arsenic, beryllium, cadmium, chromium, cobalt, mercury, and nickel) and other COIs on the surface and within the HLPs;
- Acidic drain-down solutions containing COIs entrained within the HLPs;
- Acidic drain-down solutions containing COIs stored at the base of the HLPs or contained within their associated ponds and ditches; and
- Historical spills and releases containing COIs.

The OU-8 RI/FS determined that the areas affected by Arimetco operations include the footprints of each HLP and their associated drain-down FMS components, the SX/EW Plant, and historical spill areas (CH2M Hill, 2011b). The environmental release or migration pathways of drain-down fluids are through infiltration into the subsurface from unlined areas, through tears/breaches in liner systems and FMS components, and through tears/breaches due to potential settling/structural failure of the HLP liner systems (Ecology and Environment, Inc. 2013).

On the basis of groundwater monitoring results, these impacts are thought to extend vertically down to OU-1 groundwater (CH2M Hill, 2011b). Furthermore, the OU-8 FS (CH2M Hill, 2011b) also notes that additional characterization efforts are needed to fully determine the nature and extent of contamination in: 1) OU-8 surface and subsurface soil due to releases of drain-down fluids from the Arimetco Facilities; and 2) OU-1 groundwater that may be attributed to OU-8 releases.

Numerous spills of process solution in connection with past Arimetco operations have been recorded, and all of the recorded spills report little to no confirmation sampling data or post-remedial efforts (CH2M Hill, 2010, 2011b). As noted by CH2M Hill (2011b), the spill report documentation in the HSR (CH2M Hill, 2010) only generally describes the location and type of materials spilled, along with the estimated quantity of each spill and general response action that was taken. In some instances, these records appear to underestimate the overall quantity of materials spilled. On the basis of the existing spill reports and the suspected quantities of fluids released to the environment, further investigations may be required (CH2M Hill, 2011b).

Process Areas (OU-3)

The OU-3 RI involved extensive characterization to determine the vertical extent of impacted soils beneath known source areas and above known areas of impacted groundwater in the underlying alluvial aquifer (e.g., BC, 2011a, 2014e). OU-3 RI activities have included: 1) soils and groundwater characterization in 2004-2005 (supplemented with subsequent routine OU-1 related groundwater monitoring); 2) a variety of radiometric surveys; 3) groundwater monitor well installations in 2005; 4) subsurface utility and dry well investigations during the period 2010-2013; 5) step-out (vertical and horizontal) soil characterization activities during the period 2013-2014; and 6) sampling and analysis of standing water in select locations during the 2013-2014 field program.

During the RI, a total of 3,385 samples of vadose zone alluvial soils were collected and analyzed for metals (57,764 analyses) and radiochemicals (9,172 analyses). Analytical results are discussed in the context of exceedances relative to the EPA Industrial regional screening level (RSL), a background level, and the maximum depth below ground surface that such exceedances occurred. A total of 198 metals exceedances, primarily arsenic and chromium, occurred in near-surface soils to approximately 15 feet bgs in every sub-area of OU-3. Metals exceedances also occurred at depths to 80 feet bgs at three primary waste solution conveyance ditches (Overflow, East Solution, and Calcine Ditches) and the Acid Plant Pond. Fifty-one radiochemical exceedances, primarily for radium-226 and -228, occurred in shallow soils to depths of up to five feet bgs throughout OU-3. Three exceedances occurred in the southern Calcine Ditch (now included in OU-4a) to a depth of 20 feet.

COI concentrations in OU-3 groundwater are highest beneath the Precipitation Plant (Sub-area 5 on Figure 2-1) and are typically 10 times less than the concentrations in groundwater beneath OU-4a (Figures 5-1 to 5-6). The depth to groundwater beneath OU-3 ranges from 90 to more than 120 feet bgs (Figures 4-8 and 4-9). Vadose zone alluvial materials beneath OU-3 do not differ substantially from the vadose zone alluvial materials beneath OU-4a. Thus, to the extent that insights from the OU-4a vadose zone modeling results and soil moisture profiling are applicable to OU-3, groundwater impacts beneath OU-3 are the result of past mining operations and fluid management rather than post-mining ambient Site climate conditions.

Wabuska Drain (OU-7)

The Wabuska Drain (Drain) was constructed in the late 1930s to stabilize soils adjacent to the Nevada Copper Belt Railroad and several nearby farms. Since its construction there have been no reported discharge of process fluids, other mine waters, or surface sediment to the Drain. Occasional releases of groundwater with MIW as seeps to the Drain occurred in the late 1950s. In addition, the rising water table in the late 1970s may have brought groundwater into the Drain. A summary of the historical, documented releases is described by Trelease (1958) and Seitz, et al. (1982). They are also summarized in the Site History section of the forthcoming Draft Wabuska Drain OU-7 Remedial Investigation Report.

The historical water table within the Mason Valley has declined for decades as a result of pumping for agricultural use. USGS wells installed within the valley show the decreasing water table trend since 1985. ACMS groundwater monitor wells show that the water levels have been below the southern portion of the Drain for many decades as a result of pumping. Chemical concentration maps have not shown evidence of influence to the Shallow Zone that can be attributed to OU-7 sources. Additionally, soil COI concentrations decrease with depth in samples collected during the OU-7 RI; therefore, there is no evidence in soil data of transport to the shallow groundwater system. Data collected during the groundwater and Wabuska Drain RIs, combined with monthly field observations of the Drain condition in 2017 and 2018 support the fact that there is an incomplete transport pathway of mine-influenced materials from the ACMS to the Drain. Thus, the Drain does not currently act as a source of MIW to groundwater. Recent observations indicate irrigation return water does enter the Drain but the main source of recharge to the shallow groundwater zone is from irrigation of the agricultural fields immediately north of the Site. This has been shown via numerous potentiometric surface maps for several years of groundwater monitoring reports. This irrigation does not originate at the ACMS. A further discussion is provided in the forthcoming Draft Wabuska Drain OU-7 Remedial Investigation Report.

In summary, the information presented above identifies historical sources to Site-wide groundwater from OU-4a, OU-8, and OU-3. The Final OU-4a RI Report concludes that the Evaporation Ponds (OU-4a) are not discernable current sources to groundwater. The two other OUs with Final RI Reports (OU-3, Process Areas and OU-8, Arimetco Facilities) either defer the issue of ongoing sourcing to groundwater to the OU-1 Report (i.e., OU-3) or do not report ongoing sourcing of COIs to groundwater (i.e., OU-8). The other OUs do not have approved Final RI Reports yet. As stated above, data evaluation supports the conclusion that groundwater impacts beneath OU-3 are the result of past mining operations and fluid management rather than post-mining ambient Site climate conditions. Monitor well-specific temporal trends of COIs are evaluated relative to the on-going source issue in Section 6.4.

5.2.2 Agriculture

The agricultural fields adjacent to the Site, referred to as the Hunewill Ranch (SSPA, 2014), consist of 718 acres of irrigated agricultural fields. Crop production and irrigation practices change from field-to-field and from year-to-year, in the form of crop rotation, addition of soil amendments, and irrigation practices (SSPA, 2014). These agricultural activities have had a significant impact on groundwater quality and flow as discussed further below. The large changes that have occurred in sulfate and uranium concentrations in groundwater beneath and downgradient of the fields are the result of variable agricultural practices.

Groundwater flow near the fields is dominated by the infiltration (recharge) of excess irrigation waters. On average, it has been estimated that recharge on the irrigated fields is approximately one foot per year (SSPA, 2014). Over the past 50 plus years, the amount of irrigation water that has infiltrated to the water table has been more than sufficient to replace all of the water in the Shallow Zone. As a result, the current water quality in the Shallow Zone beneath the agricultural fields is related to the quality of the irrigation water applied to the field. Major causes of changes in COI concentrations in groundwater due to agricultural activities include: 1) leaching and infiltration of fertilizers and other agricultural amendments; 2) leaching of naturally-occurring COIs in the unsaturated zone by irrigation waters; 3) changes in soil alkalinity beneath irrigated fields causing the release of uranium from sediments and soils and enhancing its mobility (Benson and Leach, 1979; Nolan and Weber, 2015); 4) evaporative concentration of COIs in applied irrigation water that infiltrates to the water table; 5) changes in groundwater flow direction from irrigation pumping; and 6) changes in groundwater flow direction from infiltration of applied water on irrigated fields and directly from irrigation ditches. The influences of agricultural practices on groundwater quality at the Site are discussed further in Section 5 of Appendix M.

The evaluation of agricultural influences on NSA groundwater performed during the RI are summarized below. Benson and Spencer (1983) noted that "agricultural practices strongly influence the concentration of uranium in Walker River and its East and West Forks. Irrigation practices induce significant losses of fluid through evaporation and evapotranspiration processes. This results in artificial increases in concentrations of uranium and other elements." Percolation of crop irrigation water through soils increases alkalinity in soil moisture, which has been shown at other sites (Jurgens et al., 2010; Brown et al., 2007) to solubilize and desorb naturally-occurring uranium from sediments resulting in elevated uranium concentrations in Shallow Zone groundwater. Application of agricultural amendments and fertilizer on crop fields contributes sulfate, calcium, nitrate, and other COIs to groundwater (Benson and Spencer, 1983; BC, 2014a, 2016b). Nitrate originating primarily from surface-applied fertilizers also plays a role in uranium solubilization leading to uranium mobilization (Nolan and Weber, 2015).

Groundwater data from the NSA show that agricultural activities contribute sulfate, uranium, nitrate, alkalinity, and other COIs to groundwater. The NSA refers to the portion of the groundwater Study Area located northeast of West Campbell Ditch and north of the Sunset Hills. Monitor wells and/or well clusters located in the NSA include B/W-10(S, D1), B/W-50(S, D1, D2, D3), B/W-53(S1, S2, B), B/W-54(S, I, B), B/W-55(S, D1, D2), B/W-56S, B/W-57(S, I, D1, D4), B/W-58(S, D1, D3, B), B/W-59(S, D3), B/W-68(S, D1, D4), B/W-69(S, D1, D2, D5), B/W-81(S, D1, D2), B/W-82R(S, I, B), YPT-MW-(6S, 15I), YPT-MW-(8S, 9I, 10B), YPT-MW-(11S, 12I), YPT-MW-13I, YPT-MW-14D1, and MMW-2. Groundwater flow data and chemical distributions from monitor wells and clusters B/W-56, B/W-57, B/W-59, B/W-68, B/W-69, and B/W-81 discussed below illustrate the effects of agricultural practices on groundwater quality near agricultural fields in the northeastern part of the Study Area. The locations of these wells are shown on Figure 3-3 and potentiometric surface maps for the alluvial aquifer (Figures 4-11[a-j]) indicate that groundwater in the NSA flows to the north and northwest.

Groundwater near these wells in the NSA is recharged by infiltration from the East Campbell Ditch and irrigation on a series of agricultural fields (BC, 2014a; SSPA, 2014). During drilling and installation of monitor wells, zonal groundwater samples were collected to profile vertical chemical gradients in the alluvial aquifer upgradient and downgradient of the fields. Chemical profiles for sulfate, uranium, alkalinity, and arsenic in groundwater are provided on Figure 5-9 for B/W-59, B/W-68, and B/W-69, and on Figure 5-10 for B/W-56, B/W-57, and B/W-81. The wells are grouped in this manner to illustrate changes in groundwater chemistry along two parallel flow paths beneath the agricultural fields.

Except for arsenic, chemical concentrations in alluvial groundwater in the Shallow through Deep 2 Zones (i.e., above 4,120 feet amsl) increase along the flow path beneath the agricultural fields (i.e., from B/W-59 and B/W-56 upgradient of the agricultural fields to B/W-68 and B/W-57 immediately downgradient of the agricultural fields). B/W-69 and B/W-81, located farther along their respective flow paths, also exhibit elevated values above 4,120 feet amsl, although the values are not as high as in B/W-68 and B/W-57 immediately downgradient of the agricultural fields. From B/W-56 to B/W-57, sulfate values exhibit more than a four-fold increase from about 30 mg/L at B/W-56 to a maximum of 137 mg/L at B/W-57. Uranium values exhibit more than a ten-fold increase from about 2 µg/L at B/W-56 to a maximum of 73 µg/L at B/W-57. Alkalinity exhibits a 2.5-fold increase from about 100 mg/L at B/W-56 to a maximum of 257 mg/L at B/W-57. From B/W-59 to B/W-68, sulfate values exhibit a two-fold increase from about 75 mg/L at B/W-59 to a maximum of 140 mg/L at B/W-68. Uranium values exhibit almost a two-fold increase from about 25 µg/L at B/W-59 to a maximum of 44 µg/L at B/W-68. Alkalinity exhibits a 20% increase from about 190 mg/L at B/W-59 to a maximum of 235 mg/L at B/W-68.

Arsenic exhibits increasing concentrations with depth at all locations, with the greatest increases observed in the Deep 3 through Deep 5 Zones (i.e., below 4,120 feet amsl). Arsenic values generally increase from about 5 µg/L in the Shallow Zone to approximately 80 µg/L in the Deep 5 Zone. Arsenic values decrease along the flow path beneath the agricultural fields (the highest values are observed in B/W-59 and B/W-56, and the lowest values are observed in B/W-69 and B/W-81).

Data from well clusters B/W-59, B/W-68, B/W-69, B/W-56, B/W-57, and B/W-81 indicate that concentrations of other COIs, in addition to sulfate, uranium, and alkalinity, also increase in groundwater above 4,120 feet amsl along the flow path beneath the agricultural fields (Table 5-

1). These COIs include TDS, calcium, chloride, magnesium, potassium, nitrate, and strontium. The average horizontal groundwater flow gradient in Shallow Zone groundwater beneath the agricultural fields is approximately 0.002 feet/foot. Vertical (downward) groundwater flow gradients beneath the agricultural fields range between 0.02 feet/foot when agricultural pumping is not occurring and 0.1 feet/foot during pumping periods (calculated using water levels in B/W-57S and B/W-57D4).

Increasing alkalinity and calcium concentrations are important controls on mobilization of naturally-occurring uranium from aquifer solids (Bernhard et al., 2001). The correlation between increasing concentrations of alkalinity and calcium associated with agricultural activities and increasing uranium concentrations as groundwater flows beneath agricultural fields in the NSA is discussed in Section 6.2.2.

Table 5-1. Concentrations of Constituents that Increase Beneath the Agricultural Fields in the North Study Area

Constituent	Units	Average Concentration in Monitor Well (May 2012 - October 2014)				
		B/W-59S	B/W-68S	B/W-68D1	B/W-69S	B/W-69D1 & B/W-69D2
Sulfate	mg/L	48	110	79	125	79
Uranium	µg/L	9	30	41	29	30
Alkalinity, bicarbonate (as $CaCO_3$)	mg/L	140	260	230	275	188
Total Dissolved Solids (TDS)	mg/L	260	530	440	540	400
Calcium	mg/L	31	49	76	78	64
Chloride	mg/L	16	20	20	25	20
Magnesium	mg/L	9	12	19	20	15
Potassium	mg/L	4	5	6	6	6
Nitrate (as N)	mg/L	0.06	5.0	2.9	1.2	0.9
Strontium	mg/L	0.3	0.5	0.6	0.7	0.6
Constituent	**Units**	**Average Concentration in Monitor Well (May 2012 - October 2014)**				
		B/W-56S	**B/W-57S**	**B/W-57I & B/W-57D1**	**B/W-81S**	**B/W-81D1 & B/W-81D2**
Sulfate	mg/L	41	62	142	70	62
Uranium	µg/L	4	11	75	4	36
Alkalinity, bicarbonate (as $CaCO_3$)	mg/L	140	174	293	160	213
Total Dissolved Solids (TDS)	mg/L	254	345	620	350	370
Calcium	mg/L	31	41	103	50	64
Chloride	mg/L	15	18	32	21	20
Magnesium	mg/L	8	11	25	13	15
Potassium	mg/L	5	3	9	5	7
Nitrate (as N)	mg/L	0.08	1.3	1.4	1.4	0.7
Strontium	mg/L	0.3	0.4	0.9	0.4	0.6

Notes: mg/L = milligrams per liter; µg/L = micrograms per liter

5.2.3 Geologic Sources

Sulfate, uranium, arsenic, and other COIs occur naturally in groundwater in Mason Valley because of groundwater contact with mineralized and/or hydrothermally-altered bedrock associated with copper deposits and/or alluvial sediments derived from mineralized bedrock (BC, 2014a, 2016b). A comprehensive evaluation of a geothermal feature in the Study Area is provided in Appendix N and briefly summarized below.

Researchers within the Great Basin have developed methods for discovering unknown or buried (hidden) geothermal systems by evaluating the regional and local geologic structure together with the local groundwater geochemistry. Faulds et al. (2011) determined that the majority of known geothermal systems are associated with only a few types of geologic structures, one of them being a step-over fault that bridges between two fault systems. They also identified certain trace elements (e.g., arsenic, boron, and fluoride) that strongly correlate with known geothermal systems.

The local geologic structure within the Study Area contains a step-over fault associated with the Sales Fault located between the Montana-Yerington Fault to the west and the Range Front Fault to the east (Figure 4-1). The step-over fault zone intersects the southeast to northwest lineament associated with the Bear Deposit, a copper porphyry deposit within the Yerington District. These copper deposits were emplaced in the Jurassic period followed by hydrothermal alteration via magmatic and non-magmatic fluids later in the Mesozoic era. The hydrothermal alteration created distinct zones defined by enrichment or depletion of trace elements, generally following the trend of heavier or less mobile elements (e.g., copper) toward the base, and lighter or more mobile elements (e.g., arsenic) toward the surface. Extensional faulting rotated these deposits toward the west-northwest by 70 degrees creating the southeast to northwest lineament (Proffett, 1977). This near-horizontal orientation of the Bear Deposit places the step-over fault zone adjacent to the Bear deposit mineralization, which is enriched with lighter elements.

Locally, naturally-occurring COIs may be associated with a fossil geothermal system inferred from geochemical modeling. Possible naturally-occurring COIs may be commingled with COIs from other sources, including MIW from the ACMS and/or agricultural water.

5.3 Background Assessment Approach

Identifying background groundwater types and defining the extent of MIW have been elements of phased OU-1 characterization activities since 2005 (BC, 2017). These characterization activities identified elevated concentrations of COIs in Study Area groundwater contributed by multiple sources including mining, natural processes, and agricultural and irrigation practices (BC, 2016a, 2017; SSPA, 2019).

However, EPA directed ARC to delineate the extent of using an approach that did not account for all anthropogenic or naturally-occurring background sources within the Study Area (EPA, 2016a, 2016b). This FRIR including Appendix N (Final GTM) addresses the acknowledged limitations and conservative assumptions that were identified by ARC, EPA, NDEP, and other stakeholders during development of the background assessment approach.

A background assessment was performed in several iterations (as documented in Section 1.3 of Appendix N) and was finalized as the Background Groundwater Quality Assessment – Revision 3 (BGQA) (BC, 2016a). The BGQA is provided in Appendix J-7. The assessment approach used in the BGQA is summarized in Section 5.3.1. Evaluation of RI data continued after the BGQA was completed. This work included preparation of the GTM (Appendix N), the PSTM (Appendix M), and revisions to the draft RI Report in response to agency comments. The post-BGQA evaluations identified elements of the BGQA approach that required revisions to the HCSM. The post-BGQA evaluation process is described in Section 5.3.2. Figure 5-11 provides the flow diagram that summarizes how 1) the BGQA in the draft OU-1 RIR was performed, 2) the post-BGQA evaluations were performed and compared with the BGQA, and 3) multiple new or revised LOEs were developed to map the extent of MIW.

The following terminology is used in this report to avoid confusion with prior documents. The areas of groundwater addressed by the BGQA are referred to as "BGQA water." The areas addressed by the post-BGQA multiple lines of evidence are referred to as "mine-influenced groundwater" (MIW). The distinction of the two terms is based on the evaluations and conclusions in this report that the BGQA boundaries include the extent of undifferentiated COIs in groundwater from other sources in addition to mine sources, whereas the extent of MIW groundwater represents only mine sources.

5.3.1 Background Groundwater Quality Assessment

The complete BGQA is provided in Appendix J-7. Key elements of the BGQA technical approach are described below.

Distinct sulfur isotope ($\delta^{34}S_{SO4}$) signatures associated with sulfate in the acidic process leach solutions were used to identify BGQA water. Early copper extraction operations at the Site (during the 1950s and 1960s) primarily relied on sulfuric acid derived from sulfur ores (i.e., pyrite) from the Leviathan Mine in California (BC, 2014a), which had a distinct sulfur isotopic signature of -17.6‰ (Taylor and Wheeler, 1994). Sulfur sources with different (higher) isotopic signatures were used during later operations in the 1970s, 1980s, and 1990s. The extent of BGQA water was identified where the $\delta^{34}S_{SO4}$ value in alluvial groundwater downgradient of the Site exceeded the minimum BGQA background value of +4.93‰.

The BGQA technical approach used a $\delta^{34}S_{SO4}$ signature thought to reflect background conditions for groundwater in the Study Area. Groundwater modeling (SSPA, 2014) indicates that recharge to groundwater in the alluvial aquifer in the Study Area results primarily from irrigation field percolation (49%), leakage from irrigation ditches (29%), and seepage from the Walker River (20%). Other minor sources of recharge include MFR and recharge from Weed Heights and Yerington wastewater treatment ponds. Eight wells that represent the dominant background groundwater types in off-Site areas were selected to represent background for the sulfur isotopic ratio, based on their locations in the groundwater flow field relative to the Walker River and agricultural features. The BGQA $\delta^{34}S_{SO4}$ background wells are B/W-15S, B/W-20S, B/W-21S, B/W-56S, B/W-59S, B/W-59D3, PLMW-2S, and PLMW-2B. Values of $\delta^{34}S_{SO4}$ in groundwater samples from these wells range from +4.93 to +6.62‰.

Areal patterns in $\delta^{34}S_{SO4}$ groundwater data are illustrated on Figures 5-19(a-g). The distinctly negative $\delta^{34}S_{SO4}$ signature associated with the sulfuric acid process leach solutions is evident in the Shallow, Intermediate, and Deep Zones beneath the Evaporation Ponds. In the Deep Zones, distinctly negative signatures are especially pronounced in samples from the W5DB well cluster. The low $\delta^{34}S_{SO4}$ values in Deep Zone alluvial groundwater beneath the Evaporation Ponds also extend off-Site toward the Hunewill Ranch, particularly in the Deep 3, Deep 4, and Deep 5 Zones (Figures 5-19[e-g]).

5.3.2 Post-BGQA Evaluations

The extent of MIW presented in the draft OU-1 RI Report, based on the BGQA, reflected the outcome of discussions during a groundwater technical meeting on June 29, 2016 and EPA's subsequent direction to conservatively establish the extent of BGQA water because it is in the best interest of the overall progress for completing the OU-1 Groundwater Remedial Investigation Report (EPA, 2016e).

EPA (2016e) noted that professional judgment is an inherent component of estimating the extent of MIW in this complex setting, and that tools used in the background assessment to estimate the area of BGQA water may not fully account for the range of chemical concentrations that occur naturally in groundwater within the Study Area. However, as noted during previous submittals (e.g., ARC, 2016b) and during groundwater technical meetings, further refinements to the MIW boundary were warranted based on additional evaluations of monitoring data and new information to account for sources of COIs in groundwater other than mine-related activities.

The following RI activities were performed after the BGQA (BC, 2016b) was finalized:

- Ongoing evaluation of a potential geothermal anomaly, including submittal of the draft GTM and the final GTM (Appendix N of this FRIR);
- Preparation and submittal of the draft and final PSTM (Appendix M of this FRIR); and
- Continued data evaluation in preparation of this FRIR.

The results of these activities provide evidence that the extent of BGQA water defined on the basis of chemical distributions that outline distinct plumes of groundwater, which have physical continuity along plausible flow pathways linked to Site features that are known sources of groundwater impacts, significantly overstates the extent of mine-influenced water. The primary reasons why the extent of mine-influenced water is overstated are two-fold:

1. Irrigated agriculture on the Hunewill Ranch fields that lie directly north of the Site have resulted in large changes in sulfate and uranium concentrations in groundwater. These changes are the result of the use of sulfate-based soil amendments and the leaching of natural occurring uranium due to increases in alkalinity caused by the agricultural activities.
2. Sulfate and uranium concentrations in shallow groundwater in some areas northwest and north of the Site may be elevated due to the natural mineralization associated with a modeled fossil geothermal system.

The BGQA did not consider the influence of agriculture and mineralization associated with a geothermal system in developing the extent of impacted groundwater. As a result, mine-influenced water is only a subset of the BGQA water.

The detailed post-BGQA-related evaluations are presented below. The primary LOEs used to map the extent of BGQA water are sulfur isotopic data and sulfate and uranium concentrations but were based on a technical approach that is not consistent with information subsequently developed in the post-BGQA evaluations as explained below. EPA directed ARC to delineate the extent of MIW primarily using isotopic data from eight monitor wells located upgradient of the

Site and near the Walker River and the East Campbell Ditch (Figure 5-12). The Walker River and East Campbell Ditch recharge shallow groundwater (Section 4.7.2), and data from these wells represent groundwater recharge of low TDS water originating from snow melt in the Sierra Nevada mountain range that has had a relatively short residence time in the groundwater flow system as opposed to monitor wells in the west, where groundwater primarily originated as MFR that has had relatively long residence times in the groundwater flow system. Increases in dissolved chemical concentrations are associated with longer periods of contact between groundwater and aquifer sediments (i.e., longer groundwater residence times). Consequently, the $\delta^{34}S_{SO4}$ background dataset used to delineate BGQA water do not represent the hydrogeologic and geochemical characteristics of un-impacted groundwater more distant from the Walker River and the East Campbell Ditch.

The post-BGQA statistical evaluations indicate that there are additional non-mining related water types in the area (i.e., geothermal and regional [agricultural/Walker River] water types) that are not represented by data from the eight BGQA $\delta^{34}S_{SO4}$ background wells listed above. The water types of the eight wells are listed in Table 5-3. The sulfur isotopic ranges of the three major water types (mine-influenced, geothermal, and regional [agriculture/Walker River]) in RI groundwater samples are illustrated in the box and whisker plot provided in Figure 5-13. The box and whisker plot illustrates that there are many geothermal and regional (agricultural/Walker River) water type samples with $\delta^{34}S_{SO4}$ values less than +4.93‰. In fact, the median values of both of these water types are less than +4.93‰, and therefore the BGQA approach classified more than half of the samples in both water type groups as MIW. Conversely, the box and whisker plot indicates that there are mine-influenced water types with $\delta^{34}S_{SO4}$ values greater than +4.93‰. Therefore, the BGQA $\delta^{34}S_{SO4}$ background value of less than +4.93‰ is not an accurate indicator of a mine-influenced water type and is also not representative of the other two non-mining water types (geothermal and regional [agricultural/Walker River]). Post-BGQA evaluations indicate that only sulfur isotopic ranges below approximately -5‰ are uniquely associated with early releases from ACMS mine sources. This evaluation is provided in Section 5.4.4.

The BGQA used sulfate and uranium concentrations to augment the delineation of BGQA water based on the $\delta^{34}S_{SO4}$ background threshold of +4.93‰. Post-BQGA evaluations indicate that sulfate and uranium are not uniquely associated with mine-influenced water and are also associated with the geothermal and regional (agricultural/Walker River) water types. Absent other lines of evidence, sulfate and uranium concentrations cannot be used individually or exclusively to map MIW.

The box and whisker plots of sulfate and uranium concentrations by water type (Figure 5-13) include the average concentrations of these COIs in the eight BGQA $\delta^{34}S_{SO4}$ background wells (3Q 2014 data). The average sulfate concentration in the eight background wells is below the quartile ranges measured in the two non-mining water types. Sulfate is a major water chemistry anion and commonly involved with the evolution of water types and rock-water interaction, especially in a sedimentary basin that contains evaporites. The observation that sulfate concentrations in the BGQA $\delta^{34}S_{SO4}$ background wells are below the quartile ranges of sulfate concentrations in the geothermal and regional (agricultural/Walker River) water types indicates that these background wells are not representative of the geochemistry of the non-mining water types at locations distant from the Walker River or East Campbell Ditch. The average uranium concentration in the BGQA $\delta^{34}S_{SO4}$ background wells is also generally below the quartile range of the two non-mining water types. These observations further confirm that the BGQA $\delta^{34}S_{SO4}$ background value is not representative of the non-mining water types at locations distant from recharge areas. In summary the BGQA $\delta^{34}S_{SO4}$ background value is not useful to identify any of the three water types over most of the basin north of the Site.

The BGQA calculated BCLs for two purposes: 1) to use sulfate and uranium BCLs to confirm the extent of BGQA water defined by $\delta^{34}S_{SO4}$ signatures and 2) to define areas within the extent of

BGQA water where chemical concentrations exceed BCL background chemical concentrations. As explained above, the BGQA $\delta^{34}S_{SO4}$ background value is not useful to identify any of the three water types. The calculations of BCLs in the BGQA do define a large area of elevated concentrations of various COIs. However, the post-BGQA evaluations indicate that the large area of elevated COI concentrations is due to three different influences: MIW, geothermal, and regional (agricultural/Walker River). The BCLs cannot be used to subdivide the large area into subareas influenced by one or more of the specific influences because of many of the COIs (especially sulfate and uranium) are common to all three potential sources.

A north-south cross section of the alluvial aquifer that illustrates the northern extent of the BGQA boundaries (with 5 x vertical exaggeration) is shown in Figure 5-14. The Shallow through Deep 2 BGQA water extends much further to the north than the BGQA water in the deeper zones. There are only two hypotheses of groundwater flow and one hypothesis of a surface water/groundwater pathway that could explain the greater migration distance in the Shallow through Deep 2 Zones relative to the deeper zones. These hypotheses are discussed and evaluated below.

The first hypothesis is that there is a confining layer(s) such as lacustrine clay sediments present between Deep 2 and Deep 3 everywhere within the BGQA boundaries. A confining layer, if present, would limit mixing of COIs in the shallower zones with the deeper zones, and perhaps result in different spatial distributions of COIs versus depth. However, borehole logging information acquired prior to and during the RI does not show an extensive confining layer(s) in the alluvial aquifer throughout the BGQA area (Section 4.6). In summary, the geologic data do not indicate the presence of a confining layer needed to support the first hypothesis.

The second hypothesis is that the hydraulic conductivity of the shallower units is much greater than the deep units, allowing greater groundwater velocities and hence larger extents of BQGA water in the shallow units. The aquifer test data indicate that the deeper zones are actually more permeable than the shallower zones. Slug test data are summarized by zone in Table 4-6 and indicate that Deep 4 and Deep 5 Zones are more permeable that the overlying units, and the hydraulic conductivities of the Deep 2 and Deep 3 Zones are similar. RI pumping tests, which stressed a larger volume of the aquifer than the slug tests, yielded a median hydraulic conductivity of 4.1 ft/day for the Shallow Zone and 77 ft/day for the composite alluvial aquifer stressed by pumping in well WDW019 (Section 4.9.5). The calibrated horizontal hydraulic conductivities developed for the groundwater model vary within each model layer, but in the area between the Site and Sunset Hills the Shallow and Intermediate Zone values are generally below 10 ft/day, and much of the deeper zone layers are assigned values of greater than 30 ft/day (SSPA, 2014; Appendix H, Figures H-1 through H-7). In summary, the hydraulic test data do not support the second hypothesis that requires greater hydraulic conductivity in the shallower zones than in the deeper zones.

The third hypothesis would require a surface water pathway to transport mine-influenced water northward, and subsequently infiltrate to groundwater. This hypothesis would require aerially extensive and persistent surface water flooding over the area and then downward infiltration to only Shallow through Deep 2 Zones. This surface water pathway has not been documented and does not appear to be physically feasible.

In summary, the post-BGQA evaluations do not support the technical basis used in the BGQA to map the extent of MIW. Identification of MIW is complicated by the fact that there are no COIs uniquely associated with, and only with, mine-influenced water. The COIs associated with the mine are also associated with other potential sources such as agriculture, geothermal, and other natural sources of mineralization.

A multiple LOE approach has been developed to map the extent of MIW that is consistent with RI data.

5.4 Extent of Mine-Influenced Groundwater

The delineation of MIW described in this report is based on the evaluation of up to six LOE utilizing physical, chemical and isotopic information as follows:

- Groundwater flow distances (Section 5.4.1);
- Extent of nitrate from agricultural activities (Section 5.4.2);
- Multivariate statistical analysis (MSA) (Section 5.4.3);
- Sulfur isotopic information (Section 5.4.4);
- Age dating of groundwater (Section 5.4.5); and
- Collocation of MIW indicators (Section 5.4.6).

The method, application, and limitations and uncertainties of each LOE are presented below. There are uncertainties associated with the development and testing of any hypothesis using the scientific method, including the LOE presented herein. Limitations and uncertainties of a specific LOE are presented for each LOE.

5.4.1 Groundwater Flow Distances

Method

This LOE uses the physical properties of the alluvium and hydraulic gradients of groundwater to estimate how far groundwater may have migrated from the UEP and LEP since the time of initial releases. The concentrations of the COIs sulfate and uranium in on-Site groundwater are highest beneath the UEP. Migration from the Sulfide Tailings (OU-4b,located to the east of the Evaporation Ponds, is not included in this analysis because 1) this pond was used for disposal of acidic water for only a short period of time; 2) concentrations of the COIs sulfate and uranium in the Shallow Zone beneath the Sulfide Tailings are more than an order of magnitude less concentrated than concentrations beneath the UEP; and 3) any prior northward migration would have been reversed back towards the Site by irrigation recharge and the current flow direction beneath the Sulfide Tailings is westward toward the UEP.

This method does not consider attenuation or retardation of specific chemicals and solely assesses the migration of water. Groundwater travel distances are calculated for water in the Shallow Zone through the Deep 2 Zone. Deeper zones are not amenable to this approach due to interception of groundwater by large capacity pumping wells.

The method multiplies the average linear velocity of groundwater by the time of flow to estimate the travel distance. The direction of travel is estimated from the groundwater potentiometric surface. Average linear velocity (*v*) is defined by the following equation, which is a derivative of Darcy's Law:

$$\boldsymbol{v = Ki/n_e}$$

where: K = hydraulic conductivity,
i = hydraulic gradient, and
n_e = effective porosity.

Application

Groundwater travel distances were calculated over a range of values for the three parameters listed above using a Monte Carlo analysis for the Shallow Zone as sufficient information is available for this zone to define probability distribution functions for the parameters. For the Intermediate, Deep 1, and Deep 2 Zones, constant parameter values were used for calculating travel distance. The Monte Carlo approach involves performing numerous calculations using values randomly selected from a probability distribution for each parameter. The resulting output from the Monte Carlo simulation provides a description of groundwater travel distance probabilities.

The probability distribution function for hydraulic conductivity was derived from hydraulic conductivity estimates from aquifer tests conducted on wells in the PWS system located near the northern boundary of the LEP or immediately to the east of the LEP. These wells are located within the area in which COIs would migrate from the UEP and LEP toward the north. The hydraulic conductivity estimates for these wells were derived from four-day aquifer tests conducted at each of the wells. For these tests, drawdowns were monitored in three observation wells located five, 10, and 20 feet, respectively, from the tested well. The drawdown data were analyzed with the distance-drawdown method to derive the estimates of hydraulic conductivity. This method is the most appropriate method for analyzing the test data from these wells as it combines the drawdown data collected at the pumped well and the three observation wells in a single consistent analysis method. The aquifer test data from the PWS wells are described and evaluated in the *Summary of PWS Aquifer Testing* memorandum report (BC, 2010f). In this report, the aquifer test data are analyzed using various analytical aquifer models, including the distance-drawdown model.

The hydraulic conductivity values estimated using the distance-drawdown method from PW-4 through PW-11 range from two to 33 ft/day and are approximately log-normally distributed. The mean and standard deviation of a log-normal distribution based on these data were used to define the probability distribution function for hydraulic conductivity.

Hydraulic conductivity data from PWS wells PW-1, PW-2, and PW-3, which are located near the northeast corner of the LEP and the northwest corner of the sulfide tailings pond, were not used in deriving the probability distribution function for hydraulic conductivity. They were not used because of their location outside of the area of interest for the travel distance calculations, which was primarily the area between the LEP and the area near B/W-32S. The hydraulic conductivity values estimated by the distance-drawdown method for these three wells are 31 ft/day, 23 ft/day, and 11 ft/day, respectively (BC, 2010f). If these wells had been included in deriving the probability distribution function for hydraulic conductivity, the average hydraulic conductivity would have been higher. The average hydraulic conductivity of the eight PWS wells that were utilized is 12 ft/day, and the average hydraulic conductivity of all eleven PWS wells is 15 ft/day. In addition, hydraulic conductivity estimates from slug tests of monitor wells located between the northern boundary of the LEP and the area near B/W-32S were not used in deriving the probability distribution function. The spatial distribution of the hydraulic conductivity estimates from the slug tests are presented on Figure 4 in Appendix J-4. The slug test data were not used because the average value of hydraulic conductivity calculated from the slug tests was considerably lower than that calculated from the aquifer tests of the PWS wells, and as slug tests are short duration tests they are generally not as reliable for estimation purposes as long-term aquifer tests with observation wells.

The probability distribution function for effective porosity (n_e) was defined using a normal distribution that is described using values for the mean (0.15) and standard deviation (0.05). This distribution for n_e is truncated at a minimum (0.05) and maximum (0.25) because values above and below those maximums and minimums are unlikely for this system. This probability distribution function was derived based on the dominant characteristic of the more permeable units within the Shallow Zone (silty sand) and professional judgment. Effective porosity used in Intermediate, Deep 1, and Deep 2 Zone groundwater calculations was 0.15. Commonly, the specific yield of the aquifer is used as a surrogate for estimating effective porosity, though technically specific yield and effective porosity are not equivalent. The specific yield at eight of the PWS wells was estimated based on the aquifer test data from these wells (BC, 2010f). Specific yield was estimated for each of the eight PWS wells from data at each of the three piezometers associated with each of the PWS wells. For each of the PWS wells, the estimated specific yield declined markedly with distance from the pumped well; the highest specific yield was estimated at the closest observation well and the lowest specific yield was estimated at the furthest observation wells. The median specific yields were 1.4, 0.68, and 0.03, respectively, for the closest, middle, and farthest observation wells from the tested PWS well. These data are not useful for estimating the effective porosity, as a specific yield greater than one is not possible. Additionally, a declining trend in specific yield with distance from the tested well indicates a problem with the estimation method, as specific yield most likely is relatively constant with distance from the tested well. Regardless, evaluations of the capture zones of the PWS system were made using average and median specific yields of 0.12 and 0.07 (Appendix J-1). The average and median specific yields were derived from the aquifer test results for the PWS wells after manually eliminating all results that were greater than 0.4 and not considering the distance bias in the calculated specific yields.

The direction of groundwater flow and hydraulic gradients used in these calculations vary based on the groundwater zone and the time period of calculations. During Site operations, pumping stresses created by the mine water supply wells lowered groundwater levels in the deeper zones of the northern part of the Site and caused groundwater flow towards the pumping wells and limited northward migration of contaminants (e.g., Seitz et al., 1982). Anaconda groundwater supply pumping ceased in 1977. As shown on Figure 3-7, there was groundwater flow in the Shallow Zone northward from the LEP during the latter years of mine operation until the early 1990s, when flow directions were altered by irrigation on the adjacent fields. Arimetco required water supply during their period of operation from 1989 to 1999, but this pumping was from a water supply well located near the pit, and thus did not affect groundwater flow directions in the northern part of the Site. Therefore, the groundwater travel distance calculations assume northerly flow in the Shallow (and conservatively, the Intermediate Zone as well) from source areas after pond operations began until 1991.

The hydraulic gradients are taken from reports describing historical groundwater elevations, calculated gradients from elevations measured at paired wells, and RI potentiometric surface maps. The potentiometric surface map for the Deep 1 Zone (Figure 4-11j) indicates that the groundwater mounding created by irrigation could result in flow directions ranging from southeast to the north, depending on the specific location. The direction of flow varies toward the west from the western perimeter to toward the north from the northern perimeter.

The mean used in the Shallow Zone hydraulic gradient distribution for groundwater travel Monte Carlo simulations during 1954 to 1976 northward flow was 0.0015, which represents measurements between potentiometric contours of 1966 water level data and measured gradients in 1976 (Figure B-1 from SSPA, 2014). The hydraulic gradient immediately adjacent to the Evaporation Ponds may have been higher during a period of northerly flow; Seitz et al. (1982, p. 23) used a hydraulic gradient of 0.002. Therefore, a standard deviation of 0.0005 was selected to adequately capture the potential range in gradient during this time, including the hydraulic gradient of 0.002 used in Seitz et al. (1982). The same values (mean=0.0015, standard deviation=0.0005) were used to describe the Intermediate Zone hydraulic gradient during the assumed northward flow period (1954-1991).

Between 1976 and 1991, groundwater elevation measurements at W5AB-2S/USGS-7 and USGS-13 provide the basis for additional Shallow Zone hydraulic gradient parameterization (Figure 3-7). The mean hydraulic gradient between W5AB-2S/USGS-7 and USGS-13 during 1977-1986 is relatively stable, and the mean of 0.00125 was used for calculations during this time period. The gradient between W5AB-2S/USGS-7 and USGS-13 consistently decreases from 1987 to 1991. After 1991, the northward hydraulic gradient fluctuates around zero. Constant hydraulic gradients from W5AB-2S/USGS-7 and USGS-13 measurements are used for each year 1987-1991 and are provided in Table 5-2.

Review of information indicates that the areas of irrigation and agricultural well pumpage north of the Site from at least the late 1980s to the present were relatively constant. This is consistent with near neutral hydraulic gradients between well pairs W5AB-2S/USGS-7 and USGS-13, indicating that a significant increase in irrigated acreage and pumping in the late 1980s resulted in a change in groundwater flow from northward to westward in the area north of the LEP. The hydraulic gradients in this area from 1976 to 2012 are shown on Figure 3-7.

Three Shallow Zone well pairs on the west and northwest side of the Evaporation Ponds were selected to evaluate the northwest-oriented hydraulic gradient after 1991. These well pairs are: MW-5S and B/W-34S; B/W-44S and MW-2002-2S; and PW-11S and LEP-MW-2S. A normal distribution was used in the Monte Carlo simulations and utilizes the mean (0.00046) and standard deviation (0.00007) of 1Q hydraulic gradients for these well pairs during 2011-2018.

Two Intermediate Zone well pairs on the west and northwest side of the Evaporation Ponds were similarly used to describe the 1991-2014 Intermediate Zone hydraulic gradient. These well pairs are: MW-5I and B/W-34I; and W5AB-3I to the midpoint between LEP-MW-4I and B/W-2I ([LEP MW-4I + B/W-2I]/2). The hydraulic conductivity of the Intermediate Zone is specified as 12 ft/day, which is equal to the average value in the Shallow Zone based on similar boring log descriptions and slug test results (BC, 2010f).

The hydraulic conductivity used for the Deep 1 and Deep 2 calculations (constant of 30 ft/day) is the average hydraulic conductivity derived from a seven-day aquifer test at irrigation well WDW019, which is screened over much of the saturated alluvium (SSPA, 2014, p. 11) and was the initial hydraulic conductivity assigned to model layers representing the Deep 1 and 2 Zones in the groundwater model (SSPA, 2014, p. 11). Based on the above discussion, the calculation of groundwater travel distances in Deep 1 and Deep 2 Zones assumes that flow of groundwater outward from the UEP location began in 1978.

The travel time computation period ends in 2014 because that was when the RI groundwater chemistry data set was collected that provides the comprehensive snapshot of information used to characterize the nature and extent of groundwater contamination.

A summary of the parameter distributions and constant values used in the Monte Carlo analysis and groundwater travel distance calculations are provided in Table 5-2.

Table 5-2. Input Parameters for Groundwater Travel Distance Calculations

Parameter	Time Period	Distribution Type (or Constant)	Min	Max	Mean (or Constant)	Calculated Travel Distances
Shallow Zone						
Hydraulic Conductivity (Kh)	1954-2014	Log-Normal	-	-	12.0 ft/day	See Figure 5-15a
Effective Porosity (n_e)	1954-2014	Normal	0.05	0.25	0.15	
Hydraulic Gradient (i) Northerly Flow	1954-1976	Normal	-	-	0.0015	
	1977-1986	Constant	-	-	0.0013	
	1987	Constant	-	-	0.001	
	1988	Constant	-	-	0.0008	
	1989	Constant	-	-	0.0006	
	1990	Constant	-	-	0.0004	
	1991	Constant	-	-	0.002	
Hydraulic Gradient (i) Northwest Flow	1992-2014	Normal	-	-	0.00046	
Intermediate Zone						
Hydraulic Conductivity (Kh)	1954-2014	Constant	-	-	12 ft/day	1,183 ft North of UEP (1954-1990); 591 ft North of LEP (1972-1990); 631 ft Northwest (1991-2014)
Effective Porosity (n_e)	1954-2014	Constant	-	-	0.15	
Hydraulic Gradient (i) Northerly Flow	1954-1991	Constant	-	-	0.0015	
Hydraulic Gradient (i) Northwest Flow	1992-2014	Constant	-	-	0.0012	
Deep 1 and Deep 2 Zones						
Hydraulic Conductivity (Kh)	1978-2014	Constant	-	-	30 ft/day	2,168 ft Deep 1 (1978-2014) 1,774 ft Deep 2 (1978-2014)
Effective Porosity (n_e)	1978-2014	Constant	-	-	0.15	
Hydraulic Gradient (i) Deep	1978-2014	Constant	-	-	0.0011	
Hydraulic Gradient (i) Deep	1978-2014	Constant	-	-	0.0009	

Note: See text for sources of data and input assumptions.

Examples of Shallow Zone parameter selections based on distributions defined in Table 5-2 and Monte Carlo simulation results over 2,500 iterations are provided in Figure 5-15a. Monte Carlo simulation results display probabilities of exceeding a specific groundwater travel distance. For example, an exceedance probability of 50% corresponds to the median (50th percentile) groundwater travel distance calculated in the Monte Carlo analysis. The 25% probability corresponds to the 75th percentile of calculated travel distances, and the 5% probability corresponds to the 95th percentile of calculated travel distances.

Figure 5-15b shows the spatial extent of northward groundwater flow calculations at the 50%, 25%, and 5% exceedance probabilities for the periods of 1954-1972 from the UEP and 1972-1991 from the LEP. The 5% exceedance probability travel distance from the UEP for 1954-1972 extends to just north of the LEP at approximately the northern ACMS boundary. The 1972-1991 Monte Carlo simulation groundwater travel distances are extended northward as a cohesive unit from the ACMS boundary to provide a conservative representation of northward groundwater travel distances.

Figure 5-15c shows the Monte Carlo simulation travel distances for northwest groundwater migration from 1992-2014. Figure 5-15d shows the composite footprints of Monte Carlo simulation results for 1954-2014 for the 50%, 25%, and 5% exceedance probabilities. The groundwater travel distances for the Intermediate, Deep 1 and Deep 2 Zones are shown on Figures 5-21b, 5-21c, and 5-21d, respectively.

Groundwater travel distances for a five-year time period were calculated using the 1992-2014 parameters in Table 5-2. This is to provide a frame of reference for the time period (2014-2019) since the RI chemistry dataset. In the shallow zone, between 50 ft (50% exceedance probability) and 200 ft (5% exceedance probability) of travel distance is calculated. In the intermediate zone, approximately 175 ft of travel distance is calculated.

Uncertainties and Limitations

Uncertainties in the use of Darcy-based equations result from errors associated with the measurement or estimation of hydraulic conductivities, hydraulic gradients and effective porosity (Freeze and Cherry, 1979, p. 427). The largest uncertainties are related to hydraulic conductivity and effective porosity. Limited data are available to estimate hydraulic conductivity in the region beneath the LEP and north of the LEP to the vicinity of B/W-32S. The available data include slug tests at most of the monitor wells in this region and 4-day aquifer tests at the wells in the PWS. The latter were deemed to provide more reliable data for estimating hydraulic conductivity. The probability distribution function for hydraulic conductivity used in the Monte Carlo analysis was based on hydraulic conductivity estimates from the eight PWS well located along the northern boundary of the LEP and to the east of the LEP as these wells located in the region of interest; the slug test data were not used. Hydraulic conductivity data were not used from three PWS wells at the northwest corner of the Sulfide Tailings Pond and the northeast corner of the UEP. The data were not used because these three locations are outside of the area of interest for the travel distance calculations. The estimated hydraulic conductivities at these three wells are within the range of hydraulic conductivity values for the eight PWS wells used in the Monte Carlo analysis. Use of these three additional wells would have increased the average hydraulic conductivity from 12 feet per day based on eight PWS wells to 15 feet per day based on all eleven PWS wells. As noted above, the probability distribution function for effective porosity was based on dominant characteristics of the more permeable units within the Shallow Zone and expert judgement as no measurements of effective porosity have been made at the Site. Thus, there is significant uncertainty associated with the mean value used for effective porosity, which is in part accounted for by the large standard deviation used in the effective porosity probability distribution function. The use of a larger mean hydraulic conductivity in the probability distribution function and a lower mean effective porosity would have produced a probability distribution of travel distances that shifted the 50% confidence intervals further north, and use of a value lower than 0.05 for the low end of effective porosity would also have shifted the 5% confidence interval farther north.

5.4.2 Extent of Nitrate from Agricultural Activities

Method

As discussed in Appendix M (Section 5, p. 13), nitrate concentrations in groundwater are a tracer of agricultural influences on groundwater quality because nitrogen-containing fertilizers are used on the fields north of the Site. Nitrate is a not a mine-related constituent.[8] A large nitrate plume is centered on the agricultural fields north of the Site boundary. This nitrate plume, as defined in February 2018, is shown on Appendix M, Figure 5. The maximum nitrate concentration greater than 40 mg/L is near the center of the irrigated fields. Nitrate concentrations in most shallow wells beneath the fields exceed 10 mg/L. The configuration of the nitrate plume reflects, in part, the groundwater flow patterns created by water infiltration on the irrigated fields. Elevated nitrate concentrations beneath the Hunewill Ranch fields extend through the Deep 2 Zone.

Nitrate concentrations within the Site boundary are generally less than 1 mg/L, and therefore nitrate concentrations are a useful indicator of the extent of the impact of agricultural practices at locations within the groundwater plume extent. As discussed in the Plume Stability Technical Memorandum (Appendix M):

“Groundwater flow near the fields is dominated by the infiltration (recharge) of excess irrigation waters. On average it has been estimated that recharge on the irrigated fields is approximately one foot per year (SSPA, 2014). Over the past 50 plus years, the amount of irrigation water that has infiltrated to the water table has been more than sufficient to replace all of the water in the Shallow Zone. As a result, the current water quality in the Shallow Zone beneath the agricultural fields is related to the quality of the irrigation water applied to the field” (Appendix M, p. 13).

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[8] Most nitrate measured in on-Site wells appears to have originated in water supplied by mine water supply wells located in the northern part of ACMS adjacent to agricultural fields.

Application

The relative influence of the agricultural operations is illustrated by the nitrate contours shown in Figure 5-16. As a result, MIW is not discernable in areas where agricultural operations are the dominant influence on current groundwater quality. Although agricultural impacts are discernable at lower concentrations, the 5 mg/L nitrate contour is used as an exclusionary criterion to indicate where MIW is not discernable. These areas are indicated on Figures 5-21(a -g).

Some wells in the area to the west of the Site also yield groundwater samples with nitrate concentrations above 1.5 mg/L. The source of nitrate in this area may originate from historical (pre-mining) agricultural activities, the Weed Heights sewer system, individual septic systems, and/or possibly mine-related water produced from mine water supply wells that intercepted the agricultural nitrate plume present to the north of the Site. Due to the potential for multiple sources to have contributed to nitrate concentrations west of the Site, nitrate is not used as a line of evidence to delineate the extent of MIW on the western side of the Site.

Uncertainties and Limitations

Uncertainties and limitations in this line of evidence include the accuracy and precision of nitrate measurements. The QAPP specifies the accuracy of matrix spike duplicates as 80% to 120% and the precision limits as 20% for nitrate analysis (ARC, 2018).

As stated above, the amount of irrigation water that has infiltrated to the water table has been more than sufficient to replace all of the water in the Shallow Zone. To the extent there may be commingling of historical mine-derived constituents in areas dominated by agricultural influences, as reflected by elevated nitrate concentrations, the presence of such constituents may not be quantitatively discernable.

5.4.3 Multivariate Statistical Analysis

Method

MSA was used to explore the groundwater chemical data to identify relationships between analytes and relationships between individual samples. Correlations between analytes signify unique geochemical processes associated with certain natural influences (e.g., geothermal) or anthropogenic activities (e.g., mine-influenced or agricultural). Relationships between samples can be used to determine the spatial extent of anthropogenically-influenced groundwater as well as the extent of naturally-influenced groundwater.

The evaluation of trends in one analyte versus another analyte is a detailed component of MSA. The distributions of pairs of conservative (non-attenuated) ions were used to evaluate the presence of dilution and mixing trends of differing water types in the Shallow Zone. Mixtures of two end members plot as a linear trend between the end members. A primary example is the dilution of MIW with the low ionic strength “background” groundwater end member. If mixing of additional end members with differing chemistries are present, they plot as separate linear trends. The evaluation of trends in one analyte versus another analyte displayed on bivariate plots is a detailed variation of MSA and allows the raw analytical data to be visualized without pre-graphing interpretation of data. Bivariate plots are a standard graphical presentation of chemical data to identify correlations between plotted parameters (Richter and Kreitler, 1993, p. 214).

Application

MSA was applied to the OU-1 groundwater data utilizing methods that eliminated the potential to introduce bias, knowingly or unknowingly, into the evaluation. These MSA methods are “blind” exploratory techniques that require no a-priori assumptions as input to the methods. Two different datasets were evaluated independently using MSA. The first dataset included 171 samples analyzed for 29 chemicals (nearly 5,000 data points) collected from the Shallow Zone.

The second dataset included 399 samples analyzed for 34 chemicals (over 13,000 data points) collected from all alluvial zones and bedrock. These MSA methods are described in greater detail in Appendix N.

The MSA results from both datasets identified correlations among analytes that reflect key influences affecting the groundwater quality. Strong correlations were observed among metals in both datasets that reflect mine-influenced groundwater; among key geothermal indicators such as arsenic and molybdenum; and among agriculturally-significant analytes such as nitrate, total organic carbon, and alkalinity. The MSA results also identified spatial correlations among samples that reflect the spatial extent of key influences on groundwater quality. Many correlations were similar between the two datasets, indicating reproducibility of the observed trends and correlations. These strong correlations suggest that the dataset can be clustered into meaningful groups.

The outcomes from these MSA exploratory evaluations were used to statistically classify by discriminant analysis all samples from one of the two datasets (399 groundwater monitoring samples with 34 analytes per sample) into 18 distinct clusters. Those 18 clusters were then grouped into three geochemically-distinct primary groundwater types based on interpretation:

- Mine-influenced groundwater;
- Geothermal groundwater; and
- Regional (includes agricultural/Walker River) groundwater

The discriminant analysis compares each sample to the 18 clusters within the primary groundwater types, based on all 34 analytes, and determines to which groundwater type the sample belongs. The discriminant analysis also calculates the probability (i.e., uncertainty) of properly classifying each sample into each cluster. Greater than 95% of the dataset (380 of 399 samples) was classified with greater than 90% likelihood of proper classification indicating little ambiguity within the dataset.

The water type assigned to each monitor well sample is listed in Table 5-3. This information was used as an independent line of evidence by mapping the water type for each well sample and then delineating the extent of the water type with a boundary line. Application of this technique for the Shallow Zone is shown in Figure 5-17. The extent of the three water types for all zones are shown in Figures 5-21(a-g).

Table 5-3. Groundwater Samples Classified by Water Type

Shallow Zone						
Mine-Influenced Water Type				**Geothermal Water Type**	**Agriculture/ Walker River Water Type**	
B/W-11S (17)	EP-GC-27 (18)	EP-GW-25 (16)	OU4-UEP-09 (17)	B/W-5RS (14)	B/W-1S (13)	B/W-64S (13)
B/W-29S (16)	EP-GC-29 (16)	EP-GW-26 (17)	OU4-UEP-11 (17)	B/W-8S (11)	B/W-2S (9)	B/W-65S (9)
B/W-36S (10)	EP-GC-30 (10)	EP-GW-27 (16)	PA-MW-2S (12)	B/W-16S (14)	B/W-3S (1)	B/W-66S (9)
B/W-38RS (12)	EP-GW-01 (17)	EP-GW-28 (10)	PA-MW-3S2 (12)	B/W-18S (5)	B/W-4S (13)	B/W-67S (13)
B/W-70S (17)	EP-GW-02 (17)	EP-GW-29 (17)	PA-MW-4S (10)	B/W-19S (11)	B/W-9S (13)	B/W-68S (9)
B/W-71S (18)	EP-GW-03 (17)	EP-GW-33 (16)	PA-MW-5S1 (10)	B/W-22S (11)	B/W-10S (3)	B/W-69S (9)
B/W-74S (8)	EP-GW-04 (18)	EP-GW-34 (17)	PA-MW-5S2 (10)	B/W-32S (5)	B/W-13S (3)	B/W-76S (1)
B/W-75S (12)	EP-GW-05 (17)	EP-GW-37 (18)	PA-MW-7S (12)	B/W-33S (14)	B/W-15S (9)	B/W-78S (1)
D5AC-1S (10)	EP-GW-06 (18)	EP-GW-38 (17)	PLMW-4S (12)	B/W-34S (14)	B/W-20S (9)	B/W-79S (9)
EP-GC-04 (18)	EP-GW-07 (18)	FMS-05S (18)	PW-11S (10)	B/W-37S (5)	B/W-21S (3)	B/W-81S (9)
EP-GC-05 (18)	EP-GW-08 (17)	FMS-06S (18)	PW-1S (17)	B/W-40S (14)	B/W-25S (13)	B/W-82RS (3)
EP-GC-06 (10)	EP-GW-09 (18)	FMS-07S (17)	PW-2S (17)	B/W-41S (11)	B/W-27S (3)	B/W-83S (13)
EP-GC-07 (18)	EP-GW-10 (10)	HLP-03S (12)	PW-3S (17)	B/W-44S (14)	B/W-28S (13)	D4BC-1S (13)
EP-GC-08 (18)	EP-GW-11 (17)	HLP-04S (12)	PW-4S (16)	B/W-51S (11)	B/W-30S (13)	LC-MW-1S (13)
EP-GC-09 (18)	EP-GW-12 (18)	HLP-08S (8)	PW-5S (16)	B/W-53S2 (11)	B/W-31S1 (3)	LC-MW-3S (13)
EP-GC-10 (18)	EP-GW-13 (18)	LEP-MW-3S (15)	PW-7S (16)	B/W-56S (2)	B/W-31S2 (13)	LC-MW-5S (3)
EP-GC-11 (10)	EP-GW-14 (18)	LEP-MW-6S (10)	PW-8S (10)	B/W-6S (14)	B/W-42S (13)	PA-MW-1S (3)
EP-GC-12 (16)	EP-GW-15 (18)	MW-2S (18)	USGS-2BS (17)	B/W-77S (5)	B/W-43S (9)	PA-MW-3S1 (13)
EP-GC-13 (10)	EP-GW-16 (18)	MW-4S (12)	UW-1S (10)	LC-MW-2S (2)	B/W-45S (13)	PLMW-2S (3)
EP-GC-14 (10)	EP-GW-17 (18)	MW-5S (18)	W5AB-2S (10)	LEP-MW-1S (14)	B/W-46S (13)	PW-6S (9)
EP-GC-15 (10)	EP-GW-18 (17)	MW-SXN (12)	W5DB-S (18)	LEP-MW-2S (11)	B/W-50S (13)	YPT-MW-11S (3)
EP-GC-17 (10)	EP-GW-19 (17)	MW-SXS (12)	OU4-LEP-04 (10)	LEP-MW-5S (5)	B/W-52S (3)	
EP-GC-18 (18)	EP-GW-20 (16)	OU4-FEP-13 (18)	OU4-UEP-10 (10)	MW2002-2S (14)	B/W-54S (3)	
EP-GC-19 (18)	EP-GW-21 (8)	OU4-FEP-15 (17)	OU4-FEP-16 (16)	PW-10S (5)	B/W-55S (4)	
EP-GC-21 (10)	EP-GW-22 (17)	OU4-LEP-03 (17)	OU4-LEP-01 (16)	PW-9S (5)	B/W-57S (13)	
EP-GC-23 (18)	EP-GW-41 (14)	OU4-LEP-05 (18)	OU4-LEP-02 (16)	W5BB-S (11)	B/W-59S (9)	
EP-GC-24 (18)	OU4-FEP-14 (16)	OU4-UEP-06 (18)		EP-GW-30 (11)	B/W-60S (13)	
EP-GC-25 (17)	EP-GW-23 (17)	OU4-UEP-07 (18)		EP-GW-31 (11)	B/W-61S (13)	
EP-GC-26 (16)	EP-GW-24 (10)	OU4-UEP-08 (18)		EP-GW-39 (11)	B/W-62S (13)	
				EP-GW-40 (11)	B/W-63S (3)	

Table 5-3. Groundwater Samples Classified by Water Type (cont.)

Intermediate Zone			Deep 1 Zone		
Mine-Influenced Water Type	**Geothermal Water Type**	**Agriculture/ Walker River Water Type**	**Mine-Influenced Water Type**	**Geothermal Water Type**	**Agriculture/ Walker River Water Type**
B/W-29I (16)	B/W-2I (6)	B/W-4I (4)	B/W-1D1 (8)	B/W-2D1 (6)	B/W-4D1 (4)
B/W-37I (15)	B/W-3I (7)	B/W-9I (4)	B/W-29D1 (8)	B/W-3D1 (6)	B/W-10D1 (4)
B/W-74I1 (8)	B/W-5RI (11)	B/W-27I (3)	B/W-71D1 (17)	B/W-5RD1 (11)	B/W-25D1 (13)
B/W-70I (15)	B/W-6I (14)	B/W-28I (13)	B/W-74D1 (8)	B/W-18D1 (6)	B/W-30D1 (4)
B/W-71I (17)	B/W-7I (11)	B/W-30I (4)	HLP-08D1 (15)	B/W-19D1 (11)	B/W-37D1 (1)
HLP-03I (12)	B/W-8I (11)	B/W-31I (4)		B/W-28D1 (6)	B/W-42D1 (3)
HLP-08I (15)	B/W-19I (11)	B/W-38RI (1)		B/W-33D1 (14)	B/W-46D1 (9)
MW-5I (10)	B/W-32I (7)	B/W-42I (4)		B/W-34D1 (7)	B/W-50D1 (4)
PA-MW-2I (12)	B/W-33I (14)	B/W-46I (4)		B/W-38RD1 (7)	B/W-55D1 (3)
PA-MW-3I (12)	B/W-34I (14)	B/W-51I (1)		B/W-40D1 (7)	B/W-57D1 (4)
PA-MW-4I (10)	B/W-41I (11)	B/W-52I (4)		B/W-44D1 (11)	B/W-58D1 (3)
W4CB-1I (15)	LEP-MW-4I (6)	B/W-54I (3)		B/W-45D1 (7)	B/W-60D1 (13)
W4CB-2I (16)	MW-4I (11)	B/W-57I (9)		B/W-66D1 (6)	B/W-61D1 (13)
W5AB-3I (15)		B/W-63I (4)		LEP-MW-2D1 (14)	B/W-62D1 (13)
W5DB-I (15)		B/W-65I (13)		MW2002-2D1 (14)	B/W-63D1 (4)
		B/W-66I (3)		W32DC-D1 (7)	B/W-64D1 (13)
		B/W-67I (4)			B/W-65D1 (4)
		B/W-74I2 (1)			B/W-67D1 (4)
		B/W-76I (1)			B/W-68D1 (4)
		B/W-82RI (3)			B/W-69D1 (4)
		LEP-MW-8I (3)			B/W-76D1 (1)
		LEP-MW-9I (4)			B/W-81D1 (4)
		W5AA-1I (4)			B/W-83D1 (13)
		YPT-MW-12I (9)			W4CB-2D1 (4)
		YPT-MW-13I (3)			W5DB-D1 (3)
		YPT-MW-15I (4)			YPT-MW-14D1 (3)
		YPT-MW-9I (3)			

Table 5-3. Groundwater Samples Classified by Water Type (cont.)

Deep 2 Zone			Deep 3 Zone		
Mine-Influenced Water Type	Geothermal Water Type	Agriculture/ Walker River Water Type	Mine-Influenced Water Type	Geothermal Water Type	Agriculture/ Walker River Water Type
B/W-11D2 (10)	B/W-18D2 (7)	B/W-1D2 (13)	B/W-1D3 (12)	B/W-2D3 (7)	B/W-29D3 (1)
B/W-44D2 (8)	B/W-25D2 (7)	B/W-9D2 (3)	B/W-60D3 (8)	B/W-27D3 (6)	
B/W-70D2 (15)	B/W-32D2 (7)	B/W-27D2 (3)	B/W-61D3 (8)	B/W-40D3 (7)	
MW-5D2 (15)	B/W-45D2 (7)	B/W-31D2 (3)	B/W-71D3 (15)	B/W-50D3 (6)	
PA-MW-4D2 (8)	B/W-41D2 (14)	B/W-50D2 (4)	MW-5D3 (15)	B/W-58D3 (2)	
		B/W-52D2 (3)	W4CB-2D3 (15)	B/W-59D3 (2)	
		B/W-55D2 (3)	W5DB-D3 (15)	B/W-63D3 (6)	
		B/W-62D2 (13)		B/W-67D3 (6)	
		B/W-63D2 (4)		B/W-83D3 (6)	
		B/W-64D2 (13)		LEP-MW-2D3 (14)	
		B/W-81D2 (9)			
		HLP-08D2 (9)			

Deep 4 Zone			Deep 5 Zone		
Mine-Influenced Water Type	Geothermal Water Type	Agriculture/ Walker River Water Type	Mine-Influenced Water Type	Geothermal Water Type	Agriculture/ Walker River Water Type
B/W-62D4 (12)	B/W-2D4 (14)		B/W-1D5 (12)	B/W-27D5 (6)	
W4CB-2D4 (8)	B/W-27D4 (6)		B/W-62D5 (8)	B/W-32D5 (14)	
W5DB-D4 (15)	B/W-41D4 (11)			B/W-60D5 (2)	
	B/W-57D4 (2)			B/W-63D5 (7)	
	B/W-68D4 (2)			B/W-65D5 (7)	
				B/W-66D5 (14)	
				B/W-69D5 (2)	

Notes:
Water types are determined in Appendix N.
Cluster ID shown in parenthesis.
Clusters were assigned to water types based on professional judgment.

Chloride is used as the abscissa on the ionic cross plots due to the conservative nature of the ion (not sensitive to pH or redox reactions or mineral precipitation in the Study Area groundwater system). The ordinate ions are the dominant mine COIs (sulfate and uranium) as well as a group of COIs with bimodal distributions similar to those of sulfate and uranium (boron, fluoride, lithium, and vanadium). The graphs are shown on Figure 5-26a.

The data points are coded by the three water types discussed above. These graphs were prepared to understand dilution trends of the MIW and geothermal ("GEO") water types and evidence of mixing between these two end members. Analyses of a water sample collected from the LEP in 1976 and reported in Seitz et al. (1982) are plotted with the MIW samples if the specific analyte was measured. End members of the GEO water type in the Shallow Zone are shown as samples from wells B/W-32S and B/W-77S. Dilution trends of the two end members with low ionic strength "background" end members are displayed on each graph.

The MIW and GEO end members plot in different areas of each graph and have different dilution mixing trends. This array of data is caused by the higher chloride content of the GEO end members relative to the MIW dilution trends and end member. No data are observed in a linear trend directly between the MIW and GEO end members.

Each graph also identifies (with a dotted circle) samples that plot along a dilution trend that is not consistent with the water type identified for the indicated sample. For example, there are blue MIW data points that plot along the low end of the GEO dilution trend on the chloride/fluoride and chloride/vanadium cross plots. There are red GEO data points that plot along the upper end of the MIW dilution trend on the chloride/boron cross plot. Additionally, there are MIW and GEO data points that plot between the low ends of the MIW and GEO dilution trends on the chloride/sulfate, chloride/uranium, and chloride/lithium cross plots. These samples are identified in Table 5-4.

Table 5-4. Mixtures of MIW and GEO Water Types Identified on Ionic Cross Plots

Well	MSA Water Type	Well sample identified as a mixture of MIW and GEO water types or plotting along a dilution trend on Figure 5-26a that is different from the MSA water type.				
		Cl v SO4	Cl v U	Cl v F	Cl v Li	Cl v
D5AC-1S	MIW	Mixture	GEO	GEO	Mixture	Mixture
LEP-MW-3S	MIW	Mixture	Mixture	GEO	Mixture	Mixture
LEP-MW-5S	GEO	Mixture	Mixture		Mixture	
PW-10S	GEO	Mixture	Mixture		Mixture	
PW-7S	MIW	Mixture	GEO	GEO	Mixture	Mixture
PW-8S	MIW	Mixture	GEO	GEO	Mixture	Mixture
PW-9S	GEO	Mixture	Mixture		Mixture	
W5AB-2S	MIW	Mixture	GEO	GEO	Mixture	Mixture

Notes:
1. Multivariate Statistical Analysis (MSA) Water Types shown in Table 5-3
2. Wells are plotted in Figure 5-26b.
3. MIW = mine-influenced water type
4. GEO = geothermal water type

The Shallow Zone wells listed in Table 5-4 are plotted in Figure 5-26b. All wells are located in the LEP and most are in the vicinity of the former PWS. The hydraulic influences of the PWS when operational may have induced flow of both MIW from the south and the GEO water type into the PWS wells. Alternatively, these areas may represent an interfingered fringe area of the two water types. In either case, diluted samples of both MIW and GEO water are found in this area along the former PWS alignment.

In summary, the ionic cross plot line of evidence indicates that wells B/W-32S and B/W-77S do not contain discernable amounts of MIW and can be used as a LOE to exclude the presence of MIW in the vicinity of those wells. Other GEO water type wells in the vicinity of these wells (i.e., B/W-18S) do not contain chloride at concentrations that allow the reliable identification of non-MIW water characteristics. The ionic cross plots indicate that some well samples previously assigned as MIW or GEO water types may be mixtures of the two water types.

Molybdenum

Review of molybdenum versus pH data as part of the ionic cross plot evaluation process provided additional insight to the evaluation of MIW and non-MIW constituent migration. The distribution of molybdenum in Shallow Zone groundwater monitor wells is shown in Figure 5-24e and in the DPT samples in Appendix B, Figure 4-25. There is an area of elevated molybdenum concentrations in the Shallow Zone that begins along the northern boundary of the LEP and extends north/northwestward in the area of the alternate model of COI migration. The following discussion evaluates whether the molybdenum originated at the ACMS.

Figure 5-27 illustrates that molybdenum was not reported in DPT samples at concentrations greater than 50 µg/L where pH was less than 6 (i.e., the orange and blue areas are not congruent). Molybdenum is an oxyanion that is mobile under alkaline conditions. A standard treatment process for removal of molybdenum from mining waste streams is acidification, contact with iron, and precipitation. The following excerpts are from US Patent 4,219,416 (Ramirez and Ramadorai, 1980), process for recovering molybdenum and tungsten from mining wastewater:

> *"Adjust pH to a value of between 2.0 to 6.0, and preferably between 3.0 and 6.0. This is accomplished by the addition of mineral acid to the wastewater… it is preferred to use sulfuric acid in view of its availability and low cost."*
>
> *"Also, a trivalent metal cation is added to the wastewater…Preferably, however, ferric sulfate is used to provide trivalent ions in view of its availability, low cost and ease of use."*
>
> *"The treated wastewater is then retained for at least 3 minutes … so as to enable the formation of insoluble ferric molybdate…"*

The acid copper recovery process used by Anaconda unintentionally replicated these key elements of molybdenum treatment (and existing information indicates the process stream was not enriched in molybdenum.· The pregnant solution was then sent to the iron launders to recover the copper. The pH and iron content in the process sequence (e.g., input and discharge for the Cementation Circuit [“Launders”]) was in the range needed for treatment of molybdenum (Table 2-1). The process effluent was discharged to the Evaporation Ponds where evaporation and infiltration occurred. Removal of dissolved molybdenum by precipitates of ferric iron also has been documented in natural systems (LeGendre and Runnells, 1975). The occurrence of ferric oxyhydroxides in the vadose zone beneath the Evaporation Ponds is discussed in Section 6.2.1. Groundwater samples with strong mining impact (pH values below 5.5) plot along the K-jarosite and schwertmannite phase boundaries and triple points in Figure 6-2.

The low to non-detect molybdenum concentrations in the Shallow Zone beneath the Evaporation Ponds are consistent with the process chemistry and the vadose zone geochemical processes described above.

The evaluation of molybdenum in the Shallow Zone beneath and north of the ACMS can be summarized as follows. MIW that historically infiltrated in the Evaporation Ponds had passed through the acid leaching copper recovery process. This process and geochemical conditions in groundwater immediately beneath the Evaporation Ponds preclude the presence of elevated molybdenum in MIW migrating in the Shallow Zone. Therefore, it is not possible for the elevated molybdenum found in the Shallow Zone north of the ACMS (Figures 5-27 and 5-25e) to have originated from ACMS mining processes. Non-acidic mixtures of MIW and non-MIW water types could contain molybdenum, however. Thus, the presence of molybdenum is not used to exclude the possibility of a water sample containing a mixture of MIW.

Uncertainties and Limitations

Please refer to Appendix N for more details regarding the datasets, statistical methods, and uncertainty of this line of evidence. Uncertainty is also discussed in Appendix O (Response to Comments, Specific Comment 34). A limitation of the MSA is the inherent subjectivity in assigning the 18 clusters to three groundwater types, including commingled groundwater, since not all of the individual constituents are unique to any one groundwater type.

Accuracy and precision of analytical measurements introduce uncertainty to the display of data on bivariate plots. Linear regression lines represent the best fit of a linear trend in the bivariate data. The uncertainties and limitations of water types discussed above applies to the data labels displayed on the bivariate plots.

5.4.4 Sulfur Isotopic Information

Method

Sulfur isotopes in groundwater were used in the BGQA to identify the leading edge of MIW in the Study Area. The delineation in the BGQA of the leading edge of MIW has been demonstrated to be incorrect because the threshold value for MIW was not appropriate given the presence of multiple water types and complexities of the groundwater system. This section describes the process used to revise the threshold value that represents early releases from the Site and then describes the areas where sulfur isotope values meet this threshold in the alluvial aquifer zones.

Application

Statistical plots of the sulfur isotopic data generated in ProUCL (EPA, 2015c) were used to identify the isotopic range of the early mine release (Figure 5-18). Sulfur isotopic values (‰) are plotted on the y-axis. The x-axis plots theoretical quantiles for the calculated normal distribution that best describe the sulfur isotopic dataset used for the Q-Q plot. The straight blue line on the plot represents the sulfuric isotope distribution that best fit the theoretical quantiles on the x-axis. The closer the datapoints plot along the best-fit line, the closer the dataset is to a normal distribution. Large jumps and breaks from the best-fit line represent potential observations coming from multiple populations.

A quantile-quantile (Q-Q) plot of the entire sulfur isotopic dataset is shown in the upper left panel of Figure 5-18. A Q-Q plot does not in itself quantitatively determine goodness of fit to a normal distribution, but comparison of the data to a best-fit normal distribution (straight line) provides a qualitative estimate. Qualitative results of Shapiro-Wilk tests for normality at the 95% confidence coefficient are additionally provided next to the Q-Q plots (Figure 5-18). The data for all subsets do not plot along a straight line or meet the requirements of the Shapiro-Wilk tests, and there are multiple inflection points indicating the presence of multiple populations within the dataset. This is not unexpected, given the multiple water types identified in the Site-wide dataset. Identification of the early mine release samples is based on defining the sulfur isotope population that represents the mixture of early mine water with the Leviathan Mine ore signature (-17.6‰) with local pre-mining groundwater. This mixture is represented by the normally distributed sample population that includes the array of strongly negative isotopic values at the lower end and the inflection point with another population at the upper end. This mixture range was identified by an iterative process of selecting increasingly narrow isotopic ranges, generating Q-Q plots, and assessing whether the selected range met the requirement of a normally distributed population of mixture between the groundwater signature of Leviathan Mine and local groundwater. Q-Q plots were generated for data < 0‰, <-2.5‰, and <-5‰ in the other panels of Figure 5-18. The <-5‰ Q-Q plot qualitatively appears closest to a normally distributed population of mixing between the groundwater signature of Leviathan Mine and local groundwater. This is confirmed by the Shapiro-Wilk test results. Therefore, evaluation of the Q-Q plots supports the selection of the < -5‰ range to identify the early release from the mine.

The rationale to select the < -5‰ $\delta^{34}S_{SO4}$ range to uniquely identify early mine releases is also supported by the differences in the sulfur isotopic signatures of ACMS and Leviathan Mine ores. The ACMS sulfide ore has a mean $\delta^{34}S_{SO4}$ of -5.45‰, based on the average of five samples reported in the literature (Field, 1966) and ranged from -4.9 to -6.3‰. Lipske (2003) reported a $\delta^{34}S_{SO4}$ value of -4.7‰ for ACMS sulfide ore. The isotopic composition of Leviathan Mine ore is more strongly negative than the ACMS ore samples; median value of -17.6‰ (Table 5-5 in Appendix J-7). Thus the < -5‰ $\delta^{34}S_{SO4}$ range is expected to represent a single population of early ACMS sulfuric acid releases and pre-existing groundwater with the ACMS ore isotopic signature, and therefore plot along a linear trend on the < -5‰ $\delta^{34}S_{SO4}$ Q-Q plot in Figure 5-18. The sulfur isotopic line of evidence would not be a useful discriminator if the sulfur isotopic ratios of sulfur from the two mines were similar.

The lateral distributions of $\delta^{34}S_{SO4}$ values < -5‰ in each groundwater zone are illustrated on Figures 5-19(a-g). The distinctly negative $\delta^{34}S_{SO4}$ signature associated with Leviathan Mine is evident in the Shallow, Intermediate, and Deep Zones beneath the Evaporation Ponds. In the Deep 3 Zone, well W5DB-D3 has a strongly negative $\delta^{34}S_{SO4}$ signature of -17.12‰ (Figure 5-19e).

The -5‰ $\delta^{34}S_{SO4}$ contour line is used as the sulfur isotope line of evidence to delineate the extent of early releases of MIW in Figures 5-19(a-g). As discussed above, later sources of sulfur used at the ACMS were less negative than -5‰ $\delta^{34}S_{SO4}$. Therefore, MIW from later releases could occur in areas beyond the areas shown these figures.

Uncertainties and Limitations

The analytical precision of $\delta^{34}S_{SO4}$ measurements is ± 0.5‰ (Table 3-8). Ball and Nordstrom (1985) published $\delta^{34}S$ measurements of elemental sulfur, “whole-rock” sulfur, and sulfate in aqueous samples, all from the Leviathan Mine. The values of individual samples range from -17.6 ‰ (aqueous sample) to -11.8‰ (elemental sulfur sample). The range of these samples are more negative than measurements of the Yerington ore discussed above (-4.9‰ to -6.3‰). As all of the Ball and Nordstrom (1985) Leviathan sulfur isotopic data are more negative than the range of Yerington ore samples, this additional information does not introduce additional uncertainty in the sulfur isotope LOE. Based on the distinctly negative $\delta^{34}S_{SO4}$ reported signatures, samples from the Leviathan Mine are the only known source of sulfur more negative than the Yerington ore samples. The additional Ball and Nordstrom data also do not introduce additional uncertainty in the statements that later sources of sulfur used at the ACMS were less negative than -5‰ $\delta^{34}S_{SO4}$ and therefore MIW from later releases could occur in areas beyond the areas shown Figures 5-19(a-g).

The Raab and Spiro (1991) research indicates that the sulfur isotopic ratio of evaporating sea water decreases from +20‰ to +18.4‰ during the halite precipitation cycle and then increases up to +20.94‰ at the end of the K-Mg-Sulfate precipitation cycle. A net increase of 0.94‰ to process water containing Leviathan sulfur during evaporation will not introduce additional uncertainty to the sulfur LOE as the sulfur isotopic ratio of the residual liquid would still be more negative than the Yerington ore samples.

Measurements of sulfur isotopic composition of water that infiltrated to groundwater from the Evaporation Ponds through time would have reduced the uncertainty in this LOE, but such data are not available.

5.4.5 Groundwater Age Estimates

Method

The age of groundwater is the time since the water was recharged to the saturated zone. There are techniques to determine whether groundwater was recharged before Anaconda operations, and hence could not contain MIW. A technique based on a combination of tritium and helium was used at the Site to assess the age of groundwater in the Study Area, and the tritium method was used to specifically assess where recharge occurred prior to Anaconda operations.

Briefly, the tritium/helium groundwater age dating method involves quantitating the ^{3}He in the water sample and accounting for the tritium-derived ^{3}He. Tritium/helium age dating is not affected by mixing between old (or sub-modern) groundwater and young tritiated water (Cook and Herczeg, 2000) because sub-modern groundwater does not contain ^{3}H or tritium-derived ^{3}He. Thus, no mixing model assumptions are required to estimate a groundwater age, unlike groundwater ages estimated from tritium data alone. The ^{3}He "clock" is reset upon exposure to atmospheric gas and "the $^{3}H/^{3}He$ clock does not start until water is below the water table and completely isolated from a gas phase" (Solomon and Cook, 2000, p. 414). A detailed discussion of the principles of groundwater age estimation using $^{3}H/^{3}He$ data is provided in Appendix H-1, Section 5.2.2.

Measurements of the concentrations of ^{4}He and neon (Ne), and the relative abundances of helium isotopes ($^{3}He/^{4}He$) and noble gases (He/Ne) in the sample, are used to estimate these amounts and to adjust the apparent ages, if needed. Two approaches are used to adjust the apparent ages: the "Ne model" relies only on neon and the "closed-system equilibration (CE) model" uses multiple noble gas concentrations. Both estimation models provide comparable groundwater ages as shown on the correlation plot in Figure H-1-10 in Appendix H.

Application

The $^{3}H/^{3}He$ data reported in Appendix H estimate groundwater ages of up to ≥ 60 years. The $^{3}H/^{3}He$ data used to map the extent of MIW were collected in 2012 and 2014. A groundwater age estimate of ≥ 60 years indicates that the water was recharged prior to 1954 (conservatively assuming all data were collected in 2014), which effectively precedes Site operations (the vat leach plant was commissioned in Nov 1953). The groundwater recharged prior to 1954 is primarily on the western side of the valley northwest of the Site (Figure 5-20).

The area of old groundwater identified by the $^{3}H/^{3}He$ technique is similar to the area of old groundwater identified by the uranium disequilibrium (Ue) method. As discussed in Section 4.9.4, high Ue values are associated with "long" periods of contact between groundwater and aquifer solids (i.e., "old" water), whereas low Ue values are associated with "short" periods of contact between groundwater and aquifer solids (i.e., "young" water). The highest Ue values commonly occur on the west side of the Study Area adjacent to the Singatse Range. The application of the $^{3}H/^{3}He$ age dating line of evidence is shown in Figure 5-20, where the area containing groundwater that was recharged to the Shallow Zone prior to 1954 is shown. This area is mapped by a line bounding the individual well samples with ages ≥ 60 years. This approach is used to delineate the extent of the $^3H/^3He$ age dating line of evidence on Figures 5-21(a-g).

Limitations and Uncertainties

The tritium age dating methodology provides quantitative estimates of groundwater age whereas the uranium isotope disequilibrium method provides non-quantitative information about groundwater age based on the activity ratio of ^{234}U and ^{238}U and excess daughter ^{234}U (referred to as uranium excess, Ue) which represents the deviation from secular equilibrium of the $^{234}U/^{238}U$ activity ratio, as described above. Ue values do not quantify the groundwater age because there is no absolute scale for relating Ue values to groundwater ages, and because changes in the nature of the uranium host phases, water rock interactions, and groundwater redox conditions alter the Ue value (Porcelli and Swarzenki, 2003) without affecting the groundwater age. Consequently, no direct correlation between tritium groundwater ages and Ue values is possible. Nonetheless, Ue values are useful for providing constraints on groundwater conditions (Cook and Herczeg, 2000; Kendall and McDonnell, 1998). Therefore, it is not inconsistent for a water sample to be older than 60 years as dated with tritium and to be classified as “young” by the uranium disequilibrium method.

Solomon et al. (1993) reports that analytical uncertainties usually result in errors in age estimates of less than 10%. Sensitivity of the $^3H/^3He$ age estimate to recharge temperature is less than 0.5 years per ^oC. Sensitivity to excess air is approximately -5.0 years per cubic centimeters per kilogram (cm^3/kg) of excess air for very young water, decreasing to -0.25 years per cm^3/kg for water approximately 25 years old. The error bars surrounding the Site-specific $^3H/^4He$ groundwater ages in Figure H-1-10 in Appendix H represent uncertainty associated with laboratory measurements and calculations. The average calculated error estimate for data less than 60 years old is 5.6 years for the "Ne Model" results and 4.4 years for the "CE Model" results (Appendix H).

However, the tritium and $^3H/^3He$ age dating methodologies are not equivalent. The measurement of helium (a decay product of tritium) is useful for estimating the high tritium concentrations that occurred in water after 1952 but is not useful for better determining low concentrations in water recharged prior to 1952. The errors for the $^3H/^3He$ age estimates in Appendix H are not representative of analytical uncertainty in the tritium only methodology. Calculation of groundwater ages greater than 60 years is beyond the dating range of the tritium only method. The average of the estimated error in the tritium measurements for samples with reported ages of > 60 years is approximately 0.05 TU (Appendix H). Increasing tritium concentrations by 0.05 TU would not shift the tritium concentrations of > 60-year-old samples into the dating range of the tritium method. The analytical error of these tritium measurements is unlikely to change the classification from > 60 years to < 60 years.

Regarding reproducibility of the tritium only results, well B/W-32S was sampled for age dating in 2011 and 2012. The measured tritium in these sampling events were 0.11±0.06 TU and 0.16±0.10 TU, respectively, resulting in groundwater age estimates of > 60 years for both measurements.

Wells yielding tritium ages of > 60 years are in a contiguous area on the western side of the valley where groundwater flow rates and opportunities for recharge are low, relative to the eastern side of the valley. The uncertainty in the delineation of the specific boundary of > 60 year-old groundwater is primarily limited by well spacing. It is possible that the actual boundary could be slightly eastward or westward of the mapped boundary and does not intersect parts of the low confidence MIW area discussed in Section 5.4.10.

5.4.6 Collocation of MIW Indicators

Method

The use of the distribution of collocated MIW COIs in the Shallow Zone as a line of evidence to estimate the extent of MIW is presented in this subsection. As discussed in Section 5.2.1, sulfate and uranium are key indicator parameters for MIW. Sulfate and uranium are more mobile in Site groundwater than other indicators of MIW that can be less mobile due to changes in pH or rock-water interaction. Although both sulfate and uranium are present due to natural and anthropogenic processes in the Mason Valley, elevated concentrations are not always collocated. Elevated concentrations of sulfate and uranium are collocated in some areas of the ACMS.

Application

The empirical selection of collocated areas where sulfate is greater than 1,000 mg/L and uranium is greater than 100 µg/L is used as a line of evidence to estimate the extent of MIW in the Shallow Zone. The distributions of sulfate and uranium in the Shallow Zone are shown in Figures 5-2a and 5-3a, respectively. Both contour maps indicate an area of elevated concentrations in the vicinity of the Evaporation Ponds that extends off-Site to the north/northwest in the vicinity of wells B/W-32S and B/W-77S.

The distribution of COIs, notably sulfate and uranium, north and northwest of the ACMS, follow the natural direction of groundwater flow during the period of mine operations. Lower concentrations of sulfate and uranium in the area north of the PWS could be the result of gradient reversal during the operation of the PWS and/or commingling of COIs from different sources. Elevated concentrations of sulfate and uranium farther north, near wells B/W-32S and B/W-77S, may be related to MIW that migrated prior to the operation of the PWS. There is a large area of elevated concentrations beneath the Evaporation Ponds and a smaller area of elevated concentrations in the vicinity of wells B/W-32S and B/W-77S. The two areas of elevated concentrations are separated by an area of lower concentrations. This bimodal pattern is also observed in the Shallow Zone DPT data as discussed subsequently.

A description and evaluation of the DPT program, including contour maps of the DPT data, are provided in Appendix B. Contour maps are included in Appendix B Figures 4-12 through 4-35. The distribution of sulfate, uranium, and pH contours in the DPT data (Appendix B; Figures 4-14, 4-32, and 4-12, respectively) are similar to those shown for Shallow Zone groundwater monitoring data in Figures 5-2a, 5-3a, and 5-1a, respectively. Evaluation of the DPT contour maps for other COIs indicates there are three types of patterns as described below (excluding calcium, the distribution of which does not fit neatly into one of the three patterns):

1. COIs with bimodal distributions similar to those of sulfate and uranium (i.e., boron, lithium, magnesium, potassium, selenium, sodium, strontium, and vanadium, which are COIs that typically are less sensitive to the range of pH found at the Site than the metals and metalloids listed in the other two groups).
2. COIs with distributions coincident with areas of pH < 6 (i.e., cobalt, copper, iron, manganese, nickel, and zinc, which are metals that are mobile in acid conditions).
3. COIs with distributions predominantly located in some areas of pH > 6 (alkalinity, arsenic, and molybdenum). Arsenic and molybdenum occur as oxyanions that can be more mobile in alkaline conditions. Alkalinity can be produced by the acid dissolution of carbonates.

The spatial patterns of COIs in the second and third groups are consistent with the general effects of pH on COI mobility summarized above. The similarity of spatial patterns of COIs in the first group to the distributions of sulfate and uranium provide support to the collocated sulfate/uranium line of evidence to map the extent of MIW.

Sulfate and uranium concentrations measured in both the DPT program (Appendix B) and in groundwater monitor wells (3Q 2014) are illustrated on the NW/SE cross section that tracks along the axis of the collocated sulfate/uranium model of MIW migration (Figures 5-25a and 5-25b, respectively). Concentrations are elevated on the southeast end of the cross section beneath

ACMS and are lowest on the northwest end of the cross section. As shown on the contour maps of monitor well data and DPT data, a zone with concentrations lower than those observed at either the ACMS or wells B/W-32S and B/W-77S is observed between the two areas. Similar patterns are observed in the cross sections illustrating the distributions of lithium and vanadium (Figures 5-25f and 5-25g, respectively).

The distributions of COIs can be interpreted to support a model of MIW migration, and this line of evidence is summarized in Figure 5-21h as the area in the Shallow Zone where 3Q 2014 groundwater monitoring data indicate sulfate concentrations are greater than 1,000 mg/L and uranium concentrations are greater than 100 µg/L.

Uncertainties and Limitations

Uncertainties and limitations in this line of evidence include the accuracy and precision of sulfate and uranium measurements. For sulfate and uranium, respectively, the QAPP specifies the accuracy of matrix spike duplicates as 80% to 120% and 75% to 125%, and the precision limit for both analytes is 20% (ARC, 2018).

5.4.7 Use of LOEs to Develop Hypotheses of MIW Extent

Hypotheses of the extent of MIW in the Shallow Zone are assembled using mutually supporting LOEs. Review of the LOE information indicates that there are two hypotheses regarding the extent of MIW in the Shallow Zone that are supported by mutually consistent evidence. One hypothesis integrates five LOEs (MSA including ionic cross plots, sulfur isotopes, groundwater travel distance, tritium age dating and nitrate) into a composite evaluation of MIW. This hypothesis is termed the High Confidence MIW Area for the Shallow Zone. There is a high probability it contains MIW from the ACMS. The second hypothesis uses two mutually supportive LOEs (conservative groundwater travel distance and collocation of MIW indicators) to assemble a composite evaluation of MIW. This hypothesis is termed the Low Confidence MIW Area. There is a lower probability that it contains MIW, and it is more likely influenced by other sources. The composite weight of evidence for the High and Low Confidence MIW Areas is further described in the two following sections.

There is one hypothesis for the Intermediate through Deep 5 Zones based on five lines of evidence (MSA, sulfur isotopes, groundwater travel distance, age dating, and nitrate). This hypothesis refers to the MIW Area and is described in Section 5.4.11.

5.4.8 High Confidence MIW Area Shallow Zone

This section presents a composite weight of evidence evaluation of the High Confidence MIW Area in the Shallow Zone. The six independent LOEs supporting this area in the Shallow Zone are mapped on Figure 5-21a. The discussion of the LOEs are grouped into the evidence that indicates the presence of MIW and evidence that excludes the presence of MIW.

Lines of Evidence that Indicate the Presence of MIW

LOEs that indicate the presence of MIW are water types developed using the MSA process, well samples with $\delta^{34}S_{SO4}$ values less than – 5‰, and 50% probability groundwater travel distance estimate. The MSA is based on the multivariate evaluation of 34 analytes in 399 groundwater samples, which brings a quantitative approach to the interpretation and collective understanding of the comprehensive water chemistry data base. The sulfur isotope LOE delineates the earliest releases from the Site. The groundwater travel distance estimates are based on a Monte Carlo simulation that characterizes travel distances associated with various probability criteria. An additional level of conservatism is added to the evaluation by identifying the limit of MIW by selecting the evidence line that extends furthest from the mine sources.

Lines of Evidence that Exclude the Presence of MIW in the Shallow Zone

LOEs that exclude the presence of MIW are agricultural nitrate and groundwater age dating information and are used to corroborate the LOEs that indicate the presence of MIW interpretations. The ionic cross plots, a subset of the MSA LOE, are also used to exclude the presence of MIW in the vicinity of wells B/W-32S and B/W-77S in the Shallow Zone.

The composite weight of evidence boundary of the High Confidence MIW Area in the Shallow Zone is shown on Figure 5-21a. Each line of evidence is mapped as a blue layer due to the density of individual data points. The more the layers overlap, the darker blue the color.

MIW is mapped beneath the UEP and LEP in the Shallow Zone. The location of the MIW beneath the UEP and LEP is consistent with the contour maps indicating highest concentrations of sulfate, uranium and other COIs beneath the UEP and LEP. On-Site MIW is also mapped in the Shallow Zone to the east and west of the Evaporation Ponds. On-Site COI concentrations in MIW decrease with distance from the UEP and LEP.

Off-Site areas containing the High Confidence MIW occur in the Shallow Zone in the following two areas contiguous to the ACMS:

1. A triangular area in the Shallow Zone on the western border of ACMS; and
2. An area in the Shallow Zone north of the LEP.

The triangular area in the Shallow Zone on the western border of ACMS is consistent with westward groundwater flow from the ACMS. The area north of the LEP may reflect northward MIW migration in the Shallow Zone before the current westerly flow direction was established by irrigation practices on agricultural fields to the east and north of the Site.

The extent of High and Low Confidence MIW Areas developed by the multiple LOEs are plotted on sulfate and uranium contour maps to compare the results of the technically rigorous multiple LOEs process to simple contour maps that are commonly used to identify the extent of contamination. The contour maps used in this comparison were developed using the "Natural

Neighbor" interpolation in ArcGIS®.[9] Contour lines were not modified, with the exception to smooth lines at the interpolation periphery, as the intent was to objectively compare the composite LOE MIW boundary to an impartial representation of the extent of the sulfate and uranium COIs. The extent of MIW is superimposed on sulfate and uranium contours for the Shallow and Intermediate Zones in Figures 5-22(a-b) and 5-23(a-b), respectively.

The High Confidence MIW Area in the Shallow Zone generally mirrors the elevated sulfate and uranium contours at the ACMS and does not extend to the elevated sulfate and uranium contours in the B/W-32S and B/W-77S area. Figures 5-22(a-b) graphically illustrate how the multiple LOEs approach can be used to delineate MIW from other sources, even when there are common COIs. The Low Confidence MIW Area in the Shallow Zone extends northward to the vicinity of wells B/W-32S and B/W-77S.

In the Intermediate Zone (Figures 5-23[a-b]), the MIW area also generally mirrors the elevated sulfate and uranium contours at the ACMS and serves as a logical indicator of the presence of MIW. The sulfate and uranium concentrations are not as elevated in the area of B/W-32I, and therefore the ability of the multiple LOE approach to distinguish various sources is not as pronounced in the Intermediate Zone as in the Shallow Zone.

In summary, the extent of the High Confidence MIW Area in the Shallow Zone shown on Figure 5-21a is based on the composite evaluation of multiple independent lines of physical, chemical, and isotopic evidence to produce a reliable estimate of the extent of MIW in each zone.

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[9] ArcGIS Desktop Version 10.5.1.7333 [Computer Software] (2019). Redlands, CA: Environmental Systems Research Institute (Esri).

5.4.9 Low Confidence MIW Area Shallow Zone

This section presents a composite weight of evidence evaluation of the Low Confidence MIW Area Shallow Zone hypothesis. The two independent LOEs that indicate the presence of MIW are the 5% probability of groundwater travel distance LOE and the collocation of MIW indicators LOE.

Lines of Evidence that Indicate the Presence of MIW

The 5% probability of groundwater travel distance LOE represents a conservative estimate of groundwater travel distances. This estimate is not representative of the central tendency of the range of travel distance estimates but is near the upper end of the probability range calculated using Monte Carlo simulations. Similar to an Upper Tolerance Limit, this value provides a reasonable upper limit of the possible range of groundwater travel distances. The extent of the 5% probability of groundwater travel distance (taken from Figure 5-15d) is displayed in Figure 5-21h.

As discussed in Section 5.4.6, elevated concentrations of sulfate and uranium are collocated in some areas of the ACMS. The empirical selection of collocated areas where sulfate is greater than 1,000 mg/L and uranium is greater than 100 µg/L is used as a LOE to estimate the extent of MIW in the Shallow Zone. The similarity of spatial patterns of other COIs to the distributions of sulfate and uranium provide support to the collocated sulfate/uranium LOE to map the extent of MIW.

Lines of Evidence that Exclude the Presence of MIW in the Shallow Zone

The Low Confidence MIW Area Shallow Zone hypothesis does not utilize exclusionary LOE, although it is not in conflict with the sulfur isotope LOE. The Low Confidence MIW Area differs from the High Confidence MIW Area shown in the Shallow Zone in Figure 5-21a only in the area immediately to the north and northwest ACMS and includes wells B/W-32S and B/W-77S.

Detailed evaluations of the COI trends indicate low certainty in the Low Confidence MIW Area, however. Ionic cross plots of various COIs indicate that the MIW and GEO water end members plot in different areas of each graph and have different dilution mixing trends. No data are observed in a linear trend directly between the MIW and GEO end members. These graphs are not consistent with MIW as the source of COIs throughout the Low Confidence MIW Area in the Shallow Zone.

5.4.10 Extent of MIW - Intermediate through Deep 5 Zones

There is only one hypothesis for the Intermediate through Deep 5 Zones based on five lines of evidence (MSA, sulfur isotopes, the 50% probability of groundwater travel distance, age dating, and nitrate). This hypothesis refers to the MIW Area.

The composite weight of evidence MIW boundaries for the Intermediate through Deep 5 Zones are shown on Figures 5-21(b-g). On these maps, the composite boundary of MIW is shown as a wide, dashed line similar to that used in the BGQA to delineate BGQA water. The extent of MIW is indicated as a band, and not a fine line, that reflects the multiple and variable factors used to estimate the extent of MIW. In all cases, the boundary of MIW is shown as a band between wells, because the extent is based on comparison of each LOE between wells, not at or near a well.

The MIW extents derived from all five LOEs are generally similar. Specific exceptions were:

- Deep 1 Zone, where the use of the mine-influenced water type at B/W-1D1 to define the extent of MIW at that location would require including as MIW three other wells (B/W-60D1, B/W-61D1, and B/W-29D1) closer to the Site that have an agricultural signature based on water type and elevated nitrate concentrations (11, 14, and 5.8 mg/L, respectively).
- If data are sparse in a zone, the extent of MIW is conservatively mapped to represent the extent in overlying and/or underlying zones (e.g., the extent of MIW in Deep 2 and Deep 4 Zones is extended northeastward to match the extents of MIW in Deep 1, Deep 3, and Deep 5 Zones). This approach provides vertical integration of data in overlying

and underlying zones and is consistent with Site data indicating that the zones are hydraulically connected.

As shown in Figures 5-21(b-g), the areal extent of MIW is greatest in the Intermediate Zones and decreases with depth (Appendix M, Table 1). MIW is assumed to occur in each zone continuously between the northern extent of MIW shown in these figures and the western, southern, and eastern boundaries of the alluvial aquifer. The northernmost limit of the MIW in each zone changes from the northwest to the northeast with depth. The change in overall northern position of the MIW boundary with depth does not change as much as the slope of bedrock in the southern part of the Site. The decrease in areal extent of MIW with depth is primarily controlled by the northerly dip of bedrock underlying the southern part of the alluvial aquifer.

MIW is mapped beneath the UEP in the Intermediate through Deep 5 Zones. MIW is also mapped beneath the LEP in the Intermediate through Deep 2 Zones, but less so in the deeper zones, in part due to the alluvial aquifer pinching out the west with depth below the LEP. On-Site MIW is also mapped to the east and west of the Evaporation Ponds. On-Site COI concentrations in MIW decrease with distance from the UEP and LEP.

Off-Site MIW occurs in the Intermediate through Deep 5 Zones in an area to the northeast of the intersection of the UEP and LEP that extends progressively farther from the Site with depth. In the Deep-3, Deep 4, and Deep 5 Zones, MIW extends beyond the B/W-1 well cluster location. The area of MIW in the Deep 3 through Deep 5 Zones is consistent with groundwater flow directions in these zones.

5.4.11 Comparison of the Extent of MIW with BGQA Maps

The extents of the High Confidence MIW Area Shallow Zone and MIW Areas Intermediate through Deep 5 Zones are presented in Sections 5.4.8 and 5.4.10 and are shown in blue for each groundwater zone on the stacked map graphic of Figure 5-29. The BGQA boundaries are also shown in yellow for comparison. The differences between the two boundaries are most pronounced in the Shallow through Deep 2 Zones, and less pronounced in the Deep 3 through Deep 5 Zones. The extent of MIW in the Shallow through Deep 2 Zones do not exhibit the unusual geometry of the extent of BGQA in those zones that is inconsistent with hydrogeologic knowledge of the Study Area.

5.4.12 Evapoconcentration of Shallow Zone Groundwater

The identification of several water types presented in Appendix N and discussed in Section 5.4.3 invites questions about sources of sulfate and other major ions in addition to geochemical reactions associated with geothermal processes. The potential for evapoconcentration processes to influence water chemistry in the Shallow Zone in the area north and west of the ACMS (and even beneath the ACMS) is discussed below. As used herein, the term evapoconcentration of water refers to both: 1) the concentrating effects as water is evaporated and leaves solutes in an ever-reducing volume of water; and 2) historical geological processes in which the former Lake Lahontan left evaporite deposits along the ancestral shoreline, which could then be resolubilized by shallow groundwater systems.

Information on the occurrence of Lake Lahontan in Mason Valley during the Pleistocene is provided in Section 4.6. Lake Lahontan formed during a more humid climate with greater mountain runoff than today. As the climate became more arid, most of Lake Lahontan "evaporated to dryness and the dissolved salts were deposited in the upper part of the sediments, producing modern playas …" (Papke, 1976, p. 7). Authigenic evaporite minerals known to occur in Nevada playa deposits include the following sulfates (Papke, 1976, p. 9):

- Gypsum ($CaSO_4 \cdot 2H_20$);
- Bassanite ($CaSO_4 \cdot 1/2H_20$);
- Anhydrite ($CaSO_4$);
- Thenardite (Na_2SO_4);

- Mirabilite ($Na_2SO_4{\cdot}10H_2O$);
- Glauberite ($Na_2Ca(SO_4)_2$); and
- Epsomite ($MgSO_4{\cdot}7H_2O$).

One of the minerals listed above (mirabilite) was mined at the Wabuska Marsh from 1925 until the early 1930s (Papke, 1976, p. 28).

Pre-Anaconda aerial photographs of the Study Area taken in 1938 indicate the presence of white deposits (Figures 5-24[i-j]). It is unknown whether the white material in the photographs is associated with Lake Lahontan, and/or evapoconcentration of modern groundwater discharge, or another process. The distributions of chloride and sulfate measured in 3Q 2014 are superimposed on the aerial photographs in Figures 5-24(i-j). Elevated concentrations of chloride and sulfate occur in the Shallow Zone in the areas of white deposits, although the sulfate trends also reflect infiltration from the subsequently constructed Evaporation Ponds.

Groundwater elevations were at or near the land surface prior to construction of the Wabuska Drain and lowering of the water table due to irrigation well pumping. Evaporation of groundwater in the arid setting would have been intensive and formation of mineral salts during this period is expected. This process could have also contributed to the white areas observed in Figures 5-24(i-j).

Water chemistry data from the Shallow Zone were evaluated for indications of evapoconcentration and sources of sulfate and other major ions. The MSA evaluation identified Clusters 5 and 11 as Geothermal Surface Evaporated and Geothermal West Evaporated, respectively. Chloride is a good indicator of evapoconcentration in the Shallow Zone as it is conservative, and the groundwater chemistry is well below the saturation index of chloride salts such as halite. If the major ions are being concentrated by evaporation, the increase in concentration should parallel the increase in chloride.

The 3Q 2014 major ion data for the Clusters 5 and 11 wells are plotted in Figure 5-28, excluding wells LEP-MW-2S, LEP-MW-5S, PW-9S, and PW-10S (which Figure 5-26a indicates contain a component of MIW). The data are arrayed in order of increasing chloride concentration. The sodium and sulfate trends parallel the chloride concentrations, which is consistent with the evapoconcentration process. The magnesium and calcium trends have a flatter slope than the chloride trends, which may reflect the precipitation of carbonate minerals. Some areas of the Shallow Zone are saturated with respect to calcite (Appendix N, Figure 7-42a).

In summary, it is probable that contact with historical evaporite minerals and evapoconcentration influence the sulfate and other ion concentrations in Cluster 5 and 11 Shallow Zone groundwater samples.

6 CONTAMINANT FATE AND TRANSPORT

The mine-related COIs in OU-1 groundwater include acidity (i.e., low pH), TDS, major ions including sulfate, metals, and radiochemicals including uranium. The physical transport mechanisms and geochemical attenuation/mobilization processes that affect the movement of COIs in Study Area groundwater are discussed in Section 6.1 and 6.2. Plume dynamics are summarized from Appendix M in Section 6.3. Temporal trends of key COIs are presented in Section 6.4.

6.1 Contaminant Transport

Transport mechanisms are physical processes that move COIs from source areas downgradient in the groundwater system. In the Study Area, COIs were introduced to groundwater from the ACMS and/or agricultural practices, and also occur naturally in groundwater in Mason Valley because of groundwater contact with naturally-mineralized and/or hydrothermally-altered bedrock associated with copper deposits and/or alluvial sediments derived from mineralized bedrock.

The dominant solute transport mechanisms for COIs in groundwater are advection and dispersion. Advective transport is the migration of the COI with the groundwater. Groundwater moves from areas of recharge (i.e., high hydraulic head) to areas of discharge (i.e., low hydraulic head) and groundwater velocities are determined by solving the groundwater flow equation, which is a function of hydraulic conductivity, hydraulic gradient, and porosity. Groundwater levels in monitor wells provide hydraulic head and groundwater gradient information that can be used to determine magnitude and direction of groundwater velocity. Hydrodynamic dispersion describes the spread of COIs around an average groundwater flow path, beyond the region they would normally occupy due to advection alone. Hydrodynamic dispersion is the sum of two processes: mechanical dispersion and molecular diffusion. Mechanical dispersion results from mixing that occurs because of local variations in groundwater velocity and the aquifer’s matrix.

Molecular diffusion results from variations in solute concentrations within the groundwater system.

The primary influences on groundwater movement in the Study Area are subsurface lithology and structure, and local groundwater pumpage and irrigation associated with agriculture. Agricultural activities influence groundwater flow rates and directions, chemical migration pathways and transport rates, and contribute chemicals to groundwater via leaching of soil amendments and leaching of naturally-occurring uranium and subsequent transport through the vadose zone to groundwater. Irrigation practices near the Site, including groundwater extraction using high-capacity wells and conveyance/irrigation of both water from the Walker River and groundwater, alter groundwater flow rates and directions during the irrigation season due to the alteration of the natural recharge/discharge rates and locations. These alterations in groundwater flow affect contaminant transport rates and directions.

6.2 Geochemical Attenuation/Mobilization Processes

Geochemical processes that affect the release and subsequent mobility/attenuation of mine-related COIs such as sulfate, uranium, and metals during groundwater transport in the Study Area have been evaluated (BC, 2016b; Appendix J-7) using: 1) Study Area groundwater data from August 2014 including field parameter measurements (i.e., pH, ORP, and DO) and chemical concentrations; 2) the EPA-approved thermodynamic database developed for geochemical modeling Site geochemical attenuation/mobilization processes; and 3) correlations between common groundwater chemicals that affect uranium mobility.

6.2.1 Groundwater Geochemical Conditions and Chemical Speciation

Geochemical oxidation/reduction (redox) conditions within alluvial aquifer groundwater are variable; however, certain general trends and patterns are observed. In general, oxic conditions (i.e., DO > 1 mg/L and higher Eh values) occur in Shallow Zone groundwater and suboxic to anoxic conditions (i.e., DO < 1 mg/L and lower Eh values) occur in Intermediate and Deep groundwater zones (Figure 6-1). Exceptions to this general pattern include the following: 1) in Shallow Zone groundwater beneath much of the Site and off-Site to the north of the Evaporation Ponds, anoxic (rather than oxic) conditions occur; 2) in Deep groundwater zones beneath the Hunewill Ranch, oxic (rather than anoxic) conditions occur; and 3) on the west side of the Study Area adjacent to the Singatse Range, oxic conditions occur in the Shallow Zone, as well as all deeper zones.

Dissolved iron and the iron-system mineralogy of all the metals present in groundwater provide the most important constraints on pH, redox, and other metal solubilities in Study Area groundwater (BC, 2016a; Appendix J-5). Ferric hydroxide solids in the aquifer sediments can adsorb significant concentrations of dissolved metals and metalloids such as uranium and arsenic, attenuating transport of these constituents. The importance of iron mineralogy and its widespread influence on the Study Area fluid chemistry are illustrated with two Eh-pH diagrams (Figures 6-2 and 6-3). The mineral stability fields in each diagram were constructed for groundwater pH values below and above 5.5, respectively, using chemical data from discrete groundwater populations. The individual samples were then plotted on each diagram. The populations were selected from similar chemical environments (samples from two or more unrelated populations are scattered and often obscure the trends).

Groundwater samples from two populations with strong mining impact (pH values below 5.5; Figure 6-2) clearly plot along the K-jarosite and schwertmannite phase boundaries and triple points. Sample alignment near and along the aqueous-mineral boundaries demonstrates the important role these minerals play in dominating and buffering the acidic groundwater at relatively oxidizing Eh levels, as noted above. Also shown on Figure 6-2 is the specific sample from which the phase boundaries in the figure were computed.

Groundwater samples from two off-Site and more alkaline populations (pH values above 5.5; Figure 6-3) are aligned along the aqueous Fe^{2+} - $Fe(OH)_3$(ppd) phase boundary. The mineral

$Fe(OH)_3(a)$ is an important adsorptive phase that limits/attenuates the concentrations of other groundwater metals.

As noted previously, the strong alignment demonstrates the impact of the ferric hydroxide phase upon the groundwater redox and pH. The phase boundaries for this diagram were computed from the cluster centroid composition. Based on the groundwater redox conditions and geochemical modeling (BC, 2016a; Appendix J-5), chemicals in Study Area groundwater exhibit the following:

- The dominant aqueous redox states determined in virtually all August 2014 groundwater samples were As(V), Fe(II), Mn(II), Se(IV), and U(VI). Vanadium was present in the V(V) redox state in all August 2014 groundwater samples from the Intermediate, Deep, and bedrock wells. In Shallow Zone groundwater, the V(IV) redox state predominated below a pH of about 5 and the V(V) redox state predominated above a pH of 5.
- Sulfur dissolved in Study Area groundwater is present as sulfate (a negatively-charged ion) and gypsum saturation is observed only in groundwater samples with sulfate concentrations exceeding approximately 1,500 mg/L.
- In the absence of sulfate reduction or gypsum precipitation, the negatively-charged sulfate ion is minimally attenuated in groundwater systems by adsorption. Consequently, in Study Area groundwater with sulfate concentrations less than approximately 1,500 mg/L, sulfate can be expected to be transported as a conservative constituent.
- U(VI) is the dominant oxidation state of dissolved uranium in all August 2014 groundwater samples. Differences in the aqueous speciation of uranium are related to the pH and availability of cations in solution, not redox conditions. Dissolved uranium in Study Area groundwater is predominantly present in complexes with sulfate, carbonate, and/or calcium that form neutral or negatively-charged ions (e.g., $Ca_2UO_2(CO_3)_3^0$, $CaUO_2(CO_3)_3^{-2}$, $UO_2(SO_4)_2^{-2}$, and UO_2SO_4; Figure 6-4). Samples with dominant calcium-uranyl-carbonate ligands reflect oxidizing, carbonate-rich groundwater conditions. Samples with dominant uranyl-sulfate ligands reflect oxidizing conditions with no detectable amounts of alkalinity and pH <5.

- Formation of neutral or negatively-charged aqueous uranium species has been shown to limit uranium adsorption and increase uranium mobility (Fox et al., 2006; Stewart et al., 2010). Since limited uranium adsorption onto hydrous ferric hydroxide solids and soils is expected in neutral to slightly alkaline groundwater (Hsi and Langmuir, 1985; Echevarria et al., 2001), uranium attenuation during groundwater transport in off-Site portions of the Study Area can be assumed to be negligible as a conservative first approximation.
- Geochemical modeling predicts the potential for jarosite mineral precipitation primarily in the Shallow Zone under the LEP, UEP, Thumb Pond, and Phase IV HLP (Figure 6-5). Uranium and other metals associated with predicted jarosite mineral phases and aquifer sediments beneath these features potentially represent a persistent source of chemicals to groundwater.
- The spatial distributions of elevated concentrations of mobile parameters such as sulfate and uranium are significantly more extensive than the spatial distributions of acidity (as indicated by pH) and metals. The limited spatial extent of low pH values and elevated metal concentrations indicates that geochemical attenuation processes in Study Area groundwater reduce acidity and limits the mobility of metals relative to the more mobile chemicals such as sulfate and uranium. Elevated metal concentrations in groundwater correlate strongly with low pH.

6.2.2 Controls on Uranium Mobility in North Study Area Groundwater

As noted previously in Section 5.2.2, the concentrations of numerous COIs in groundwater increase along groundwater flow paths beneath agricultural fields in the NSA. The correlation between increasing concentrations of alkalinity and calcium, associated with agricultural activities, and increasing uranium concentrations as groundwater flows beneath agricultural fields in the NSA is shown on Figure 6-6. Increasing alkalinity and calcium concentrations are important controls on the formation of uranium species that have a low tendency to bind to aquifer solids (Bernhard et al., 2001) and, thus, uranium is mobilized from aquifer solids to adjacent groundwater as alkalinity and calcium concentrations increase. In addition, the observed correlation among increasing concentrations of alkalinity, calcium, and uranium is consistent with the previously-noted geochemical modeling predictions of uranium speciation in Study Area groundwater.

The upper panel in Figure 6-6 displays data for all the alluvial monitor wells in the NSA that are located near agricultural fields. The lower panel in Figure 6-6 displays a subset of the data presented in the upper panel. Specifically, this lower panel displays data along a groundwater flow path from wells upgradient of the NSA agricultural fields (i.e., wells B/W-56 and B/W-59) to wells/well clusters located downgradient of the NSA agricultural fields (i.e., well cluster B/W-57 and well YPT-MW-12I). In wells located upgradient of the NSA, uranium concentrations are less than 12 µg/L. However, as groundwater flows beneath the NSA agricultural fields and concentrations of alkalinity and calcium increase, naturally-occurring uranium attached to aquifer solids is released. At well B/W-57I, elevated uranium concentrations range from approximately 48 to 72 µg/L, and at well B/W-57D1 elevated uranium concentrations range from 73 to 110 µg/L.

Well YPT-MW-12I is located hydraulically downgradient of well cluster B/W-57, which provides additional information about chemical loading to alluvial groundwater associated with agricultural activities in the NSA. The influence of agricultural activities on chemical concentrations in YPT-MW-12I is shown on Figure 6-7. Although concentrations of uranium in groundwater at YPT-MW-12I have increased recently (upper panel), the trends in chemical concentrations in groundwater at this well are consistent with geochemical projections based on the control that calcium and alkalinity have on uranium mobility and are not related to the Site.

As indicated in the lower panel, sulfate concentrations in well YPT-MW-12I are greater than 71 mg/L and exhibit seasonal variability, with elevated concentrations occurring in February of each year. Plume advancement cannot account for the magnitude of sulfate concentrations or the seasonality observed in this well because sulfate concentrations are lower in wells to the southwest that demarcate the leading edge of MIW (i.e., well clusters B/W-10, B/W-52, and B/W-

55). Instead, the concentrations of sulfate in well YPT-MW-12I can only be accounted for by sulfate concentrations in upgradient groundwater, such as sampled at well B/W-57I, which is impacted by agricultural activities.

6.2.3 Site-Specific Distribution Coefficients

As groundwater migrates within the Study Area, certain geochemical reactions occur between the COIs in groundwater and the alluvial aquifer sediments. These reactions determine the relative rates at which chemicals migrate with respect to the advective front of groundwater. The primary process associated with the attenuation of inorganic COIs, particularly metals, in groundwater is adsorption onto the surfaces of minerals or organic material in the aquifer sediments.

The Site-specific K_d discussed in Section 3.3.5 may be used to quantify adsorption reactions between COIs in groundwater and the alluvial aquifer sediments during groundwater flow and chemical transport simulations. It should be noted that the distribution coefficient is a bulk measurement and provides only indirect information on the type of adsorption interactions taking place on the alluvial aquifer sediments. Summary statistics for the revised K_d values calculated using zonal groundwater sample data are shown in Table 6-1.

Table 6-1. Summary Statistics for Zonal Sample K_d Values

Analyte	Distribution Coefficient Statistics (L/kg)				
	Minimum	Maximum	Mean	Geometric Mean	Median
Sulfate	0.05	0.62	0.18	0.15	0.16
Uranium	0.42	289	49	17	33
Arsenic	108	6,412	1,763	957	800

Notes: L/kg = liters per kilogram

K_d values less than 1 L/kg, as is the case for sulfate, indicate little if any partitioning of this negatively-charged ion from the dissolved (i.e., liquid) phase to the solids phase. The uranium K_d values are higher than K_d values for sulfate, indicating some tendency for uranium adsorption on aquifer sediments. Dissolved uranium in Study Area groundwater is predominantly present in neutral or negatively-charged complexes (Figure 6-4), which has been shown to limit uranium adsorption (Fox et al., 2006; Stewart et al., 2010). However, limited uranium adsorption onto hydrous ferric hydroxide solids and soils is expected in neutral to slightly alkaline groundwater (Hsi and Langmuir, 1985; Echevarria et al., 2001; Li and Kaplan, 2012), and likely explains the higher uranium K_d values.

K_d values for arsenic are substantially greater than 1 L/kg. These high K_d values reflect much higher concentrations of those chemicals in the solid phase versus the dissolved phase, which indicates a strong tendency for arsenic to partition from the dissolved to the solid phase.

Summary statistics for the K_d values calculated for the 25 chemicals and compounds using monitor well groundwater samples are shown in Table 6-2.

Table 6-2. Summary Statistics for Monitor Well K_d Values

Analyte	Distribution Coefficient Statistics (1)				
	Minimum	Maximum	Mean	Geometric Mean	Median
Aluminum	10,850	214,670	88,830	64,851	98,350
Arsenic	18.4	1,311	461.2	277.2	421
Boron	0.40	9.26	4.39	3.15	3.36
Barium	93.8	6,676	1,937	959	1,000
Calcium	0.38	20.6	7.62	3.47	4.74
Chloride	0.07	1.56	0.22	0.18	0.17
Cobalt	192.3	4,643	2,462	1,869	2625
Chromium	316	1,182	689	626	579
Copper	83.3	13,684	4,645	2,662	3,333
Fluoride	1.56	17.2	4.52	3.73	3.26
Iron	8.6	541,880	22,112	146,061	242,733
Potassium	5.1	292.4	109.0	72.7	97.5
Lithium	46.0	379.3	126.6	116.1	113.8
Magnesium	0.34	76.8	26.2	12.5	23.5
Manganese	56.8	3,838	795	351	291
Molybdenum	0.14	2,130	176.9	21.0	63.1
Sodium	0.21	1.3	0.78	0.64	0.88
Nickel	300	1,736	1,142	1030	1,200
Nitrate	0.03	4.0	0.30	0.16	0.15
Lead	1,000	48,100	14,983	8,355	9,600
Selenium	35.9	9,180	3,084	1,202	2154
Sulfate	0.03	1.04	0.186	0.15	0.17
Uranium	1.0	238.2	41.7	20.2	25.6
Vanadium	43.5	4,717	1,115	673	917
Zinc (2)	1,385	1,385	1,385	1,385	1,385

Notes:
1. L/kg = liters per kilogram.
2. For zinc, only one set of co-located sediment and water samples were available for calculating a K_d.

The majority of the K_d values calculated using the monitor well groundwater samples are greater than 1 L/kg. Chloride, nitrate, and sulfate (negatively charged ions in groundwater) and sodium were the only chemicals where the K_d values were consistently less than 1 L/kg. Boron, calcium, and fluoride had K_d values in the range of 1 to 10 L/kg. All other chemicals and compounds had K_d values ranging from 10 L/kg up to approximately 500,000 L/kg.

The values at the high end of the range are influenced by the presence of non-detects in the data for the groundwater concentrations. In particular, groundwater concentrations for aluminum, iron, and lead were almost universally below the detection limits. However, the concentrations in the soil samples for those same chemicals were nearly all above the detection limit. In these cases, the groundwater concentrations used to calculate K_d values typically represented the reporting limit for that particular chemical. Given that, the K_d values can be thought of as upper bounds for that particular chemical and indicate little mobility in groundwater.

The variability between the K_d values for the same chemical or compound was, in general, consistent. Only a few chemicals had K_d values with variabilities of more than one order of magnitude. Typically, the wider-ranging K_d variabilities were expressed for those chemicals with non-detect results that were incorporated into the calculation.

Statistical parameters characterizing the uranium K_d values derived from zonal and monitor well groundwater sample data are similar. Uranium K_d values based on monitor well data vary by approximately two orders of magnitude. A correlation of uranium K_d values with Site geochemical data shows that uranium adsorption varies as a function of changes in pH and concentrations of alkalinity, calcium, and magnesium. Consequently, use of a constant K_d approach to simulate uranium adsorption during predictive transport modeling may not accurately represent actual adsorption processes.

Instead, a surface complexation model (SCM), such as the general composite approach described by Davis et al. (2009), may be more suitable for modeling adsorption processes during transport at the Site because it can describe changes in adsorption reactions at mineral surfaces as chemical conditions and aqueous speciation(s) change. However, in off-Site portions of the Study Area where aqueous concentrations are lower and hydrous ferric hydroxide solid concentrations are more sporadic, it may be appropriate to assume negligible uranium attenuation during groundwater transport as a conservative first-order approximation.

6.3 Mine-influenced Groundwater Plume Dynamics

One of the deliverables to NDEP under the IAOC (SOW, Section 6.1 [b]) is a PSTM:

> *ARC shall submit to NDEP for approval a Plume Stability Technical Memorandum. This memorandum is intended to provide the methodology and results of an evaluation of the stability of the groundwater plume. This will include statistical analysis of monitoring data as well as other lines of evidence such as movement or stability of centers of mass of the various COI plumes. The results of the plume stability evaluation will be used to assist in the evaluation of remedial options for the groundwater FS.*

The Final PSTM is provided in Appendix M and is summarized below. The PSTM describes an evaluation of the stability of the MIW plume at the Site. Plume stability has not been thoroughly evaluated in the area of wells B/W-32S and B/W-77S. This area of possible commingled groundwater COI sources will be evaluated in the risk assessment and FS and will be considered in the development of a long-term monitoring program as part of the remedy selected by NDEP in the Record of Decision. The evaluation assessed if the MIW at the Site is increasing or decreasing in volume and spatial extent due to the presence of COIs from Site operations. Multiple approaches consisting of groundwater flow rates and direction, center-of-mass calculations, total mass and average concentration calculations, trend analyses at individual monitor wells, and groundwater transport modeling were used to evaluate short- and long-term plume stability. The results of these evaluations indicate that, in general, the areal extent of MIW plume of COIs due to Site operations is changing very slowly. Evaluations of the center-of-mass of sulfate and uranium indicated no movement over the past six years in the centers-of-mass. Under current groundwater flow conditions, groundwater in the Shallow, Intermediate, and Deep 1 Zones at the northern Site boundary, with few exceptions, is from the fields north of the Site toward the Site and not from the Site towards the fields, which limits potential migration in these zones. Migration will occur in the deeper aquifer zones in the future, but rate of expansion of the plume of MIW in these zones will be very slow. The MIW plume stability is the result of very slow groundwater velocities and attenuation by dilution.

Particle tracking with the groundwater flow model, a transient model that incorporates seasonal irrigation pumping and annual variations in the availability of surface water, was used to evaluate potential groundwater flow away from the boundary of the mine-influenced plume over the next fifty years. The results of this analysis, which depicts groundwater movement near the boundary of the mine-influenced plume in each of the aquifer zones over the next fifty years, is shown on Figure 6-8a. The variable lengths of the particle tracks on Figure 6-8a, each of which represents potential migration over the next 50 years, illustrates the variability in groundwater velocities. The average groundwater flow velocities along the northern boundary of the plume of MIW are listed on Figure 6-8a. Groundwater velocities in the Shallow Zone along the northern boundary range from four to 39 ft/yr. In the Intermediate Zone velocities range from seven to 70 ft/yr, and in the Deep 1 Zone velocities range from 14 to 93 ft/yr. In the Deep 2 through Deep 4 Zones, groundwater velocities along the northern boundary range from 24 to 189 ft/yr. In the Shallow, Intermediate, and Deep 1 Zones, little movement occurs beyond the current plume boundary over the next fifty years, except in the eastern portion of the plume in the Intermediate and Deep 1 Zones in the vicinity of the former sulfide tailings pond. Similar flow patterns have likely existed for decades. In the deeper alluvial zones, some migration to the north-northeast is calculated. Calculated groundwater migration velocities at various locations in each of the alluvial aquifer zones are listed on Figure 6-8a. For purposes of evaluating plume stability, it is essential to evaluate groundwater concentrations of COIs at the downgradient extent of the groundwater plume of MIW, as expansion of the plume, if it occurs, will occur as the result of transport of COIs with groundwater. The particle flow paths shown on Figure 6-8a for each of the aquifer zones were used to define locations downgradient of the plume for purposes of stability analysis.

Centers of mass for sulfate and uranium were calculated for the Shallow, Intermediate, Deep 1, Deep 2, Deep 3, Deep 4, and Deep 5 Zones of the alluvial aquifer for each year in the seven-year period from 2012 through 2018. The calculated centers of mass in each year in the Shallow Zone are shown on Figure 6-8b, and the calculated centers of mass in the Deep 1 Zone are shown on Figure 6-8c. The calculated centers of mass in the other groundwater zones are shown on figures in Attachment C in Appendix M.

The calculated centers of mass for sulfate and uranium are located beneath the Evaporation Ponds, except for the sulfate and uranium centers of mass in the Deep 5 Zone. In the Deep 5 Zone, the sulfate and uranium centers of mass are located beneath the agricultural fields within a few hundred feet of the Site boundary.

Perimeter wells were selected to evaluate the migration of the plume in each of the alluvial groundwater zones. These wells were selected to be generally downgradient of the perimeter of the plume of MIW as determined using the results of particle tracking with the groundwater model. When downgradient wells were not available or determined to be influenced mainly by processes other than the Site (i.e., agriculture), wells within but near the plume boundary were selected as perimeter wells. These particle tracking results are shown on Figure 6-8a.

The perimeter wells for each of the alluvial groundwater zones are shown on Figures 6-8(d-j) along with trends in sulfate concentrations at each monitoring location. Wells with increasing trends in sulfate concentrations are shown with an upward facing red triangle, wells with decreasing trends in sulfate are shown with a downward facing green triangle, and wells with no trend in sulfate are shown with a grey dot. Similar figures depicting trends in uranium concentrations are shown separately from the sulfate trends on Figures 6-8(d-j). The perimeter well network that was chosen is appropriate for assessing both potential horizontal and vertical migration of COIs.

In the perimeter monitor wells in all zones, with some exceptions, both sulfate and uranium concentrations do not have upward trends. In the Deep 2 Zone (Figure 6-8g), there are two perimeter wells that display increasing trends for both sulfate and uranium (B/W-9D2 and B/W-25D2) indicating that there is some potential that expansion of the groundwater plume is occurring in this zone. These data indicate that expansion of the groundwater plume may be occurring in the vicinity of these wells, but concentrations of sulfate are far below its secondary Maximum Contaminant Level (MCL) and concentrations of uranium are far below its MCL. In the Deep 5 Zone (Figure 6-8j), perimeter wells B/W-27D5 and B/W-65D5 have increasing trends for sulfate and uranium. These wells, though, also have increasing trends for most of the COIs evaluated (Appendix M), suggesting something more complex is occurring at these locations than simply migration of MIW. In addition, these wells have elevated concentrations of molybdenum, and B/W-65D5 also has elevated concentrations of arsenic and fluoride suggesting a geothermal source. Sulfate concentrations are increasing in the Deep 4 Zone (B/W-27D4), but uranium concentrations are stable (Figure 6-8i). These exceptions indicate some temporal variability in plume geometry.

As stated in the PSTM (Appendix M, p. 33), the evaluation of plume stability based on trend analysis at the perimeter wells was weighted more heavily than trends in non-perimeter wells in the evaluation of potential evidence of plume expansion. However, concentration trends in the interior of the MIW area can inform assessments of ongoing sources, as discussed in Section 6.4. Trends of increasing concentrations of sulfate or uranium measured in wells located within the extent of MIW are evaluated in the PSTM Addendum (Appendix M). The results are summarized in Table 6-3 and include the calculated rate of increase listed in Table 4 of the PSTM (Appendix M). These results are utilized in the evaluation of ongoing sources presented in the following section.

Table 6-3. Wells Located Inside Extent of MIW with Increasing Trends of Sulfate, Uranium, or Both

Zone	Wells with Increasing Sulfate Trend	Rate of Change (mg/L/yr)	Wells with Increasing Uranium Trend	Rate of Change (µg/L/yr)	Wells with Increasing Trends of Both Sulfate and Uranium
Shallow Zone	B/W-34S	9.4	B/W-22S	3.5	None
	MX-SXN	220	B/W-76S	0.9	
	MX-SXS	27	LEP-MW-3S	140	
	PW-8S	770	PA-MW-1S	3.7	
			PA-MW-2S	3.7	
			PA-MW-7S	3	
			USGS-2BS	0.85	
			W5AB-2S	4	
Intermediate Zone	None	N/A	PA-MW-4I	13	LEP-MW-8I
					LEP-MW-9I
					W4CB-1I
Deep 1 Zone	W4CB-2D1	51	None	N/A	None
	W5DB-D1	21			
Deep 2 Zone	None	N/A	None	N/A	B/W-62 D2
Deep 3 Zone	B/W-29D3	14	None	N/A	B/W-1D3
Deep 4 Zone	None	N/A	W4CB-2D4	83	None
			W5DB-D4	24	
Deep 5 Zone	None	N/A	None	N/A	B/W-62 D5
					B/W-65 D5

Notes:

1. Individual sulfate and uranium trends are from PSTM, Table 4; and PSTM Addendum, p. 2 (Appendix M)
2. Listing of wells with combined increasing trends are from the PSTM, pp. 34-43 (Appendix M)
3. N/A = Not Applicable

6.4 Ongoing Source Evaluation

Sources of COIs to Site-wide groundwater are discussed in Section 5.2. Of the mine facilities, the most elevated concentrations of COIs occur beneath OU-4a (Section 5.2.1). The following analysis was performed to augment OU-specific RIs and further evaluate the potential for on-going sources from mine facilities to Site-wide groundwater. In this subsection COI and water level trends in monitor wells located within the footprint of the High Confidence MIW Area in the Shallow Zone are assessed to help evaluate whether mine-related COIs in on-Site materials or the vadose zone are continuing to migrate to groundwater at measurable flux rates.

The Shallow Zone is assessed because any ongoing sources of COIs in surficial materials or the vadose zone would first influence the Shallow Zone. The assessment includes the Evaporation Ponds in OU-4a and whether episodic, severe storm events infiltrate and transport COIs from the vadose zone to groundwater. When such events occurred during the period of the RI (such as January 2017[10]), and if these storm events resulted in transport of measurable quantities of COIs to groundwater, then it is logical that "spikes" of COI concentrations would be evident in the chemical trends of wells in this area. It is also possible that the potentiometric surface beneath the Evaporation Ponds would have increased in response to infiltration of large storm events during this period of record. The RI data record is reviewed relative to these two hypotheses in the following discussion.

Sulfate and uranium are mine-related COIs and are mobile. Concentration trends of sulfate and uranium in monitor wells located within the MIW are plotted in Figures 6-9(a-b) and 6-11(a-b), respectively. Visual review of the temporal trends indicates variable, but range-bound concentrations in individual wells, and the composite evaluation of all wells indicates stable to declining trends in all but a few wells. There are no "spikes" or other discernable increases in concentration that could be attributed to storm events.

A review of sulfate and uranium trends in Figures 6-9(a-b) and 6-11(a-b) indicates that wells PW-8S and MW-SXN, and well LEP-MW-3S exhibit visibly increasing trends for sulfate and uranium, respectively. Concentrations of sulfate appear to be increasing in samples from well LEP-MW-6S since 3Q2015. The increasing trends in these wells (shown in black in Figures 6-9 and 6-11) are consistent with the statistical evaluation of sulfate and uranium trends listed in Table 6-3, except

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[10] During a series of storm events that occurred between January 3 and January 13, 2017, a total of 2.4 inches of cumulative precipitation fell on the ACMS during this eleven-day period, or almost half of the annual average precipitation at the Site (Copper Environmental Consulting and Broadbent and Associates, 2019).

well LEP-MW-6S.[11] Wells PW-8S and MW-SXN, and well LEP-MW-3S have calculated annual increases for sulfate or uranium that are one to two orders of magnitude greater than the other wells in Table 6-3. Thus, it is expected that the rate of increase in these three wells would be visibly evident in Figures 6-9 and 6-11.

Concentrations of sulfate, uranium, alkalinity, nitrate, and pH are plotted for each of these wells in Figures 6-10 and 6-12 to evaluate whether there are likely mechanisms responsible for the increase in sulfate or uranium. The visible trends for the specific COIs are summarized in Table 6-4.

Table 6-4. COI Trends in MIW Wells with Visibly Increasing Trends in Sulfate or Uranium

Well	Sulfate	Uranium	Alkalinity	Nitrate	pH
PW-8S	**Increasing**	Stable	Increasing	Stable	Decreasing
MW-SXN	**Increasing**	Stable	Decreasing	Stable	Decreasing
LEP-MW-6S	**Increasing**	Decreasing	Decreasing	Increasing/ Decreasing	Decreasing
LEP-MW-3S	Stable	**Increasing**	Increasing	Increasing	Stable

Notes:
1. The increasing trends in sulfate or uranium are statistically significant for PW-8S, MW-SXN, and LEP-MW-3S (Table 6-3).
2. The trends in alkalinity, nitrate and pH are based on visual inspection and may not be statistically significant.
2. Trends are plotted on Figures 6-10 and 6-12.

The increasing concentrations of uranium in LEP-MW-3S are associated with apparent increasing trends of alkalinity and nitrate. Increasing trends of these three COIs are commonly associated with agricultural inputs. LEP-MW-3S is located near the northern boundary of the LEP. Given the current westerly flow direction in the Shallow Zone in this area, it is possible that the regional (agricultural/Walker River) water type may encroaching in this area. LEP-MW-3S was classified as the mine-influenced water type using the 3Q 2014 dataset.

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[11] The PSTM trend evaluation used the entire period of record and calculated a stable trend for sulfate in this well.

The increasing trends in sulfate concentrations in the other three wells do not appear to be consistently associated with increases or decreases in the other parameters. There are no pronounced increases in concentration in any of these wells that could be attributed to the effects of storm events.

As discussed above, potentiometric levels are evaluated for indications of increases in water levels associated with episodic, severe storm events. Well W5DB-S is located between the UEP and the LEP near the area where water ponds in the UEP during and after storm events, and is screened in the Shallow Zone (i.e., first water-bearing zone) beneath the UEP/LEP. Wells USGS-2BS and B/W-11S are Shallow Zone wells located approximately upgradient and downgradient, respectively, of the groundwater flow line (i.e., perpendicular to the potentiometric contour) passing through well W5DB-S, and provide reference points for potentiometric level changes at W5DB-S. The potentiometric levels at these three locations are plotted in Figure 6-13, along with available precipitation data. Potentiometric trends in wells USGS-2BS and B/W-11S (Figure 6-13) reveal the characteristic intra- and interannual water level fluctuations due to seasonal and irrigation effects discussed in Section 4.0. The water level trends in W5DB-S exhibit a subdued and dampened response to these seasonal and irrigation effects. The W5DB-S potentiometric trends are gradual and smooth; there are no sharp increases or “spikes” in water levels, except for a small spike (2.5 inches) following a large storm event in January 2017, which marked the maximum rainfall in one month (4.55 inches) over the period of record. All other storm events shown on the graph are not associated with a spike in water levels. The January 2017 water level may represent uncertainty in the measurement, such as likely occurred with the drop in water level reported for May 22, 2014 (a decrease of 7.8 inches relative to precedent and antecedent measurements). Storm events do not result in perceptible water level increases in the water level measurements collected from this Shallow Zone well.

In summary, evaluations of the water chemistry and potentiometric trends in MIW Shallow Zone wells do not reveal discernable increases in COI concentrations or potentiometric levels that could be caused by infiltration and transport of COIs from the mine-related material or vadose zone to groundwater in response to episodic, severe storm events.

7 HYDROGEOLOGIC CONCEPTUAL SITE MODEL

The Site is a former copper mine located in the west-central portion of the Mason Valley (Figure 1-1). Mining and ore processing operations were conducted by various owners from 1953 to 1999. Anaconda began producing copper in 1953 and divested the Site in 1978. During this time period the pit was excavated, mine water supply wells were installed and pumped, and process facilities and Evaporation Ponds were constructed. Arimetco operated heap leach facilities from 1989 to 1999. Key historical operational events relevant to groundwater conditions are summarized in Table 7-1. The generalized mine operational sequence and groundwater conditions through time are illustrated in Figure 7-1. The CSM incorporates relevant historical data from the RI.

Table 7-1. Key Operational Events Relevant to OU-1

Operational Processes and Events	Time Period
Crushed oxide ore processing in vat leach tanks, disposal of spent ore in the Oxide Tailings area, copper recovered from recirculated acid leach solutions in precipitation plant, waste liquid conveyed in unlined ditches to Evaporation Ponds.	1953 – 1978
• UEP in service	1954 – 1978
• LEP in service	1972 – 1978
Sulfide ore processing in froth floatation system, copper concentrates shipped to off-Site smelter.	1961 - 1978
Sulfuric acid used to leach oxide ore produced in fluosolids and acid plant.	1952 – 1978
• Leviathan Mine sulfur ore trucked to Site to produce sulfuric acid.	1952 – 1962
• Stockpiled Leviathan Mine ore blended with liquid sulfur (primarily from refineries). Percentage of Leviathan Mine ore used through this time period is not available.	1963 - 1978
Groundwater pumped by Anaconda to dewater the pit, for copper ore beneficiation, and for residential use in Weed Heights. Groundwater obtained from the wells near the pit, five wells in Evaporation Pond Area, well WW-10 in Process Area (beginning in 1960), and three off-Site area wells beginning in 1968.	1952 - 1978
• Oxide ore leaching only (lower groundwater use)	1952 - 1960
• Oxide and sulfide ore processing (higher groundwater use)	1961 - 1978
• Peak groundwater pumpage (~ 7,100 gpm)	1974
• Groundwater pumpage last full year of Anaconda period (~ 5,000 gpm)	1977
Arimetco pumping of groundwater from well WW-36 located north of east end of the pit (average rate of ~660 gpm)	1989 - 1999
Irrigation Pumping on the Hunewill Ranch	
• Approximate annual pumpage 300 gpm	1961- 1986
• Approximate annual pumpage 860 gpm (pumping capacity increased significantly in 1987)	1987 -2009

Note: Source of information is Section 2.0.

Geology

Mason Valley is a Basin-and-Range-type graben filled with up to 1,000 feet of unconsolidated valley-fill deposits of Tertiary and Quaternary age lying unconformably on a weathered surface of hydrothermally-altered Tertiary volcanic and Mesozoic intrusive bedrock. The valley is bordered to the west by the Singatse Range, to the east by the Wassuk Range, and to the north by the Desert Mountains. Mineralization and hydrothermal alteration associated with porphyry and skarn copper deposits occur in the Singatse Range and nearby portions of the Mason Valley.

Stratigraphy

The major stratigraphic features in OU-1 are the unconsolidated valley fill deposits and underlying bedrock. The unconsolidated valley fill deposits were derived primarily from erosion of the uplifted mountain blocks and from deposition in the floodplain of the Walker River. In addition, ancestral Lake Lahontan extended into the Study Area. Depositional processes have resulted in a complex interbedded sequence of alluvial sediments.

Within the Study Area, the alluvial sediments are up to 700 feet thick and comprise a single aquifer. The alluvial aquifer is subdivided into a Shallow, an Intermediate, and a Deep Zone (which is further subdivided into Deep Zones 1 through 5). These groundwater zone designations are based on elevation and used only to identify and group monitor wells with similar screen intervals at the same relative depth in the aquifer. Clay layers or other low-permeability sediments are laterally discontinuous except in areas such as the “50-ft Clay” (discussed in the next paragraph) in the north area of the LEP. Groundwater occurs in unconfined or semi-confined conditions in the Study Area.

Drilling activities conducted in the vicinity of the Evaporation Ponds prior to the RI documented the presence of an approximately 10-foot thick clay layer in some boreholes occurring at an elevation of 4,300 ft amsl, which corresponds to the Shallow and Intermediate Zone boundary.

This clay layer was termed the "50-ft Clay" because it is situated around 50 feet below the LEP. The lateral continuity and extent of the 50-ft Clay was undetermined at that time.

Extensive drilling conducted during the RI indicates that the 50-ft Clay appears continuous beneath a portion of the LEP and north of the LEP. The 50-ft Clay is inferred to extend approximately 2,900 feet north of the ACMS boundary (Figure 4-7b). Vertical groundwater flow from the Shallow to the Intermediate Zones is limited in this area. Clay lithology sections (Figures 4-7[c-g]) show that the "50-ft Clay" is not continuous beyond this area. As a result, this layer is not a laterally extensive confining unit across the Study Area.

Groundwater Flow and Changes Over Time

Historically, regional groundwater flow patterns in the Mason Valley prior to Anaconda mining operations were similar to current conditions with groundwater moving generally from south to north and discharging to surface seeps and geothermal springs in the northern portion of the basin. Local flow patterns have been significantly altered from historical patterns by agricultural and mining activities. The alluvial aquifer in the Mason Valley is very productive and groundwater is currently pumped extensively for irrigation (Carroll et al., 2010; SSPA, 2014).

Historical changes in groundwater flow patterns at the Site due to mining included 1) construction and subsequent post-mining water infilling of the mine pit; 2) groundwater pumping from mine water supply wells; and 3) to a lesser extent, operation of the PWS. Excavation of the mine pit and associated mine pit dewatering during the Anaconda operational period reversed the general northward groundwater flow in the vicinity of the pit (Figure 7-1). The pit continues to be a groundwater sink maintained by evaporative losses and is projected to have a steady-state long term level that is more than 100 feet lower than pre-mining groundwater levels. Thus, a groundwater divide currently exists in the Shallow Zone in the area east of Weed Heights. South of the divide, flow is to the south toward the mine pit.

Significant amounts of water were pumped by the mine water supply wells, which included wells used for pit dewatering and wells in the northern part of the Site constructed solely for water-supply purposes. A portion of this mine supply water was placed in the Evaporation Ponds as waste process solution. The pond water was lost to evaporation or infiltration to the groundwater and likely created mounded water table conditions beneath the ponds during operations.

The mine water supply wells were screened across much of the alluvial aquifer. Based on historical potentiometric maps and response of the alluvial aquifer to long-term RI pumping tests, the cones of depression created during pumping were large enough to create reversals of groundwater flow directions towards the pumping locations when operational. Anaconda pumping of mine water supply wells ceased in 1978. As shown on Figure 3-7, there was groundwater flow in the Shallow Zone northward from the LEP during the latter years of mine operation until the early 1990s, when flow directions were altered by irrigation on the adjacent fields.

Detailed documentation of hydraulic gradients and flow directions is not available prior to the RI. The capture zone of the PWS, which operated from 1986 to 2009, was limited to the Shallow Zone in the northeast corner of the UEP from 1986 through 1998, and expanded with the addition of wells at the north end of the LEP in 1999 to include the area beneath and immediately north of the LEP until the system was shut down in March 2009, as discussed in Section 3.3.1 and in Appendix J-1.

The alluvial aquifer is primarily recharged by downward percolation from irrigated fields (49%), leakage from irrigation ditches (29%), infiltration from the channel of the Walker River (20%), and MFR (2%), consisting of infiltration through bedrock of precipitation that falls on surrounding mountain ranges and minor tributary surface flows in ephemeral drainages (SSPA, 2014).

Recharge from precipitation falling directly on the valley floor is negligible (Huxel and Harris, 1969; Lopes and Allander, 2009a, 2009b).

Currently, alluvial groundwater near the Site generally flows down the valley, but flow directions are locally affected by bedrock outcrops within and along the margins of the valley, drawdown from pumped wells (particularly in deeper zones of the alluvial aquifer), and recharge sources including the Walker River, the West Campbell Ditch and irrigation on the Hunewill Ranch and other agricultural fields. Groundwater flow in the Shallow Zone beneath the Evaporation Ponds is to the west before eventually flowing to the northwest and then north. Groundwater in the alluvial aquifer moves slowly; aquifer hydraulic properties and flow rates are discussed in Sections 4.9.5 and 5.4.1.

As described in the OU-2 FSAP (Copper Environmental Consulting and Broadbent and Associates, 2018b, pp. 11-12), groundwater inflows to the Pit Lake are estimated to be slightly greater than the current rate of evaporation, based on a water balance study from 2007 (BC, 2017). Although the lake level is slowly rising, projections indicate that the lake will reach a steady-state level before 2027, when net water inflow and outflow (i.e., evaporation) are equal. Compared to the pre-mining groundwater elevation range of 4,350 to 4,375 ft amsl (Gill, 1951), the estimated steady-state Pit Lake level will be at least 100 ft lower than the pre-mining groundwater level due to the large amount of evaporation the Pit Lake surface experiences (Figure 3-8). Consequently, the Pit Lake is and will remain a groundwater sink resulting in a trough of depression (extending north up to the Process Areas) with local groundwater flowing toward the pit. Because the Pit Lake does not and will not in the future discharge into the Site-wide groundwater system, the Pit Lake is not a source of COIs to Site-wide groundwater.

Groundwater Pumping

Historical and present groundwater pumping rates over time for mining and Arimetco operations, remedial measures (i.e., PWS), and agricultural irrigation north of ACMS are summarized in Table 7-2. These groundwater extraction rates are used in the local domain area of the Site groundwater model (SSPA, 2014).

Table 7-2. Summary of Historical and Present Groundwater Pumpage

Time Period	ACMS	Off-Site Irrigation
1953 – 1959	3,614	0
1960 - 1969	5,253	3,098
1970 - 1979	5,866	4,874
1980 - 1989	18 (PWS)	3,117
1990 - 1999	25 (PWS) 1,065 (Arimetco)	3,806
2000 - 2012	58 (PWS)	5,320

Notes:

1. Units are acre-feet/year
2. Data from Appendix C of SSPA (2014) and Appendix A, Table 3-2 of this document.
3. Off-Site irrigation pumpage is from wells located inside the local domain of the groundwater model.

The ACMS pumping increased when the demand for water supply increased after the addition of the copper sulfide ore floatation circuit. Evaporation Pond Area wells were installed during the period 1959-1961 (WW-8, WW-9, and WW-11), in 1965 (WW-12C) and in 1968 and 1969 (WW-29, WW-22, WW-35, and WW-26 – the latter three wells are located just beyond the Site boundary) to provide additional water. Well logs indicate perforated casing intervals were large and straddle the alluvium-bedrock boundary. Based on perforated casing positions relative to alluvium-bedrock contact depths these wells drew water from Deep 3 through Deep 5 Zones. Pumping of these Evaporation Pond Area wells likely created significant downward hydraulic gradients and associated downward migration of COIs. Increased pumping on the Hunewill Ranch adjacent to the Site probably had a similar effect until these wells ceased operation in 2009. Arimetco pumping in the 1990s focused on well WW-36 located in the southern part of ACMS and likely had negligible influence on groundwater flow in the northern part of the Site.

Groundwater Travel Distances

Physical properties of the alluvium and hydraulic gradients of groundwater are used to estimate how far groundwater may have migrated from the UEP and LEP since the time of initial releases up to 2014 as described in Section 5.4.1. The estimated distance of groundwater travel in the

Shallow Zone based on the 50% and 5% exceedance probabilities (i.e., probability based on Monte Carlo simulation that the actual travel distance is greater than estimated) are shown on Figure 5-15d. The groundwater travel distances for the Intermediate, Deep 1, and Deep 2 Zones are shown on Figures 5-21b, 5-21c, and 5-21d, respectively. The 5% exceedance probability travel distance to the north through 2014 in the Shallow Zone extends to the north of well B/W-32S.

Groundwater travel distances for a five-year time period were calculated using the 1992-2014 parameters in Table 5-2. These calculations provide a frame of reference for the time period beyond that estimated by the Section 5.4.1 calculations (i.e., 2014-2019). In the Shallow Zone, between 50 ft (50% exceedance probability) and 200 ft (5% exceedance probability) of travel distance is calculated. In the intermediate Zone, approximately 175 ft of travel distance is calculated.

Evapoconcentration

The potential influence of evapoconcentration processes on water chemistry in the Shallow Zone in the area north and west of the ACMS (and beneath the Evaporation Ponds) is discussed in Section 5.4.12. Pre-Anaconda aerial photographs indicate the presence of white mineral deposits (Figures 5-24[i-j]) in this area. It is unknown whether the white material in the photographs is associated with Lake Lahontan, evaporite minerals, evapoconcentration of modern groundwater discharge, and/or other processes. The distributions of chloride and sulfate measured in 3Q 2014 are superimposed on the aerial photographs in Figures 5-24(i-j). Elevated concentrations of chloride and sulfate occur in the Shallow Zone in the areas of white deposits, although the sulfate trends also reflect infiltration from the subsequently constructed Evaporation Ponds. The evaluation of sodium, sulfate, and chloride concentrations in the Shallow Zone are consistent with the evapoconcentration process. It is probable that contact with historical evaporite minerals and evapoconcentration influence the sulfate and other ion concentrations in Shallow Zone groundwater samples north and west of the ACMS.

Pumpback Well System

The PWS operated from 1986 through early 2009, when it was shut down with EPA approval. The system generally had three operating wells from 1986 through 1988, all near the northeast corner of the UEP, and generally had 11 operating wells from 1989 through early 2009. A total of 340 million gallons were pumped from the Shallow Zone during the operation of the system, and about 10,000 tons of sulfate were removed from the aquifer. Of the total amount of sulfate removed, about 13% was removed by the wells located along the northern boundary of the LEP and 87% was removed by the wells near the northeast corner of the LEP. The effectiveness of the PWS when operational is detailed in Section 3.3.1 and in Appendix J-1, p. 24. The annual pumping rates for each of the wells are listed on Table 3-10, and the annual average sulfate concentrations are listed on Table 3-11. The total mass of sulfate removed at each of the wells annually was calculated based on the pumping rates and sulfate concentrations and are listed on Table 3-12. The capture zones shown on Figures 5 and 6 of Appendix J-1 indicate that the PWS was capturing some of the MIW along the northern margin of the Site and partially limited migration of this groundwater to areas beyond the Site boundary. The generally east-to-west groundwater flow direction across the northern end of the Site restricted the area of capture to an elongated strip roughly defined by the capture zones of the northern wells and to the limited downgradient and cross-gradient capture areas of the eastern wells. MIW that is south of the northern wells and to the west of the eastern wells was not within the capture zones of the PWS.

Bedrock Characterization

Bedrock characterization information indicates: 1) a high degree of fracture heterogeneity and vertical hydraulic connection between the bedrock and alluvial groundwater systems; and 2) low horizontal hydraulic connectivity and transmissivity of bedrock fractures over horizontal distances that are relevant to the scale of the Site and the surrounding Study Area. In addition to a high degree of three-dimensional (i.e., anisotropic) variability in hydraulic characteristics and hydrologic tracer signatures, the bedrock groundwater system also exhibits three-dimensional spatial variability in chemical concentrations. COI concentrations in bedrock groundwater are 10 to over 1,000 times lower than the values in overlying alluvial groundwater. In addition, areas of elevated mine-related COIs in bedrock groundwater are small in comparison to the alluvial aquifer and found mostly on-Site. The small areas of elevated mine-related COIs in bedrock groundwater indicate that bedrock fractures have low hydraulic connectivity and transmissivity over horizontal distances relevant to the scale of the Study Area. Bedrock groundwater flow velocities, average volumetric flow rates, and chemical fluxes through the bedrock groundwater system are low, and the bedrock groundwater system is not considered to be an important migration pathway at the Site. EPA (2015a) stated that sufficient data had been collected to conclude that bedrock is not an important migration pathway at the Site and requested preparation of a technical memorandum to update the bedrock HCSM.

Agriculture

Anthropogenic activities within the Study Area, especially agricultural activities, influence groundwater recharge, flow rates and directions, and chemical migration pathways and transport rates. These activities contribute chemicals to groundwater via evapoconcentration and leaching of soil amendments and naturally-occurring chemicals in alluvial sediments, and subsequent transport through the vadose zone to groundwater. Irrigation practices near the Site, including groundwater extraction using high-capacity wells and conveyance/irrigation of both water from the Walker River and groundwater, alter groundwater flow rates and directions during the irrigation season due to the alteration of the natural recharge/discharge rates and locations. Agricultural pumping seasonally results in strong downward vertical gradients that are often 10 to 100 times greater than horizontal gradients.

Geologic Background

Sulfate, uranium, arsenic, and other COIs occur naturally in groundwater in Mason Valley because of groundwater contact with mineralized and/or hydrothermally-altered bedrock associated with copper deposits and/or alluvial sediments derived from mineralized bedrock (BC, 2014a, 2016b). A comprehensive evaluation of a geothermal feature in the Study Area is provided in Appendix N and summarized in Section 5.2.3.

Sources of COIs

Sources of COIs to Study Area groundwater include multiple sources: 1) the mine waste facilities (past source), which have been grouped into the various Site OUs; 2) groundwater interaction with geothermal alteration minerals and evapoconcentration (past and current source); and 3) agriculture (past and current source). The major past mine-related sources of COIs to groundwater include: 1) OU-4a, the Evaporation Ponds (BC, 2014a, 2014d; Copper Environmental Consultants and Broadbent and Associates, 2019); 2) OU-8, the Arimetco Facilities (CH2M Hill, 2010, 2011a, 2011b); and 3) OU-3, the Process Areas (BC, 2014a, 2014e). OU-7, the Wabuska Drain, as a potential source is discussed in Section 5.2.1 and the OU-7 RI is in preparation at this date.

Mine-related COIs include acidity (i.e., low pH) and elevated concentrations of TDS; major ions, including sulfate and metals; and radiochemicals, including uranium. Concentrations of mine-related chemicals in groundwater are most elevated in the Shallow Zone of the alluvial aquifer beneath OU-4a features that include the LEP, UEP, Finger Ponds, Thumb Pond, and the Calcine Ditch. Concentrations of mine-related COIs such as sulfate and uranium within the mapped extent of MIW generally decrease with horizontal distance from these facilities (Figures 5-22[a-b] and 5-23[a-b]).

In 1955, the flow rate to the Evaporation Ponds averaged approximately 2,000,000 gpd or 1,385 gpm, and water quality characteristics of the fluid showed a free acid concentration of 1.0 g/L, total soluble salts concentration of 171 g/L, and total iron concentration of 37.5 g/L (Nesbitt, 1955; Dalton, 1998). The flow rate to the Evaporation Ponds varied considerably over the period of operations, however. A quantitative water balance of inflow and evaporation is not available. Therefore, a comprehensive and quantitative estimate of the total flux of COIs to the subsurface from the Evaporation Ponds is not possible. The COI maps presented in Section 5.0 provide reliable information needed to assess the current nature and extent of COIs.

The COIs were consistent through Anaconda operations. Extraction processes rarely undergo significant changes during mine life as efficiency is better under constant conditions. Many mines take significant efforts to blend ore from different parts of their pit to maintain a constant copper grade, rock hardness, and rock type to ensure their extraction process does not require fundamental changes on a regular basis. The ACMS was no different. For example, the Anaconda Company's mine in Butte, Montana operated the same floatation circuit from the 1950s through 1982 to process copper ore from the Berkeley Pit and the Continental Pit. "Sour" ore (containing acid-generating minerals) was either blended with sulfidic ore prior to floatation or sent to leach pads to not disrupt the delicate balance within the floatation circuit.

Nature and Extent of Contamination

The spatial distributions of elevated concentrations of mobile parameters such as sulfate and uranium are significantly more extensive than the spatial distributions of acidity (as indicated by pH) and metals. The significantly smaller spatial extent of low pH values and elevated metal concentrations indicates that geochemical attenuation processes in Study Area groundwater reduce acidity and limit the mobility of metals relative to the more mobile chemicals such as sulfate and uranium (EPA, 2007b; BC 2016a, 2016b). Elevated metal concentrations in groundwater correlate strongly with low pH.

Sulfate, uranium, and arsenic (and other COIs) occur naturally in groundwater in Mason Valley because of groundwater contact with naturally-mineralized and/or hydrothermally-altered bedrock associated with copper deposits and/or alluvial sediments derived from mineralized bedrock (BC, 2014a, 2016b). To the west of the Site where sulfate and uranium concentrations are low, elevated concentrations of arsenic are associated with MFR. In addition, sulfate, uranium, and other COIs are sourced to groundwater due to agricultural activities (BC, 2016b; EPA, 2016c). The assessment of background groundwater quality identified an area of groundwater in the NSA that has been impacted by agricultural activities rather than mining activities.

The spatial distributions of COIs in Site groundwater measured in 3Q 2014 are illustrated in Figures 5-1a through 5-8f, Figures 5-24(a-h), and Figures 5-25(a-g). The distributions of sulfate and uranium in groundwater in August 2014 are specifically illustrated on Figures 5-2(a-e) and 5-3(a-e), respectively. The most elevated concentrations of sulfate and uranium are observed in Shallow Zone groundwater beneath the Evaporation Ponds and downgradient of the Phase IV VLT HLP and VLT Pond, and concentrations generally decrease laterally and vertically away from these features by orders of magnitude. The current extent of sulfate and uranium in the Shallow Zone are shown in Figures 5-24(k-l). There are elevated concentrations of sulfate (Figure 5-2a) and uranium (Figure 5-3a) in the Shallow Zone north of the Site in the vicinity of well B/W-32S. Both sulfate and uranium concentrations are lower immediately north of the LEP relative to concentrations beneath the Evaporation Ponds and concentrations farther north near B/W-32S. One hypothesis to explain decreased concentrations of sulfate and uranium in the Shallow Zone in the area between wells B/W-32S and B/W-77S and the ACMS is hydraulic gradient reversal during the operation of the PWS, and/or commingling of COIs from different sources.

The extent of nitrate in the Shallow Zone in 3Q 2014 is shown in Figure 5-5a. Nitrate concentrations exceed 10 mg/L in most Shallow Zone wells beneath the agricultural fields north of the Site. Nitrate occurs at low to non-detected concentrations in the Shallow Zone beneath the Evaporation Ponds and much of ACMS to the south of the Ponds and in the area north of the

LEP. Nitrate concentrations are variable in the area west of the Evaporation Ponds, ranging from non-detect to over 10 mg/L.

Sulfate (Figure 5-2b) and uranium (Figure 5-3b) distributions exhibit a northeast longitudinal orientation in Deep 3 Zone groundwater from the northern portion of the Site to beneath the Hunewill Ranch toward former agricultural wells used seasonally to extract groundwater for crop irrigation.

The extent of MIW is evaluated using up to six lines of physical, chemical, and isotopic evidence (as explained in the indicated Sections):

- Groundwater flow distances (Section 5.4.1);
- Extent of nitrate from agricultural activities (Section 5.4.2);
- MSA to identify multiple water types (Section 5.4.3);
- Sulfur isotopic information (Section 5.4.4);
- Age dating of groundwater (Section 5.4.5); and
- Collocation of MIW indicators (Section 5.4.6).

The uncertainties and limitations of each LOE are discussed in the subsections identified in the bullets listed above.

Hypotheses of the extent of MIW in the Shallow Zone are assembled using mutually supporting LOEs selected from the list above. Review of the LOE information indicates that there are two hypotheses that are supported by mutually consistent evidence. One hypothesis integrates five mutually supportive LOEs into a composite evaluation of MIW using MSA (including ionic cross plots), sulfur isotopes, the 50% probability of groundwater travel distance, age dating and nitrate LOEs and excluding the collocation of MIW indicators LOE. This hypothesis is termed the High Confidence MIW Area in the Shallow Zone as there is high probability it contains MIW from the ACMS. The other hypothesis uses two mutually supportive LOEs to assemble a composite evaluation of MIW (using the conservative 5% probability of groundwater travel distance LOE and the collocation of MIW indicators LOE), although the sulfur isotope LOE does not contradict the hypothesis of low confidence MIW extent. This hypothesis is termed the Low Confidence MIW Area in the Shallow Zone. There is a lower probability that it contains MIW, and it is more likely influenced by other sources.

There is only one hypothesis for the Intermediate through Deep 5 Zones based on the first five LOEs listed above, and this hypothesis defines the extent of MIW in each of these zones. The extents of MIW by zone are shown in Figures 5-21(a-h); in the Shallow Zone, High Confidence MIW Area is shown in Figure 5-21a and Low Confidence MIW Area is shown in Figure 5-21h. The areal extent of MIW, regardless of which hypothesis for the Shallow Zone, is greatest in the Shallow and Intermediate Zones and decreases with depth (Appendix M, Table 1). MIW is assumed to occur in each zone continuously between the northern extent of MIW shown in these figures and the western, southern, and eastern boundaries of the alluvial aquifer. The northernmost limit of the MIW in each zone changes from the northwest to the northeast with depth. The change in overall northern position of the MIW boundary with depth does not change as much as the slope of bedrock in the southern part of the Site. The decrease in areal extent of MIW with depth is primarily controlled by the northerly dip of bedrock underlying the southern part of the alluvial aquifer.

The MIW is mapped beneath the UEP in all zones. MIW is also mapped beneath the LEP in the Shallow through Deep 2 Zones, but less so in the deeper zones, in part due to the alluvial aquifer pinching out to the west with depth below the LEP. The location of the MIW beneath the UEP and LEP is consistent with the contour maps indicating highest concentrations of sulfate, uranium, and other COIs beneath the UEP and LEP. On-Site MIW is also mapped to the east and west of the Evaporation Ponds. On-Site COI concentrations in MIW decrease with distance from the UEP and LEP.

Off-Site MIW occurs in the following three areas contiguous to the ACMS:

1. A triangular area in the Shallow Zone on the western border of the ACMS;
2. An area north of the LEP in the Shallow Zone (this area extends further north in the Low Confidence MIW Area [Figure 5-21h] encompassing the area that includes wells B/W-32S and B/W-77S, sometimes referred to as the "Hot Spot," than in the High Confidence MIW Area [Figure 5-21a]); and
3. An area to the northeast of the LEP that extends progressively further from the Site with depth. In the Deep-3, -4 and -5 Zones, MIW extends beyond the B/W-1 well cluster location.

Fate and Transport

Geochemical evaluations of groundwater quality data (BC, 2016a) indicate that sulfate and dissolved uranium form neutral or negatively-charged aqueous ionic species in groundwater throughout the Study Area (except beneath the Evaporation Ponds) and thus undergo little (if any) geochemical attenuation due to mineral precipitation or adsorption to aquifer materials during groundwater transport. Dissolved uranium in Study Area groundwater is predominantly present in complexes with carbonate plus or minus calcium, which reflect oxidizing, carbonate-rich groundwater conditions. Locally, dissolved uranium is also complexed with sulfate in Shallow Zone groundwater beneath the Evaporation Ponds where pH is less than 5 and sulfate concentrations exceed 1,500 mg/L.

Geochemical evaluations of groundwater quality data (BC, 2016a) also indicate the likely precipitation of solid mineral phases (e.g., jarosite – a sulfate-potassium-iron mineral) primarily in the Shallow Zone beneath the LEP, UEP, Thumb Pond, Finger Ponds, and Calcine Ditch. Of all the groundwater metals, dissolved iron and the iron-system mineralogy provide the most important constraints on pH, redox, and other metal solubilities in Study Area groundwater attenuating transport of these constituents. Ferric hydroxide solids in the aquifer sediments can sorb significant concentrations of dissolved metals and metalloids such as uranium and arsenic, attenuating transport of these constituents. The importance of iron mineralogy and its widespread influence on the Study Area fluid chemistry are demonstrated with Eh-pH diagrams in which groundwater samples from two MIW populations (pH values below 5.5) plot along the K-jarosite and schwertmannite phase boundaries and triple points (Figure 6-2). Sample alignment along the aqueous-mineral boundaries demonstrates the important role these minerals play in dominating and buffering the acidic groundwater at relatively oxidizing Eh levels. Groundwater samples from off-Site and more alkaline populations (pH values above 5.5) are aligned along the aqueous Fe^{2+} - $Fe(OH)_3(a)$ phase boundary (Figure 6-3). The strong alignment demonstrates the impact of the ferric hydroxide phase upon the groundwater redox and pH. $Fe(OH)_3$ppd) is an important sorptive phase which limits/attenuates the concentrations of other metals in groundwater.

Plume Stability

Updated estimates of the mass of sulfate and uranium within the MIW (High Confidence MIW Area in the Shallow Zone and MIW in the deeper zones) are provided in Appendix M, Section 6. The total volume estimate of MIW is 153,000 acre-feet, and it contained in 2014 an estimated 326,000 tons of sulfate and 48 tons of uranium. No irrigation or municipals wells are currently located within the extent of MIW. MIW does not discharge to surface water.

The PSTM describes an evaluation of the stability of the MIW plume at the Site. The evaluation assessed if the MIW at the Site is increasing or decreasing in volume and spatial extent due to the presence of COIs from Site operations. Multiple approaches consisting of groundwater flow rates and direction, center of mass calculations, total mass and average concentration calculations, trend analyses at individual monitor wells, and groundwater transport modeling were used to evaluate short- and long-term plume stability. The results of these evaluations indicate that, in general, the areal extent of MIW plume of COIs due to Site operations is changing very slowly. Evaluations of the center of mass of sulfate and uranium indicated no movement over the past six years in the centers of mass. Under current groundwater flow conditions, groundwater in the Shallow, Intermediate, and Deep 1 Zones at the northern Site boundary, with few exceptions, is from the fields north of the Site toward the Site and not from the Site towards the fields, which limits potential migration in these zones. Limited migration will occur in the deeper aquifer zones in the future, but rate of expansion of the plume of MIW in these zones will be very slow. The MIW plume stability is the result of very slow groundwater velocities and attenuation by dilution.

North Study Area

Groundwater quality in the NSA has been impacted by agricultural activities rather than mining activities (BC, 2016b) based on multiple LOEs including groundwater flow patterns, chemical distributions, groundwater age estimates, hydrologic tracers, and sulfur isotopes. These agricultural practices have resulted in concentrations of sulfate and uranium that are elevated above background values and/or MCLs and/or exhibit increasing trends.

Monitor well data from the NSA indicate that concentrations of nitrate, sulfate, calcium, uranium, and alkalinity in alluvial groundwater in the Shallow through Deep 2 Zones increase along the flow path beneath the agricultural fields. Increases in nitrate are associated with fertilizer application on crop fields. Increases in sulfate and calcium concentrations are associated with application of gypsum ($CaSO_4 \cdot 2H_2O$) as a soil amendment on crop fields, which dissolve in irrigation water that percolates down to the water table. Increases in uranium concentrations are associated with crop irrigation. Percolation of irrigation water through soils increases alkalinity in the soil moisture, which mobilizes and desorbs naturally-occurring uranium from sediments (as soluble complexes with sulfate, carbonate plus, or minus calcium) resulting in elevated uranium concentrations in groundwater (Jurgens et al., 2010; Brown et al., 2007). Alluvial sediments in Mason Valley contain naturally-occurring uranium (BC, 2009b). These issues are discussed further in Appendix M, pp. 21-22. The impact of agricultural activities on uranium mobility in NSA groundwater within or near irrigated land is illustrated by the strong correlation between uranium, alkalinity, and calcium. Increases in alkalinity and calcium are associated with increases in uranium concentrations that can exceed 100 µg/L.

8 RISK EVALUATION

The purpose of the RI is to define and delineate the nature and extent of Site-related impacts. The Risk Assessment identifies sources, pathways of exposure, and current and reasonably likely future receptors. In a groundwater technical meeting held on June 30, 2016 attended by ARC, EPA, and other stakeholders, EPA directed ARC to proceed with preparation and submittal of this OU-1 FRIR without including the risk characterization. The IAOC has the same requirement. The risk characterization will be provided in the OU-1 Baseline Human Health Risk Assessment.

A screening-level ecological risk assessment (SLERA) may not be required for OU-1 because potential ecological receptor exposure pathways to groundwater from within the Low Confidence MIW area may be deemed incomplete or insignificant. Results of the OU-1 RI confirm that groundwater within the Low Confidence MIW area is too deep to be encountered by burrowing animals or other ecological receptors, and groundwater is not expressed at the ground surface at any location in this area. Nor do any agricultural irrigation wells draw groundwater from within Low Confidence MIW area. To the extent groundwater from within the Low Confidence MIW area continues to be used for non-potable, residential lawn and garden watering, such use is limited to a relatively small number of properties and likely represents an incomplete or insignificant seasonal exposure pathway for a limited number of ecological receptors that may pass through or forage intermittently on residential properties within their home range . The OU-1 Human Health Risk Assessment Work Plan will provide additional information explaining why preparation of a SLERA may not be necessary due to the lack of complete and significant ecological exposure pathways. As the HHRA and ecological risk CSM is completed during the development of the HHRA Work Plan, NDEP will evaluate the potential ecological receptor risks and pathways at that time and determine if a SLERA will be required.

This OU-1 FRIR, in conjunction with the OU-1 HHRA, will provide the basis for ARC to identify RAOs and potential remedies for future FS scoping discussions.

9 CONCLUSIONS

The FRIR fulfils the study elements and DQOs presented in the Revised Groundwater RI Work Plan (BC, 2014a), and thereby completes the RI activities for OU-1. Human health risks will be addressed comprehensively in the OU-1 HHRA Report[12].

The FRIR presents two hypotheses of the extent of MIW in the Shallow Zone: 1) the High Confidence MIW Area Shallow Zone and 2) the Low Confidence MIW Area Shallow Zone. Hypotheses of the extent of MIW in the Shallow Zone are assembled using mutually supporting LOEs. One hypothesis integrates five mutually supportive LOEs into a composite evaluation of MIW (using MSA, sulfur isotopes, the 50% probability of groundwater travel distance, age dating and nitrate LOEs and excluding the collocation of MIW indicators LOE). The extent defined by this hypothesis is termed the High Confidence MIW Area Shallow Zone. There is a high probability it contains MIW from the ACMS. The other hypothesis integrates two mutually supportive LOEs into a composite evaluation of MIW extent (using the conservative 5% probability of groundwater travel distance LOE and the collocation of MIW indicators LOE). The extent defined by this hypothesis is termed the Low Confidence MIW Area Shallow Zone. There is a lower probability that it contains MIW, and it is more likely influenced by other sources. There is only one hypothesis for the extent of MIW in the Intermediate through Deep 5 Zones based on five LOEs (MSA, sulfur isotopes, groundwater travel distance, age dating, and nitrate). The extent defined by this hypothesis in each of these zones is termed MIW.

The Low Confidence MIW Area Shallow Zone and the MIW Areas in the deeper zones will primarily be used in risk assessment with additional context provided by the COIs in the High

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[12] Please see Section 8.0 for discussion of the SLERA.

Confidence MIW Area Shallow Zone that will be presented in the HHRA Work Plan. All three MIW areas will be considered in the implementation and evaluation of future groundwater monitoring programs. Both High and Low Confidence MIW Areas in the Shallow Zone and the MIW Area in the deeper zones will be considered in the remedial alternative analysis, as further defined in the FS.

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