

## Technical Review Memorandum

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Prepared for: Save Our Cabinets, Earthworks, and the Clark Fork Coalition  
Prepared by: Ann Maest, PhD  
Date: 21 December 2011  
Re: Evaluation of Geochemical Issues in the Montanore Project, Montana, Supplemental Draft Environmental Impact Statement (SDEIS)

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### I. Introduction

The Montanore Project Supplemental Draft Environmental Impact Statement (SDEIS; USDA Forest Service and Montana Department of Environmental Quality, 2011) is a proposal for an underground copper and silver mine inside the Cabinet Mountains Wilderness Area boundary in northwestern Montana. The mine proponent is Montanore Minerals Corporation (MMC). The general issues addressed in this memorandum are the first two key issues identified in the SDEIS: 1: Potential for acid rock drainage and near neutral pH metal leaching, and 2: Effects on quality and quantity of surface water and groundwater resources. More specifically, this review addresses: the lack of geochemical testing information; leaching of contaminants from wastes and to groundwater; baseline water quality; and issues related to monitoring, additional evaluation, and water treatment.

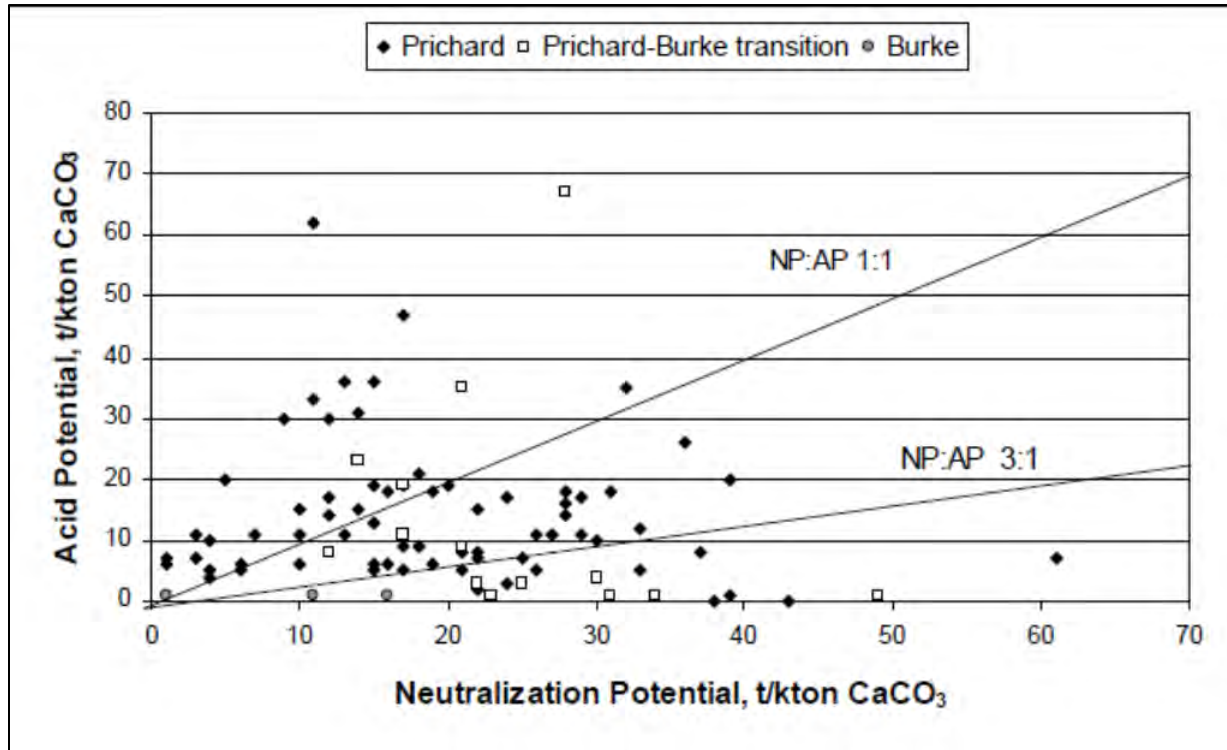
The review concludes that the Montanore Project is not ripe for permitting of the entire project, largely because of the lack of usable geochemical data on which to base assumptions about potential adverse effects to groundwater and surface water resources. In addition, non-degradation limits for arsenic could be exceeded in groundwater under the tailings impoundment under the preferred alternative.

### II. Lack of adequate and reliable geochemical testing information

The SDEIS and associated studies recognize that there are very few geochemical studies on Montanore ore, waste rock, and tailings (Enviromin, 2007, p. 2). While 190 acid-base accounting (ABA) tests have been performed on material from the Montanore Project, none were conducted in the tailings material (Enviromin, 2007, Table 3-1). The ABA results indicate that the majority of the waste rock has an AP:NP ratio <3 and is potentially acid generating (Figure 1). Only 69 whole rock tests were conducted, and the vast majority (66) only determined lead. Only 4 short-term leach tests were performed, and they used methods (Toxicity Characteristic Leaching Procedure, or TCLP; and the EP Toxicity test) that are no longer in general use for mined materials.

Even though Montanore deposit, at 135 million tons, is over twice as large as the Troy deposit – and rock cores were available for kinetic testing – only five long-term leach tests were conducted, and none used Burke waste rock or Montanore tailings material. The limited kinetic testing was conducted almost 20 years ago. The five tests lasted for 20 weeks, and minor and trace constituent concentrations were determined only during two of the 20 weeks (weeks 6 and 12). Recent Nevada Bureau of Land Management (BLM; Nevada BLM, 2010) guidance recommends that 20 weeks is the absolute minimum for kinetic testing, but tests could last a year or longer. Therefore, very limited site-specific information on the long-term environmental behavior of contaminants is available for the Montanore Project.

Figure 1. Acid production potential and neutralization potential for Montanore Project waste rock samples.



Source: Enviromin, 2007. Samples above the NP:AP 3:1 line are considered potentially acid generating.

As with the Rock Creek Project, the Montanore Project leans on geochemical testing and mine water composition from the Troy Mine in Montana. I summarized some of the important differences between the Rock Creek Project and the Troy Mine in an earlier report prepared for the Rock Creek Alliance (Maest, 2002). Although there are a number of similarities, the Montanore SDEIS plays down one of the most important differences – the Montanore deposit has approximately twice as much ore as the Troy deposit (Balla, 2000), and that much more waste will be produced. The comparisons between Rock Creek and Troy or Montanore and Troy presuppose that adequate sampling has been conducted at the Troy Mine. However, this is not the case. In addition to an abysmally low amount of geochemical testing data (e.g., no ABA testing of waste rock, only 4 ABA tests on tailings, no short-term or long-term kinetic testing of waste rock, no long-term kinetic testing of tailings, only one SPLP test (on tailings); see Table 3-1 in Enviromin, 2007 and Table 83 in the SDEIS), sampling of adit water and other mine pool water is similarly deficient. It is clear from the existing data that concentrations of metals, including copper and lead, in mine water are highest during spring snowmelt (Maest, 2002), but not enough samples have been collected during this time of rapidly changing concentrations to monitor peak concentrations over the years of operation. For many of these analyses dissolved concentrations were not measured, only total or total recoverable. Again, this points to a shortcoming in the monitoring data for the Troy Mine. Stream standards in Montana are based on unfiltered, total recoverable rather than dissolved concentrations (MDEQ, 2010), and the high concentrations of copper and lead in adit discharge are a

concern for surface water, especially during spring snowmelt. If Troy is a good environmental analogue for the Montanore deposit, one must also assume that high concentrations of base metals will be released under snowmelt conditions at the Montanore Project.

Comparisons of Rock Creek, Troy, and Montanore deposits rely solely on testing of the solid samples, not on the leaching characteristics. The shortcoming in this approach is that rocks with very similar percent sulfur values, ABA results, and whole rock chemistry can nonetheless produce very different leachate characteristics – even within the same formation at the same mine site. The inadequate amount of short-term and long-term leach test data at the Troy and for the Montanore Project makes it impossible to know if the Troy Mine is a good environmental analogue for the Montanore Project.

The Troy Mine has been operating for 20 years, and acid drainage has not been a problem. As mentioned in the SDEIS, the Coeur d'Alene deposits are in the same formation (Revett Fm.) and resulted from the same mineralizing events that are responsible for creating the Rock Creek, Troy, and Montanore deposits. At the Coeur d'Alene silver, lead, and zinc mines in Idaho, acid drainage is not a major environmental issue, with the notable exception of the Bunker Hill area, where pyrite gangue was backfilled into workings above the water table (see, e.g., Dames & Moore, 1991). Even though acid drainage is not a problem throughout most of the Coeur d'Alene area, leaching of high concentrations of lead, zinc, cadmium and other metals continues to occur throughout the basin, and high concentrations of these metals are discharging from mine adits and in seeps associated with tailings and waste rock disposal areas (Dames & Moore, 1991). Although the majority of the deposits were extracted under historic mining conditions and the area is much larger, the seeps and mine water chemistries do demonstrate that elevated concentrations of known aquatic and drinking water toxins can and have leached from rocks with similar lithology and mineralogy.

*Summary: The SDEIS and associated studies emphasize the similar geology, stratigraphy, and mineralogy of the Troy and Montanore deposits. These similarities could make the Troy an acceptable geologic and geochemical analogue for the Montanore deposit, but the SDEIS has failed to show that the Troy Mine is a good environmental analogue for the Montanore deposit. The paucity of geochemical testing results and the inadequate monitoring of mine water at the Troy Mine undermine its use as a true environmental analogue for any proposed mining project.*

### **III. Assumptions about leaching of acid and other contaminants from wastes, and potential effects to groundwater resources**

#### *Acid-Base Accounting Measurements and Assumptions*

Enviromin (2007) states that acid will not be generation from most copper-iron sulfides in the Montanore deposit, with the exception of chalcopyrite ( $\text{CuFe}_2\text{S}_3$ ), yet they provide no supporting evidence for this statement. Enviromin (2007, p. 17) further suggests that the acid production values for the Montanore Project should be reduced to account for the presence of copper sulfide minerals that do not produce acid. Plumlee (1999) and the GARD guide (2011), which is cited in the DSEIS, state that bornite ( $\text{Cu}_5\text{FeS}_4$ ), one of the primary copper ore mineral at Montanore (DSEIS, p. 200), will produce acid when oxidized by oxygen. Chalcopyrite is not a primary copper mineral in the Montanore deposit, and it

produces acid only when oxidized by ferric iron, which is more likely to exist under very low pH conditions (Plumlee, 1999; DSIES). The ABA results for Troy and Rock Creek in Enviromin (2007; Figure 3-4) were adjusted to lower acid production values, using an assumption that all sulfide was chalcocite. There is no support for this approach in the GARD Guide, an industry-sponsored website, or any other reputable source. The only adjustment made for sulfur analysis in ABA testing should be to use sulfide sulfur rather than total sulfur. However, this approach is only suggested if a good relationship can be established between the two forms of sulfur through testing. Use of total sulfur is the most conservative approach, but the amount of sulfur associated with sulfate and organic sulfur should be discounted if information on sulfur speciation is available (GARD Guide, 2011). The method used for estimating acid production potential (total sulfur vs. sulfide sulfur) for the Montanore deposit ABA results was not reported explicitly (although it appears that sulfide sulfur was used), but unless a good mathematical relationship can be established, total sulfur values should be used in ABA measurements and the interpretation of ABA testing results.

#### *Leaching from the barren lead zone*

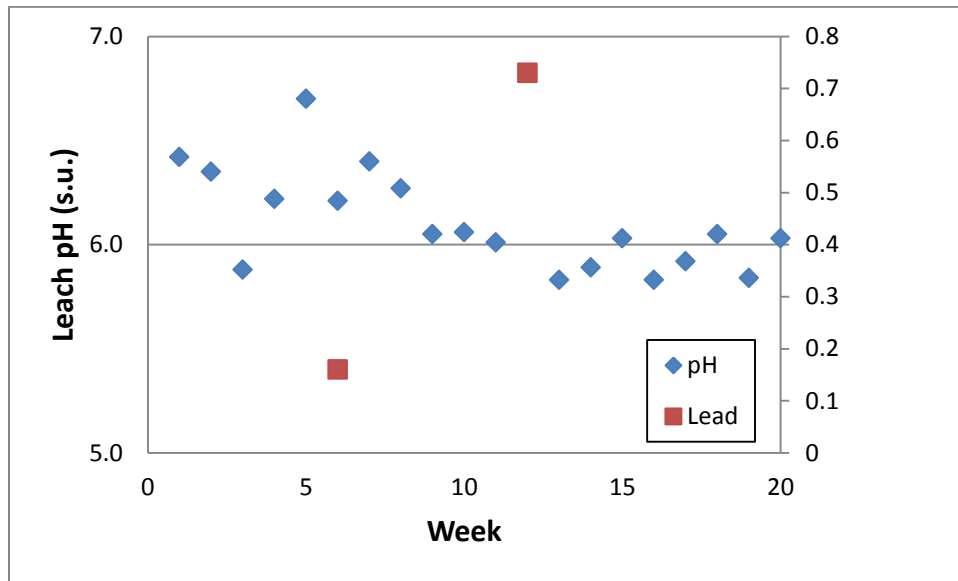
One of the biggest concerns from a water quality perspective for the Montanore Project is leaching of lead from the barren lead zone. This zone does not have a close analogue at the Troy Mine (although there is a peripheral lead zone, it is not located within two target ore zones, as in the Montanore deposit). Therefore, waste or leachate samples from the Troy Mine do not inform potential impacts related to mining of the barren zone at Montanore.

Whole rock lead concentrations in this zone are quite elevated compared to other sampled rock. For example, whole rock lead concentrations from Lower Revett Montanore barren zone rock were 350 ppm, while concentrations in Rock Creek waste rock were 13 ppm (Enviromin, 2007, App. B-1).

Lead leachate concentrations were elevated in the few short-term leach tests (only TCLP) performed on Montanore material. Weak-acid extractable leachate concentrations for Montanore rock ranged from 50 to 640 µg/L, with the highest concentrations in the “waste zone,” which includes barren (lead) zone material (Enviromin, 2007, App. B-2). Although efforts would be made to minimize disturbance of this high-lead zone during extraction, isolate the material, and store the waste underground, the likelihood that MMC can effectively do so is not currently known (Enviromin, 2007, p. 3).

The barren lead zone also has the potential to generate acid, based on limited results from one humidity cell test (HCT) with barren zone rock (SDEIS, p. 219), where the pH was consistently lower than the non-barren zone samples. Leachate pH for the 20-week humidity lead barren zone HCT (cell 4) ranged from 5.83 to 6.70, while pH values of the other four cells were generally above 7 and as high as 9 (Geomatrix, 2007, Table B-3). The two lead concentrations in the short 20-week barren zone HCT were substantially higher than any other measured lead leachate concentrations, and the limited sampling certainly did not indicate that lead concentrations were decreasing or stabilizing over time (Figure 2). More geochemical testing is needed on the barren lead zone material during the EIS phase to increase the understanding of its environmental behavior and improve waste management approaches.

**Figure2. Humidity cell test results (pH and lead) for Montanore barren lead zone sample.**



Data source: Geomatrix, 2007.

*Leaching of arsenic from tailings to groundwater*

According to a 20-year old order from the Montana Board of Health (SDEIS, Appendix A), and in response to a petition by Noranda, degradation of existing groundwater and surface water quality around the Montanore Project is allowed, up to certain limits, for a number of constituents that could derive from leaching of Montanore mined materials. Tables 103 and 104 in the DSEIS list the allowable limits and standards for surface water and groundwater, respectively. As shown in Table 1, the Board of Health and Environmental Sciences (BHES) order allows substantially higher concentrations of chromium, copper, zinc, and nitrate in groundwater than it does in surface water. In areas where groundwater discharges directly to surface water, such as areas downgradient of the proposed Poorman Impoundment but upgradient of Little Cherry, Poorman, and Libby creeks, groundwater limits should be established that are also protective of aquatic biota in surface water under low-flow stream conditions.

Unlike the limits set for a number of potential contaminants from the Montanore Project, arsenic concentrations are not allowed to increase in groundwater, unless MMC obtains an agreement from the Montana DEQ (DSEIS, Section 3.13). Using a mass-balance calculation, the DSEIS predicts that the arsenic concentration in groundwater under the impoundment under Alternative 3 will be <0.003 mg/L - (DSEIS, Table 108) under operational and post-closure conditions. The mass-balance calculation for surface water (same applies to groundwater) is described as follows in the DESIS (p. 309): "...a representative wastewater quality at an estimated flow rate was mixed with a representative surface water quality at an estimated flow rate to estimate a final surface water concentration." The baseline groundwater concentrations for arsenic are also reported as <0.003, and there is considerable uncertainty in the amount of arsenic that would leach to groundwater from the tailings impoundment.

In fact, the mass-balance modeling predicts that tailings leachate will actually slightly dilute arsenic concentrations in groundwater.

**Table 1. Comparison of BHES non-degradation limits for surface water and groundwater near the Montanore Project, Montana.**

Parameter	Groundwater Limit (mg/L)	Surface Water Limit (mg/L)
Chromium	0.02	0.005
Copper	0.1	0.003
Zinc	0.1	0.025
Nitrate + Nitrite as N	10	1

Source: SDEIS, Tables 103 and 104.

Figure 3. Results of mass-balance modeling of groundwater under the Poorman tailings impoundment under post-closure conditions.

**Poorman Impoundment Area Well Data Used for Existing Conditions  
Alternative 3**

**Mass Balance Calculations for groundwater below TI**

Parameter	Existing Groundwater Conditions		Representative Tailing Water Input from Seepage		Projected Final Mixing Concn.		Exceedance	Groundwater Standard or BHES Order Limit (mg/L)
	Conc. (mg/l)	Flow (gpm)	Conc. (mg/l)	Flow (gpm)	Conc. (mg/l)	Flow (gpm)		
TDS	60	41	245	5	80	46		200
Nitrate	0.070	41	13	5	1.5	46		10
Antimony	<0.0030	41	0.015	5	<0.0043	46		0.006
Arsenic	<0.0030	41	<0.0021	5	<0.0029	46		0.01
Cadmium	<0.00010	41	<0.00098	5	<0.00020	46		0.005
Chromium	<0.00074	41	<0.0010	5	<0.00077	46		0.02
Copper	<0.0012	41	0.024	5	<0.0037	46		0.1
Iron	<0.010	41	<0.067	5	<0.016	46		0.2
Lead	<0.00028	41	<0.0025	5	<0.00052	46		0.015
Manganese	<0.077	41	0.54	5	<0.13	46	True	0.05
Mercury	<0.000030	41	<0.000055	5	<0.000033	46		0.002
Silver	<0.00050	41	<0.00050	5	<0.00050	46		0.1
Zinc	<0.0064	41	<0.010	5	<0.0068	46		0.1

Source: DSEIS, Appendix G, p. G-42.

There is some evidence that arsenic concentrations are elevated under the Troy tailings impoundment. Table 3-11 in the Troy Mine Revised Reclamation Plan Draft EIS shows the typical water quality of wells installed in the tailings impoundment area. Wells MW-1 through MW-4 are completed in the deep aquifer, but wells MW 95-4 and MW 95-5 are shallow wells completed 10 to 15 feet below ground surface. The wells are located near the toe of the tailings dam, generally west and south of the

impoundment. This table shows no exceedence of groundwater standards, but arsenic was not included in the list. Appendix H of the Troy DEIS (Hydrometrics, 2001) does show arsenic concentrations in these wells (Tables 1 and 2 in App. H), and concentrations in the shallow wells (MW 95-4 and -5) were 0.003 mg/L in Sept 2000 and 0.008 and 0.004 mg/L in March 2001.

As noted earlier in this memorandum, there are no ABA, whole rock/total metals, TCLP, EP Toxicity, SPLP, or kinetic leach testing data for the Montanore tailings. Only one metallurgical test was conducted on Montanore tailings, and only major element chemistry was reported (see Enviromin, 2007; Table 3-3). In a total contradiction of these results, the SDEIS (p. 173) states, "The waste material (tailings) has been analyzed using the best conventional sampling methods and analytic techniques to determine its chemical and physical stability characteristics."

According to tables in Appendix G of the DSEIS, the arsenic concentrations in baseline groundwater are <0.003 mg/L, representative tailings seepage is predicted to have an arsenic concentration of <0.0021 mg/L, and concentrations of arsenic in groundwater under the impoundment during mining and under post-closure conditions would be <0.0027 and <0.0029 mg/L (DSEIS, Appendix G. pgs. G-41 and G-42; also Table 108 in DSEIS). The basis for the "representative" arsenic concentration in tailings seepage is not discussed in the DSEIS or Enviromin (2007). This value is below the current reporting limit for arsenic (0.003 mg/L), below the measured arsenic concentration in Troy tailings impoundment water of 0.02 mg/L total arsenic and <0.005 mg/L dissolved arsenic (Enviromin, 2007; Table 3-3),

Enviromin (2007) states that long-term average concentrations for the Troy tailings water, as summarized in Table 3-3, are appropriate to use as mass loading modeling inputs for tailings characterization, yet it appears that lower values were used in the modeling effort. Under operational conditions, 25 gpm of tailings leachate are expected to reach groundwater, while only 5 gpm are predicted to reach groundwater under post-closure conditions (DSEIS, Appendix G. pgs. G-41 and G-42).

With all the uncertainty noted above and the lack of adequate geochemical testing of the tailings, especially leach tests, and the noted detections of arsenic under the Troy tailings impoundment, it is not supportable to assume that a 1-ug/L difference between background groundwater arsenic concentrations and tailings leachate concentrations meets non-degradation requirements. The results of the mass-balance modeling show that tailings leachate would dilute arsenic concentrations in groundwater under the proposed Poorman impoundment. This is an unsupportable outcome, and the mass balance modeling should be rerun using more realistic arsenic concentrations in the tailings leachate, including seasonal higher concentrations of arsenic in tailings impoundment water.

*Summary: The DSEIS and associated documents made a number of assumptions that will underestimate the potential of the Montanore deposit to generate acid and leach contaminants. Enviromin (2007) proposes to reduce the measured acid generation potential by assuming no copper sulfide minerals will produce acid. Such an approach is not based in empirical evidence and should not be used. Mass-balance modeling of groundwater concentrations under the tailings impoundment for the DSEIS found that tailings seepage will actually dilute ambient arsenic concentrations in groundwater. This unsupportable result derives from assuming that no arsenic will leach from tailings and does not match with detected*

*arsenic concentrations below the Troy tailings impoundment. Limited leach tests conducted on the barren lead zone demonstrate that elevated concentrations of lead and depressed pH could result from mining of this material. These results should be incorporated in mass-balance modeling to reach more realistic estimates of mining water quality.*

#### **IV. Assumptions about baseline water quality**

For the mass-balance modeling in the DSEIS, baseline groundwater quality estimates were based on measured concentrations from 2005 to 2009 after removing data outliers. For water quality parameters with no below detection limit values, the median concentration was used. The DSEIS does not explain how below-detection values are treated, except for parameters with >70% below detection limit values – and in this case they use the detection limit (DSEIS, p. 312). This approach will overestimate groundwater concentrations, especially if the detection limits are unreasonably high.

For baseline groundwater quality below the land application and disposal area (LAD), only one well was used, WDS, 1V (SDEIS, Appendix K, Table K-4). They used 16 samples from the well, and for arsenic 13 were below detection. Therefore, they used the detection limit value as the groundwater concentration, which is <0.003 mg/L. The data for the LCC Area Well (LCTM-8V) are shown in the next set of columns, and 100% of those analyses were below detection, which is shown as 0.003 mg/L. This is a fairly high detection limit for arsenic, which should be less than or equal to 0.001 mg/L. If a better analytical method were used, the detection limit would be lower, and there would either be more above dl values, or the final value used for baseline groundwater would be lower.

*Summary: Baseline water quality estimates suffer from limited monitoring data and poor analytical detection limits. Baseline concentrations for certain parameters could be substantially lower than estimated in the DSEIS if detection limits were closer to modern, easily achievable lower values. Analytical detection limit strongly affect baseline water quality estimates, which in turn affect the non-degradation analysis and the mass-balance calculations. Because of these issues, the non-degradation analysis and mass-balance modeling results could underestimate the potential for mine water and waste leachate to adversely affect groundwater and surface water resources.*

#### **V. Issues related to monitoring, additional evaluation, and water treatment**

This section addresses a number of issues related to environmental monitoring, characterization, and mine water management.

There is very little mention of adaptive management in the DSEIS or the monitoring plan (DSEIS, Appendix C). An adaptive management plan should be required with the EIS that includes specific actions that will occur if unexpected (but predictable, based on other mines) issues arise.

The reporting limit for arsenic in the monitoring plan should be lowered to at least 1 µg/L to allow detection of arsenic at lower levels, as discussed in the previous section, and to establish an action level for arsenic, which does not currently exist for groundwater (DSEIS, Appendix C, Table C-15).



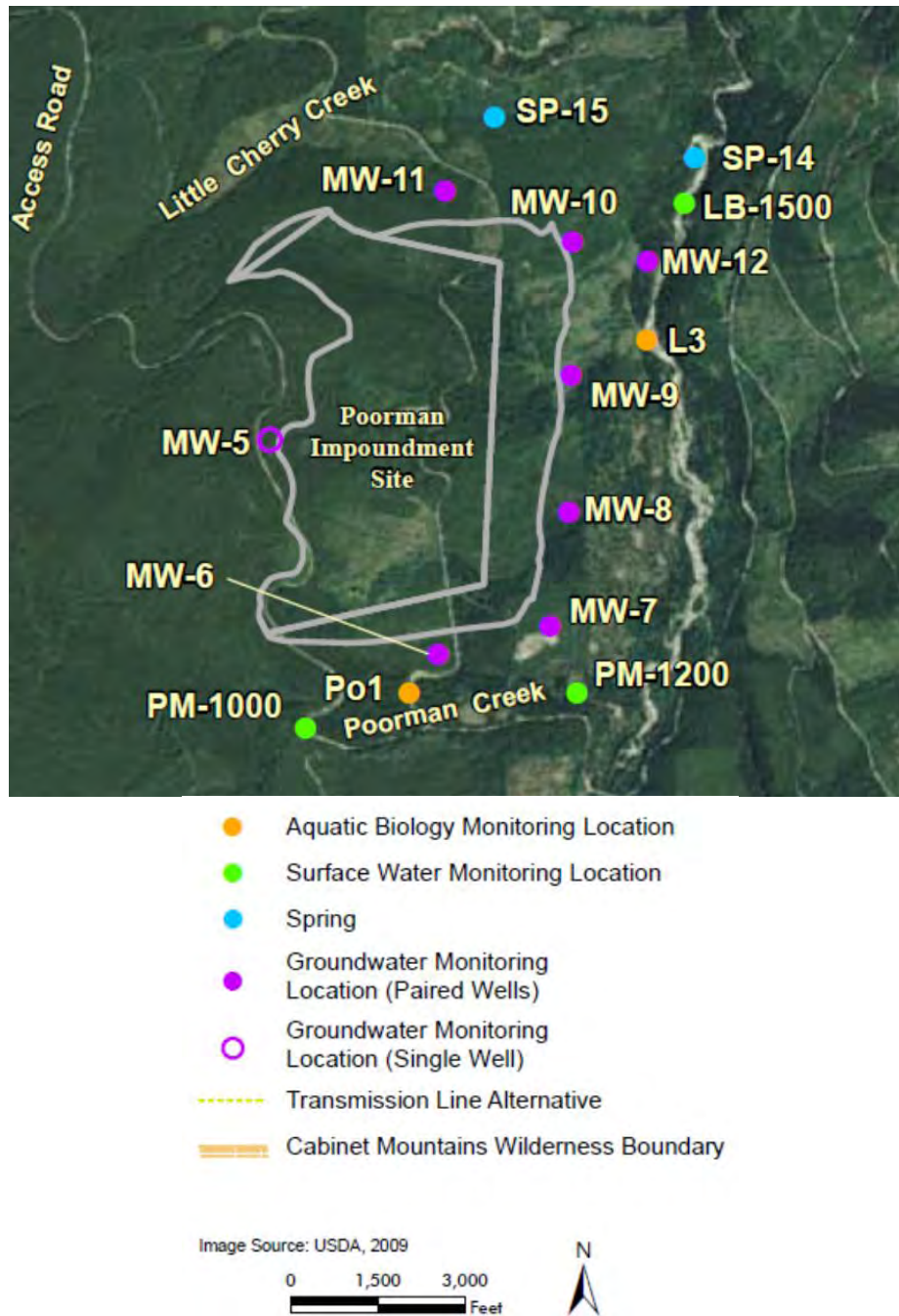
Even though the Rock Creek ROD (U.S. Forest Service, 2003) assumes that perpetual water treatment would be needed at the geochemically analogous Rock Creek Project, very few details on water treatment are included in the Montanore DSEIS. For the Montanore Project, the timeframe for water treatment and management is estimated to be decades (DSEIS, p. 28). The DSEIS states that water would be treated to remove nitrate and ammonia, but there are no plans to treat for metals removal (DSEIS, p. 52). Leaching of the barren lead zone could easily produce water that would require metals removal before disposal. Considering the elevated concentrations of lead in leachate samples from this material, and uncertainties about MMC's ability to handle the material in a way that would ensure environmental protection, detailed plans for a metals removal treatment plant should be prepared at the EIS stage.

The proposed groundwater monitoring plan does not include any groundwater monitoring between the Poorman impoundment and Little Cherry Creek (Figure 4). More multi-level monitoring wells that could easily be converted to pumpback wells (4-in diameter wells) should be proposed on the northern and western sides of the impoundment.

Appendix C of the DSEIS contains a decision matrix for geochemical sampling and analysis (Figure C-1). Text inside the boxes in the flowchart is indicated below by quotation marks. The figure contains a number of weak approaches and vague statements, including:

- "Evaluate need for mineralogical analyses based on geological observations."  
Mineralogic analyses should be required and used to help interpret the static and kinetic testing results – there are no arrows from this box to any interpretation. Mineralogic analysis should be required for each kinetic column test and generally for interpretation.
- "Identify key constituents (potential exceedences)" only refers to whole rock data. Key parameters of concern should also be identified after short-term and longer term leach testing.
- "Review statistics with baseline data to determine adequacy of sampling" It is not clear what this statement refers to. According to Enviromin (2007, p. 3), baseline data to be used in the analysis include mineralogy, whole rock geochemistry, acid base accounting tests, HCTs, in-situ monitoring of water quality, and metallurgical data. These data will come not only from Montanore but also from the adjacent deposit Rock Creek, as well as its geological analog at Troy.
- "Develop sampling or compositing plan for low-S SPLP tests." Why are short-term leach tests (SPLP) only proposed for low-sulfur wastes? SPLP tests will often underestimate contaminant concentrations in underlying groundwater and underestimate long-term leachate concentrations (Townsend et al., 2006; Maest et al., 2005).
- "Evaluate need for WTP." A detailed plan for a water treatment plan should be prepared before mining begins at the EIS stage.

Figure 4. Proposed environmental monitoring for the Montanore Project.



Source: Adapted from DSIES, App. C, Figure C-7.

Table C-6 of the DSEIS, Appendix C, presents the proposed additional geochemical testing during the evaluation phase. Some comments on the plan follow:

- No kinetic testing is proposed for the Burke, Revett ore, or tailings. The wastes and ore cannot be assumed to have a low acid-generation potential and contaminant leaching

potential based on the few geochemical tests performed to date. Long-term kinetic testing should be conducted on each geochemical test unit (see Maest et al., 2005) to evaluate the potential for neutral leaching of contaminants.

- Only two kinetic tests are proposed for the barren lead zone: one unsaturated and one saturated. This zone could leach very high concentrations of lead and other metals over time. More samples are needed for long-term leach testing to obtain an estimate of the range of leachate concentrations, especially an estimate of maximum leachate concentrations of lead, to assist in developing waste management strategies.
- SPLP tests can be used to estimate short-term runoff leachate concentrations, but SPLP results should not be used for determining long-term placement of wastes; for this, long-term kinetic testing is needed.
- SPLP tests for tailings are not appropriate as the only measure of potential seepage concentrations. Multiple kinetic tests are needed for all elements of the Montanore deposit listed as column headings in Table C-6: Prichard, Burke, Revett Halo (non-lead), Revett Barren Lead, Revett Ore, and Tailings.
- Composite testing is most appropriate for homogeneous materials such as tailings, or when average characteristics of a well-defined unit are of interest. For all other waste types, especially waste rock, sub-units within the three waste rock sources (i.e. Revett halo, Revett barren zone, Prichard, Burke) should be identified based on mineralogy and weathering characteristics, and composite created within those geochemical testing units (Maest et al., 2005).

*Summary: The conceptual monitoring plan contains many shortcomings that, if implemented without modification, will prevent an adequate understanding of the environmental behavior of mined materials at the Montanore Project. Improvements should include more proposed geochemical testing, especially long-term kinetic testing of all geochemical units, improved detection limits, and a more detailed plan for a water treatment plant.*

**Recommendations:**

1. Unless a good mathematical relationship can be established between total sulfur and sulfide sulfur, total sulfur values should be used in ABA measurements and the interpretation of ABA testing results.
2. Mass balance modeling should be rerun using more realistic arsenic concentrations in the tailings leachate, including seasonal higher concentrations of arsenic in tailings impoundment water.
3. Non-degradation limits for groundwater that is immediately upgradient of surface water should be lowered to ensure protection of surface water resources.
4. An adaptive management plan should be required with the EIS that includes specific actions that will occur if unexpected (but predictable, based on other mines) issues arise.
5. The reporting limit for arsenic in the monitoring plan should be lowered to at least 1 µg/L to allow detection of arsenic at lower levels for baseline monitoring.
6. Detailed plans for a metals removal treatment plant should be prepared at the EIS stage.

7. More multi-level monitoring wells that could easily be converted to pumpback wells (4-in diameter wells) should be proposed on the northern and western sides of the tailings impoundment area.
8. Mineralogic analysis should be required for each kinetic column test and generally for interpretation.
9. Key parameters of concern should also be identified after short-term and longer term leach testing.
10. Long-term kinetic testing should be conducted on each geochemical test unit, especially for the barren lead zone, to evaluate the potential for neutral leaching of contaminants and maximum potential leachate environmental concentrations.

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