



**Mine Unit 1
Alternate Concentration Limit (ACL) Application
Smith Ranch-Highland Mine**

**Cameco Resources
PO Box 1210
Glenrock, Wyoming 82637**

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Executive Summary

Cameco Resources is requesting a license amendment to establish site-specific Alternate Concentration Limits (ACLs) of 51 mg/L for dissolved uranium, 0.18 mg/L for dissolved arsenic, and 0.2 mg/L for dissolved selenium for Mine Unit 1 (MU1). Dissolved radium concentrations are significantly lower than before In Situ Recovery (ISR) mining, and dissolved selenium concentrations are below the Environmental Protection Agency (EPA) drinking water standard of 0.050 mg/L for all 19 restoration zone monitor wells as of the latest round of stability sampling in March, 2017. However, well B13 tested at 0.021 mg/L for dissolved selenium, which is above the NRC maximum value of 0.010 mg/L for that constituent as described in NRC 10 CFR, Appendix A to Part 40, criterion 5c. Hence only Ra does not require an ACL.

Transport modelling indicates that both dissolved uranium, arsenic, and selenium concentrations in certain limited areas of MU1 will increase in concentration over a time scale of centuries, but little change in these concentrations will occur in the first several decades after closure. Future increases in uranium and arsenic concentrations are predicted because of a large sorption sink of ferric solids within the restoration zone. Significant quantities of ferric solids result from the oxidation of naturally-occurring pyrite during mining, though some ferric solids and other sorption media (such as clays or organics) are naturally-occurring in the ore body. The waters of MU1 were not potable before ISR mining was undertaken and remain so.

A monitoring period of three years is proposed for MU1. All of the “B” wells and the down-gradient monitoring ring (wells M1-10 and M24-25) will be sampled twice per year for dissolved uranium, chloride, selenium, and arsenic. Corrective action alternatives are discussed and spot reverse osmosis treatment is suggested in the unlikely event of an exceedance of an ACL.

As Low As Reasonably Achievable (ALARA) scenarios and calculations are presented that show over \$30,000 has been spent per person-rem averted over the entire mine unit up to the present. A future spot-cleaning scenario demonstrates that an additional \$17,000 per person-rem avoided would need to be spent if restoration were to continue. Both past and future expenditures greatly exceed the proposed NRC limit of \$5,100 per person-rem avoided.

Transport modelling demonstrates that no constituents of concern will reach the aquifer exemption boundary, which is the Point of Exposure (POE). The mass balance of available pyrite and other potential reducing agents or sorption sinks is much greater than the amounts needed to immobilize any dissolved uranium, arsenic or selenium. Therefore, there is a large margin of safety built into the modelling that provides further assurance that down-gradient groundwater will be protected.

Because Cameco Resources has shown that the proposed ACLs are ALARA, after considering practicable corrective actions, and that neither uranium nor arsenic nor selenium will pose a substantial present or potential hazard to human health or the environment as long as the alternate concentration limits are not exceeded, the NRC should grant the license amendment establishing the site-specific ACLs for these constituents.

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1. General Information

1.1 Introduction

1.1.1 Mine Unit 1 (MU1) began producing uranium in June, 1997 as part of the Smith Ranch ISR mine. Production continued until September, 2006 when the mine unit entered restoration. Sampling to determine the geochemical stability of MU1 started in September, 2014. All pumping of the mine unit other than that necessary to obtain samples of water for laboratory analysis from individual wells ceased before this stability period began. Total uranium recovery (including both production and restoration phases of operation) from the mine unit was about 980,000 pounds expressed as the equivalent amount of U_3O_8 .

1.1.2 Full geochemical analyses (the “Guideline 8” suite of tests) taken for the “B” wells when MU1 entered restoration could not be located, so concentrations for constituents of concern (uranium, arsenic, selenium, etc.) start at dates later than September, 2006. For example, the earliest dissolved uranium concentrations used for this report are from December, 2009. The impact of the missing data is negligible since the ALARA calculations described in Section 5 of this report show the cost per person-rem avoided far exceeds that needed to justify termination of restoration efforts.

1.2 Facility Description

1.2.1 To mine, water containing dissolved uranium was pumped from the production wells of MU1 to the Central Processing Plant (CPP). There the uranium was extracted as it passed through columns containing ion-exchange (IX) resin. After extraction the water was fortified with dissolved oxygen and carbon dioxide before being re-injected into the aquifer. The only chemicals added to the groundwater during mining were these dissolved gases and 2 moles of chloride emitted from the IX resin for every mole of uranium recovered. The process is closed-loop with the exception of an approximately 1% bleed so that the same water is repeatedly produced, processed, re-fortified and re-injected. The bleed water creates a cone of depression ensuring that the mining solution (known as “lixiviant”) does not migrate from the production zone.

1.2.2 When full, the IX resin was stripped of uranium (eluted) using an aqueous solution containing sodium chloride and sodium carbonate. The resulting uranium-bearing solution was treated with sulfuric acid to remove the carbonate, and hydrogen peroxide was then added to precipitate uranyl peroxide (“yellow cake”). In addition to the added sulfuric acid, acid was generated as a side reaction during the precipitation, and this was neutralized using ammonium hydroxide (until 2012) or sodium hydroxide (used from 2012 to the present). These chemicals (other than the two moles of chloride per mole of uranium described in the preceding paragraph) were sent to a deep disposal well after being used to process the product and thus never reached the mine unit. Figure 1 illustrates the steps used in processing uranium at the CPP.

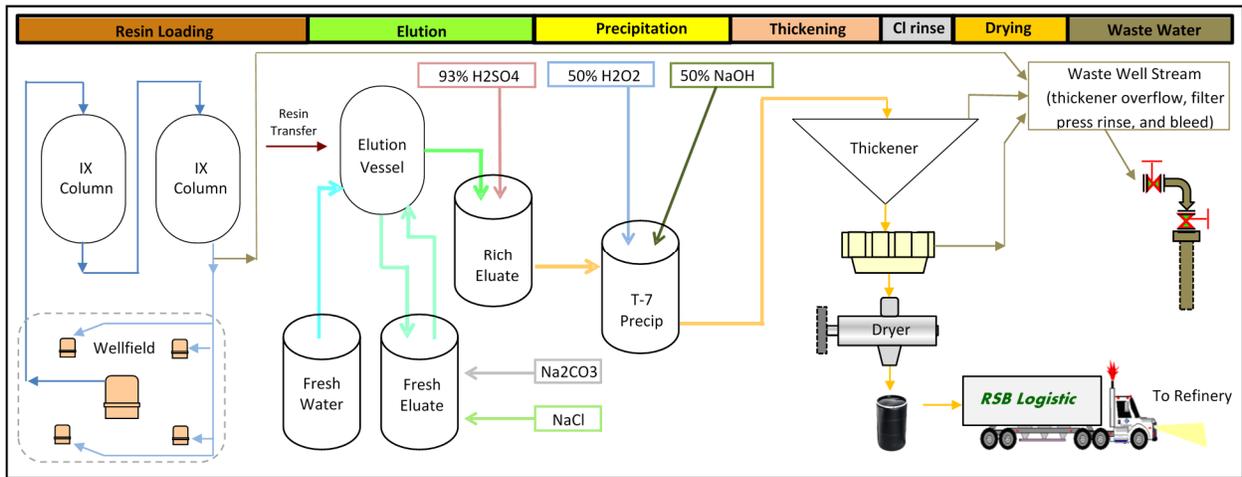


Figure 1. A schematic showing the process from uranium extraction in Mine Unit 1 to the making of ore concentrate in the Central Processing Plant (CPP). During production the only chemical that emanates from the CPP to the mine unit are two moles of chloride given up by the ion-exchange resin for each mole of uranium captured as discussed in 1.2.2.

1.2.3 MU1 itself is geographically isolated from mine units other than MU2, which lies immediately to the northwest of it (see Figure 2A/2B). The aquifer exemption boundary extends 150 feet beyond the ring monitor wells, and the transport modelling done for this ACL application terminated at this boundary (Appendix C). In other words, the aquifer exemption boundary was taken to be the point of exposure (POE) for every modeled flow path. The Points of Compliance are the 19 "B" monitor wells in the mined and restored zone. MU1 is completed in the Q-Sand while MU2 is in the O-Sand (see USEPA Region 8 letter dated Mar 27, 2014 to Mr. Kevin Frederick of the Wyoming DEQ/WQD, Ref 8P-W-UIC).

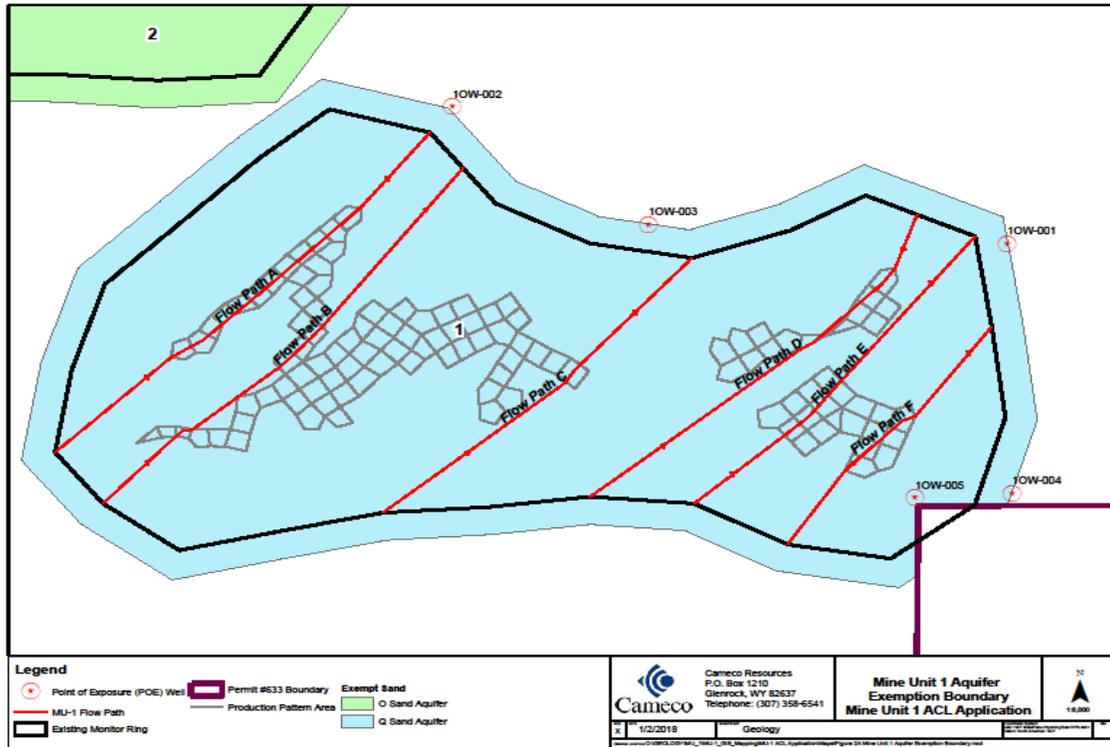


Figure 2A. Mine Unit 1 layout indicating Monitor Ring (black), Aquifer Exemption Boundary (shaded blue), Transport Modeling Flow Paths (red "A-F"), and Observation Wells ("1OW").

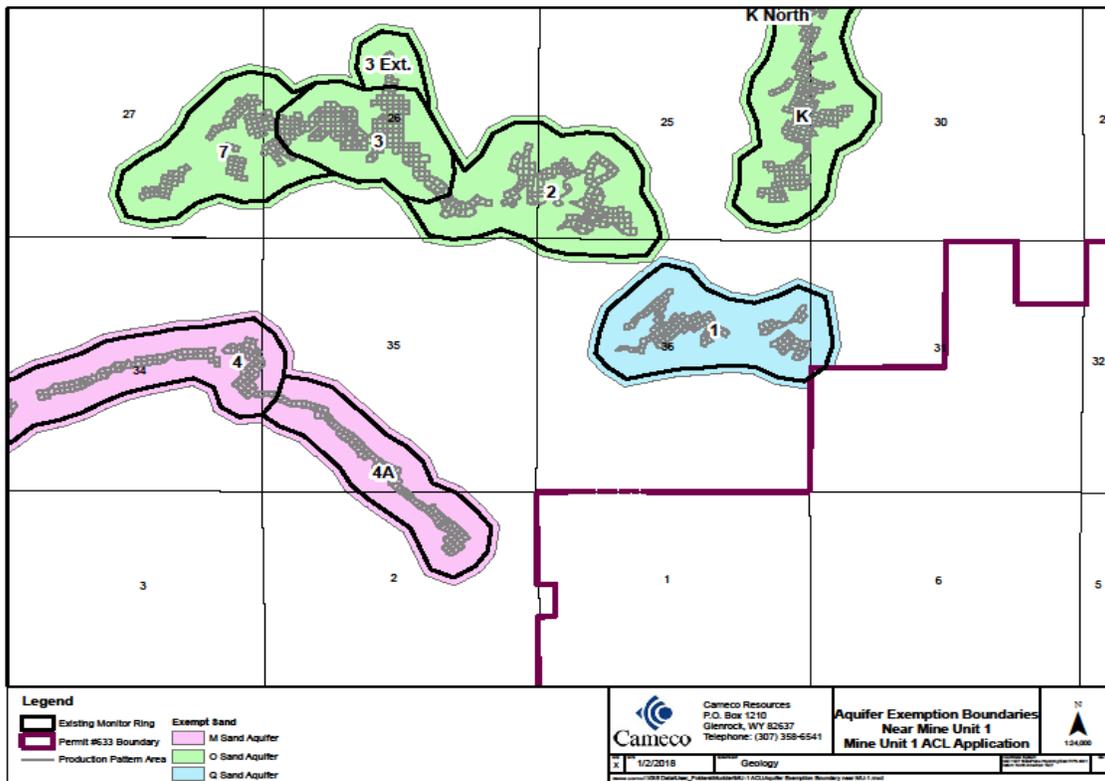


Figure 2B. Mine Unit Aquifer Exemption Boundaries relative to Mine Unit 1.

1.3 Extent of Groundwater Contamination

1.3.1 The surface area of MU1 within the ring monitor wells is 173 acres, and the pore volume of the Q-Sand groundwater is about 62.8 million gallons as described in the license to mine. This volume includes a flare factor of 1.56. All wells in MU1 have single-completion intervals into the Q-Sand at depths beneath the ground surface ranging from 421 ft to 561 ft. No excursion was ever observed at any of the ring monitor wells nor the underlying and overlying monitor wells during the entire operation of MU1 from the beginning of ISR mining through stability monitoring.

1.4 Current Groundwater Protection Standards

1.4.1 The pre-mining baseline water chemistry data are in the attached workbook entitled "**MineUnit1BaselineData.xls**". The waters for all 19 B wells (monitor wells completed in the mining zone) were not potable before mining because of high radium-226 activity concentrations. The radium at MU1 (and indeed all mine units at the Smith Ranch mine) is primarily radium-226. Radium-228 activity concentrations are typically two orders of magnitude lower than those for the lighter isotope. Well B4 had the lowest activity concentration at 15 pCi/L, or three times the drinking water standard of 5 pCi/L. All of the remaining 18 wells tested at much higher values ranging from a few hundred pCi/L to significantly over 1000 pCi/L. In addition to high radium activity concentrations, most of the B wells exceeded the EPA drinking water standard of 0.030 mg/L for natural isotopic abundance uranium (Unat). Only wells B16 and B17 had baseline uranium concentrations below 0.030 mg/L. Other elements of possible interest, such as arsenic and selenium, were not present in concentrations above drinking water standards for any of the B wells (drinking water standards of 0.050 mg/L for selenium and 0.010 mg/L for arsenic).¹

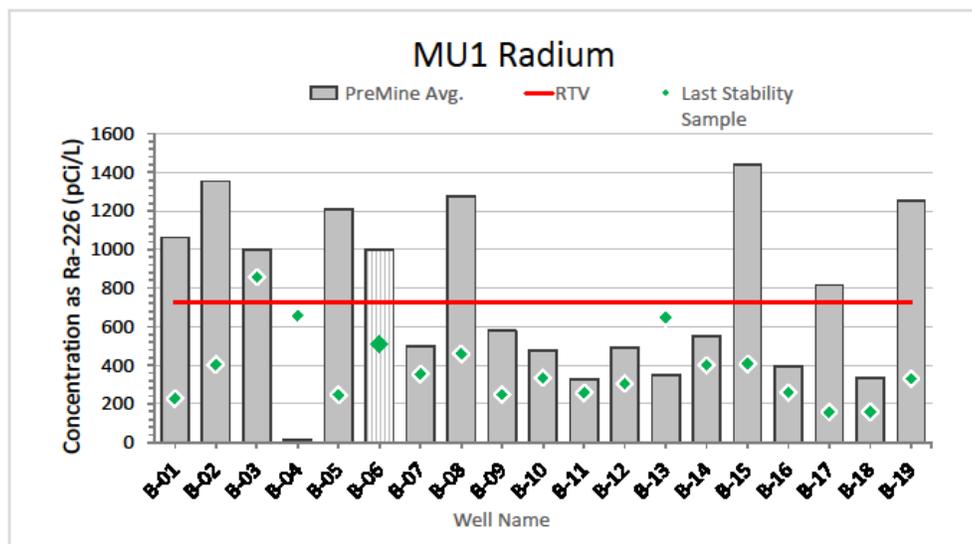


Figure 3A. Mine Unit 1 Radium-226 Pre-Mining concentrations, RTV and stability values from the last round of sampling in March, 2017

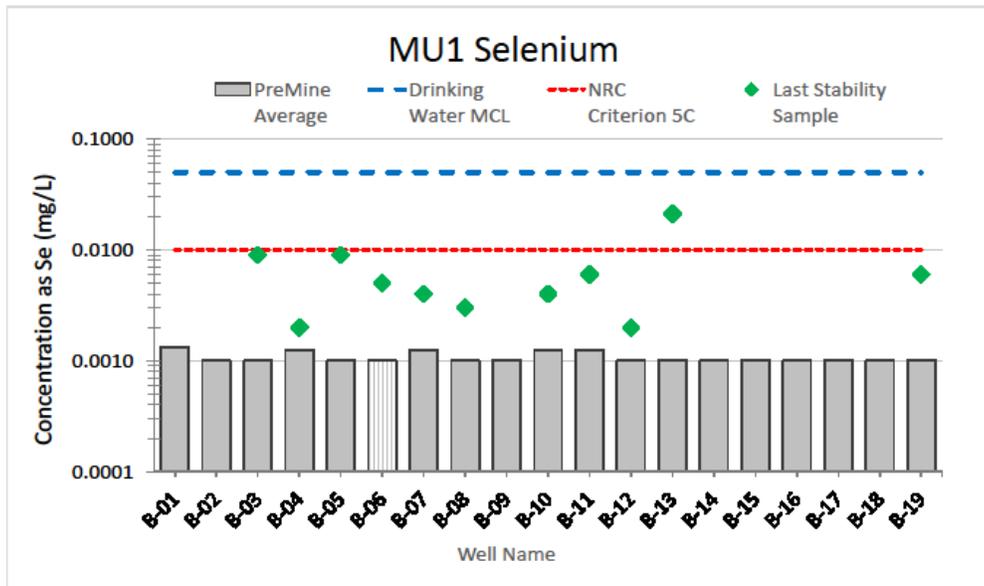


Figure 3B. Mine Unit 1 Selenium Pre-Mining concentrations, Drinking Water MCL (0.05 mg/l), RTV and stability values from the last round of sampling in March, 2017. Note that the laboratory detection limit for Selenium is 0.001 mg/L and some wells were below the detection limit at the last sample.

1.4.2 During the application process for mining, well B6 was conservatively excluded from data sets for the calculation of restoration target values (RTVs) for all elements of concern. No regulatory correspondence concerning the well was found. It was sampled four times over a period of about two-and-a-half months in late 1996 and consistently tested at concentrations of 5 to 6 mg/L of dissolved uranium. That is approximately two orders of magnitude greater than was typical for the other B wells.

1.4.3 Years before the installation of MU1 a Q-Sand pilot project was conducted in the vicinity of B6 by Rio Algom Mining Corporation (see Figure 4 for a map of the pilot patterns overlain on MU1 and see Attachment 1 for the pilot project report dated October, 1992). This is the same sand in which MU1 was completed. The proximity of B6 to this Q-Sand project strongly suggests that the outlier uranium concentrations observed at this well during the establishment of baseline were a consequence of the pilot project. It should be noted that as of the last round of post-restoration sampling in March, 2017, the waters of B6 contain less than 0.4 mg/L of dissolved uranium. No impact of the pilot project is evident since the area around B6 was restored.

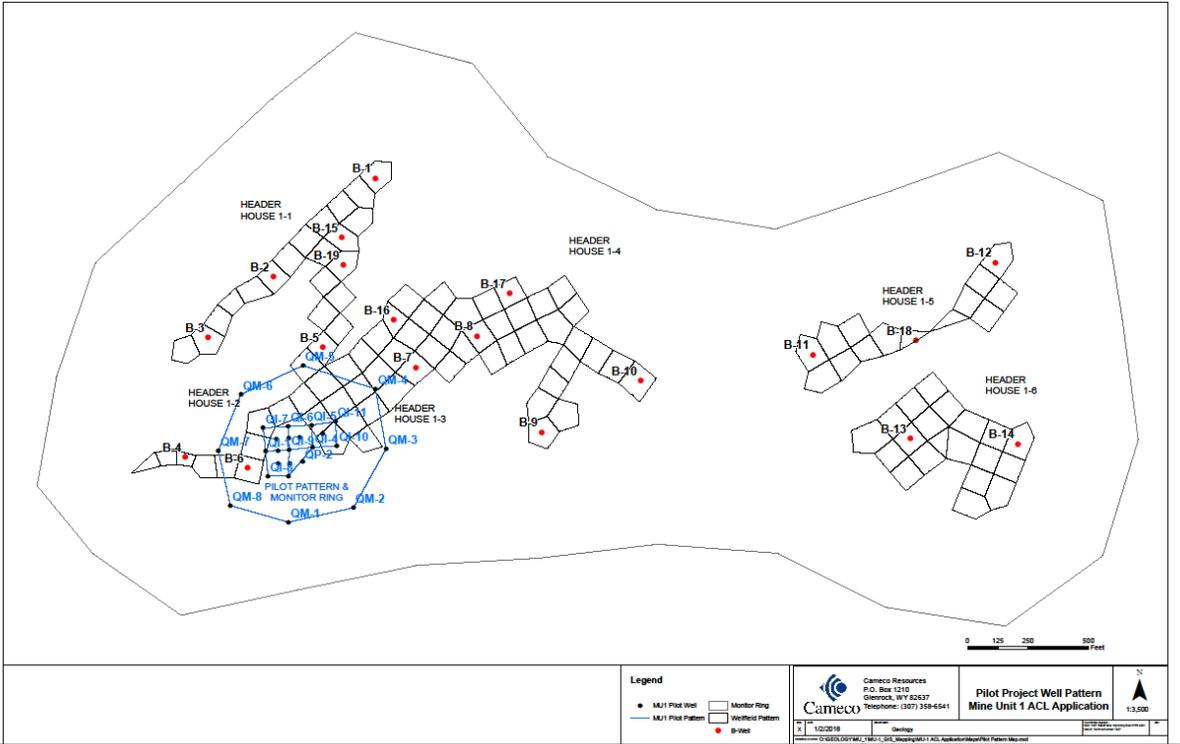


Figure 4. Q-Sand pilot project relative to the Mine Unit 1 (Q-Sand) wellfield.

1.4.4 In the 1970s an underground uranium mine operated in the O-Sand some 300 feet deeper than the Q-Sand (see Figure 5). While all of the B wells showing statistically significant increasing trends for dissolved uranium concentrations are in the vicinity of the underground mine, it is unlikely that it has any impact on MU1 water quality given the large difference in depth and an intervening shale unit.

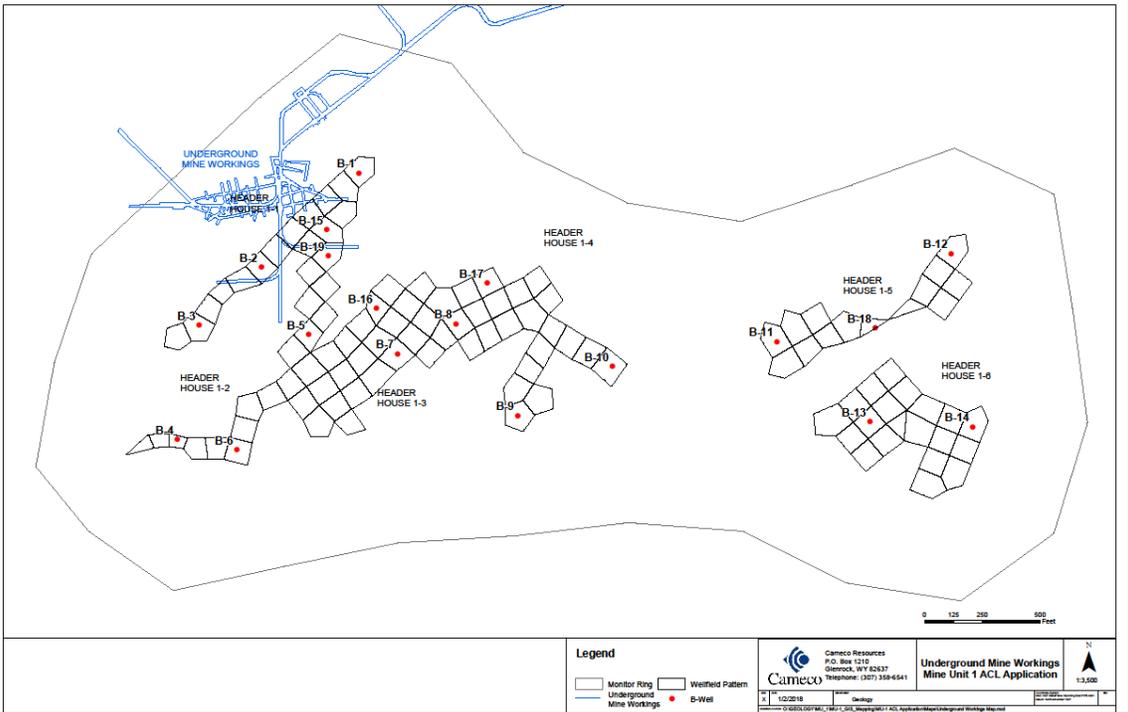


Figure 5. Previous underground mining location relative to Mine Unit 1 in-situ leach.

1.5 Proposed Alternate Concentration Limits

1.5.1 Cameco will propose ACLs for uranium, arsenic, and selenium with the actual concentrations and rationales for those concentrations presented in paragraph 5.1 below. The proposed ACL for dissolved uranium is 51 mg/L while that for dissolved arsenic is 0.18 mg/L and dissolved selenium is 0.20 mg/L. Arsenic, iron, manganese, radium, selenium, and uranium were included in the transport model (Appendix C). Radium activity concentrations are broadly lower than was the case before ISR mining having decreased from a median value of 588 pCi/L in the last round of baseline sampling to 360 pCi/L at the March, 2017 stability monitoring point (see Figure 3A above for well-by-well comparisons of pre-mining and stability radium activity concentrations). In addition, transport modeling results demonstrate that radium concentrations at the POE locations will be unaffected by MU1 groundwater concentrations. Under the prevailing geochemical conditions in the restored zone, dissolved iron and manganese are present in the ferrous and manganous states at higher concentrations than was the case before ISR mining. However, iron and manganese are of such low toxicity that they have only secondary maximum contaminant levels (secondary MCLs). Furthermore, ferrous iron is capable of reducing sorbed or dissolved uranium under geochemical conditions likely to occur within the aquifer exemption boundary.²⁻⁴ Selenium concentrations in the restored zone currently are all below the EPA drinking water standard with one well (B13) above criterion 5c as mentioned in the executive summary for this document. Transport modeling predicts that concentrations will remain very low within MU1, and selenium will not be transported to the POE locations.

2. Hazard Assessment

2.1 Source and Contamination Characterization

2.1.1 General

2.1.1.1 As described in Section 1.2.3, MU1 is situated in relative isolation from other mine units. The aquifer exemption boundary is 150 feet beyond the ring monitor wells, but actually extends within the monitor ring in the southeastern portion of MU1 (see the maps of Figure 2).

2.1.1.2 Five observation wells were drilled and completed as part of the source characterization, and their locations are shown on the map of Figure 2A. The wells were sampled using 4" diameter Hydrasleeve™ containers suspended from the boom of a data-logging truck. All of the observation wells, including 10W-005 at the corner of the aquifer exemption boundary that is within the monitor ring, showed no evidence of contamination from mining. The laboratory results for these samples are included in Attachment 2 of this report.

2.1.1.3 Pre-mining, baseline water quality data can be found in the workbook "**MineUnit1BaselineData.xls**" included with this report. Attachment 3 includes all of the laboratory reports available for the stability phase that began in September, 2014. Also included for examination is the workbook "**MU-1 Ground Quality Summary.xlsx**". That workbook has the stability phase data arranged in a fashion that makes for easy input into statistical packages such as the EPA program ProUCL for data visualization and analysis.

2.1.2 Geology

2.1.2.1 The uranium host production zone in MU1 is designated as the Q-Sand of the Paleocene-age upper Fort Union Formation. The Q-Sand is one unit of the site specific stratigraphic nomenclature developed for the Smith Ranch-Highland Uranium Project (SRH). The Q-Sand consists of fine- to very

coarse-grained, poorly to moderately sorted arkosic sandstone, which was deposited in a fluvial environment. The Q-Sand is continuous across MU1 and varies in thickness from 10 to 75 feet, with an average thickness of 37 feet. The R-Shale overlies the Q-Sand and the P-Shale underlies the Q; both shale units are composed of bentonitic claystone, fissile shales and siltstone with some discontinuous sand lenses and minor lignitic stringers. The R-Shale and P-Shale vary in thickness across the mine unit but are present throughout MU1. The R-Shale separates and hydrologically confines the Q-Sand from the overlying S-Sand while the P-Shale separates and confines the Q-Sand from the underlying O-Sand. Regional and site specific stratigraphic charts are presented in Figure A-3 in Appendix A.

2.1.2.2 Although the Q-Sand is continuous across the MU1 wellfield, it thins or “pinches out” in areas outside of the mine unit, and the P and R-Shales subsequently coalesce in nearby mining areas. The discontinuous nature of the production sand in the vicinity of MU1 can be observed in geologic cross-sections and in the Q-Sand isopach map (Figures A-6 through A-10, Appendix A).

2.1.2.3 The subsurface strata at MU1 dips at a low angle toward the north. Structural cross sections of MU1 (Figures A-13 through A-15, Appendix A) do not indicate significant folding or the presence of bedding displacement by faulting. Evidence of faulting was not noted in the correlation of hundreds of geophysical logs during the resource evaluation and wellfield planning stages of MU1. Faulting or a fault related fracture system was not detected during pre-mining pumping tests performed at MU1. The aquifer testing demonstrated a lack of hydrologic communication between the production sand aquifer and the underlying aquifer (O-Sand) and the overlying aquifer (S-Sand). The lack of hydrologic communication between aquifers indicates the absence of vertical preferential flow paths through the P-Shale and R-Shale confining units. Additionally, natural fracturing was not observed in tightly cemented sandstone or well-indurated shale core samples recovered during the recent coring program at MU1. Petrographic work conducted on core from MU1 did not indicate the presence of micro-fracturing in the samples analyzed.

2.1.2.4 The roll-front type uranium ore deposits at SRH and MU1 are found at the interface of naturally occurring chemical boundaries between reduced and oxidized sandstone facies. The mineralization occurs in several vertically superimposed (stacked) subsidiary roll fronts or sub-rolls, caused by small permeability differences in the sandstone. Seven sub-rolls were identified in the Q-Sand roll front system of MU1 and are designated, from shallowest to deepest, the Orange, Green, Red, Blue, Pink and Purple horizons or mineralized zones (Figure A-16 Appendix A).

2.1.2.5 The number of mineralized zones increases towards the eastern side of the mine unit which can be attributed to the increasing thickness of the Q-Sand. The general orientation of the roll-front system indicates that groundwater flow at the time of the formation of the sub-rolls was south-southwest to north-northwest (Figure A-29 Appendix A).

2.1.3 Hydrology

2.1.3.1 MU1 is underlain by a sequence of permeable sands separated by lower permeability claystone, siltstone and shale confining units of the Wasatch and Fort Union Formations. Aquifers of interest include the Q-Sand (production aquifer), O-Sand (underlying aquifer), and the S-Sand (overlying aquifer) of the Fort Union Formation. As previously described, the Q-Sand is separated from the underlying O-Sand by the P-Shale confining unit, and from the overlying S-Sand by the R-Shale confining unit. Groundwater in the Q-Sand aquifer is recharged pre-dominantly by infiltration of precipitation in outcrop areas west of MU1, and also by limited downward flow of groundwater from the overlying confining unit (R-Shale). Natural groundwater discharge occurs by evapotranspiration and seasonal discharge to ephemeral tributaries of the Cheyenne River drainage northeast of the SRH permit area, and also by limited downward flow of groundwater from the Q-Sand aquifer into the underlying P-Shale. The approximate distance from MU1 to the Q-Sand outcropping ranges from about 6.5 to 8.5 miles to the

northeast. Discharge from the Q-Sand aquifer also occurs through wells operated as part of ISR production, restoration and monitoring activities, and historically as part of the Bill Smith underground mine dewatering program.

2.1.3.2 The regional extent of the Q-Sand aquifer at SRH is illustrated in Appendix B, Figure B-6, as interpreted through correlation of well and borehole logs, and by projection of the Q-Sand bottom elevation into outcrop locations northeast and west of SRH, and in the southwestern portion of the permit area. The Q-Sand is considered to be relatively continuous and hydraulically connected over the area defined in Figure B-6. The Q-Sand is not present south or northwest of the permit area, or in the eastern portion of the Highland property, and is not hydraulically connected to the Highland Pit Lake or underground mines in the eastern portion of the permit area. Locally, the Q-Sand pinches out and is not present over portions of Mine Unit 2 and Mine Unit 3 northwest of MU1 as described above.

2.1.3.3 The hydraulic conductivity of the Q-Sand aquifer calculated from aquifer test analyses ranges from 1.4 to 4.4 ft/day, with an average hydraulic conductivity of 3.0 ft/day deemed representative of the Q-Sand aquifer in MU1. The storage coefficient calculated from MU1 aquifer test analyses varies from 2.4×10^{-5} to 1.3×10^{-4} , with an average storage coefficient of 5.0×10^{-5} deemed representative of the Q-Sand aquifer in MU1. Core samples collected from the Q-Sand aquifer in MU1 consist primarily of poorly consolidated, moderately to well-sorted, medium- to coarse-grained sand with less than 10 percent silt and clay. Total porosity ranges from 25.3 to 33.2 percent, with an average total porosity of 30 percent. Effective porosity ranges from 19.4 to 28.5 percent, with an average effective porosity of 24 percent. Groundwater currently flows northeast across MU1 with an average hydraulic gradient of 0.003.

2.1.3.4 Overlying and underlying confining units (R-Shale and P-Shale) are continuous across MU1 and over the extent of the Q-Sand aquifer in the permit area. The vertical hydraulic conductivity of confining units at SRH calculated from laboratory core testing ranges from 2.6×10^{-7} to 3.8×10^{-5} ft/day. Vertical hydraulic conductivity calculated from Neuman-Witherspoon analysis of aquifer test data ranges from 1.1×10^{-5} to 2.8×10^{-4} ft/day.

2.1.3.5 Over the last 45 years, groundwater flow in the vicinity of MU1 has been influenced significantly by ISR mining and restoration activities, and by the dewatering and recovery of the underground mine. Groundwater flow conditions in the vicinity of MU1 are currently influenced by regional aquifer recovery and pre-restoration groundwater withdrawals in Mine Unit J northeast of MU1 (Appendix B Figure B-2). Significant recovery of the Q-Sand aquifer is expected following the cessation of restoration activities in Mine Unit J in approximately 2023.

2.1.3.6 A groundwater flow model of the Q-Sand was constructed and calibrated to observed historical flow conditions and aquifer test drawdown in order to project future groundwater conditions in MU1. The calibrated model was run forward in time to include the Mine Unit J restoration period (2018 to 2023). Water levels in the Q-Sand aquifer in MU1 are projected to rise by approximately 74 feet from levels observed in March, 2017, and they are projected to reach steady-state (equilibrium) in approximately 50 to 70 years. The computed, current groundwater velocity in the Q-Sand aquifer ranges from 7 to 14 ft/yr in MU1, which is approximately four times greater than the projected steady-state groundwater velocity of from 1.8 to 5.5 ft/yr. This difference can be attributed to a significant decrease in hydraulic gradient across the mine unit as the aquifer recovers. Following complete recovery of the aquifer system, the computed groundwater flow direction is essentially the same as the historic and current flow direction (see Section 6.5.2 of Appendix B).

2.1.3.7 Additional information about the regional hydrology and further details of site hydrogeology and groundwater flow conditions is provided in Appendix B.

2.1.4 Characterization of Core Materials

2.1.4.1 A total of eleven core holes were drilled and samples taken in MU1 to provide data for the transport modelling described in Section 3.1 of this cover document. A detailed discussion of the methods of core analysis can be found in Section 3.0 of Appendix C (“Geochemical Evaluation and Groundwater Transport Model for the Smith Ranch-Highland Uranium Project Mine Unit 1”). Given the abundance of clays in the flow paths through the restoration zone, a significant part of the core characterization effort was the study of their mineralogy using such techniques as X-Ray Diffraction (XRD), Energy Dispersive Spectroscopy (EDS), and Scanning Electron Microscopy (SEM). Equally important were a series of sequential wet chemical extractions performed on portions of core. These shed light on the ion-exchange properties of aquifer sediments as well as the ability of various core components to act as sorption sinks for radio-isotopes and other species of interest.⁵ Ferric solids can act as a sorption sink and are generated by the reaction of dissolved oxygen added for mining with naturally occurring pyrite found in the ore body. Aquifer sediments and organic materials can also sorb significant amounts of uranium. A large fraction of the oxidized uranium in the aquifer (i.e., uranium(VI)) was found to be adsorbed on ferric solids and clays, and the implications of this are discussed in Sections 2.1.5 and 4 of this report. Arsenic was also found to have a strong affinity for adsorption to ferric solids.

2.1.5 Stability

2.1.5.1 A detailed stability report concerning MU1 was submitted to the Nuclear Regulatory Commission (NRC) and Wyoming Department of Environmental Quality (WDEQ) (dated 24 July 2017). That report was discussed in a 4 October 2017 public meeting that included staff from the NRC, WDEQ, and Cameco Resources. At that meeting, NRC staff advised that inter-well averaging for a given constituent of concern is not appropriate unless each individual restoration zone monitor well data set (i.e., intra-well data set) for that constituent of concern is stable. Note that a well with a statistically significant *decreasing* trend (SSD) is unstable just as is one with an increasing trend (SSI).

2.1.5.2 The intra-well data sets for all of the major ions (including uranium and arsenic) were analyzed using the EPA statistics program ProUCL for all 19 “B” wells as part of that stability report. Trending was done using the Mann-Kendall method. Table 1 below shows the results for arsenic, radium, selenium, and uranium.

Well Name	Arsenic Stability	Radium Stability	Selenium Stability	Uranium Stability
B-01	0	+1	0	+1
B-02	0	0	0	-1
B-03	0	0	0	+1
B-04	0	0	0	0
B-05	0	+1	0	+1
B-06	0	0	-1	-1
B-07	0	0	0	0
B-08	-1	0	0	0
B-09	0	0	-1	-1
B-10	-1	0	-1	0
B-11	0	+1	0	0
B-12	0	0	-1	0
B-13	0	0	0	0
B-14	0	0	0	0
B-15	0	-1	0	0
B-16	0	0	0	-1
B-17	0	0	0	0
B-18	0	0	0	-1
B-19	0	0	0	+1

Table 1. Trends for parameters of importance on an individual B-well basis. +1 implies an upward trend (SSI), 0 implies no statistically significant trend, and -1 indicates a downward trend (SSD). All results include data through the March 7-8, 2017 sampling period and were determined using Mann-Kendall as implemented in ProUCLVersion 5.0 at 95% confidence.

For the case of uranium, four wells (B1/3/5/19) have SSIs, while five wells (B2/6/9/16/18) display SSDs. The slopes of these data sets were estimated using the Thiel-Sen method included in ProUCL, and the results are found in Table 2.

Well	Thiel-Sen Slope (mg/L/day)	Upper Confidence Limit	Lower Confidence Limit
B1	+6.0959E-5	+9.8576E-5	+1.2159E-5
B2	-1.78E-3	-5.129E-4	-3.87E-3
B3	+8.8575E-4	+2.4176E-4	+1.7407E-4
B5	+1.5187E-4	+2.8739E-4	-5.768E-5
B6	-1.17E-3	-4.047E-4	-2.21E-3
B9	-3.880E-4	-3.971E-5	-1.86E-3
B16	-1.264E-4	-3.758E-6	-3.486E-4
B18	-1.859E-4	-7.638E-5	-5.154E-4
B19	+8.8960E-4	+1.24E-3	+3.3388E-4
SUM	-1.66E-3		

Table 2. Dissolved uranium Thiel-Sen analysis slopes for wells showing an SSI or SSD for uranium.

Note that the sum of the Thiel-Sen slopes in Table 2 is a negative number, and is negative by a wide margin when compared to the magnitude of any of the individual slopes that are positive. This suggests that the overall trend for uranium is downward over the aggregate of the mine unit assuming similar pattern pore volumes.

2.1.5.3 MU1 is not geochemically stable for uranium on an intra-well basis, and the primary reason for this is that a relatively large fraction of uranium(VI) in the restored aquifer is adsorbed on aquifer solids as discussed in Section 2.1.4. Adsorbed uranium(VI) is expected to gradually desorb on a time scale of centuries (see Appendix C, Figure C-5-12, Section 5.3 for a discussion of uranium desorption), and stability is thus unattainable without removing the adsorption sink of ferric solids and clays. Significant amounts of arsenic are also adsorbed to aquifer solids, and arsenic desorption over time is also predicted in portions of MU1 (Appendix C, Section 5.5).

2.2 Constituents of Concern

2.2.1 As discussed in Sections 1.5 and 5.1, this document proposes ACLs for uranium, arsenic, and selenium. As mentioned in Section 1.5.1, radium activity concentrations are less in most parts of MU1 than they were before mining began though radium remains the greatest potential radiological risk posed by drinking the waters of MU1,⁶ as it was before mining. No rise in uranium, arsenic, selenium, or radium concentrations is predicted by the transport model of Appendix C to occur at the aquifer exemption boundary. Elevated dissolved calcium concentrations can increase the mobility of dissolved uranium,⁷⁻⁸ but the concentrations of this element have decreased to levels similar to those observed before ISR mining began. Relatively minor calcium concentration changes are predicted within MU1 by the transport model (Appendix C, Section 5.1). Small increases in concentrations of Total Dissolved Solids (TDS), manganese, and iron are predicted at the aquifer exemption boundary during the thousand-year time window covered by the model of Appendix C (Sections 5.1 and 5.2), but these constituents have only secondary drinking water standards and do not represent a health risk.

2.3 Health and Environmental Risks of Constituents

2.3.1 A detailed description of the environmental risks associated with each constituent of concern for MU1 can be found in Section 2.3 of Appendix E.

3. Exposure Assessment

3.1 Transport and Pathway Assessment

3.1.1 Appendix B, Figure B-6 shows the surface foot print of the Q-Sand as well as the area where the aquifer reaches the surface in the down-gradient direction at an estimated distance of 6.5 to 8.5 miles from MU1. The sand is not hydraulically connected to features such as Mine Unit C or the Exxon pit on the far eastern side of the Smith Ranch mine. Since the transport model of Appendix C predicts attenuation of any residual concentrations of dissolved uranium, arsenic, or selenium before the aquifer exemption boundary is reached, there is no realistic scenario by which such constituents of concern reach surface waters. The only exposure pathway would be via drilling a well into the restoration zone.

3.1.2 Existing wells, including those used for watering livestock, were assessed as a possible pathway for exposure. This was done using records kept by the Wyoming State Engineer's Office (WSEO), and the results are and summarized in the workbook "**MU-1 ACL Well Inventory of Stock and Domestic Wells.xlsx**" that accompanies this application, and the well locations are mapped in Figure 6. All entries from the WSEO database were verified by actually visiting the wells. All wells are included in the workbook that are within 2 km of the MU1 ring monitor wells. All of these wells are shallower than the Q-Sand by at least 100 feet, and all have been sampled on a quarterly basis (when they produce water) for many years under the provisions of the mining license. As expected, no evidence of impact from mining has been observed.

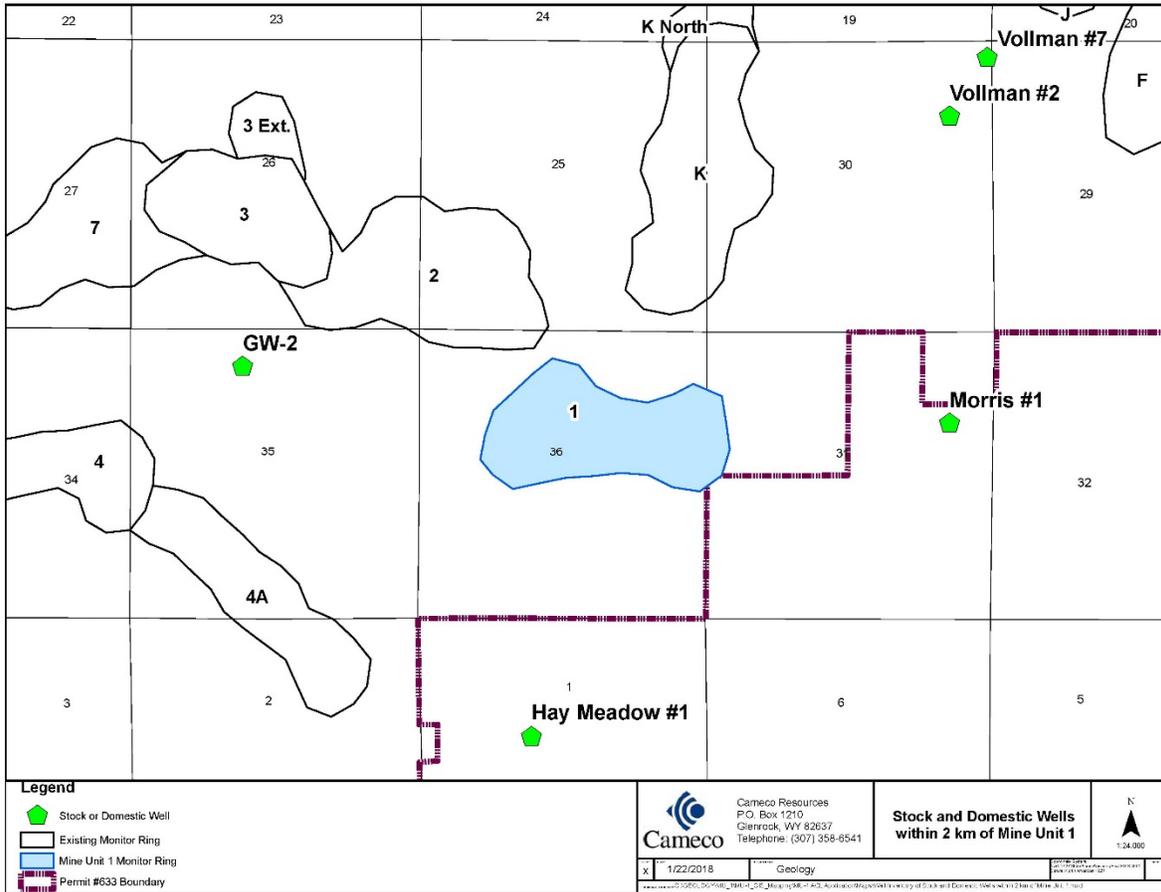


Figure 6 Stock and domestic water well sources within 2 Km of MU1

3.2. Human Exposure Potential (Including a Discussion of Institutional Controls)

3.2.1 A detailed, quantitative radiological and toxicological assessment of the potential for human exposure to uranium, arsenic, selenium, and radium is given in Appendix E, Section 3. While no Alternate Concentration Limit is needed for radium since it is lower in activity concentration post-restoration than it was before ISR mining, its inclusion in the analysis reveals that the cancer morbidity risk from it is still greater than that from uranium. Note that this assumes a dissolved uranium (future) concentration of 51 mg/L, because that is the proposed ACL for uranium as discussed in Section 5 of this report. There is no increased risk caused by consuming the groundwater at the POE since no rise in the concentrations of dissolved uranium or arsenic are predicted to occur there.

3.2.2 As was the case before mining, groundwater from the restoration zone is not potable. Scenarios are discussed in Appendix E whereby impacted water at nominal constituent concentrations (uranium = 40 µg/L; arsenic = 0.001 mg/L; selenium = 0.001 mg/L; Ra-226 = 1.0 Bq/L) is used for watering cattle and irrigating crops followed by human consumption of the beef or crops in addition to a discussion of the risks of humans using the water as a drinking source or bathing with it. The potential dose and risk from using groundwater at the maximum existing constituent concentrations at the Point of Compliance (POC) in MU1 were calculated only for ingestion of cows consuming groundwater followed by consumption of the beef. Given the institutional controls (see Section 3.5 below), other uses of the groundwater at the POC are unlikely. Not surprisingly, the potential for human exposure to the uranium, selenium, and arsenic decreases markedly if the water is used only for food production and not for drinking. While informative, these scenarios are unlikely to occur because groundwater can be found at much shallower depths than the mining zone. That zone is from 420 to 560 feet below the

surface depending upon location in MU1. The stock wells tabulated in the workbook “**MU-1 ACL Well Inventory of Stock and Domestic Wells.xlsx**” are completed in a range from only 32 feet to 200 feet beneath the surface.

3.3 Environmental Exposure Potential

3.3.1 Appendix E Section 5 also includes a discussion of the potential for environmental exposure and assesses the risk to mammals such as deer and antelope should they drink from stock tanks containing water from the restored zone. Since well B4 currently has the highest concentration of dissolved uranium, the median value for uranium concentration for this well during the stability period (3.77 mg/L) was used to assess the impact on wildlife. The analysis indicates there are no significant impacts on animals consuming the water at this concentration. At an arsenic concentration of 0.0275 mg/L, a deer would have to drink 43,000 Liters of water in a day to receive a lethal dose. For flow path A described in Appendix C maximum arsenic concentrations within the restoration zone are predicted to be 0.050 mg/L, so the amount of water a deer must drink to consume a lethal dose of arsenic at peak concentration would be $(0.0275 \text{ mg/L} / 0.05 \text{ mg/L}) \times 43,000 \text{ liters per day} = 23,650 \text{ liters per day}$.

3.4 Consequences of Exposure

3.4.1 Appendix E, Section 3 reviews the potential consequences and risks of exposure to arsenic, selenium, uranium, and radium. The risks of exposure to arsenic are well known and described in Appendix E. The primary risk from consuming dissolved uranium is kidney damage. The radiological hazard from this constituent is relatively small since uranium is only weakly radioactive. Because the half-lives of the two most abundant uranium isotopes (U-235 and U-238) are very long (about 700 million years and 4.5 billion years respectively), the specific activity of natural isotopic abundance uranium is low.

3.4.2 Arsenic is both carcinogenic and toxic, and its toxicity is greater as As^{3+} . It can cause cancers of the lung, skin, and bladder. It is also implicated in elevated rates of such ailments as diabetes, cardiovascular disease, and pulmonary disease. The transport model of Appendix C predicts that arsenite (H_2AsO_3^-) will be the predominant species of arsenic in the restoration zone, and this species implies the As^{3+} oxidation state. It is possible that some of the arsenic in MU1 is present as a bio-methylated arsenic, which is less toxic than inorganic forms of the element, but no testing was done to address this question. Therefore, the risk assessment of Appendix E Section 5.5 is predicated on the assumption that all arsenic is inorganic As^{3+} . This is the worst-case scenario and is thus a conservative estimate.

3.4.3 In humans, selenium is an essential element in trace quantities, however at excessive levels, selenium may be toxic. Selenium is not known to be a carcinogen. If mildly excessive amounts of selenium are eaten over long periods, brittle hair and deformed nails can develop. In extreme cases, people may lose feeling and control in arms and legs. There are no obvious, significant differences in the toxicity of selenate or selenite, the two predominant selenium species in the groundwater for Mine Unit 1.

3.4.4 The principal radiation risk is from dissolved radium (see Section 3.2.5 above), even though that risk is less now than it was before ISR mining. The most common consequence of excessive consumption of dissolved radium is bone cancer.

3.5 Institutional Controls to Limit Exposure Potential

3.5.1 One facet of a discussion about the potential for human and environmental exposure are institutional controls that can be maintained by governmental agencies. The lead agencies in Wyoming are the Wyoming State Engineers Office (WSEO) and the Wyoming Department of Environmental Quality (WDEQ). A meeting was held on December 12, 2017 that included representatives from Cameco Resources, the WSEO, and the WDEQ to discuss this issue.

3.5.2 The WSEO requires that a permit be obtained before any drilling is done. They have the capacity to keep a record of aquifer exemption boundaries and will provide a fact sheet to potential developers of a site indicating the existence of an aquifer exemption boundary. The sheet suggests that the WDEQ be contacted for further information. WSEO will not deny access to groundwater even if it is not potable. The WSEO database shows the locations of abandoned wells, the completion intervals, and the depths of those intervals.

3.5.3 The WDEQ-Land Quality Division (-LQD) Rules and Regulations, Chapter 11, In Situ Mining, Section 10 (h) (i) states the following to ensure the locations of the abandoned wells are adequately identified:

The boundaries of each wellfield and the location of the monitoring well ring around each wellfield shall be recorded as a deed notice with the appropriate county.

Like the WSEO, the WDEQ will not deny access to groundwater, which would require a deed restriction rather than deed a notice. However, that agency will apprise prospective drillers of the quality of the water. To determine the depth below the ground surface and the completion interval, the WSEO database must be examined.

3.5.4 The WSEO and the WDEQ were, as of December 12, preparing for a discussion of these issues with their legal advisors at the Wyoming Attorney General’s Office. Cameco representatives suggested that any notices or institutional controls extend to the aquifer exemption boundary rather than stopping at the monitor ring.

4. Corrective Action Assessment

4.1 Previous and Current Corrective Action Programs

4.1.1 Eleven pore volumes of water were treated using groundwater sweep and reverse osmosis (RO) for the restoration of MU1. During four of these pore volumes sodium sulfide was added as a reductant. This includes a flare factor of 1.56 as described in the surety bond. The total cost of restoring MU1 is \$1.9 million, including the installation of 42 wells as replacements for wells that failed mechanical integrity testing. See Table 3 below for a detailed enumeration of the costs. Since September, 2014 when restoration activities there ended, the MU1 has not been subjected to any pumping other than that required for stability sampling.

Description of cost	Cost
42 replacement wells (re-drills)	\$677,292
Ground Water Sweep (1x pore volume)	\$136,398
RO (9x pore volumes)	\$962,885
RO with sodium sulfide (1x pore volume)	\$113,328
Total Cost	\$1,889,903

Table 3. Cost expenditure for the restoration of MU1. Costs are based upon the most recent approved surety bond for the Smith Ranch mine and are expressed in 2016 dollars.

These costs can be combined with changes in dissolved uranium concentrations to calculate dollar costs per rem-avoided. The graphs of Figure 7 below shows the median dissolved uranium concentrations for MU1 as a function of time and the pore volumes treated as a function of time. As described in Section 2.1, median concentrations are used only for ALARA calculations and not for trending.

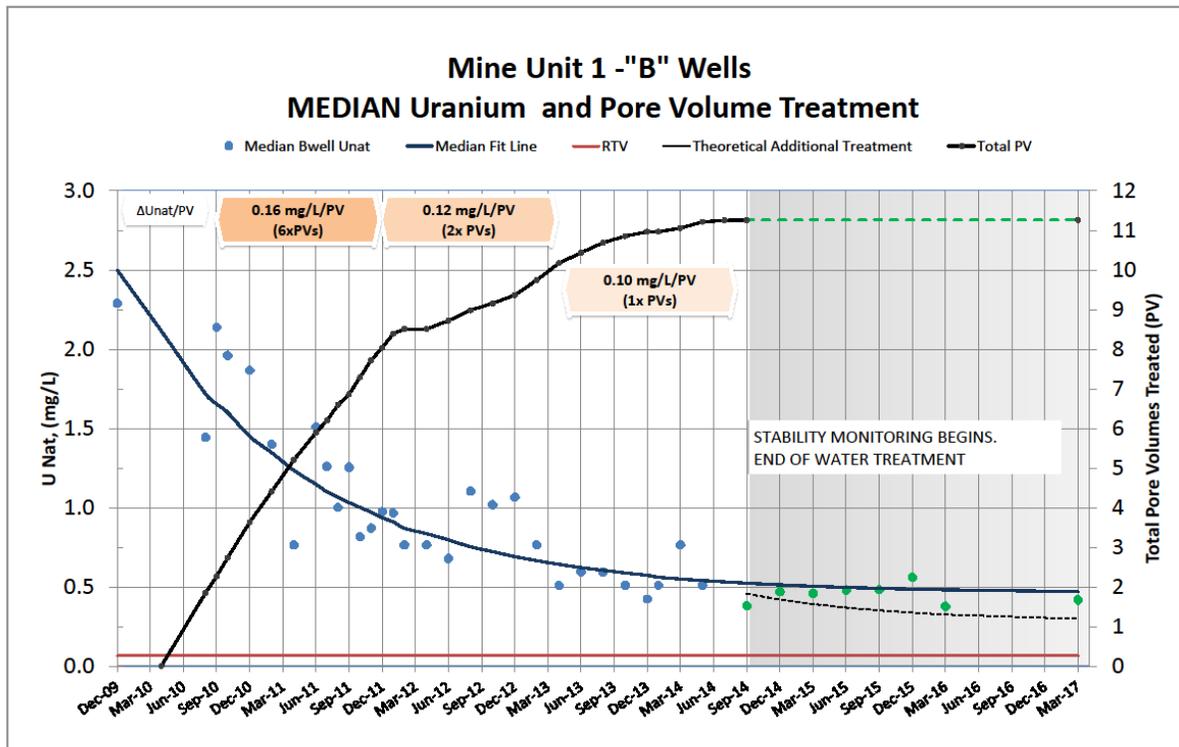


Figure 7 Median values of "B" wells during the restoration treatment and stability monitoring of Mine Unit 1. The Median Fit line was estimated using a declining function based on the formula $C(t)=C_0 e^{-k*\Delta t} + b$

The last Pore Volume (PV) Treated resulted in a change in dissolved uranium of -0.1 mg/L/PV. This amount was used in estimating additional near term treatment so that each PV treatment might result in a decline of 0.1 mg/L of dissolved uranium. The dashed line indicates the estimate if two additional PV are treated. Note that the results over time become more and more gradual for each PV treated.

The median dissolved uranium concentration for MU1 was 2.29 mg/L in December, 2009 and is, as of the latest round of stability testing in March, 2017, 0.42 mg/L. Inter-well medians are used for this graph to set the stage for ALARA calculations described in Section 4.4 of this report and are not intended for use in formal statistical trending.

4.1.2 Sodium sulfide was added to reverse the effects of oxidation, but the rate of addition was limited by clogging to about 10 mg/L of treated water. Note that the Smith Ranch surety bond bases reductant treatment costs on a 12 mg/L addition rate (see section 4.3.5 below), which later field experience demonstrated to be somewhat optimistic. Clogging occurred despite the removal of significant amounts of TDS before sulfide treatment began. To introduce the chemical, it was first dissolved in permeate (i.e., clean water) from an RO unit. The resulting solution was then blended with mine unit water before being injected. Even at limited injection rates the sulfide raised the pH of the treated stream to high enough values that carbonate scale precipitated in the plant pipes. Pure permeate was used to dissolve the scale, and this required interruptions in the addition of sulfide to the groundwater. The attached workbook "**MU-1 Sulfide Addition Summary.xlsx**" shows the amount of sodium sulfide added during each month of addition. Note that a total of 17,571 pounds of sodium sulfide was applied during the course of the restoration despite the challenges posed by carbonate precipitation.

4.1.3 The workbook "**CURRENT Mine Unit 1 Data – Restoration Parameters.xls**" contains data and graphs summarizing restoration progress for each of the 19 B wells beginning in August, 2010 and including the stability period. Solid blue lines signify the period of time where restoration treatment was occurring and the dashed green lines denote the stability period. The criterion for ending restoration treatment for individual patterns was a drop in the specific conductivity below the

Restoration Target Value (RTV) for this parameter, which was 573 μ -mho/cm. Continuing RO treatment to significantly lower conductivity values increases the tendency of metals and other constituents of concern to leach from aquifer solids.

4.1.4 Broadly speaking, uranium(VI) in the restored aquifer can be divided into two categories. One category is dissolved uranium(VI) while the other is adsorbed on the surfaces of aquifer solids. Sequential extraction data has allowed for estimates of the amount of uranium(VI) adsorbed on core material. When compared to estimates of the amount of dissolved uranium(VI), it is obvious that in some localized regions of the aquifer (notably near well B4), the mass of uranium(VI) adsorbed to aquifer solids is much greater than that dissolved in the pore waters. Note that this is not true as a generalization of the entire volume of aquifer.

4.1.5 Early in the restoration process, dissolved uranium that is associated with the lixiviant plume is prominent. Its concentration often positively correlates with that for chloride. This is because natural chloride is usually present at less than 10 mg/L (i.e., the background concentration is low), and the ion-exchange process used to recover uranium adds chloride to the groundwater until concentrations of the latter reach 150 mg/L or higher. Chloride hardly interacts with the aquifer solids, so it is highly mobile and acts as a tracer that can be used to track the plume. As restoration proceeds, the concentration of chloride tends to drop rapidly. If the primary source of dissolved uranium is this plume, then the drop in concentrations of the two constituents of concern will tend to occur in concert. However, if uranium previously adsorbed to aquifer solids desorbs after most of the chloride has been removed by RO treatment, then chloride and uranium concentrations tend to decouple. In other words, they do not correlate as strongly. Correlation can be tested by plotting chloride vs uranium concentrations for each well.

4.1.6 Figure 8 shows such plots for all the B wells. The axes for the plots are normalized to the highest concentrations of uranium and chloride for each well to make for a more straightforward comparison between graphs. Stability data are included in the plots with those data points displayed as squares (green) while points taken during restoration are indicated as diamond (blue) points. As discussed in the immediately preceding paragraph, the graphs support the contention that lixiviant is not present in significant quantities near any of the restoration zone monitor wells.

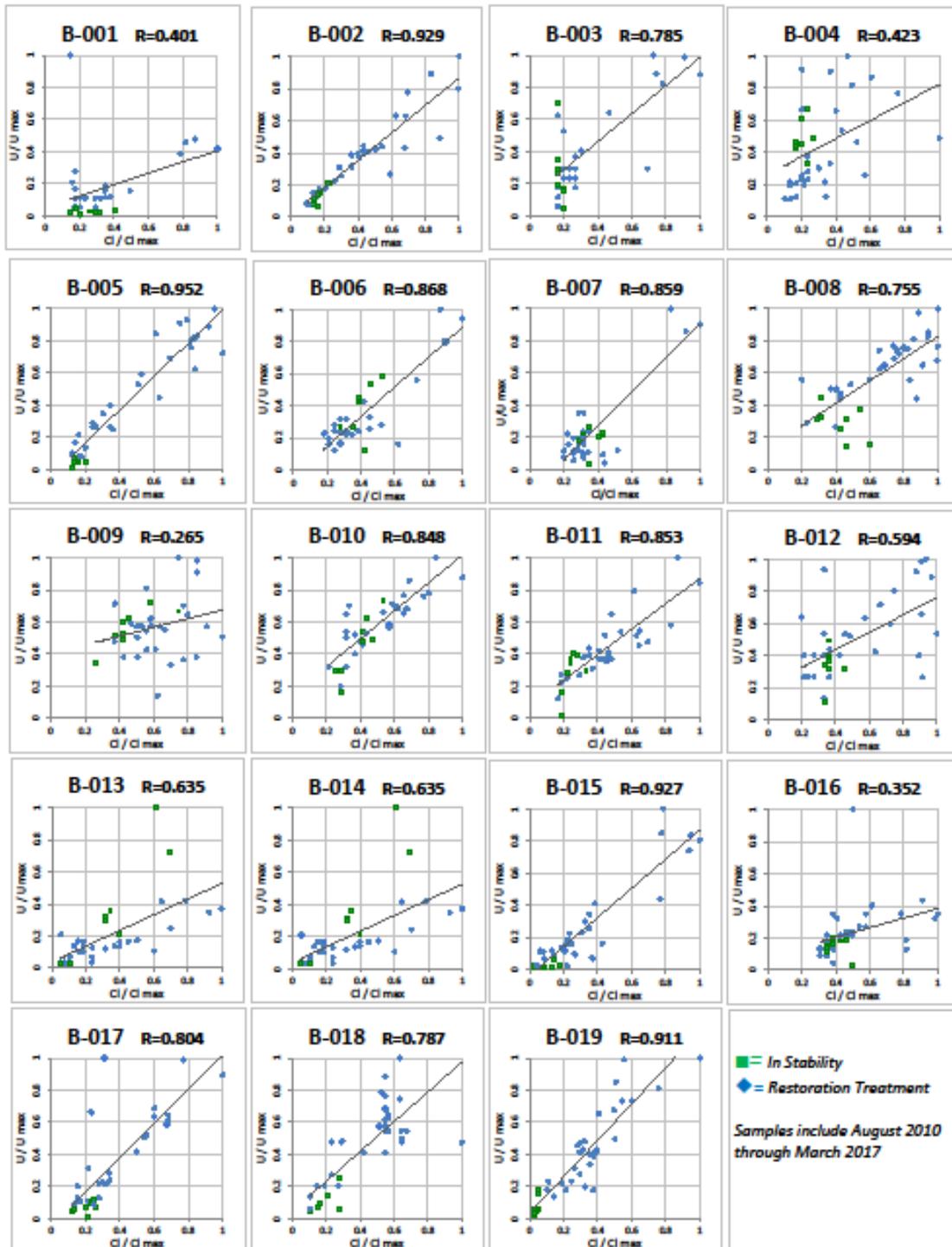


Figure 8 Correlations of normalized Uranium to normalized Chloride during restoration and stability in MU1 for “B” wells.

4.2 Potential Corrective Action Alternatives

4.2.1 In the unlikely event that the ACLs proposed in Section 5 of this document should be exceeded during the closure monitoring period, spot RO treatment would be done in the region of the effected wells. To ensure that the spot-treatment option is readily available, wells in the restoration zone of MU1 continue to be subjected to Mechanical Integrity Testing (MIT) on the schedule prescribed in the

mine license and permit. Wells will be immediately plugged and abandoned following a successful conclusion to the three year closure monitoring period.

4.2.2 There is a high probability that any such an exceedance would be attributable to a small volume of lixiviant that might be missed during the sweep of the mined aquifer conducted for restoration. As is described in Section 4.5.2 of this document, the quantity of uranium(VI) adsorbed to aquifer solids is substantial and probably much greater than would be the mass of dissolved uranium(VI) even in the event of ACL exceedance. Chloride is included in the monitoring-closure program proposed for the mine unit to provide insights into the relative importance of desorption vs. lixiviant should a rise in uranium or arsenic concentrations threaten to breach the ACL values for these elements of concern.

4.2.3 The remainder of Section 4.2 is devoted to a discussion of alternatives to RO treatment that might be used as part of a corrective action plan. The scientific literature has reports of alternative water treatment methods for removing dissolved uranium that rely on precipitating or adsorbing it. Phosphate addition can immobilize uranium in the presence of calcium, but a large portion of the uranium is adsorbed on the surface of precipitates.⁹ It offers no obvious advantages for areas of MU1 where most of the uranium is already adsorbed on ferric solids. Furthermore, the concentrations of phosphate required (typically gram/L concentrations) suggest that well-clogging could be an issue. Zero-valent iron has also been proposed for the removal of dissolved uranium, but no instances of its successful use for uranium ISR remediation are known to us.¹⁰ Potassium metavanadate or similar vanadium salts have been suggested as a way to precipitate uranium in a durable solid,¹¹ but the cost of vanadate salts and their toxicity when dissolved make it unclear whether they could be used as a practical groundwater treatment for uranium ISR restoration.

4.2.4 Another possible alternative is bio-stimulation of the restoration zone.¹²⁻¹⁴ This technique is promising in the context of restoring aquifers ISR-mined for uranium, but it has been employed with mixed results at Mine Unit C of the Smith Ranch mine.¹⁵ Well clogging was found to be significant, and concentrations of bicarbonate increased during the latter stages of the bio-stimulation. Both of these outcomes may be the result of failing to optimize the quantity and type of nutrient.¹⁶ Research is being conducted to increase the efficiency of restoration at Smith Ranch, including the possible use of bio-stimulation.¹⁷⁻¹⁹ Until that research is completed, the detailed understanding necessary to execute a successful bio-stimulation is lacking.

4.2.5 Perhaps the most important attribute that a potential corrective action alternative should possess is the ability to dissolve adsorbent substrates such as ferric iron *in a realistic time frame*. Desorption occurs so slowly that chemicals added to precipitate uranium will be flushed from the restoration zone long before the adsorbed uranium will desorb and thus be available for aqueous reaction.

4.2.6 Given the uncertainties associated with the alternatives discussed in 4.2.3 and 4.2.4, the obvious choices are more RO treatment in localized areas of the aquifer and/or the further addition of a chemical reductant (such as sodium sulfide). These are discussed in the next section.

4.3 Feasibility of Corrective Action Alternatives

4.3.1 As of the latest round of stability testing in March, 2017 the concentration of dissolved uranium for B4 is 4.65 mg/L (the highest in MU1), which is little different from the concentration in December, 2009 (4.94 mg/L). This is despite the fact that nearly 58 pattern pore volumes of water treatment have been performed there (see Table 3 and the workbook **MU-1 HH PV Treatment.xlsx**). Pattern pore volumes include a flare factor of 1.56, which is taken from the surety bond for the entirety of MU1. As discussed in 4.1.5 above, the low correlation between chloride and uranium concentrations for the B4 data set strongly suggests that dissolved uranium there is primarily the result of desorption. Not coincidentally, core samples taken from the vicinity of well B4 contain the highest concentrations of iron of all the samples tested (see Appendix C, page 41). In other words, the ferric sink is relatively large near well B4, which is why flow path A of Appendix C is discussed extensively in this report along with the fact that analysis of the other five flow paths yields similar results to that of flow path A. The almost total lack of progress in decreasing the dissolved uranium at well B4 likely stems from the slow desorption rate of

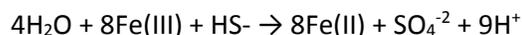
uranium from the ferric sink, and that slow rate has also contributed to B4 being classified as statistically “stable” for dissolved uranium over the time span of stability sampling. Hence further RO treatment is unlikely to either lessen the current concentration of dissolved uranium or to stabilize uranium concentrations over a time frame of centuries by removing significant quantities of adsorbed uranium from the ferric solids.

4.3.2 For a situation unlike that of well B4, where rising dissolved uranium concentrations are accompanied by strongly correlated increasing chloride concentrations, the inference would be that further RO treatment was warranted if the concentration of dissolved uranium approached the ACL value.

4.3.3 Further addition of sodium sulfide or a similar reductant should also be examined as a possible corrective action alternative. The objective of sodium sulfide addition is the dissolution of the ferric sorption sink for uranium(VI), and if successful would be followed by further RO treatment to remove newly-mobilized uranium(VI). A realistic scenario would be the addition of sodium sulfide for the treatment of HH1-1 and HH1-2 since this area of MU1 includes all wells exhibiting statistically significant increasing trends for uranium and well B4, which is stable but has the highest concentration of dissolved uranium as of the latest round of sampling.

4.3.4 The amount of sulfide required to remove the ferric solids can be estimated from the sequential extraction data for the restoration zone (see Table 3-11 of Appendix C). The highest iron content of any core is 0.3743 moles/kg of core (ST-5, core segment 494-495 feet). Note that ST-5 is the nearest core taken to well B4, where the role of uranium(VI) desorption was discussed in 4.3.1. The median value of ferric iron content for the entire mine unit restoration zone is 0.0778 moles of iron/kg of core sediment. Using the median value, a conservative estimate of the amount of ferric solids can be obtained.

4.3.5 To determine the mass of sodium sulfide needed to dissolve the ferric solids, the following redox equation was used:



Note that 8 moles of ferric iron are reduced for each mole of sulfide.²¹ This is highly conservative since it is likely to underestimate the actual amount of sulfide needed because many ferric minerals are reduced at the rate of 6 moles per mole of sulfide or even less. Furthermore, side reactions such as the formation of native sulfur instead of sulfate often plague remediation efforts and further decrease restoration efficiency. Finally, only the mass of aquifer within idealized pattern volumes is used for the estimates of ferric solids to be treated that follow (i.e., no flare is assumed for this calculation). Even with these optimistic assumptions, about 572,000 pounds of sodium sulfide would be required to remove most or all of the ferric solids in the two header houses. The sodium sulfide used at Smith Ranch contains three waters of hydration (i.e., $\text{Na}_2\text{S}\cdot 3\text{H}_2\text{O}$), which implies that about 965,000 pounds of the hydrated chemical would be required. Assuming a sodium sulfide addition rate of 12 mg/L and a combined flow rate for the two header houses of 600 gallons per minute, it would take 6590 days (about 18 years) of continuous, uninterrupted pumping to add the required reductant. 12 mg/L of sodium sulfide was chosen for the calculations because the surety bond specifies 0.00017 pounds of sodium sulfide per gallon of treated water (presumably as $\text{Na}_2\text{S}\cdot 3\text{H}_2\text{O}$ as purchased in bulk at the mine), which translates to 12 mg/L of Na_2S . See Appendix D for the calculations used to obtain the results described in this paragraph.

4.4 Costs and Benefits of Corrective Action Alternatives

4.4.1 The current surety bond for Smith Ranch assigns costs for restoration water treatment: \$0.21 per 1000 gallons (k-gallon) of water delivered by the down-hole well pump; \$0.09 per k-gallon for sodium sulfide addition; \$0.76 per k-gallon for pumping water from the Central Processing Plant (CPP) back to MU1; \$0.64 per k-gallon of RO treatment; and \$1.20 per k-gallon for disposal of waste water. These can be used to estimate the cost of removing the entire ferric sink. No RO treatment would be done during the time Na_2S was being added, so the cost per k-gallon of treatment would be: $\$0.21 + \$0.09 + \$0.76 =$

\$1.06. The total gallons pumped would be about 5.7 billion gallons, or 5.7 million k-gallons. The surety assigns a cost to this of \$6,042,000. See Appendix D for the calculations done to arrive at this estimate.

4.4.2 In the next phase of corrective action, the two header houses would be treated by RO to remove the uranium mobilized by the dissolution of the ferric sink. RO units at Smith Ranch typically recover 80% of the feed stream as permeate (“clean”) water with the remainder being the reject (sometimes called “brine”) that must be sent to disposal. The surety bond presently assumes 9 pore volumes are required for cleaning. Using this assumption, the total volume to be treated by RO is 173 million gallons. The cost for treating a k-gallon is $\$0.64 + \$0.76 + \$0.21 + (0.2 \times \$1.20) = \$1.85$ per k-gallon where 0.2 is the fraction of the incoming k-gallon that is sent to disposal. Therefore 173,000 k-gallons of treatment would cost \$218,000.

4.4.3 The two phases (reductant addition followed by RO treatment) carry a combined estimated cost of over \$6.26 million. Recall that this is for only two of the six header houses in MU1. If the ferric solids were removed, it is likely that the aquifer zone would be more stable from the perspective of dissolved uranium concentrations. However, the benefit accruing from the additional restoration effort is counter-balanced by the loss of 1.8 pore volumes of groundwater or 34 million gallons to disposal, the addition of more sodium to the aquifer that may not return during RO treatment because of strong interaction with formation clays, the expenditure of significant amounts of electricity with all the environmental costs that entails, and the cost of millions of dollars. And the water is still unlikely to be potable, just as it was before ISR mining occurred. Complete removal or even removal of a significant fraction of the ferric sink is clearly impractical.

4.5 As Low As Reasonably Achievable Demonstration (ALARA)

4.5.1 The ALARA calculations begin with a calculation of how much each person-rem averted has cost in the restoration effort to date over the entire mine unit. A detailed outline of the scenarios and calculations done to inform this discussion can be found in Appendix E, Section 4. Inter-well medians are used for these calculations, but not for trending purposes as discussed in Section 4.1. The median dissolved uranium concentration for MU1 was 2.29 mg/L in December, 2009 and is, as of the latest round of stability testing in March, 2017, 0.42 mg/L. So the drop in dissolved uranium concentration is 1.87 mg/L (2.29 mg/L – 0.42 mg/L). This drop in dissolved uranium concentration translates into a decrease of 0.620 person-rem avoided using the scenario of Appendix E. The total cost is taken from Table 3, Section 4.1 of this document (\$1,889,903). The resulting dollar cost per person-rem avoided using the scenario of 100 persons consuming beef watered from the restoration aquifer of Appendix E, Section 4 is \$30,482/person-rem avoided over the entire mine unit for the restoration done until now.

4.5.2 The discussion that follows addresses ALARA as it relates to the further treatment of only a select portion of MU1, namely flow path A of the transport model of Appendix C. The narrative centers on flow path A because it is near well B4, which has the highest concentration of dissolved uranium, and because it is near the region for which ferric solids were found in the highest concentrations in core testing.

4.5.3 Assigning a dollar cost per person-rem averted for further treatment is challenging given the fact that uranium adsorbed to surfaces in an aquifer hundreds of feet beneath the surface of the ground is not accessible for consumption by humans or plants and animals but will slowly be desorbed over time scales of centuries. Only when desorption occurs is there a plausible path for ingestion.

4.5.4 As a prelude to addressing this challenge, the fraction of uranium(VI) that is dissolved in the pore volume as compared to the total amount of uranium(VI) both adsorbed and dissolved was estimated. This was done using the ferric sequential extraction data from the core studies (see Appendix C, Table 3-11, page 40 for the sequential extraction data) and the dissolved uranium stability data for the last round of sampling in March, 2017 at MU1 (**MU-1 Groundwater Quality Summary.xlsx**). The dissolved fraction of uranium(VI) was found to be 2% to 9% of the total uranium(VI) budget in a unit volume of restored aquifer. Appendix D explains how estimates of the total uranium content of a unit volume of aquifer were obtained.

4.5.5 A realistic scenario to examine the costs of continuing to restore might involve further RO treatment only in HH1-1 and 1-2. No more sodium sulfide addition would occur because of the costs and other difficulties outlined in Section 4.4. These header houses include all of the wells in MU1 that show statistically significant increasing trends (B1/3/5/19) in addition to the well with the greatest remaining concentration of uranium (B4).

4.5.6 To be conservative, we assume that 9% of the total (sorbed + dissolved) uranium is removed if an additional pore volume is treated. Note that 9% is the high end of the estimated fraction of uranium(VI) that is dissolved. We further assume that a 9% decrease in the total mass of uranium(VI) will result in a 9% decrease in the concentration of dissolved uranium for HH1-1 and 1-2. This assumption is likely to significantly over-state the decrease in dissolved uranium concentrations, as will be described later in Section 4.5.9.

4.5.7 To determine the concentration of dissolved uranium that would be removed by further RO treatment, inter-well median values for the “B” wells in the two header houses as of the latest round of stability sampling in March, 2017 were used (see Table 4 below). As mentioned in Section 2.1 of this report, NRC staff have issued guidance that inter-well averaging (including means or medians) should not be used for aggregate stability trending.²² Note that the inter-well median values used for the ALARA calculation here are not for this purpose but are intended only to estimate the amount of total uranium removed (thus the person-rem averted) from HH1-1 and HH1-2 for the ALARA scenario calculations.

Well	Dissolved U (mg/L)
B1	0.0829
B2	0.852
B3	0.901
B4	4.65
B5	0.399
B6	0.361
B15	0.108
B19	1.00

Table 4. Dissolved uranium concentrations for all of the “B” wells along flow path A as of the latest round of sampling in March, 2017.

The inter-well median value of the dissolved uranium concentrations shown in Table 4 is 0.63 mg/L. 9% of this median value is 0.057 mg/L, and this is the decrease in dissolved uranium concentration attributed to one additional pore volume of further RO treatment. The surety bond specifies a cost of \$112,000 per pore volume of RO treatment for the entirety of MU1, and this translates into \$31,500 for the pore volume of the two header houses combined. Combining the decrease in dissolved uranium concentration with the cost of further RO treatment results in an estimate of \$17,000 in cost for every person-rem avoided if an additional pore volume of treatment is done on these two header houses (see Appendix E, pages 31-32 for the details of the calculations). That is well above the \$5,100 per person-rem avoided that is the threshold beyond which further treatment is not needed.²³

4.5.8 Restoration experience with well B4 offers insights into the effectiveness of RO treatment is to remove uranium that is slowly desorbing from aquifer solids. As described in 4.1.5 of this document, dissolved uranium in this well is largely attributable to the desorption of uranium(VI) from aquifer solids rather than a lixiviant plume. In December, 2009 the dissolved uranium concentration at B4 was 4.94 mg/L, and as of the latest round of stability sampling in March, 2017 the concentration was 4.65 mg/L. The lack of substantial progress in decreasing dissolved uranium concentrations at B4 is despite treatment of 58 pattern pore volumes (see workbook **MU-1 HH PV Treatment.xlsx**), including groundwater sweep, RO, sodium sulfide application phases.

4.5.9 The inescapable conclusion from the example of B4 is that the ALARA scenario assumption that 9% of the total uranium would be removed with each succeeding pore volume of aquifer treatment is unrealistically optimistic. This is because the kinetics of desorption are slower than the rate at which RO pump-and-treat can remove dissolved uranium. The 58 pore volumes of groundwater were not treated in a continuous, uninterrupted fashion, but the interruptions that did occur were not sufficient lengths of time to allow the dissolved uranium(VI) to reach equilibrium with adsorbed uranium(VI). In summary, the \$17,000 per person-rem avoided discussed in 4.5.7 is a significant underestimate because the calculation cannot capture the costs of waiting for uranium(VI) desorption to occur. Therefore, even the spot treatment of selected regions of MU1 are not justified.

5. Alternate Concentration Limits

5.1 Proposed Alternate Concentration Limits

5.1.1 The highest concentration of dissolved uranium is predicted to occur in the vicinity of well B4 at a time some five centuries in the future (see Figure C.5-12 Appendix C). B4 was chosen as the worst-case scenario for the rise in dissolved uranium concentrations, and it should be emphasized that future dissolved uranium concentrations are not expected to increase as much in other parts of MU1 as in this particular location. This is because core ST5, which is nearest well B4, had the highest uranium content of any of the core obtained and characterized for the transport model described in Appendix C. In addition, as described in Appendix C (Section 5.3), the predicted maximum uranium concentration at B4 is conservatively high.

5.1.2 As Figure C.5-12 illustrates, the uranium concentration increase at B4 is very slow and is largely because of a gradual rise in pH as the up-gradient water begins to flush the lower pH fluid from the restoration zone. The pH decreased during ISR mining because of the injection of carbon dioxide and because of the oxidation of pyrite.²⁴ As pH rises, smaller fractions of anionic species (such as uranyl dicarbonate) remain bound to the surfaces of ferric solids.²⁵ In other words, anionic desorption becomes more favored as pH increases though this effect is not as dramatic for arsenic. Another mechanism that can increase the rate of desorption is an influx of higher TDS waters, but this is not likely to occur in MU1 because water treatment was terminated when restoration zone TDS concentrations were at or near pre-mining levels.

5.1.3 Field pH measurements were made during the restoration period for the “B” wells of MU1, and the results are summarized in the workbook “**Mine Unit 1 Field pH Measurements.xlsx**”. All measurements were made using a YSI handheld model 556 meter and a flow-through cell to minimize the loss of dissolved carbon dioxide at ambient atmospheric pressure. Such carbon dioxide losses can result in a systematic bias towards higher measured pH values. Because a range of field pH values was measured at well B4, the sensitivity of predicted uranium concentrations to initial pH was assessed using the transport model (Appendix C, Section 5.3)

5.1.4 Dissolved uranium eventually reaches a concentration of 51 mg/L at well B4 if the initial pH of the local aquifer is taken to be 6.3. Since 6.3 is representative of the low end of the range of field pH measurements at well B4, it is used to determine the maximum, future dissolved uranium concentration as a worst case scenario. See Figures 9A and 9B immediately below for graphical representations of the impact long term desorption has on stability and the ultimate dissolved uranium concentration. The graphs were obtained using the transport model of Appendix C. Figure 9A shows how little dissolved uranium concentrations increase in the first half century from time zero of the transport model. The millennial time scale for dissolved uranium concentrations is shown in Figure 9B.

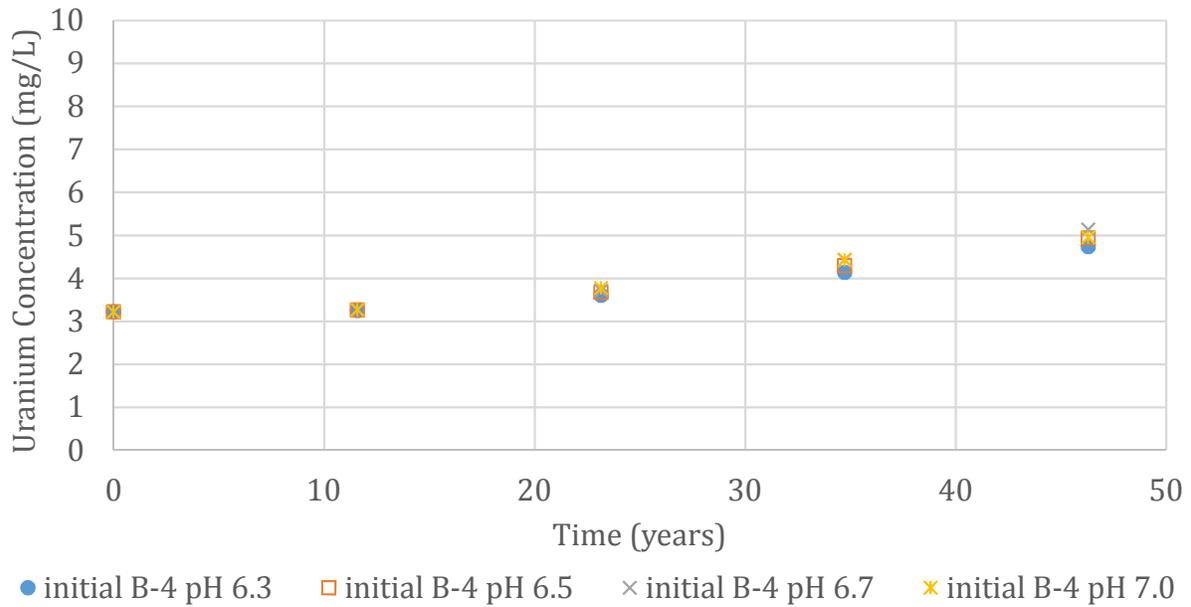


Figure 9A. Dissolved uranium concentrations over a span of 50 years as a function of the initial pH. As can be seen, there is little difference in the rate of desorption over this relatively short period of time. The graph was constructed using the transport model of Appendix C.

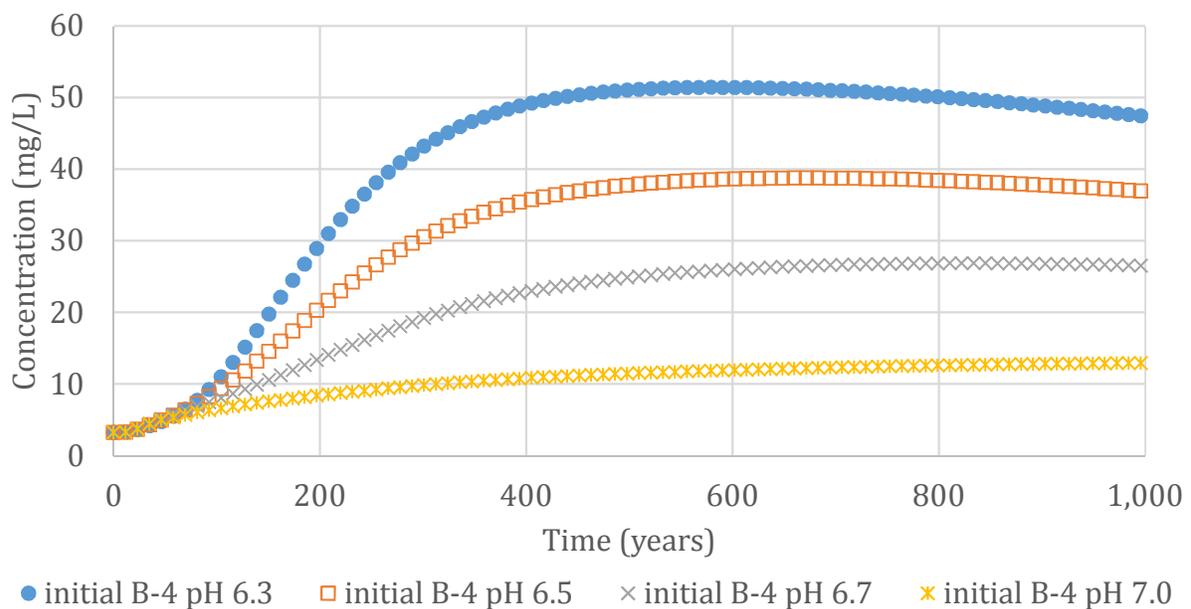


Figure 9B. Dissolved uranium concentrations over time for a millennium as a function of the initial pH. The graph of Figure 9A is the first 50 years of this plot.

5.1.5 The transport model (Appendix C, Section 5.3) shows that the amount of dissolved uranium that can be immobilized by down-gradient sediments is much greater than that emanating from the MU1 now or in the future. Mass balance calculations show that there is an excess of anywhere from 188 to 2,726 times the amount of available pyrite to precipitate dissolved uranium on any of the flow paths of MU1 (see Appendix C, Table 5-11). This assumes median values for core pyrite content and dissolved uranium concentrations. In addition to the pyrite, there are large amounts of clays along the flow paths

that have been shown to aid in immobilizing uranium.³ Furthermore, carbonaceous materials such as lignite are also commonly encountered, and these can act as reducing agents for dissolved uranium (VI).²⁶

5.1.6 Given the facts that down-gradient migration of uranium(VI) beyond the aquifer exemption boundary is highly unlikely (see Figure 5-11a of Appendix C for a graph showing dissolved uranium concentrations as a function of distance traveled for flow path A) and that the water within MU1 was not potable to begin with, an Alternate Concentration Limit of 51 mg/L is proposed for dissolved uranium.

5.1.7 The pH dependence of future dissolved arsenic concentrations for well B4 is shown in the graph of Figure 10 below. The concentration of arsenic shows some dependence upon the assumption of the initial pH at well B4, but the effect is not as strong as is the case for dissolved uranium.

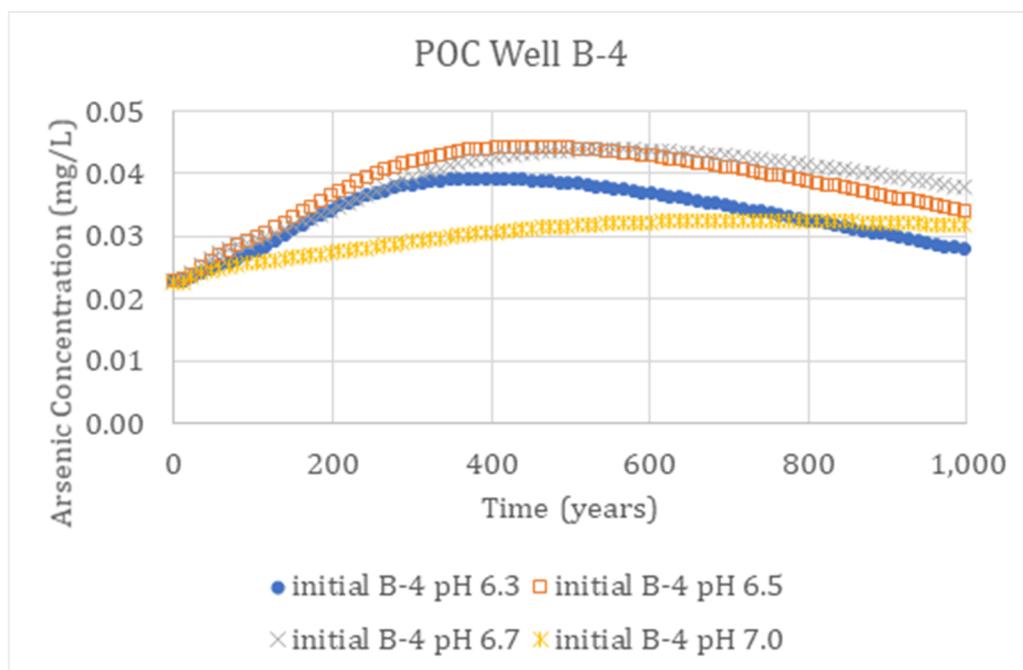


Figure 10 The concentration of arsenic at the point of compliance as a function of initial pH at well B-4. Results are calculated using the transport model of Appendix C.

The highest arsenic concentration measured during the stability period in all of MU1 was 0.048 mg/L at well B9 on 9 September 2015 (see “**MU-1 Groundwater Quality Summary.xlsx**” attached to this report for the stability data). To provide a safety margin in the event of a patch of lixiviant missed during restoration, an assumption of an initial restoration zone arsenic concentration of 0.1 mg/L for arsenic leads to the prediction that the maximum dissolved arsenic concentrations will reach 0.18 mg/L in 320 years at wells B1 and B4 (Appendix C, Section 5.5). At this concentration, the model shows that arsenic never reaches the aquifer exemption boundary. 0.18 mg/L is therefore the proposed ACL for arsenic.

5.1.8 Like arsenic, long-term dissolved selenium concentrations are not expected to be as strongly impacted by pH changes as those for uranium. As of the last round of stability sampling in March, 2017, all wells are either stable or trending downwards for dissolved selenium. The highest concentration for dissolved selenium among the B wells is 0.021 mg/L at well B13. However, well B2 reached a concentration of 0.099 mg/L of selenium for the June, 2015 stability sampling round. Since then, selenium concentrations have sharply curtailed at B2 and many other wells. Figure 11 illustrates the maximum concentrations of dissolved selenium that are predicted by the modelling of Appendix C at wells B3, B5, and B13 using initial concentration assumptions representative of current conditions. In addition, modelling was done assuming an initial much higher dissolved selenium concentration of 0.2

mg/L for each of the wells. Figure 11 (taken from Appendix C) shows how dissolved selenium concentrations are predicted to evolve over time at each of the three wells using these initial concentration assumptions.

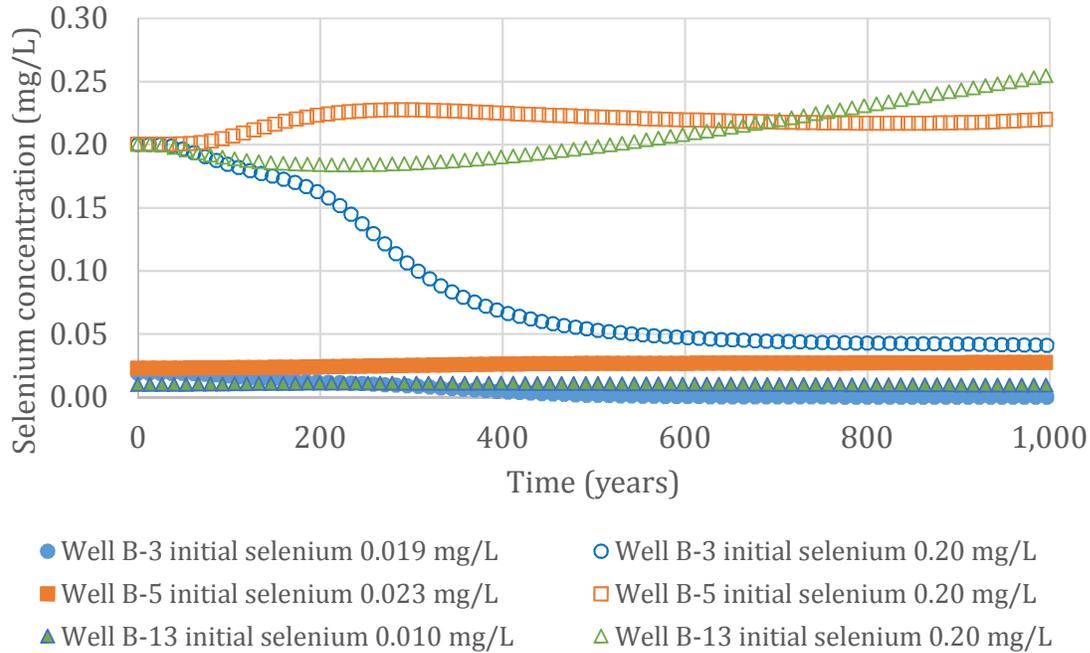


Figure 11 Predicted selenium concentrations at Wells B-3, B-5 and B-13 as a function of time and initial selenium concentrations. The initial pH was the field pH values at each well in March 2016.

B-3 initial pH 6.1, increased throughout modeling period, maximum pH was 6.8

B-5 initial pH 6.45, stayed pretty constant throughout modeling period

B-13 initial pH 6.6, increased throughout modeling period, maximum pH was 6.8

These wells are respectively along flow paths A, B, and E of the transport model, and they are representative of the impact of initial selenium concentration assumptions for wells along all six flow paths. Note that in no case do elevated dissolved selenium concentrations occur at the aquifer exemption boundary for any of the scenarios or any of the six flow paths A through E. Because of these considerations, the proposed ACL for selenium is 0.20 mg/L.

5.2 Proposed Implementation and Ground Water Monitoring Measures

5.2.1 Sample collection has typically been done on a quarterly basis for MU1, but the slow rate of migration of uranium, arsenic, and selenium demonstrated by the transport model of Appendix C suggests a less frequent schedule. Furthermore, there is little point in sampling up-gradient ring monitor wells in the absence of the pumping done during mining or restoration. Finally, no evidence of excursion has ever been observed at any of the monitor wells in the layers directly above or below the restored zone. As a consequence, there is no reason to sample any of these wells during the monitoring period.

5.2.2 The duration of the monitoring measures is also informed by the transport model. The predicted long-term rise in dissolved uranium and arsenic concentrations occur on a time scale that cannot realistically be captured by a monitoring program (see Section 5.1 above). Any program encompassing even a multi-decadal time window is likely to only observe isolated patches of lixiviant plume missed by the groundwater treatment rather than wholesale desorption of uranium(VI) from aquifer solids. Isolated volumes of lixiviant, or even volumes encompassing a large fraction of MU1, are not a threat to down-gradient water for uranium, arsenic, or selenium.

5.2.3 The proposed monitoring program encompasses all 19 B wells and the down-gradient ring monitor wells M1-10 and M24-25. All samples will be analyzed for uranium, arsenic, selenium, and chloride, and collection will be done at six-month intervals for a period of three years.

6. Conclusion

As shown above, Cameco Resources' request for a license amendment to establish site-specific Alternate Concentration Limits (ACLs) of 51 mg/L for dissolved uranium, 0.18 mg/L for dissolved arsenic, and 0.20 mg/L for dissolved selenium for MU1 should be granted in accordance with 10 CFR Part 40, Appendix A, Criterion 5B(6). The restoration efforts in MU1 are protective of the surrounding aquifers, human health, and the environment. Cameco Resources has shown that the proposed ACLs are ALARA, after considering practicable corrective actions, and that neither uranium nor arsenic nor selenium will pose a substantial present or potential hazard to human health or the environment as long as the alternate concentration limit is not exceeded.

7. References

- 1) All of the MCLs discussed here can be found at the following URL: <https://www.epa.gov/ground-water-and-drinking-water/national-primary-drinking-water-regulations#Radionuclides>
- 2) Boland, *et al*, Reduction of U(VI) by Fe(II) during the Fe(II)-Accelerated Transformation of Ferrihydrite, *Environmental Science & Technology*, **2014**, 48, 9086-9093.
- 3) Tsarev, *et al*, Uranium Reduction by Fe(II) in the Presence of Montmorillonite and Nontronite, *Environmental Science & Technology*, **2016**, 50, 8223-8230.
- 4) Xin, *et al*, Reduction of Uranium(VI) by Soluble Iron(II) Conforms with Thermodynamic Predictions, *Environmental Science & Technology*, **2011**, 45, 4718-4725.
- 5) The use of sequential extractions in characterizing aquifer solids was discussed by Mike Truex of Pacific Northwest National Laboratories in his talk at the recent FRTR Semi-Annual General Meeting, November 8, 2017: "Overview of Remediation Technologies for Radionuclides in Soil and Groundwater", <https://frtr.gov/meetings1.htm>.
- 6) Ruediger, *et al*, An Evaluation of Health Risk to the Public as a Consequence of In Situ Uranium Mining in Wyoming, USA *Journal of Environmental Radioactivity*, **2015**, 150, 170-178.
- 7) Stewart, *et al*, Impact of Uranyl-Calcium-Carbonato Complexes on Uranium (VI) Adsorption to Synthetic and Natural Sediments, *Environmental Science & Technology*, **2010**, 44, 928-934.
- 8) Dong, *et al*, Influence of Calcite and Dissolved Calcium on Uranium (VI) Sorption to a Hanford Subsurface Sediment, *Environmental Science & Technology*, 2005, 39, 7949-7955.
- 9) Mehta, *et al*, Effect of Reaction Pathway on the Extend and Mechanism of Uranium(VI) Immobilization with Calcium and Phosphate, *Environmental Science & Technology*, 2016, 50, 3128-3136.
- 10) Yan, *et al*, Uranium(VI) Removal by Nanoscale Zerovalent Iron in Anoxic Batch Systems, *Environmental Science & Technology*, 2010, 44, 7783-7789.
- 11) Tokunaga, *et al*, Potential Remediation Approach for Uranium-Contaminated Groundwaters Through Potassium Uranyl Vanadate Precipitation, *Environmental Science & Technology*, 2009, 43, 5467-5471.
- 12) Lee, *et al*, Biogenic Formation and Growth of Uraninite (UO₂), *Environmental Science & Technology*, 2010, 44, 8409-8414.
- 13) Bernier-Latmani, *et al*, Non-uraninite Products of Microbial U(VI) Reduction, *Environmental Science & Technology*, 2010,44, 9456-9462.
- 14) Bargar, *et al*, Uranium Redox Transition Pathways in Acetate-Amended Sediments, *Proceedings of the National Academy of Sciences*, 2013, 110, 12, 4506-4511.
- 15) Confidential correspondence TFN 5 6/97, December 15, 2010, "Mine Unit C Bioremediation Report" submitted to Wyoming Department of Environmental Quality, Department of Land Quality.
- 16) Druhan, *et al*, Timing the Onset of Sulfate Reduction Over Multiple Subsurface Acetate Amendments by Measurement and Modeling of Sulfur Isotope Fractionation, *Environmental Science & Technology*, 2012, 46, 8895-8902.
- 17) A pilot-scale research project involving the use of conventional chemical reductants (sodium dithionite and sodium sulfite) and bio-stimulation is being conducted at Smith Ranch in collaboration with Dr. Paul Reimus of Los Alamos National Laboratory as a Cooperative Research and Develop

Agreement between LANL and Cameco Resources, Inc. The bio-stimulation investigation is expected to begin in early 2018.

18) Another pilot-scale research project that deals exclusively with bio-stimulation is reported by Willford, *et al*, concerning Header House 4-6, Mine Unit 4 at Smith Ranch: "Field Studies to Assess Bio-Stimulation to Remediation of Radionuclides and Heavy Metals at In Situ Leach Mine Sites". This was presented at the Federal Remediation Technologies Roundtable on November 8, 2017, <https://frtr.gov/meetings1.htm>.

19) Gallegos, *et al*, "Persistent U(IV) and U(VI) Following In-Situ Recovery (ISR) Mining of a Sandstone Uranium Deposit, Wyoming, USA", *Applied Geochemistry*, 2015, 63, 222-234.

20) The inability of sodium sulfide treatment to rapidly reduce U(VI) in solution under the geochemical conditions at Smith Ranch has been observed in the field repeatedly over the years. This has been confirmed by formal laboratory testing done at Los Alamos National Laboratory by Dr. Paul Reimus. This is likely the result of a kinetic barrier since reduction appears to be thermodynamically favored.

21) Poulton, *et al*, "A Revised Scheme For the Reactivity of Iron (Oxyhydr)oxide Minerals Towards Dissolved Sulfide", *Geochimica et Cosmochimica Acta*, 2004, 68 (18), 3703-3715.

22) See NRC memorandum dated October 27, 2017 describing a public meeting held on October 4, 2017 concerning the Cameco stability report on MU1, Docket #40-8964, License #SUA-1548.

23) NUREG-1530, Revision 1, August, 2015.

24) For a discussion of pyrite oxidation and its impact on pH, see Stumm and Morgan, "Aquatic Chemistry: Chemical Equilibria and Rates in Natural Waters", page 690-691, Third Edition, John Wiley & Sons, 1996.

25) Stumm and Morgan, page 544.

26) WoldeGabriel, *et al*, Characterization of Cores From an *in situ* Recovery Mined Uranium Deposit in Wyoming: Implications for Post-Mining Restoration, *Chemical Geology*, 2014, 390, pp. 32-45.