# Report as of FY2008 for 2007MT152B: "Sediment and Heavy Metals Source Determination and Reduction at a Reclaimed Abandoned Mine Site, Alta Mine, Jefferson County, MT"

## Publications

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## Report Follows

# SEDIMENT AND HEAVY METALS SOURCE DETERMINATION AND REDUCTION AT A RECLAIMED ABANDONED MINE SITE
## ALTA MINE, JEFFERSON COUNTY, MT

by

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# ABSTRACT

Abandoned hardrock metal mines can have an antagonistic effect on soil productivity, vegetation, and water quality. Specifically, abandoned mines that actively generate acidity in soil are phytotoxic due to low pH and increased bioavailability of heavy metals. Arsenic concentrations in mine soils are often elevated, but may not be as mobile as heavy metals at low pH. Acid mine drainage migration from abandoned mines is problematic because it leads to water quality impairments that limit water use for certain activities (i.e. stock watering and irrigation). In this work, a previously reclaimed abandoned lead and silver mine (Alta Mine Jefferson County, MT) was characterized for its persistent impacts on soil, vegetation, and water quality. A progressive monitoring effort linked offsite water quality impacts to deep underground mine workings, shallow ground water, and metalliferous soils found at the Alta mine. Water quality impacts were of importance because they significantly hindered the attainment of Total Maximum Daily Load (TMDL) allocations that were established shortly before this study. Vegetative cover on the previously reclaimed portion of the mine was measured in 16 transects in conjunction with 30 soil pits excavated on the reclaimed site. By regression and analysis of variance, sparse vegetative cover was significantly ($p<0.1$) linked to pH and acid generation potential. To overcome acidic soil conditions, lime and compost amendments were tested on site. The amendments significantly ($p<0.1$) neutralized soil acidity; however, a corresponding increase in vegetative cover was not observed. Erosion of the bare unstable slopes caused greater than anticipated seed bank loss that precluded vegetation establishment. It is anticipated that successful establishment of a dense vegetative cover on the abandoned mine could prevent erosion and water quality impacts due to sedimentation. Vegetation may also have minor impacts on landscape sources of arsenic and heavy metals that were identified in the study; but the most significant source of water quality impairment, deep underground mine workings, will persist at the Alta mine under any land treatment.

# INTRODUCTION

The science of land rehabilitation integrates principles of soil physiochemistry, plant physiology, and water interactions. Rehabilitation implies the return to a previous condition; thus, scientists must examine soil characteristics, plant growth requirements, and the hydrologic regime of disturbed lands if they are to return them to their original state. In application, land rehabilitation is the practice of attenuating earth disturbances and revegetating them by ameliorative, adaptive, or agricultural treatment (Ye et al. 2000, Tordoff et al. 2000). Mining, timber harvest, agriculture, road building, and other construction activities commonly result in land disturbances that require rehabilitation. Watershed processes are explicitly linked to land use; thus, the same suite of disturbances that often require land rehabilitation may also lead to impairments in bodies of water. Poor agricultural practices, such as over-fertilization or cattle feeding operations, may lead to excess nutrient loads in streams; also, mine waste may be an important non-point source of stream impairment by heavy metals. Nearly all anthropogenic land uses, including those previously named, are potential sources of excess sediment. The interconnectedness between land disturbance and watershed impairment is important for scientists, land managers, and regulatory agencies to understand if they are to implement effective land rehabilitation and watershed restoration strategies.

## Problem Statement

The specific problems addressed in this research are the factors that limit the rehabilitation of lands disturbed by historic, acid-producing, hard rock metal mines and the persistent impact that these mines have on water quality on the watershed scale. Sulfide mineral oxidation resulting in low pH and increased heavy metal toxicity reduces soil productivity and plant vigor; however, it is not the only process that limits the successful rehabilitation of mined lands. Soil texture, drought, erosion, lack of organic matter (OM), low nutrients, and steep slopes exemplify the universe of limitations on effective mine rehabilitation. In addition, site specific limitations may occur due to abandoned mine features (i.e. adits, shafts, dumps), site accessibility, and budget constraints. When chemical, physical, logistical, and fiscal limitations are not overcome, land rehabilitation is not effective; thus, the potential exists for water quality impairments downstream. Impairments such as sediment, heavy metals, arsenic, sulfates, and pH may limit water use for drinking, swimming, irrigation, fish and other organisms.

# Water Quality in Mined Watersheds

## Acid Mine Drainage Impairments

Acid drainage is a term assigned to solutions produced from sulfide mineral weathering that are characterized by low pH and often contain high levels of potentially toxic trace elements (e.g. Cd, Zn, Pb,Cu, As). Mining disturbances are a common source of exposed sulfide minerals; so acid drainage is frequently referred to as acid mine drainage (AMD). A copious amount of work has been published regarding the formation of AMD. Comprehensive manuscripts (OSU 1971, Evangelou 1995) discuss the production chemistry, kinetics, role of microorganisms, and control of AMD. Pyrite ($FeS_2$) is the most common acid-producing Fe-sulfide mineral and is found in many geological environments (OSU 1971). The oxidation of pyrite by $O_2$ and subsequent acid production is the summation of 3 intermediate reactions: 1) dissolution of $FeS_2$; 2) oxidation of $Fe^{2+}$ to $Fe^{3+}$; and 3) hydrolysis of $Fe^{3+}$. Reactions 1 and 3 produce a total of 5 moles of $H^+$; while 1 mole of $H^+$ is consumed in reaction 2, resulting in 4 moles of acidity produced for every mole of $FeS_2$ reacted. Also, at low pH (below 3.5) ferric iron ($Fe^{3+}$) oxidation of pyrite can accelerate acid production to a rate of 16 moles of acid per 1 mole of pyrite reacted (Jennings and Dollhopf, 1995). Reactions assigned to AMD production, though they are charge and mass balanced, have no molecular, mechanistic, or rate reaction meaning because pyrite oxidation involves many other metastable species (Evangelou 1995).

The products of AMD have a significant impact on water quality that is not limited to the spatial confines of mine disturbances. Rather, historic metal mine disturbances produce water quality impairments that persist throughout entire watersheds (Sullivan et al. 2000, Caruso 2003, Herr et al. 2003, U.S. EPA 2006). Sediment, metal hydroxides, and toxic trace elements (e.g. As, Cd, Cu, Pb, and Zn) from abandoned mine sources have been shown to cause impairments in fluvial ecosystems (Henry et al. 1999, Soucek et al. 2000a, Soucek et al. 2000b, Niyogi et al. 2001, Niyogi et al. 2002).

## AMD Characterization

Products of AMD and evidence of their negative ecological impact may extend to all reaches within a watershed; however, several factors make it difficult to characterize and/or model AMD systems. Heavy metal and As transformations and pH shifts can occur rapidly in AMD systems at the confluence of waters not impacted by AMD or by rapid dilution during snowmelt or rainfall events (Brooks et al. 2001, Sullivan et al. 2001, MacDonald et al. 2007). Mixing with alkaline or neutral streams causes distinct changes in the chemical signature of AMD (Broshears 1996). Metal hydroxide precipitation in mixing zones is important in determining downstream aqueous chemical composition.

Heavy metal concentrations decrease in mixing zones due to coprecipitation with metal hydroxides (Runkel and Kimball 2002, Butler II 2006).

In addition to chemical changes brought on by hydrologic influences, the character of AMD impacted streams is dependent on biological and physical factors. Reduced stream flows due to evapotranspiration by riparian vegetation can amplify heavy metal concentrations, as photoreduction and microbial reduction of Fe(III) to Fe(II) is increased during periods of low flow (Sullivan and Drever 2001, Butler II 2006). Concentrations of heavy metals in the water column increase when Fe is reduced, either microbially or by photoreduction, because of decreased adsorption on Fe-hydroxides (Sullivan and Drever, 2001).

Abandoned metal mines that produce AMD have the potential to discharge As and heavy metals from a variety of sources. Heavy metals and As may be transported in leachate from tailings, waste rock, and contaminated soils or may be discharged from mine workings such as portals, shafts, and adits. For this reason, site specific factors are possibly the most important in determining how products of AMD are transported in streams. Kimball et al. (2002) identified numerous inflows springs, bogs, diffuse sources, seeps, and adits along a 12 km reach of Cement Creek, Colorado. Depending on the nature of each source, inflows either attenuated or contributed to the transport of heavy metals and other products of AMD. Recent monitoring and modeling efforts concluded that abandoned mine features (point sources) were likely the most significant sources of AMD products (Runkel and Kimball 2002, Kimball et al. 2002, Caruso 2003, and Herr et al. 2003).

## Sediment from Abandoned Mines

As early as the beginning of the 20th century, impacts of excess sediment from mining sources were assessed (Gilbert 1917). Hydraulic gold mining led to the production of nearly 8 billion $m^3$ of mining debris in the Yuba, Bear, and American river watersheds, some of which negatively impacted the San Pablo, San Francisco, and Suisun Bays. The spatial extent of sediment delivery from abandoned hard rock mines is not as vast as that from large hydraulic mine sources but resulting overbank deposits and silted stream channels are the same. Soda Butte Creek drains the historic New World gold mining district near Yellowstone National Park. A tailings impoundment failure in 1950 resulted in exposed over-bank tailings deposits as far as 25 km downstream (Marcus et al. 2001). The source of the sediment was unmistakably linked to the tailings impoundment failure by sand-sized pyrite grains, strong oxidation, and sparse vegetative cover. The tailings also filled abandoned channels as deep as 0.7 m.

Bonta (2000) monitored 3 Ohio watersheds for suspended sediment in their natural condition, during active mining and reclamation, and after final reclamation. Maximum suspended sediment concentrations were the highest in each watershed during mining and reclamation activities; while, minimum concentrations were always in the undisturbed condition. In one watershed, average post reclamation suspended sediment loads were decreased by 75% of the load in the undisturbed condition. The decrease was attributed to increased vegetative cover and not lack of stream power because average stream flows had actually increased following reclamation.

## Limits on Soil Productivity and Vegetation Establishment

Establishing a vegetative cover on land disturbed by historic hard rock mining is an effective way to reduce erosion, limit heavy metal and As leachate, and improve water quality within a watershed (Brown 2005); however, there are many chemical and physical limitations to vegetation establishment on these sites.

### Mine Waste

Tailings deposits and waste rock dumps are common remnants of abandoned mine disturbances that produce AMD. Tailings are the by-product of the ore milling and concentrating process; as such, they are generally made of fine-grained material (Tordoff 2000). Residual sulfide ore minerals and metals content is high in historic mine tailings because of inefficiencies in past separation processes (Alloway 1995). Poor vegetation establishment on fluvially deposited tailings on the banks of the upper Clark Fork River and on the historic Keating gold and copper mine tailings in Montana are just 2 of the countless examples of the phytotoxicity of tailings (RRU-BRI 2004, Neuman et al. 2005). Further, pyrite oxidation occurring as deep as 1 m within mine tailings heaps may lead to a decline in vegetative cover where increased sulfur oxidizing bacteria are present and neutralization potential is depleted (Schippers et al. 2000).

In contrast to tailings deposits, waste rock piles are typically composed of larger rock fragments that have been removed from their native location but have not been milled or processed. Waste rock piles also have the potential to produce AMD by sulfide mineral oxidation (Lefebvre et al. 2001, Sracek et al. 2004). It cannot be assumed that tailings or waste rock pile removal will result in successful remediation of AMD. As a result of AMD solution leaching from waste rock, native soils beneath waste rock piles may be enriched with metals in soluble secondary minerals (Zhixum 1997). Low pH in mine waste substrates and soils in contact with mine waste is a crucial limiting factor to vegetation establishment.

### Revegetation and Rehabilitation

Rehabilitation is successful when acid-metalliferous substrates (i.e. impacted soils, tailings, waste rock) cease to inhibit plant growth and are no longer mobile in the environment. In situ amendment of substrate that reduces toxic trace element bioavailability and raises soil pH has emerged as a commonly practiced treatment in abandoned acid-metal mine rehabilitation

Amendments are chosen for the ameliorative effects that they will have on soil and substrate character. Organic amendments such as biosolids, peat, and compost are added to substrate to increase water and nutrient holding capacity, provide a source of nutrients, increase cation exchange capacity, and form organic complexes with potentially toxic trace elements (Tordoff 2000). Calcium carbonate (agricultural lime), steel sludge, furnace slag, zeolites, and red mud by-product from alumina production are among the host of inorganic materials available for acid-metalliferous substrate amendment (Chen 2000, Friesl et al. 2003). The principle of inorganic amendment addition is to reduce heavy metal availability by raising soil pH and binding or precipitating heavy metals. Recent work has confirmed the effectiveness of both organic and inorganic amendments at increasing productivity of abandoned acid metal mine substrates (Friesl et al. 2003, Walker et al. 2004, Brown et al. 2005, Neuman et al. 2005, O'Dell 2007).

## Purpose of Work

The purpose of this work was to characterize a previously reclaimed abandoned Pb and Ag mine for site specific limitations on vegetative succession and sources of impairment on water quality. A vast area of sparse vegetation on the reclaimed site was studied for its soil chemical and physical composition and other environmental factors that have often been attributed to poor vegetative cover in the current land rehabilitation literature. The extent of heavy metal and As discharge from soil, shallow groundwater, and deep underground mine workings were examined through an extensive water quality monitoring effort. The water quality monitoring plan was designed such that the temporally variable contribution of contaminants from the mine could be compared to Montana's aquatic life standards (MT DEQ 2006) and recently established federally mandated maximum contaminant load allocations.

Two (2) hypotheses were tested by our site characterization:

1. Waters that drain the site are the significant source of impairment to downstream water quality in terms of total maximum As, Cd, Cu, Pb, Zn, and sediment load;
2. A chemical or physical variable or combination of physical and chemical variables consistent with acid mine disturbed lands control vegetative cover on site, despite previous reclamation attempts.

The first hypothesis was tested by direct comparison of monitored site discharge to measured background loads of As and heavy metals in receiving waters. Hypothesis 2 was tested by statistical significance (linear regression and ANOVA) of pH, soil heavy metal and As concentrations, OM, nutrients, EC, acid-base potential, and slope aspect control on vegetative cover.

Beyond site characterization, a third hypothesis tested the effectiveness of lime and compost amendments at increasing soil productivity and vegetative cover in a field experiment. A healthy vegetative cover would not only indicate effective mine rehabilitation but could potentially reduce sediment, heavy metal, and As loads from soils and shallow groundwater. The third hypothesis was:

3. Lime and compost amendments will increase soil productivity and vegetative cover on experimental treatment plots on the disturbed site.

This hypothesis was tested by the statistical significance of end of growing season soil and vegetative cover characteristics compared to end of growing season controls.

The results of this work are important to private, state, and federal stakeholders who may be responsible for the rehabilitation of similar mines in roughly the same geographic region. This work also exemplifies the complex link between water quality impairments and acid mine land disturbances. Watershed planners and conservation districts may consider the results of this work when assessing the impact of abandoned mines in watersheds of comparable size and character.

## Study Site

### Location

The study site, located approximately 30 km south of Helena, MT, consists of the abandoned Alta Mine complex, a small stream referred to as the Alta tributary, and Corbin Creek immediately above and below its confluence with the Alta tributary (Figure 2). The abandoned Alta mine complex is included in the Colorado Mining District of Jefferson County, MT (Pioneer 1994). As an abandoned mine, the Alta is not unique. In fact, it is one of an estimated 6,000 inactive mining and milling sites across Montana (Pioneer 1994).

The Alta mine complex is divided into an upper mine site and a lower mine site. The lower site is located in Section 10, T7N, R4W; and the upper site is distributed between Sections 9 and 10, T7N, R4W. The Alta tributary drains the Alta mine diagonally from SW to NE across Section 10, T7N, R4W, until it reaches a confluence with Corbin Creek in the SW1/4 of SW1/4 Section 2, T7N, R4W. The Alta tributary and Corbin Creek are found at the southern edge of the Lake Helena Watershed in the Upper Missouri River Drainage Basin. Corbin Creek drains an estimated 442 ha above its confluence with the Alta tributary. The Alta tributary subwatershed accounts for approximately 19% (82 ha) of the drainage area. The streams drain to Prickly Pear Creek, which is the largest stream in the Lake Helena Watershed.

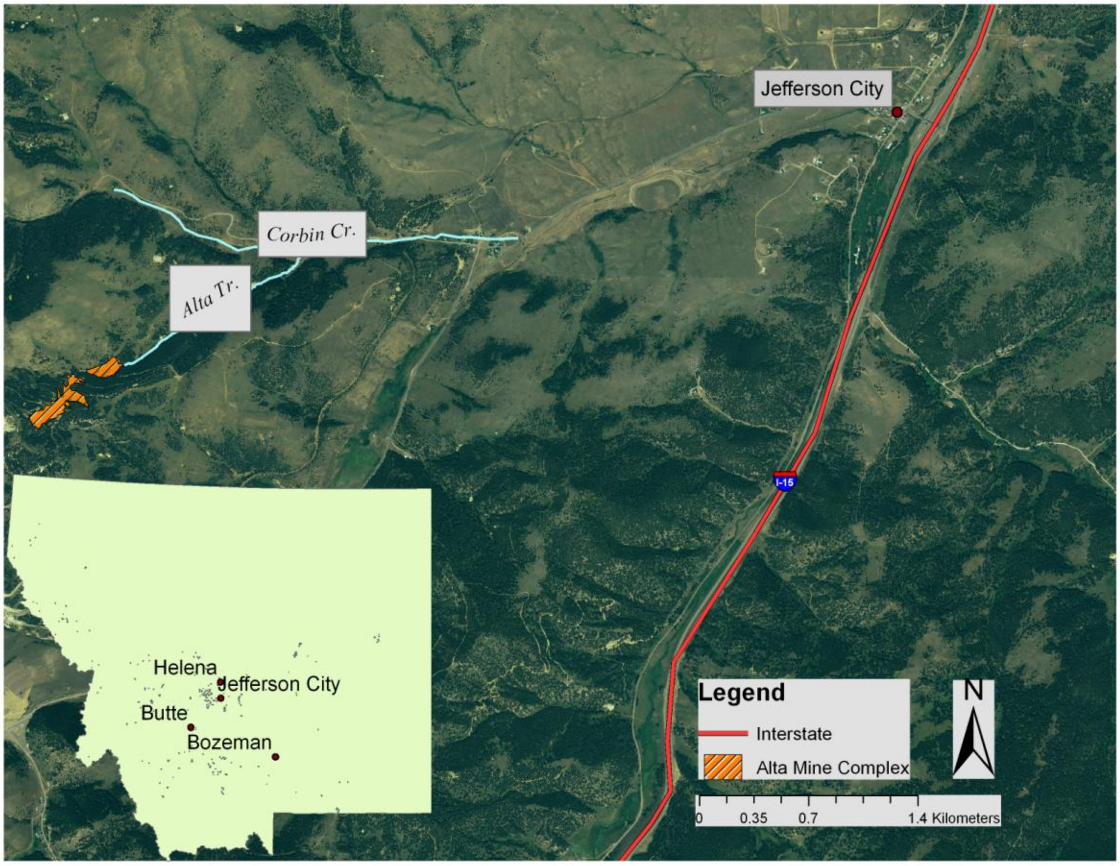

Figure 1. Alta Mine Site Location. Points on Montana inlay represent abandoned mines.

## Geology and Climate

The elevation of the study site ranges from approximately 1,500 m above mean sea level (amsl) at the confluence of the Alta tributary and Corbin Creek to 1,905 m-amsl at the summit of Alta Mountain. Snow in the winter, rain in the spring, and intermittent thundershowers throughout summer and fall account for precipitation at the site. Snow in the Alta drainage accumulates primarily on the slopes with northern aspect; while, snow on the south facing slope tends to melt or sublimate rapidly. The 30-year average cumulative annual precipitation at the nearest weather station in Boulder, MT is 28.7 cm. Cumulative precipitation for the latest complete year of record (2006) was 29.3 cm. Both Deckler (1982) and Tetra Tech (1997) suggested increased precipitation at the Alta Mine due to orographic effects; and observations made in the summer of 2007 seem to corroborate this assumption. Cumulative precipitation at the Alta Mine—approximately 190 m above the Boulder weather station—was nearly 6% greater than that observed in Boulder. Site temperature varies from extreme cold in winter to extreme heat in summer. Average Monthly temperatures (from Boulder weather station) range from -12 $^{o}$C to 0 $^{o}$C in January and from 7 $^{o}$C to 26 $^{o}$C in July (Tetra Tech 1997).

The study site is at the north end of the Boulder Batholith and Elkhorn Mountain volcanics formations. Both formations are igneous and Cretaceous in age; however, the Batholith is intrusive, while the Elkhorn Mountain volcanics are remnant of lava flows and volcanic ejecta (Veseth and Montagne 1980). Rocks present on site are representative of both formations. Quartz monzonite is the primary batholithic rock; while, rhyolite and dacite tuffs and lava flows account for much of the volcanics (Veseth and Montagne 1980, Tetra Tech 1997).

## Mining History

The Alta mine was the most productive underground lead and silver mine in the area, owing to an extensive ore body of galena (presumably argentiferous galena), pyrite, tetrahedrite, and sphalerite (Becraft et al. 1963). Ore was discovered there in 1869; and the mine operated at its greatest capacity from 1883 to 1896. Minimal amounts of silver and lead ore were produced after 1896; however, low grade ore, used as silica flux in the East Helena smelter, was mined in the 1950s. When mining ceased, the site was abandoned as a pile of waste rock, 13 levels of underground mine workings, and a series of shafts and adits that opened to the surface on the east side of Alta Mountain. Most notably, the #8 shaft was a continuous source of metals laden AMD that discharged from beneath the lower waste rock pile. The waste rock piles consisted of ore minerals sparsely and heterogeneously distributed within volcanic and batholithic rock materials. In addition to sulfide minerals from within the Alta ore body, limonite, chalcopyrite, and quartz were deposited on the lower Alta waste rock dump after exploration in a separate vein immediately to the southwest. Ore samples from this vein contained 11 % copper. Samples of crude ore from the Alta ore body were shown to contain 18 – 35 % lead; and concentrated samples had 12 % zinc.

## Reclamation History

Of the estimated 6,000 abandoned mines in Montana, a list of 270 priority abandoned mine sites was compiled (Pioneer 1994). In 1993, nearly all of the 270 mines were characterized for their risk to human health and the environment by chemical degradation and hazardous abandoned structures. Upon characterization, sites were compared and ranked according to severity of risk. Alta was ranked 17th amongst the most threatening mine sites due to metals laden AMD that exceeded nearly all of Montana’s numeric water quality standards and for its abundance of actively eroding waste rock dumps with elevated levels of Pb and As (Pioneer 1994).

Reclamation of priority abandoned mine sites is funded largely by the Surface Mining Control and Reclamation Act (SMCRA) of 1977 and is administered by the Montana Department of Environmental Quality (MT DEQ 2005). MT DEQ reclaimed the Alta mine in the summer of 1999. Work done during the reclamation effort included: removing 117, 741 $m^3$ of waste rock; transporting waste rock to a 2.3 ha repository;

reconstructing 274 m of stream channel; re-contouring the 40% slopes with terraces; and fertilizing, seeding, and mulching the re-contoured area (Tetra Tech 2002). Risk-based As, Pb, and Mn concentrations were used to determine which portions of the waste rock to remove. All of the waste rock removal and subsequent reclamation activities were done exclusively on the lower Alta mine site; while, waste rock piles at the upper mine and in the Alta tributary below the reclaimed area were left in place. The reclamation effort did not include surface water or groundwater treatment; however, a failed attempt to seal the #8 shaft was made in order to eliminate surface discharge (Tetra Tech 2002).

## Watershed Restoration

The Clean Water Act (CWA), as amended in 1977, includes programs that aim to limit the discharge of pollution from industrial point sources, regulate effluent from waste water treatment plants, and reduce nonpoint source pollution impacts (Gallagher and Friedman 2001). Section 303(d) of the CWA maintains that each state must assemble a list of waters that are impaired and the contaminants that impair them. Each state must assess contaminant sources (point and nonpoint) for a given body of water and determine the reduction in contaminants needed to support beneficial water uses. The reduced load is known as a Total Maximum Daily Load (TMDL).

Corbin Creek, the receiving waters of the Alta tributary, has been found to be impaired by As, Cd, Cu, Pb, Zn, and sediment from its headwaters to its mouth (U.S. EPA 2006). TMDLs have been established for each of these constituents, along with the current estimated daily load and the percent reduction required for TMDL attainment (Table 2). Abandoned mines are estimated to account for 59%, 99%, 96%, 86%, 98%, and 5% of the As, Cd, Cu, Pb, Zn, and sediment loads in Corbin Creek, respectively. The remainder of the current load is attributed to dirt roads, timber harvest, natural sources, and stream bank erosion. Extensive mining and reclamation activities have taken place in the Corbin Creek watershed above the Alta tributary but not below. Of the many abandoned metal mines with a potential impact on Corbin Creek, the Alta mine complex is mentioned by name in the Framework Water Quality Restoration Plan for the Lake Helena Watershed Planning Area (US EPA 2006); however, the mine’s contribution of sediment and heavy metals is not directly estimated. Moreover, the document suggests that the #8 shaft had been sealed during reclamation. While the document recognizes that limited revegetation of the Alta mine may be a source of sediment and toxic trace elements, it accounts for less than 0.5 ha of bare ground.

Table 1. Corbin Creek TMDL Attainment Goals and Estimated Current Maximum Loads (US EPA 2006).

| Constituent | Estimated Current Load (kg $day^{-1}$) | % Reduction Required | TMDL (kg $day^{-1}$) |
|---|---|---|---|
| Arsenic | 0.06 | 25% | 0.045 |
| Cadmium | 0.11 | 97% | 0.003 |
| Copper | 1.32 | 89% | 0.141 |
| Lead | 0.12 | 66% | 0.041 |
| Zinc | 72.72 | 97% | 2.045 |
| Siltation | 0.59 | 23% | 0.455 |

## Precedent Research

More than a decade before the massive abandoned mine characterization effort launched by the state of Montana, Deckler (1982) characterized soils and substrates at the Alta mine. Consistent with the efforts of the state, he found the site to include 2 large waste piles that were uninhabited by vegetation. The upper Alta waste rock pile was less acidic (pH from 5.5 to 6.2), coarse textured (and homogenous), and contained lower concentrations of potentially toxic trace elements than the lower pile. Erosion, nutrient availability, and overly drained soils were suspected to limit vegetation on the upper pile. The lower pile was finer textured and more acidic (pH=3.0). Extractable metal levels were high on the lower waste rock pile but soluble metal levels were low. There was evidence that metals had leached from upper portions of the waste profile; thus, it is possible that they reached native soils below. Deckler made several suppositions about the future reclamation of the site. He concluded that: sufficient cover soil depth must be maintained on lower slopes to sustain vegetation; upper waste rock material may be used as a barrier between lower waste rock and cover soil; and mulches would provide nutrients and decrease incident solar radiation from the bare waste rock piles. Finally, Deckler (1982) noted that tailings from the nearby Bertha mine were located directly in the Corbin Creek channel above the Alta tributary at the time of his study. These tailings were removed in 2002 in a reclamation project separate from the Alta reclamation effort (MT DEQ 2008).

Schroth (2001) and Schroth and Parnell (2005) provide a thorough account of geochemical transformations within the Alta waste rock pile prior to its removal and in the Alta channel following relocation of the pile. They found that schwertmannite precipitation was an important metal sink within the waste rock pile. It was further discovered that metal hydroxide precipitation increased upon pile removal; however, trace element concentrations did not actively decrease by coprecipitation with schwertmannite. Pre and post waste rock removal concentrations of selected trace elements in the Alta tributary immediately above Corbin Creek remained relatively constant (Figure 3).

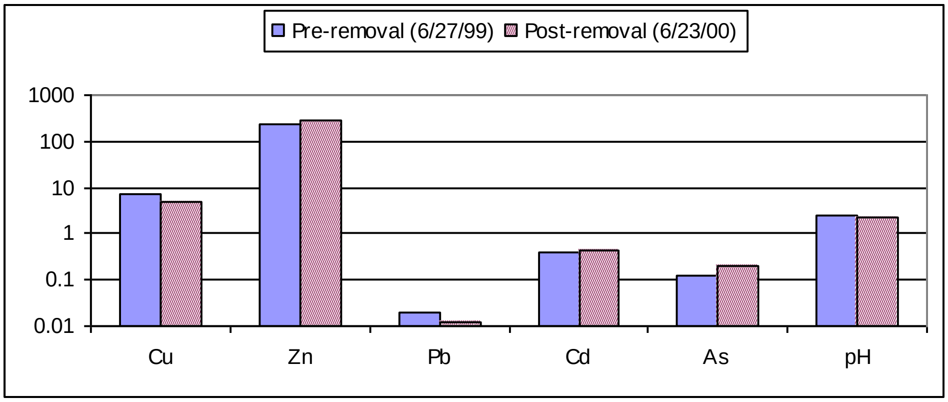

Figure 2. Heavy Metal and As Concentrations (mg kg $^{-1}$) in Alta Tributary Before and After Waste Rock Removal. Data are from Schroth (2001).

In effect, Schroth and Parnell (2005) concluded that trace element attenuation by the waste rock pile was equal to that of attenuation by coprecipitation with Fe-hydroxides in the channel. All trace elements except for Cu were found to decrease in the direction of flow in the Alta tributary both before and in the summer after pile removal (Figure 4).

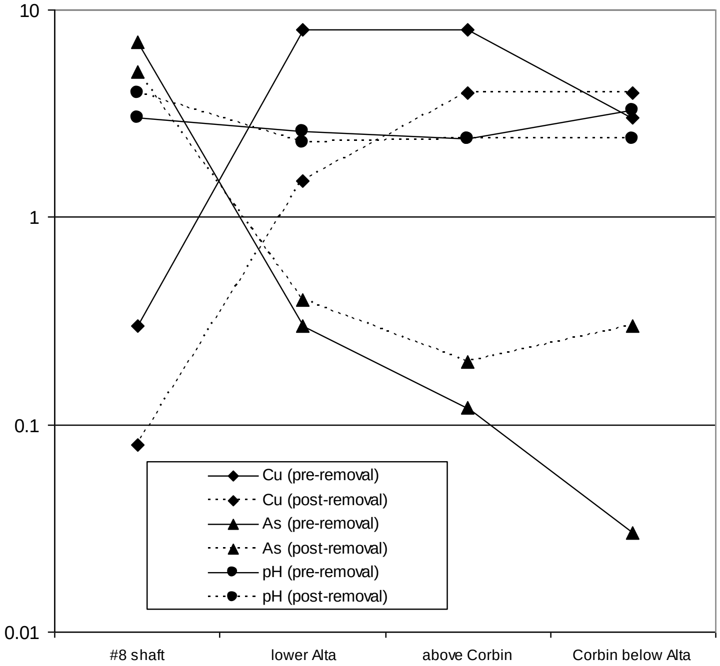

Figure 3. Cu, As, (mg L$^{-1}$) and pH (standard) Along Longitudinal Profile of Alta Tributary Before and After Waste Rock Removal. Data are from Schroth (2001).

The longitudinal profiles substantiate the idea that pile removal did little to influence downstream concentrations of heavy metals and arsenic in the first year after reclamation. Hydrologic regime appears to play an important role in determining concentrations in Corbin Creek. No surface flow was observed in the creek above the Alta tributary in 2000. As a result, pH below the Alta tributary was 1 unit less than in

1999. Concentrations of Cu and As decreased after mixing with Corbin Creek surface flows in 1999 but increased or remained constant in 2000—when no flow was observed in the creek above the confluence with the Alta tributary.

The findings of previous researchers are a time-specific representation of conditions, processes, and events at the Alta mine. The relevance of each work was dictated by prevailing site conditions at the time of study. As a means to assess reclamation alternatives, Deckler (1982) examined the characteristics of two waste rock piles that had been abandoned for nearly 30 years. Schroth (2001) provided a narrative of geochemical processes that occurred as a result of transient conditions during the reclamation effort in the summer of 1999. The current study expands the temporal coverage of our knowledge about the abandoned Alta mine to include a characterization of post reclamation site conditions and water quality impacts. The study is relevant because of an apparent vegetation failure on reclaimed slopes and because of recently established TMDLs in Corbin Creek.

# METHODS AND MATERIALS

## Soil Survey

### Soil Pit Excavation

Post-reclamation soils were surveyed on the Lower Alta reclaimed site in the summer of 2006. Sharpshooter shovels were used to dig 30 soil pits to completed depths between 0.6 m and 0.75 m. In most cases, pits were started by scraping surface-exposed material from the edge of a terrace (Sobek et al. 1978); then, pits were completed by excavating through topsoil of the adjacent lower terrace to the desired depth. Soil profiles were described in field notes according to soil color, mottling, moisture, root zone, and apparent texture. The pits were located on an approximate 25 m grid of the reclaimed area (Figure 5).

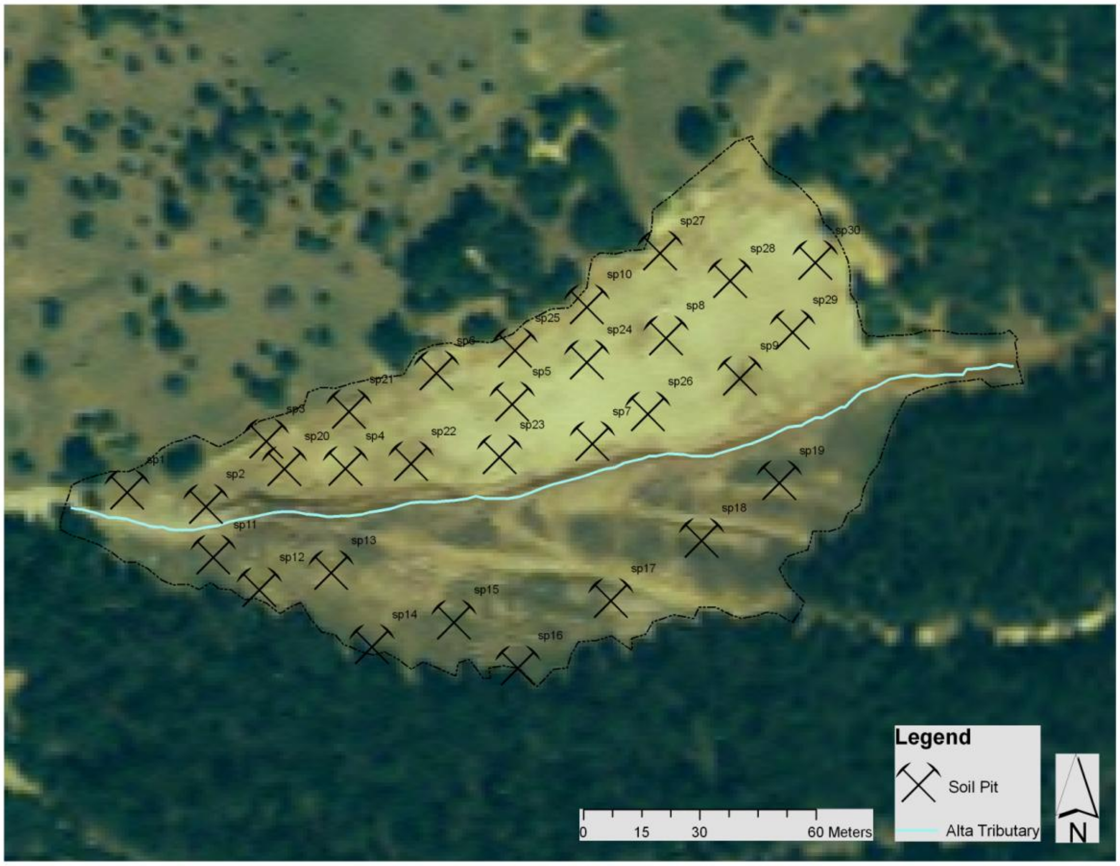

Figure 4. Soil Pit Locations on the Lower Alta Mine Site.

To prevent cross contamination between soil pits, they were dug with 2 alternating shovels. Upon completion of a pit, the first shovel was washed and allowed to air dry while another soil pit was completed with the second shovel. Shovels were decontaminated with Alconox® detergent dissolved in ambient temperature de-ionized water at 10 g/L. Decontamination included scrubbing with a wire brush; scrubbing with a synthetic-bristled brush; and rinsing with de-ionized water for 10 seconds. The decontamination station was located on the tailgate of our field vehicle. It consisted of a 19 L tub of detergent solution and 15 L of de-ionized rinse water. Shovels were decontaminated at the end of each sampling day to prevent uncontrolled offsite transfer of impacted soils.

### Soil Sample Collection

Each of the pits was sampled at two intervals: 0 - 15 cm; and 60 - 75 cm. Sampling at these intervals was completed to remain consistent with the EEECA sampling protocol (PRC EMI 1997). Between 1 and 1.5 kg of soil was collected from each interval with multiple scoops of a Corona CT3020 polished aluminum trowel. Soils were stored and transported in sealed plastic bags, each labeled with soil pit name, date, and depth interval. As was done with the sharpshooter shovels, trowels were washed between samples. Four trowels were used for soil sample collection such that two trowels were in use while the other two were washed. Trowels were carried from the decontamination station in a separate, clean, dry plastic bag. In addition to soils, controlled 1.2 mm silica sand was collected with trowels that had been decontaminated at the time of sampling. The sand, which was transported to the site in a sealed, clean, dry, plastic bag, underwent all further sample preparation and analysis steps as a blind control. Like soil samples, it was sampled in 1 to 1.5 kg increments. Trowels were decontaminated at the end of the day to prevent uncontrolled offsite transfer of impacted soils.

### Soil Sample Preparation

Soil samples were first homogenized by mixing within the sealed plastic bags in which they were originally collected and transferred; then, they were removed from their bags and spread onto sheets of brown kraft paper (Sobek et al. 1978) to air dry for at least 24 hours. Each sample was placed on a separate piece of paper and labeled with its soil pit name, date, and depth interval. While on the kraft paper, each soil sample was mixed with its own clean spatula in order to accelerate drying and to further homogenize the samples.

Once samples were dry, they were transferred to clean aluminum pans and weighed on 10-kg capacity scale. The total weight of each sample was recorded before it was sieved through a USA Standard No. 10 sieve with 2 mm openings. Each sample was shaken by hand for approximately 1 minute or until the sample was sufficiently separated. The samples, less the fraction with particle diameter greater than 2 mm, were returned to their aluminum pans, weighed, and recorded. Only the fraction of each sample with particle diameter less than 2 mm was retained for soil chemical and textural analyses. Retained samples were returned to clean, dry, plastic bags and labeled by soil pit name, depth interval, and date collected. All samples underwent this process as a minimum of preparation. Additional preparation and methods of specific soil analyses are outlined below.

## Soil Sample Analyses

A multitude of soil physical and chemical properties, known to affect vegetative succession, were considered in the characterization of the Lower Alta reclaimed site. The list includes Acid Base Accounting (ABA), heavy metals and arsenic, pH, EC, texture, rock content, nutrients, and organic matter. Some of the analyses were performed on the campus of Montana State University; while, others were completed by an independent laboratory—namely Energy Laboratories Inc. of Billings, MT (Energy Labs).

Rock Content. Rock content of Alta soils was determined as the amount of material that did not pass through a USA Standard No. 10 (2 mm) sieve (USDA, NRCS 2007). Specifically, rock content of each sample was expressed as a percent of the total sample by mass.

Soil Texture. Soil texture and class were determined by the hydrometer method described in Tan (2005) and the USDA Soil Texture Triangle. A mechanical blender was used to blend 100 g of soil, distilled water (to within 10 cm of the top of a stainless steel blender cup), and 10 ml of 0.25 M $(NaPO_3)Na_2O$ (sodium metaphosphate) solution for 15 minutes. The mixture was transferred to a glass ASTM soil testing cylinder; and distilled water was added until the suspension reached the 1205 ml level. A stirrer was used to thoroughly mix the suspension; then, a hydrometer was placed in the cylinder. Hydrometer readings were taken twice at 40 s and again after 2 hrs. The averaged 40 s readings were taken to be the silt and clay fraction of the soil matrix; and the 2 hr reading was clay alone. The temperature of the suspension was recorded and used to correct hydrometer readings to the calibration temperature of 68 $^{o}$F ( 20$^{0}$C).

Paste pH and EC. Saturated pastes (2:1, soil:water) were prepared with soils from each of the 30 soil pits and sampled depth intervals. The pastes were prepared in 200 ml paper cups according to Sobek (1978), except that less than 10 mesh soils were used instead of less than 60 mesh soils. The soils were allowed to wet by capillary action before being stirred to a thin paste with a spatula. The spatula was rinsed with a jet of de-ionized water before stirring each sample.

EC was measured with a YSI 3100 Benchtop Conductivity Meter. Before making EC measurements, the cell constant of the instrument was set at 1.00/cm and it was calibrated with 8,974 μS and 2,764 μS standard solutions. Additional calibrations were performed after every 10 soil samples. The YSI 3100 was set to automatically adjust EC readings with changes in temperature. Measurements were recorded when the digital EC reading was stabile. Between measurements, the probe and cell were rinsed with a jet of and stored in a beaker of de-ionized water.

Paste pH was measured with a Fisher Scientific Accumet 15 pH Meter on the same set of saturated pastes used for EC measurement. Measurements were recorded when the digital pH reading was stabile. The probe and electrode were rinsed with de-ionized water between samples. The pH meter was calibrated prior to use and after every 10 measurements with pH 4.0 and pH 7.0 standard buffer solutions.

Heavy Metals and Arsenic. Both total and soluble heavy metal and arsenic concentrations were determined for Alta mine soils. Energy Labs performed the analysis with Inductively Coupled Plasma Mass Spectroscopy (ICPMS). Total As, Pb, and Zn were determined for each of the 30 soil pits at both the 0 – 15 cm interval and the 60 – 75 cm interval. Sub-samples of approximately 150 g of less than 10 mesh air-dried soils were sent to the lab, where they were digested according to SW-846 EPA Method 3050 (US EPA 1995) before ICPMS analysis.

Soluble metals in the 0 – 15 cm interval were determined for 18 of the 30 soil pits. The analysis was limited to soils from the 0 – 15 cm depth because this was the observed rooting depth of vegetation on site. Also, only 18 samples had a sufficient amount of soil remaining—for additional preparation—once the soils had been sub-sampled for prior analyses. Additional sample preparation for soluble metals and arsenic included: making 2:1 saturated soil pastes; obtaining soil water extracts (SWE) by centrifugation; filtering the SWE; and acidifying the filtrate. The 2:1 saturated soil pastes were made in the same manner as those used in pH and EC determination, except these were made directly in clean 250 ml centrifuge tubes. First, approximately 200 g of soil were weighed in the tubes; then, approximately 100 g of water was added. The soils were allowed to wet by capillary action before being stirred to a thin paste with a spatula. The spatula was rinsed with a jet of de-ionized water before stirring each sample. SWE was obtained by centrifuging saturated pastes at 9,000 rpm for 10 minutes in a refrigerated centrifuge. The separated SWE was then evacuated from the centrifuge tubes with 60 ml BD syringes. From the syringes, the SWE was filtered through 0.2 μm membranes into 15 ml Falcon tubes. Each 0.2 μm membrane was used only once and discarded; however, the 50 ml BC syringes were used for multiple samples. Before reuse, each syringe was flushed 3 times with de-ionized water. Finally, filtrate was acidified with $HNO_3$ to $pH < 2$ and stored at $< 4\ ^0C$ until samples were shipped to Energy Labs for ICPMS analysis.

Acid Base Potential. Acid base potential (ABP) was completed by Energy Labs with the Modified Sobek Method (Sobek et al. 1978). The method considers both the neutralization potential (amount of neutralizing bases in a sample) and the maximum acid generating potential by total sulfur determination. Total ABP was calculated as the difference between neutralization potential and acid generating potential in T-$CaCO_3$/1kT-soil.

Nutrients and Organic Matter. Soils sent to Energy Labs were analyzed for N, P, K, and OM. Nitrate, as N, analysis was done with Method 38-8.1 (ASA, SSSA 1982). P was extracted by Method 24-5.4 (ASA, SSSA 1982) and analyzed with Method E365.1 (EPA, NERL 1993). K was extracted with ammonium acetate according to Method 13-3.5 (ASA, SSSA 1982) and analyzed with ICPMS. Finally, OM was extracted with the Walkley-Black Procedure, Method 29-3.5.2 (ASA, SSSA 1982), and analyzed by spectrophotometry.

## Vegetation Survey

Concurrent with soil pit excavation, a pair of 50 m transects were centered, perpendicular to each other, at each of the first 16 soil pits. Vegetation transects were only established at just over half of the soil pits; yet, they provided adequate coverage of the Lower Alta reclaimed area without overlapping each other. Canopy cover (Daubenmire 1959) was estimated in eight 20 cm x 50 cm (0.1 $m^2$) frames per pair of transects. Average vegetative cover of the eight frames was calculated for each pair of transects. The average was considered representative of the vegetative cover for the soil pit at which transects were centered.

## Soil Amendments and Revegetation

### Materials Handling and Plot Construction

The efficacy of lime and compost amendments at increasing vegetative cover was tested on 16 - 54 $m^2$ plots on the lower Alta mine site. Restricted access, remoteness of the site, and rough steep terrain limited the type of materials and number of agronomic practices available. Rather than having lime and compost delivered to the site in bulk, the amendments were purchased in smaller more manageable units. Some materials, 64 bags of lime (23 kg each) and 6 rolls of silt fence (30 m each), were stockpiled at the lower Alta site prior to construction. Compost, 20 super sacks (272 kg each), was transported to the lower Alta mine site on a ¾-ton flatbed truck. In order to ensure safe passage of the transport vehicle, the access road was improved with a Link-Belt 2700 excavator and a John Deere skid steer. Improvements included smoothing, widening, and removing rocks in the road that would cause undue stress on the transport truck. Upon completion of road work, compost was transported from an offsite staging and loading area established near the confluence of the Alta tributary and Corbin Creek. The skid steer was used to load super sacks at the staging area and the excavator was used to unload them at the lower Alta site.

Steep rough terrain made it impossible to prepare treatment plots with conventional agricultural equipment; instead, an excavator was used. A 1 m x 1 m trench was dug approximately 30 m upslope and parallel to the area chosen for plot construction in order to protect treatments from excessive erosion. Also, silt fences were installed below treated plots to prevent soil loss from reaching the Alta tributary. Terraces on the area to be treated were returned to the approximate original contour of the hillside. As terraces were excavated, all substrate that had the appearance of topsoil (i.e. more apparent organic matter, finer texture) was stored upslope from the plots. The apparent topsoil was the last substrate returned to each plot to be used as the seed bed. For treatments that required amendments, the teeth on the bucket of the excavator were used to incorporate lime and/or compost into the top 15 – 20 cm of soil. All work performed on the excavator was done by Arrowhead Reclamation, a private contractor. Plot construction was completed on April 30, 2007.

### Compost

Compost made from sawdust, agricultural wastes, and dairy, steer, and horse manure was purchased from Earth Systems Compost. The compost, which was analyzed by an independent laboratory, was found to contain 32.9% OM, 0.793% N, 0.393% P, and 2.16% K. Plots amended with compost received approximately 2 m$^3$ each, which corresponded to an addition of roughly 2.25% OM in the top 15 cm of the soil profile. Accounting for background OM concentrations, a target of 2-6% OM was set in the 0 -15 cm interval on plots that received compost.

### Lime

Lime product used on treatment plots was 97.5% pure agricultural lime ($CaCO_3$). Nominal grain sizes listed by the manufacturer were between 0.297 and 0.8 mm. An actual fineness factor (FF) of 61% was determined by the method of Whitney and Lamond (1993). By this method, grains < 0.25 mm are 100% reactive, grains between 0.25 - 2.4 mm are 50% reactive, and grains > 2.4 mm are not considered. Lime product was applied at a rate of 1.45% (14.5 T-lime 1kT-soil$^{-1}$). The purity ($CaCO_3$ equivalence) and FF reduced the effective rate of calcium carbonate addition to 8.6 T-$CaCO_3$ 1kT-soil-1$^{-1}$. A target pH of 6.5 was the goal of lime amendment.

### Seeded Species

Plots were seeded on May 4, 2007 with a bulk seed mix that contained equal portions of 5 native grass species and 1 forb species (Table 3). Each species was included in the seed mix for a specific adaptation to the Alta mine site or to AMD site conditions in general.

Spring wheat was also seeded in each of the 16 plots as a cover crop. All plots received 34 kg-pls ha$^{-1}$ of bulk seed mix and 42 kg-pls ha$^{-1}$ spring wheat. Alfalfa seed (5.5 kg-pls ha$^{-1}$) was applied to half of the plots as an additional cover crop. Seed was broadcasted onto the soil surface with 9-kg capacity Solo® model 421S portable seed spreaders and then lightly raked into the soil by hand.

Table 2. Seeded Species of Grasses, Gorbs, and Trees.

| Common Name | Scientific Name | Variety | Reason |
|---|---|---|---|
| *Grasses* | | | |
| Bluebunch wheatgrass | *Pseudoroegneria spicatum* | Goldar | drought tolerant, bunchgrass |
| Idaho Fescue | *Festuca idahoensis* | Joseph | found on site, |

| | | | |
|---|---|---|---|
| | | | bunchgrass |
| Big bluegrass | *Poa ampla* | Sherman | acid tolerant |
| Canada wildrye | *Elymus canadensis* | Mandan | drought and acid tolerant |
| Slender wheatgrass | *Agropyron trachycaulum* | Copperhead | acid tolerant |
| | | | |
| *Forbs* | | | |
| Lewis flax | *Linaria lewisii* | Appar | found on site |
| | | | |
| *Cover Crops* | | | |
| Alfalfa | *Medicago sativa* | falcata | N-fixing cover crop |
| spring wheat | *Triticum spp.* | | cover crop |
| | | | |
| *Trees* | | | |
| Douglas-fir | *Pseudotsuga menziesii* | harvested near site | |
| Limber Pine | *Pinus flexilis* | harvested near site | |
| Aspen | *Populus tremuloides* | harvested near site | |

The possibility of successfully establishing deciduous and coniferous trees on the lower Alta mine site was also examined. Roots of aspen trees from 2 separate aspen stands within 100 m of the lower Alta site were collected in early June 2006. Seventy (70) root cuttings were harvested and cultivated in a greenhouse according to MacDonald (1986). Saplings were removed from the greenhouse in the fall and exposed to winter conditions in a standard horticultural lath house; thereby, trees entered a dormant state and acquired winter hardiness. Freshly fallen Limber pine and Douglas fir cones were collected on August 21, 2006 from tree stands adjacent to and at the same elevation as the Alta mine. Seeds were plucked from the cones with tweezers, planted in individual growth containers, and vernalized according to MacDonald (1986). Following the 6 week vernalization, conifer seeds were planted in 2.5 x 15 cm containers filled with standard greenhouse potting soil. Containerization was completed in December 2006. Emerging seedlings were watered weekly and held at $25^{o}$C until they were planted on site. At the time they were transplanted, seedlings were approximately 2.5 – 6 cm tall. Aspen saplings and Douglas fir and Limber pine seedlings were transplanted to the lower Alta reclaimed area on May 4, 2007. Saplings and seedlings were planted into unamended soils above and below treatment plots.

## Experimental Design

The 16 plots consisted of 4 treatments, each replicated 4 times. Treatments included: compost alone, lime alone, lime and compost, and seeded controls. Replicates were equally divided and randomly applied to either of 2 blocks: trees; or alfalfa. Plots in the alfalfa block were seeded with bulk seed mix and an additional 5.5 kg pls ha$^{-1}$ alfalfa. The trees block received only the bulk seed mix; but the conifer seedlings and deciduous saplings were planted in unamended soils around its perimeter (Figure 6).

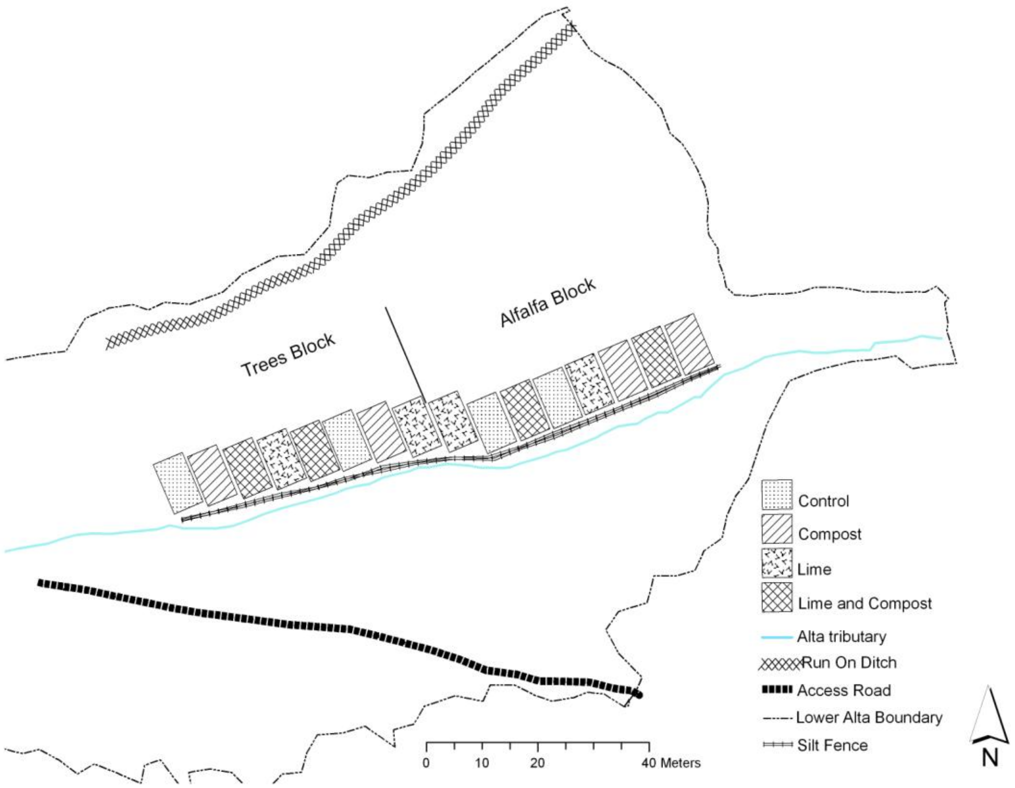

Figure 5. Treatment Plot Layout for Experimental Revegetation Trials.

## Quantification of Vegetative Cover

As was done in the baseline vegetation survey, canopy cover at the end of the 140-day growing season was measured with the Daubenmire method (1959). A single diagonal transect from the southeast corner to the northwest corner of each treatment plot was established. Cover was determined by species in 6 – 0.1 m$^2$ frames spaced approximately 1.8 m apart on each transect. Tree mortality was determined by counting the number of living and dead trees at the end of the growing season.

## Quantification of Soil Productivity

A composite soil sample from the amended soil depth was collected for each plot from 4 locations, spaced approximately 2.7 m apart, on the diagonal transect. Samples were collected in 1 L disposable cups and transferred to plastic bags. Samples were homogenized by a thorough mixing within the plastic bags and during the drying process of Sobek et al. (1978).

Soil samples derived from the treated plots were analyzed for OM, pH, EC, and and water extractable Cd, Cu, Pb, Zn, and As. Methods of analysis were consistent with those previously described except for OM and water extractable heavy metals and As. OM was determined by the loss on ignition method (Nelson and Sommers 1996) because it could be done cheaply in-house. Soils subject to loss on ignition (LOI) were ground first, for 2 minutes in a SPEX Certiprep 8000D ball mill, with 1.27 cm diameter ceramic balls. Water extractable heavy metals and As were determined from a 1:10 soil solution because subsamples that remained after LOI were too small to yield a viable extract from a 2:1 saturated paste; however, ICPMS was stilled used to determine heavy metal and As concentrations.

## Water Quality Monitoring

### Surface Water Monitoring Locations

Paired stream flow measurements and sediment, As, and heavy metals samples were collected at 6 surface water monitoring locations in both the Corbin Creek watershed and Alta subwatershed throughout the summer of 2007 (Figure 7).

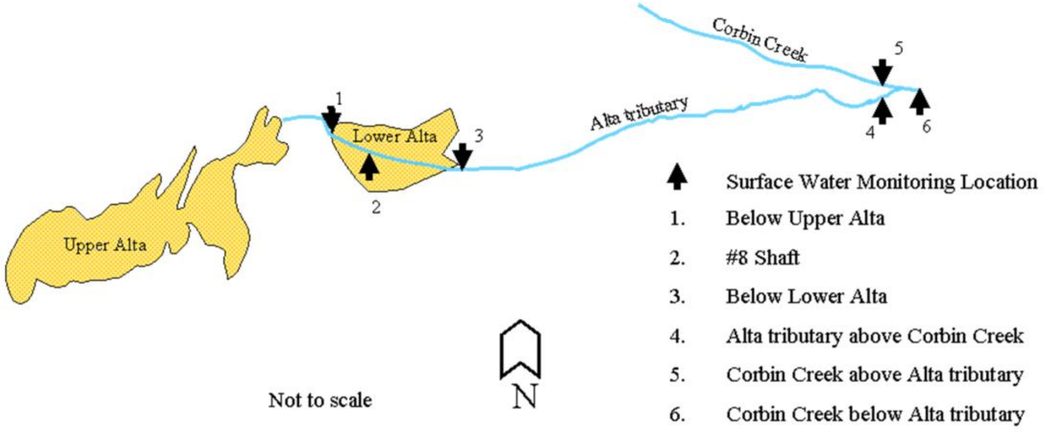

Figure 6. Summer 2007 Alta Sub-Watershed Surface Water Monitoring Locations.

The 6 sampling locations were: below the Upper Alta waste rock pile; at the #8 shaft; below the Lower Alta reclaimed area; above the confluence of the Alta tributary and Corbin Creek (in the tributary); above the confluence of the Alta tributary and Corbin Creek (in the creek); and below the confluence of the 2 streams. Monitoring locations in the Alta tributary were situated in the approximate location described by Scroth (2001). While the exact geographic location of 1999 and 2000 sample points was not known, the locations proposed above are likely a close fit. The monitoring location in Corbin Creek below the Alta tributary was situated well below the mixing zone (Thomann and Mueller 1987) determined by maximum stream velocity, depth, and width. Monitoring locations allowed for comparison of 2007 As and heavy metal concentrations to heavy metal and As concentrations present during reclamation (1999) and 1 year removed (2000). In addition, paired solute concentrations and discharge measurements taken over the course of summer 2007 at the monitoring locations above and below the confluence with the Alta tributary in Corbin Creek were used to determine the Alta tributary's contribution to Corbin Creek water quality impairments by As and heavy metals. Metal loads were compared to TMDL goals and previously estimated maximum loads for Corbin Creek (US EPA 2006) No direct measures of water quality impacts on aquatic life were monitored; however, in stream concentrations of As and heavy metals were compared to acute and chronic aquatic life standards (MT DEQ 2006). Results of the water quality monitoring effort are included in Appendix A.

### As and Heavy Metal Samples and Analysis

Heavy metals and As samples were frequently collected at each of the 6 sampling locations, if sufficient flows permitted, from March 23 to September 14, 2007. Grab samples were collected in clean 250 ml plastic bottles from the center of flow in either the Alta tributary or Corbin Creek. Bottles were rinsed 3 times with stream water before a sample was collected on the 4th grab. Stream pH was measured directly with a handheld model IQ150 pH meter at the time of sample collection.

All samples were analyzed for total recoverable (TR) metals and As. A limited subset of samples was analyzed for total soluble (TS) metals and As. TR and TS samples were prepared by acidifying to pH<2 or by filtering (0.2 µm) and acidifying to pH<2, respectively.

### Hardness

Hardness was measured exclusively at each of the two monitoring sites in Corbin Creek in order to draw comparison with acute and chronic aquatic life standards (ALS) for Cd, Cu, Pb, and Zn instituted by MT DEQ (2006). Toxicity of heavy metals and subsequent standards for aquatic life are a function of hardness, (Table 4), due to competition of benign calcium (Ca) and magnesium (Mg) cations with toxic heavy metal cations. In general, as hardness increases, toxicity of heavy metals decreases. If hardness is <25mg $L^{-1}$ as $CaCO_3$, the number 25 is used in the relationships; likewise, 400 is used in the relationships for hardness >400 mg $L^{-1}$ as $CaCO_3$ (MT DEQ 2006).

Table 3. Hardness Relationships and Coefficients for Acute and Chronic Aquatic Life Standards (MT DEQ 2006)

Hardness relationships

$$\text{Acute ALS} = e^{\{ma[\ln(\text{hardness})]+ba\}}$$

$$\text{Chronic ALS} = e^{\{mc[\ln(\text{hardness})]+bc\}}$$

| | Coefficients | | | |
|---|---|---|---|---|
| | ma | ba | mc | bc |
| Cadmium | 1.0166 | -3.924 | 0.7409 | -4.719 |
| Copper | 0.9422 | -1.7 | 0.8545 | -1.702 |
| Lead | 1.273 | -1.46 | 1.273 | -4.705 |
| Zinc | 0.8473 | 0.884 | 0.8473 | 0.884 |

Hardness (as $CaCO_3$) was determined by analysis of water samples for both Ca and Mg cations by ICPMS. A single sample per month of study was randomly selected for hardness determination and considered representative of hardness for the stream until subsequent samples were taken.

### Stream Gaging

The height of the water column was gaged at all surface water sampling locations except for the #8 shaft and Corbin Creek above the Alta tributary. These sites were excluded due to a limited number of gaging instruments and because the #8 shaft was not a suitable location for instrumentation. Gage height was read with Ecotone WM water level monitors at 15 minute intervals during the summer of 2007.

Discharge (Q) was determined at each monitoring location by 1 of 2 methods: 1) a calibrated bucket and stopwatch; or 2) the continuity method and a Swoffer 3000 stream velocity meter. As a rule, Q was determined by calibrated bucket at monitoring locations at the lower Alta reclaimed site; and the velocity area method was used for sites lower in the watershed. Sedimentation dams constructed at the lower Alta site in 1999 provided confined flows over spillways that were easily sampled by calibrated bucket; further, rip rap placed in the channel during the reclamation effort negated our ability to use a velocity meter there. Monitoring locations at lower sites had no such spillways but were free of large rocks that would preclude usage of a velocity meter. Gage heights were related to stream flow by power function rating curves.

### Suspended Sediment Samples and Analysis

Suspended sediment concentrations were monitored at each location except for the #8 shaft. The height of the water column in both the Alta tributary and Corbin Creek was insufficiently high to accommodate the use of standard USGS sampling equipment, US DH-48 and US U-59 samplers. Instead, suspended sediment samples were collected in an open bottle (US DH-48 variety), as suggested by Edwards and Glysson (1999). Samples were analyzed according to standard methods (American Public Health Association 1989) for Total Suspended Solids.

### Precipitation

A tipping bucket rain gage with a Hobo® event recorder was deployed on northeast corner of the lower Alta reclaimed site at approximately 1,710 m amsl. The gage was situated on a level plane away from objects that would obstruct rain and wind. The gage was used to record cumulative precipitation, event frequencies, and durations from March 23 to October 1, 2007.

### Snow Water Equivalent

Snow water equivalent was estimated in bi-weekly snow course measurements from January 22 to April 15, 2007. Measurements were made according to Colbeck et al. (1990) on a snow course established on the lower Alta reclaimed area. Slopes of south and north aspect were represented in 10 established measuring locations within the snow course. The snow course was established on the lower Alta reclaimed area because the site was not covered with tree canopy; but trees nearby provides some resistance to wind. Also, the lower Alta site was situated at a moderate elevation in the Alta subwatershed.

## Groundwater

Shallow groundwater on site was characterized for heavy metal and As concentrations by installing and sampling a network of shallow monitoring wells (Figure 8). The wells were systematically located near seeps and abandoned mine features that had the appearance of potential groundwater movement and on the flanks of the Alta tributary, where hyporheic mixing was possible. Aside from the #8 shaft, the most notable sources of water at the lower Alta reclaimed area were the portal, a horizontal adit south of the #8 shaft, and the bathtub springs, a small spring to the north of the #8 shaft that was not impacted by mining activity. In the absence of a seep or leaky mine feature, most of the slopes of the Alta subwatershed were unsaturated to depths below that which could be reached by shallow monitoring wells. Shallow wells were installed near each of the 3 surface water monitoring sites near the confluence of the Alta tributary and Corbin Creek.

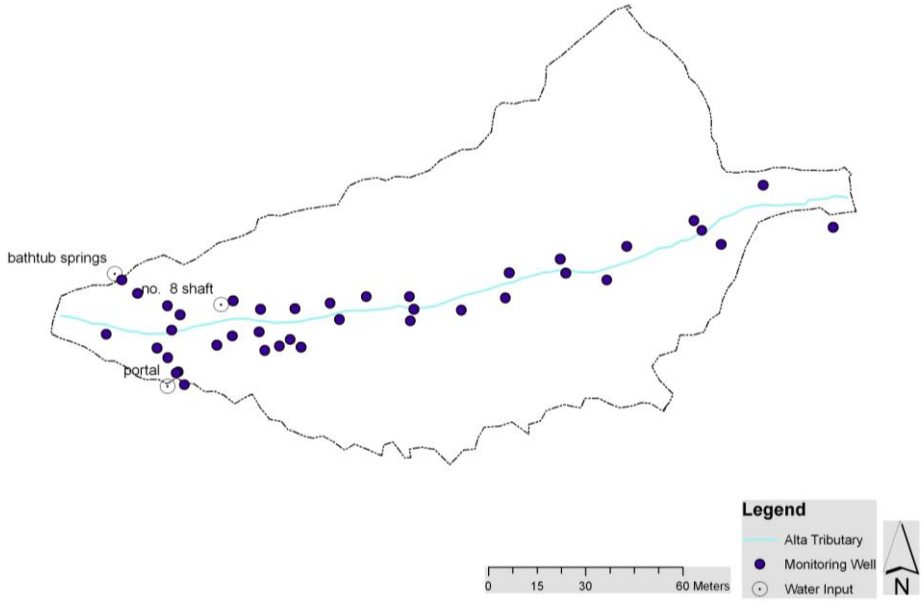

Figure 7. Network of Shallow Monitoring Wells at Lower Alta Reclaimed Area.

Well Construction. Holes used in well construction were made with a 5 cm diameter conical tipped steel pipe. The pipe was driven into the ground with a post driver and extracted with a hi-lift jack. Wells were cased with 2.54 cm diameter PVC pipe screened over an approximate 30 cm interval from completed depth. PVC pipe was capped at the bottom and lowered into completed holes. Annular space around the casing was filled with 2 mm silica sand to the top of the screened interval. A 25 cm bentonite clay seal was layered over the sand in each well. Soils found on site were used to fill the remainder of the annular space. Each well was capped with a 3.8 cm PVC sleeve to prevent rainfall from entering the casing. Completed depths of the shallow wells ranged from 114 cm to 200 cm below the ground surface.

Groundwater Sample Collection and Analysis. Shallow groundwater was drawn from 12 of the monitoring wells in the summer of 2007. First, each well was purged with a peristaltic pump to confirm a hydraulic connection with groundwater. Approximately 1 hour after purging, samples were pumped into clean 250 ml bottles from wells that exhibited adequate recovery. The pH of each sample was measured with a handheld Model IQ150 pH meter and recorded at the time of collection. Samples were filtered (0.2 μm), transferred to a separate clean sample bottle, and acidified to pH < 2. Samples were analyzed by Energy Laboratories, Inc. of Billings, MT with inductively coupled plasma mass spectroscopy (ICPMS).

Static Water Level Measurements. A Solinst® water level monitor was used to establish groundwater depths below the ground surface. Water level measurements were used to indicate water level recovery during groundwater sampling events. Also, bi-monthly measurements were made throughout the summer of 2007.

## Statistical Methods

### Z* test Statistic

Existing soil As and Pb concentrations in the 0 – 15 cm interval were compared to soil screening levels used in the 1999 reclamation project. The null hypothesis (Ho) was that observed mean soil As and Pb levels were equivalent to targeted screening levels. The alternative hypothesis (Ha) being that soil As and Pb concentrations in 2006 were indeed higher than the screening levels. The hypothesis was tested on the basis of a Z statistic (Devore and Farnum 2005). Prior to analysis, normal probability plots of the data were examined and a logarithmic data transformation was made to normalize the data.

### ANOVA Approach to Linear Regression

Single and multifactor linear regression models were constructed in R 2.5.1 to determine any correlation between soil physical and chemical properties and vegetation on site. Vegetative cover was used as the dependent variable; and independent variables included aspect, pH, ABP, OM, nutrients, heavy metals, and arsenic. Association between dependent and independent variables was measured with the coefficient of correlation (r); and regression models were considered significant at $p<0.1$. The aptness of linear regression models was diagnosed with normal probability plots and plots of residuals against fitted values. Significant single and restricted variable regression models were compared to multifactor models by Analysis of Variance (ANOVA), using

R version 2.5.1 (2007), to account for variability in vegetation due to cumulative factor effects. Like regression models, ANOVA models were considered significant at $p<0.1$.

### Wilcoxon Rank Sum Test

The Wilcoxon Rank Sum Test was used to compare the relative quality of soil in the 0 – 15 cm interval to that of soil in the 60 – 75 cm interval for each of the 30 soil pits. The comparison was made on the basis of total Pb, As, Zn, OM, ABP, N, P, K, and pH. It was expected that the test would establish patterns of incomplete waste removal, upward movement of contaminants, or erosion of topsoil cover material.

The Wilcoxon Rank Sum Test was also used to compare lime and compost treatment effectiveness at increasing pH, decreasing EC, increasing vegetative cover, and decreasing available heavy metals and As. The test was used because the assumption of normality was not appropriate with only 4 replicates of each treatment type. R 2.5.1 was used to perform statistical computation.

# RESULTS AND DISCUSSION

## Water Quality Monitoring

### Hydrograph

Stream flow in the Alta tributary and Corbin Creek was gaged and discharge hydrographs for summer 2007 were approximated at three locations: at the lower Alta reclaimed site, in the Alta tributary above Corbin Creek, and in Corbin Creek below the Alta tributary (Figure 9). Low intermittent flows at the monitoring site between the upper Alta waste rock dump and lower Alta reclaimed area prevented the establishment of a hydrograph and limited the monitoring effort at that location. A discharge hydrograph was not established for the #8 shaft, due to diffuse flows that were not measurable by a calibrated bucket, stream velocity meter, or Ecotone® water level monitor. As such, flows recorded at a confined overflow point 50 m downstream were considered representative of #8 shaft discharge. Corbin creek above the Alta tributary was not gaged; although, stream flow measurements were made as surface water samples were collected. Surface water flows were not observed above the confluence with the Alta tributary from 6/9/2007 to 9/14/2007.

Flow patterns in the Alta tributary immediately above Corbin Creek and in Corbin Creek immediately below the tributary were typical of low order mountain steams. Peak discharge at both sites corresponded to spring rain and snowmelt. Seasonal snow accumulation had completely disappeared from the lower Alta snow course by early April. Snow water equivalent accounted for 3.5 cm of the total 25.5 cm of precipitation received in the Alta sub-watershed. At the lower Alta reclaimed site, discharge was highly variable; and flow patterns were controlled more by mine shaft discharge than by surface hydrologic processes. Discharge at all locations was low and easily quantified in L min$^{-1}$. Peak flow at all sites occurred on May 22, as the result of an extended late spring rainfall event. Peak flows were estimated as 239, 137, and 91 L min$^{-1}$, respectively, at Corbin Creek below Alta, the Alta tributary above Corbin Creek, and at the lower Alta reclaimed site.

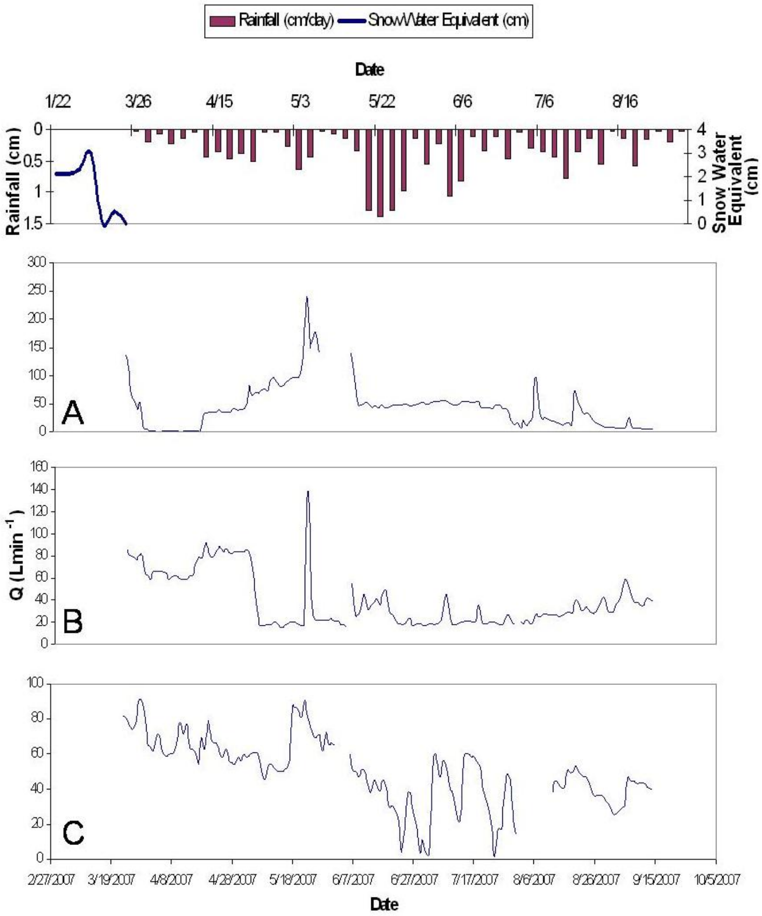

Figure 8. Rainfall Hyetograph, Snow Water Equivalent, and Mean Daily Discharge for A.) Corbin Creek Below the Alta Tributary, B.) Alta Tributary Above Corbin Creek, C.) Alta Tributary Below Lower Alta Reclaimed Area.

## Surface Water pH

Each monitoring location had a distinct pH signature. Generally, pH was greatest in surface flows of Corbin Creek above the confluence with the Alta tributary. Measurements made at the upper Corbin Creek monitoring location ranged from 7.16 – 8.0 before the stream dried up in early June 2007. Neutral pH at this site might be the result of lime applied to soils and limestone rip-rap used in the stream channel during the reclamation of the Bertha tailings. This is a crude indicator that removal of the acidic tailings (pH < 3.0, Deckler 1982) may have mitigated water quality impacts in Corbin Creek above the contribution of the Alta tributary.

Stream flow at all other monitoring locations exhibited pH that was well below neutral. A single pH measurement of the intermittent surface flow between the upper Alta waste rock pile and lower Alta reclaimed site was 3.5. Trends in pH followed the hydrograph at the remainder of the monitoring locations (Figure 10). Generally, pH was highest for observations made in the spring and during precipitation events. When peak hydrograph conditions subsided in early June, a distinct drop in pH was observed at all locations. The pH drop was likely attributable to less mixing with neutral snow-melt and shallow meteoric waters.

Decreased pH was nowhere more noticeable than in Corbin Creek below Alta, where pH dropped by more than 1 unit (from 3.7 to 2.5). The decrease in pH at this location was due to vanished or reduced mixing with neutral surface waters from above. Discharge from the #8 shaft had the highest pH (as high as 4.63) of the monitoring locations in the Alta tributary; but it decreased rapidly in the downstream direction. The rapid decrease was probably due to sulfide oxidation or the precipitation of Fe-hydroxides. The stream channel immediately below the #8 shaft was blanketed with ferric hydroxide precipitates. It was common for pH to have dropped more than 1 unit by the time flows exited the lower Alta reclaimed site, roughly 180 m downstream. Further reductions in pH occurred in the downstream direction. Surface flows with the lowest pH were generally found in the Alta tributary immediately above Corbin Creek.

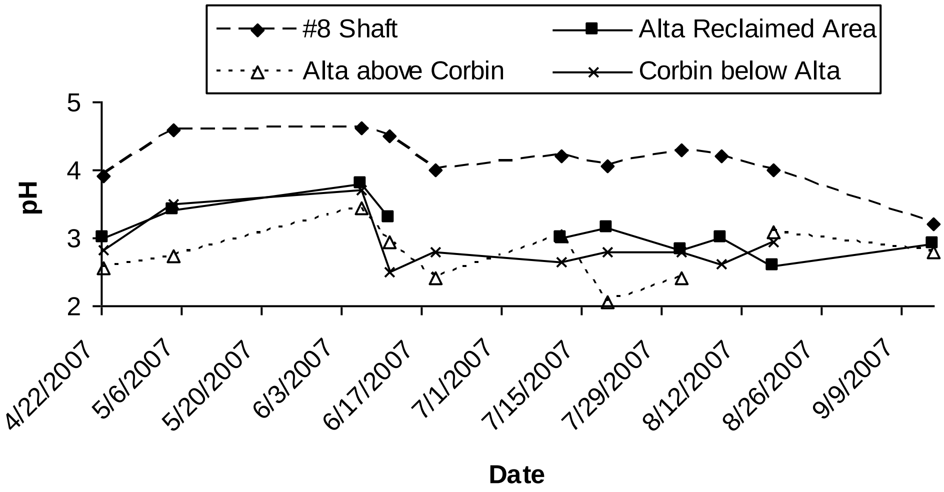

Figure 9. Temporal Trends in Alta Sub-Watershed pH by Site.

Heavy Metals and As.

Sources of heavy metal and arsenic varied by element and were controlled by pH, stream flow patterns, and apparent reactive processes. Heavy metal and As monitoring was only meaningful at or below the #8 shaft. Intermittent stream flow between the upper Alta waste rock pile and lower Alta reclaimed area was minimal during visits to the site. Before the flows ceased in early June 2007, two (2) observations of paired flow and heavy metal and As concentrations were made at the monitoring location between the upper and lower Alta (Table 5). Loads greater than those observed were likely to have occurred during peak runoff and high intensity rainfall events; but low aqueous metal and

As concentrations would still result in significantly lower transport rates than those measured at downstream monitoring locations.

Table 4. Heavy Metal and Arsenic Loads From Upper Alta Waste Rock Pile.

| | | | Element Load kg day$^{-1}$ | | | | |
|---|---|---|---|---|---|---|---|
| Date M/DD | Time HH:MM | Flow L min$^{-1}$ | As | Cd | Cu | Pb | Zn |
| 23-Mar | 16:00 | 8.18 | ND* | ND* | 0.12 | 0.04 | 0.62 |
| 22-Apr | 13:00 | 1.57 | 0.07 | 0.005 | 0.18 | 0.33 | 5.04 |

*ND=Element not detectable by ICPMS

Arsenic was predominately discharged from the #8 shaft and was naturally attenuated in the downstream direction (Figure 11). As concentrations observed in the #8 shaft discharge ranged from 4.37 – 10.5 mg L$^{-1}$ but decreased by an average of 73% at the monitoring location approximately 180 m downstream. Enriched As concentrations in #8 shaft discharge resulted in the highest loads of any site in the Alta tributary. Stream flow increased in the downstream direction from March to June; so dilution may have accounted for some of the decrease in As concentration. However, the decline in As load was likely due to co-precipitation with noticeable Fe-hydroxides in the channel of the lower Alta reclaimed site. No measures of Fe-hydroxide production or As sorption were made in the monitoring effort.

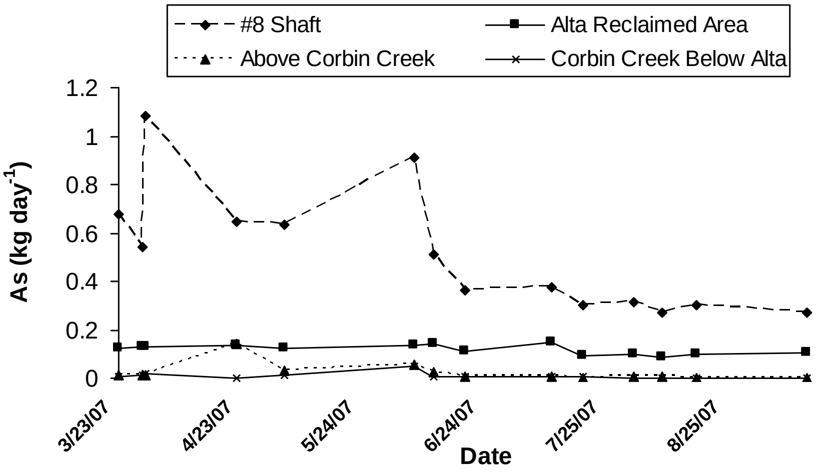

Figure 10. Temporal Trend in Alta Sub-Watershed As Loads by Site.

Cu load in the tributary was similar in magnitude to As load; but it exhibited an opposite trend (Figure 12). Concentrations emerging from the #8 shaft were 0.1 – 0.3 mg Cu L$^{-1}$ during spring and early summer and no Cu was observed in shaft discharge (by ICPMS) from June 11 to September 14, 2007. Cu load increased in the downstream direction, which implicated an input of leachate from Alta mine soils or waste rock in the lower reach of the tributary. The highest Cu loads were typically measured at the Alta tributary above Corbin Creek monitoring site; however, the observed peak Cu load for all monitoring locations and all sampled times occurred in Corbin Creek below the Alta tributary during a June 6th precipitation event. Peak loads measured at upper monitoring locations (#8 shaft, and Below Reclaimed Area) were also at times of hydrologic input of either snowmelt or rainfall. The response of Cu transport to hydrologic input further suggests that it is flushed to the Alta tributary in soil pore water.

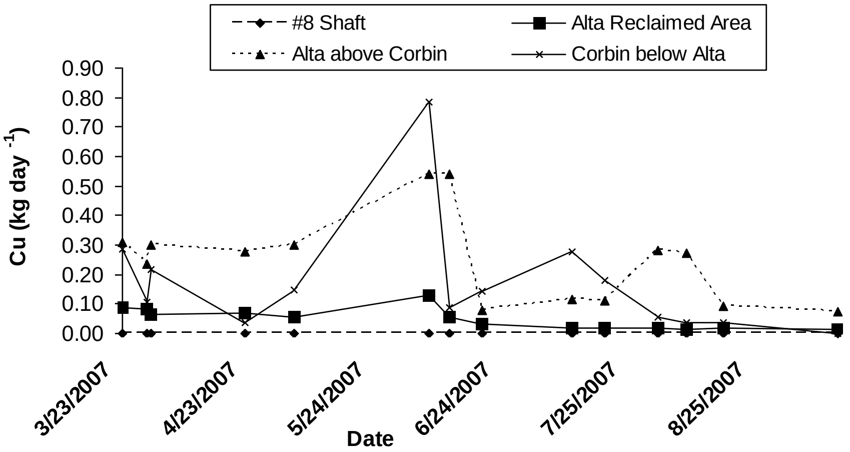

Figure 11. Temporal Trend in Alta Sub-Watershed Cu Loads by Site.

Maximum Cd and Pb loads observed at any location in the Alta tributary were an order of 10x less than those of Cu and As. Higher Pb loads were observed at the #8 shaft; while, Cd accumulated in stream flows throughout the lower reach of the Alta tributary (Figure 13). Both metals, however, appeared to originate from point source deep underground mine workings and diffuse landscape contributions. Cd loads observed at both the #8 shaft and below the lower Alta reclaimed area remained relatively constant over all hydrologic conditions monitored. Cd loads at lower monitoring locations followed trends similar to Cu and Pb, in that they varied with hydrologic input.

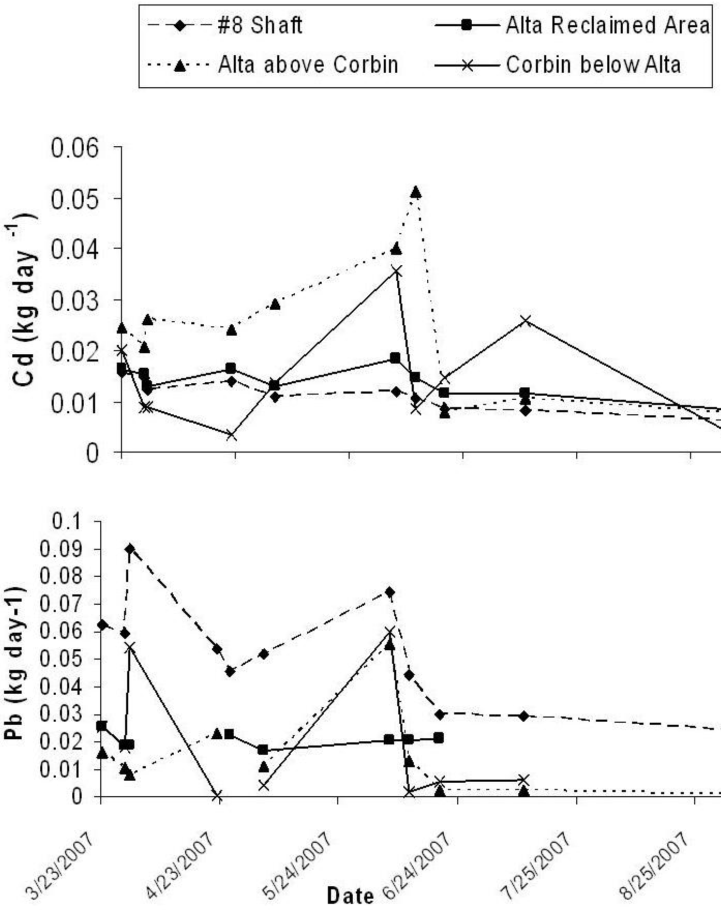

Figure 12. Temporal Trends in Alta Sub-Watershed A.) Cd Load and B.) Pb Load by Site.

By suspected co-precipitation with Fe-hydroxide or other chemical transformations, #8 shaft Pb loads were mitigated by an average of 60% by the time they reached the monitoring location below the lower Alta reclaimed site; however, Pb loads at downstream monitoring locations did not appear to be controlled by loads sourced from above. Pb loads more closely followed hydrologic patterns. During a freeze thaw cycle that occurred in late April, Pb loads were as low as $7 \times 10^{-4}$ kg day$^{-1}$ in the Alta tributary above Corbin Creek. At the same monitoring location, Late spring rains elevated Pb loads to $5.6 \times 10^{-2}$ kg day$^{-1}$, despite a near constant Pb discharge from the lower Alta reclaimed area.

Zn loading rates in the Alta sub-watershed were far greater than that of any other element under study. Loads as high as 45 kg day$^{-1}$ were observed from the #8 shaft (Figure 14). The shaft generally transported the highest rates of Zn; but concentrated discharge that occurred during spring rains also led to elevated Zn loads at lower monitoring locations. Zn loads from the #8 shaft responded only mildly to hydrologic inputs and decreased continually throughout the monitored interval. Zn loads below the lower Alta reclaimed area were relatively constant throughout the study and did not decrease with decreasing Zn input from the #8 shaft above.

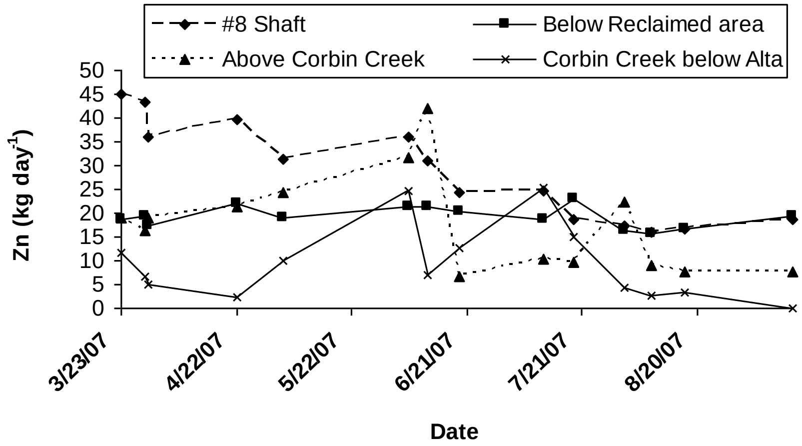

Figure 13. Temporal Trends in Alta Sub-Watershed Zn Load by Site.

Heavy Metals and As in Soil-Water and Groundwater.

Concentrations of As, Cu, and Zn were determined in both soil-water and in shallow groundwater for a random sub-sample of soil pits and monitoring wells, respectively, on the lower Alta reclaimed area in order to determine the contribution of the elements from the mine-impacted landscape. Groundwater samples were collected on August 17th and limited to wells constructed at the lower Alta reclaimed area because wells at the Corbin Creek and Alta tributary above Corbin Creek monitoring locations were dry at the time of sampling. Soil and groundwater source determination was particularly important in the case of Cu loading to the Alta tributary, as the #8 shaft was found to be a minimal source of that element. By examination of surface water concentrations of other heavy metals, it appeared that Cd, Pb, and Zn came from both deep underground sources and landscape sources. Zn was the only of the three metals measured in soils and groundwater; and its behavior was treated as a surrogate for Cd and Pb behavior. As concentrations in soil and shallow groundwater were determined to test the assumption that As was primarily discharged from the #8 shaft and because As behaves markedly different in soil and aqueous systems than heavy metals (Inskeep et al. 2002).

Patterns of soil-water and shallow groundwater As, Cu, and Zn concentrations differed greatly by element (Figure 15). Maximum water extractable Zn concentrations in soil were lower than groundwater concentrations or surface water quantities at either the #8 shaft or below the lower Alta reclaimed area; but considerably higher than soil water concentrations of the other elements. On the date sampled, Zn concentrations at the shaft and below the reclaimed area were nearly equal. Shallow groundwater Zn concentrations in wells immediately adjacent to the Alta tributary were indicative of concentrated surface waters mixing with dilute soil leachate that had accumulated in the saturated hyporheic zone. The connection of surface water and shallow groundwater is further evidenced by Cu concentrations in soil-water and shallow groundwater.

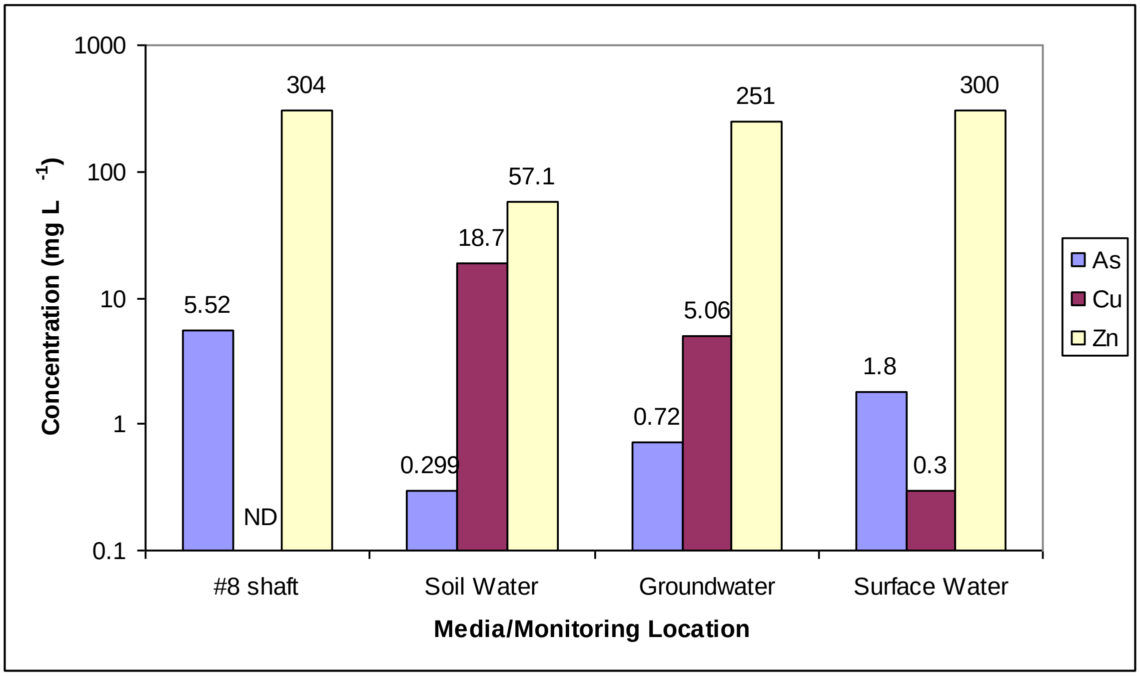

Figure 14. Peak As, Cu, and Zn Concentrations in Soil-Water, Shallow Groundwater, and Surface Water at the Lower Alta Reclaimed Area (August 17, 2007).

An apparent gradient of peak Cu concentrations from soil to shallow groundwater and ultimately to Alta tributary surface flow was identified. The peak soil water-extractable Cu concentration (18.7 mg $L^{-1}$) was determined in a soil pit on the south facing slope of the Alta reclaimed area near were Becraft (1963) described an exploratory mining effort of a Cu-rich vein that resulted in waste deposits on the lower Alta dump. Cu on the landscape of the lower Alta reclaimed area is likely the result of residual Cu-bearing minerals left during the 1999 reclamation project; or it is leached from the collapsed exploratory adit. The average Cu concentration in shallow groundwater of 6 monitoring wells was 2.04 ±1.53 mg $L^{-1}$. The relatively dilute Cu concentrations in the saturated subsurface near the Alta tributary were the result of mixing with Cu-deficient waters discharged from the #8 shaft. During the August 17th sampling effort, the #8 shaft Cu concentration was not detectable by ICPMS and the in stream concentration of discharge below the lower Alta reclaimed area was 0.3 mg $L^{-1}$.

Water extractable As concentrations in soil were low, averaging 0.06 ± 0.08 mg $L^{-1}$ and peaking at 0.299 mg $L^{-1}$. Low As mobility was expected in Alta soils because As is highly sorbed to soils and metal hydroxide solid phases in low pH systems (Manning and Goldberg 1997, Goldberg 2002). Concentrations of As in shallow groundwater were slightly more elevated (0.14 ± 0.22 mg $L^{-1}$) than soil-water concentrations, likely due to mixing with surface waters. Consistent with observations made at surface water monitoring locations, the #8 shaft appears to be the prominent source of As in the Alta tributary.

## Alta Contribution to Corbin Creek Loads

In order to directly address the hypothesis that the Alta tributary is the significant source of As and heavy metals to Corbin Creek, tributary loads were compared to background concentrations in the steam. Heavy metal and As loads carried by Corbin Creek above the Alta tributary and by the Alta tributary above Corbin Creek were not conservative beyond the confluence of the 2 streams; instead, co-precipitation of the elements with Fe-hydroxide precipitation in the mixing zone resulted in decreased transport of the elements. Background loads of each element in Corbin Creek and loads delivered to Corbin Creek by the Alta tributary were expressed as the mean fraction of the total load in Corbin Creek below the confluence with the Alta tributary. Fractionation normalized the data to account for changes in hydrologic regime throughout the monitoring period. Loads in Corbin Creek above the confluence with the Alta tributary were only 3.8% - 36% of the load in Corbin Creek below the mixing zone with the Alta tributary, depending on element (Figure 17). In contrast, Alta tributary loads ranged from 286% to 1,492% of the total As, Cd, Cu, Pb, and Zn in Corbin Creek. This result provided compelling evidence that the Alta tributary accounts for a significant fraction of the total As and heavy metals in Corbin Creek.

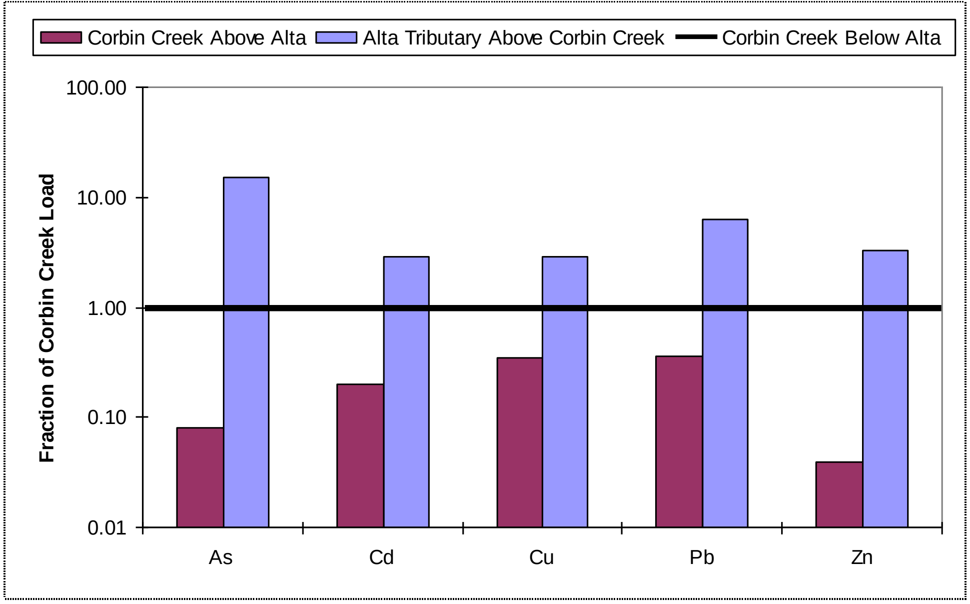

Figure 15. Alta Tributary and Corbin Creek above Alta Tributary Heavy Metal and As Loads as Fraction of Total Corbin Creek Below Alta Tributary Load.

## Alta Contribution to Aquatic Life Standard Exceedences

Above the Alta tributary, hardness ranged from 347 – 392 mg $L^{-1}$ $CaCO_3$. These values were within the range of 25 – 400 mg $L^{-1}$ $CaCO_3$ and were used to determine acute and chronic ALS in Corbin Creek above the Alta tributary. Below the Alta tributary, hardness ranged from 556 – 1,330 mg $L^{-1}$ $CaCO_{3.}$ The maximum value (400 mg $L^{-1}$ $CaCO_3$) was used to determine ALS below the Alta tributary. Heavy metal and As concentrations measured in the summer of 2007 were compared to ALS established for each element (Figure 18).

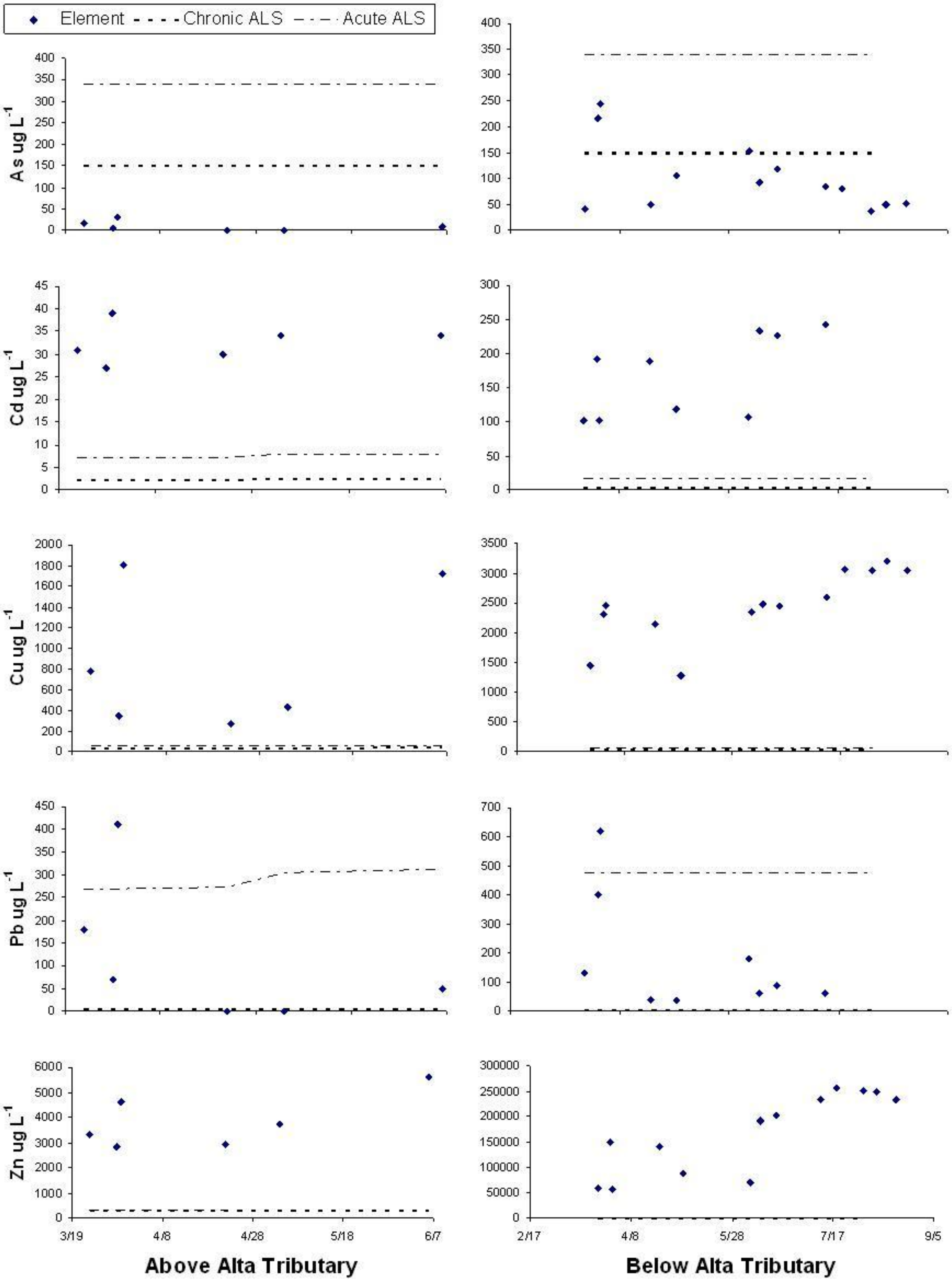

Figure 16. Heavy Metal and As Concentrations in Corbin Creek Above and Below the Alta Tributary Compared to Aquatic Life Standards in Summer of 2007.

Trace element concentrations that exceeded acute and chronic ALS were common at both monitoring locations in Corbin Creek for all elements with the exception of As. Only below the confluence with the Alta tributary were exceedances of ALS observed in the summer of 2007. Exceedances of only the chronic ALS, which is the lower standard, were observed at the monitoring location in Corbin Creek below the Alta tributary. This result suggests that of all of the elements considered in this study, As may have the least amount of impact on the health of aquatic life.

All observed heavy metal concentrations in Corbin Creek at monitoring locations above and below the Alta tributary exceeded at least the chronic ALS. Pb was the only metal not concentrated at levels above the acute ALS for all observations made. Even so, Pb was concentrated to levels above the acute ALS during early spring runoff. The level of exceedance for Cd, Cu, Pb, and Zn was far greater below the Alta tributary than in Corbin Creek above, as evidenced by the y-axes in the plot above. Again, the greater exceedances can be attributed to the disproportionately large loads contributed by the Alta tributary.

### Alta Contribution to TMDL Exceedances

As and heavy metal loading rates measured in Corbin Creek above and below the Alta tributary were also compared directly to Corbin Creek TMDLs. Loads measured in the summer of 2007 at both of the Corbin Creek monitoring sites were plotted with the envelope of TMDL goals and previously estimated maximum daily loads (US EPA 2006) (Figure 19). Patterns of TMDL exceedances varied by element; but more and greater exceedances were observed in Corbin Creek below the Alta tributary. Pb and As loads exceeded the TMDL only during observed periods of high flow and only in Corbin Creek below the Alta tributary; although, Pb loads in Corbin Creek above the Alta tributary were very near the TMDL during periods of high flow. Like Pb and As, Zn loads only exceeded the TMDL in Corbin Creek below the Alta tributary. There, Zn loads routinely, almost daily, exceeded the TMDL under all hydrologic conditions.

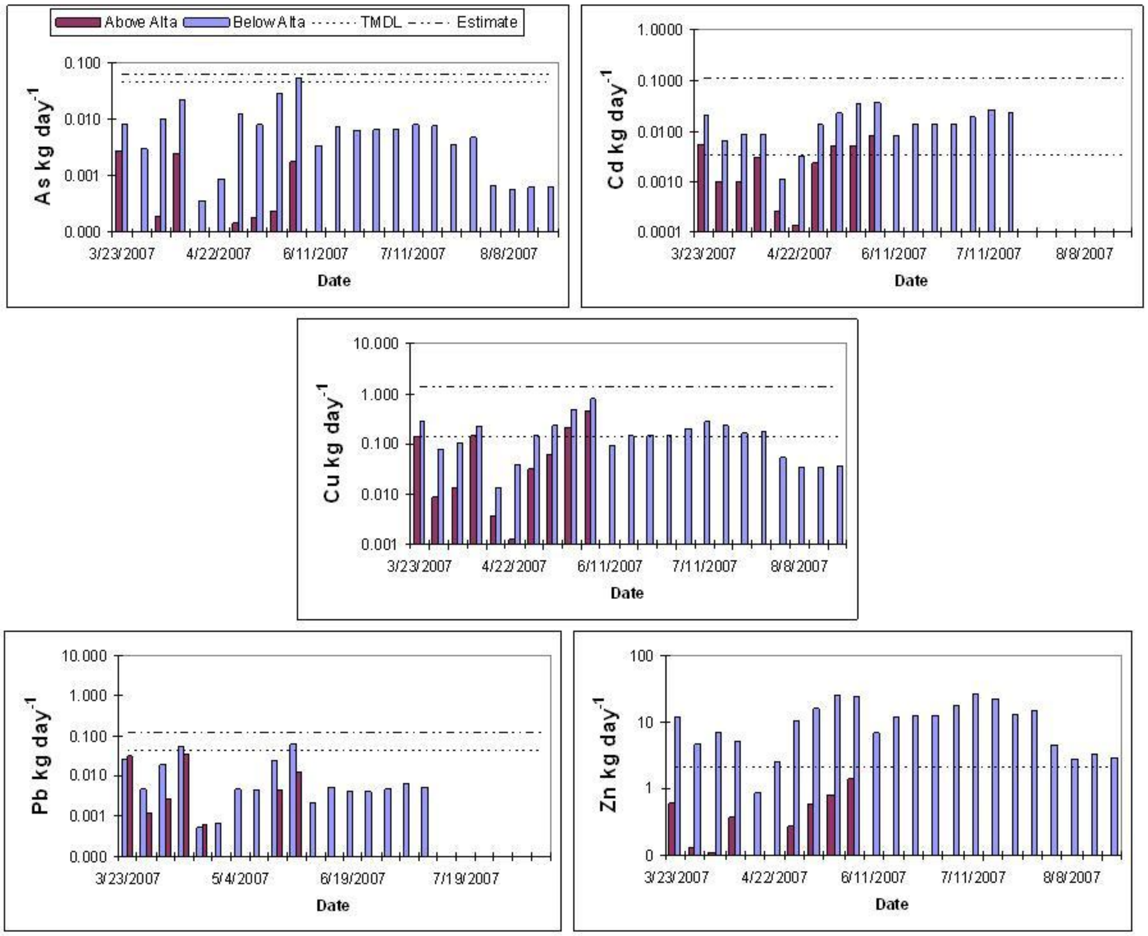

Figure 17. Corbin Creek Heavy Metal and As Loads Above and Below the Alta Tributary Compared to TMDL Goals and EPA Estimated Maximum Loads (US EPA 2006).

The plots provide evidence that TMDLs were not often exceeded by Corbin Creek background As and heavy metal concentrations alone; but that the Alta tributary contribution to Corbin Creek resulted in loads that exceeded TMDLs during some or all hydrologic conditions. Cu and Cd loads measured above the Alta tributary during the greatest observed flow events were the only exceptions. The combined results of the Alta tributary load to Corbin Creek background load comparison, ALS comparison, and TMDL comparison provide evidence to accept the hypothesis that the Alta tributary is not only the significant source of heavy metals and As but that it hinders attainment of TMDLs.

## Sediment

Sediment transport, estimated by total suspended solids (TSS), was lower than estimated daily maxima for Corbin Creek (US EPA 2006) throughout the summer at all monitoring locations. The highest sediment loading rates observed came during the June

6, 2007 rainfall event or early spring runoff measured on March 30, 2007 (Table 6). Concentrations of TSS measured during low flow or baseflow conditions were often below the detectable limit of the analytical method (APHA 2001). Difficulties in determining suspended sediment concentrations below the lower Alta reclaimed area arose because Fe rapidly precipitated in sample bottles before samples could be transported to the lab; thus, the analysis of suspended sediment was restricted to the Alta tributary above Corbin Creek and Corbin Creek above and below the Alta tributary.

Table 5. Summer 2007 Alta Tributary and Corbin Creek Sediment Loads and TMDLs.

| Monitoring Location | TMDL kg day$^{-1}$ | EPA Estimated Maximum kg day$^{-1}$ | Measured Maximum kg day$^{-1}$ |
|---|---|---|---|
| Corbin Creek Below Alta | 0.45 | 0.59 | 0.065 |
| Corbin Creek Above Alta | 0.45 | 0.59 | 0.037 |
| Alta Tributary Above Corbin Creek | -- | -- | 0.018 |

No TMDL exceedances were measured at either of the Corbin Creek monitoring locations. Also, the maximum observed sediment load carried by the Alta tributary was not above the TMDL established for Corbin Creek. During the monitoring effort, substantial accumulation of sediment was observed behind each of the sediment traps installed during the 1999 Alta reclamation project and the 2002 Bertha tailings removal. It is likely that sediment aggradation is the result of debris flows from low frequency high intensity rainfall events (Benda 1990) that occurred without observation or prior to the study. More monitoring data that includes these events are needed to construct a new sediment transport model or to validate the existing one (US EPA 2006) in Corbin Creek below the Confluence with the Alta tributary.

## Soil Characterization

Soil physical and chemical factors were successfully analyzed as outlined in methods and materials. A summary of the results of the soil survey are presented below. Complete results are listed in Appendix B. Where applicable, soil parameters in subsoil were compared to parameters in topsoil on the basis of the Wilcoxon Rank Sum Test.

### Soil Texture

All soils analyzed for texture, from both the surface and at depth, were found to be either sandy loam or loamy sand. Sand, silt, and clay fractions ranged from 56.1 – 84 %, 8 – 42.5 %, and 0 – 15.6 %, respectively. Rock fragments (> 2 mm) ranged from 24 – 70 % by mass of each sample. The texture, rock content, elevation, and appearance of Alta soils are consistent with the *Comad series* (Veseth and Montagne 1980) of well-drained, weak-structured, granitic soils. Soils of this type may be limiting to a revegetation effort because of their high erodability and inability to hold moisture.

## Soil Nutrients and Organic Matter

Soil fertility on the basis of N was typically low on the Alta reclaimed site; and P levels were low to medium (Table 7). K concentrations were near or below the range for disturbed soils (150 mg $kg^{-1}$ – 500 mg $kg^{-1}$); and OM was found to be in the low (2.1 – 3.5 %) to very low (<2 %) range (Munshower 1994). The lack of N and P in Alta soils may be explained conceptually by a cumulative effect of their low OM and coarse texture. Nutrients are leached from the well-drained soils when there is not sufficient OM to bind them. Sterile soils will not support vegetation; thus, they will not receive input of additional OM due to natural plant decay. Eventually, the structureless bare soils will erode. Merely through the textural and nutrient analyses of Alta soils, there was impetus for soil amendment with OM.

Table 6. Nutrient (mg $kg^{-1}$) concentrations and OM (%) in Alta mine soils

|  | Macronutrient |  |  |  |
|---|---|---|---|---|
| Interval | Nitrate (N) | P-Olsen | K-extractable | OM |
| 0 - 15 cm | 2.6 ± 1.07 a | 11.1 ± 5.7 a | 107.3 ± 41.2 a | 2.046 ± 0.84 a |
| 60 - 75 cm | 2 ± 1.6 a | 14.7 ± 7.8 a | 63.7 ± 26.4 b | 1.33 ± 0.5 b |

Values followed by same letter within columns are not significantly different ($p<0.1$).

## pH and EC

The pH of Alta soils ranged from 2.32 – 7.23 in the 0-15 cm interval and from 2.53 – 6.21 in the 65 – 70 cm interval. More instances of near neutral pH occurred in the upper interval, which is indicative of remnants of topsoil from the 1999 reclamation effort. Overall, pH of surface soils was significantly greater than that of subsoils (Table 8); however, many soil samples in both intervals were in the range of acute toxicity to plants (Rengel 2002).

Table 7. pH (standard units) and EC (μS $cm^{-1}$) of Alta mine soils.

| Interval | pH | EC |
|---|---|---|
| 0 - 15 cm | 4.45 ± 1.2 a | 535.8 ± 491.1 a |
| 60 - 75 cm | 3.45 ± 0.9 b | 854 ± 546.7 b |

Values followed by same letter within columns are not significantly different ($p<0.1$).

By EC measurement, salinity of Alta soils was found to be sufficiently lower than published inhibitory thresholds. Depending on species, salinity is not expected to cause a reduction in yield of grass or other forage except at levels > 2,300 µS/cm (Cardon et al 2003). EC was lower in surface samples than at depth, which suggested that plant exposure to salts is limited. The low level of available salts may only prove to be significant if they contain phytotoxic trace elements. Salts of $CdCl^+$ were increasingly taken up by wheat and Swiss chard *(Beta vulgaris)* when dissolved NaCl salts were added to soil solution (Weggler-Beaton et al. 2000).

## Heavy Metals and Arsenic

Total Metals and Arsenic. Prior to reclamation in 1999, mean Pb and As concentrations in the top 75 cm of waste were 3,238 mg $kg^{-1}$ and 1,063 mg $kg^{-1}$, respectively (PRC EMI 1997). In the reclaimed condition, respective mean concentrations over the entire 75 cm interval were reduced to 373 mg Pb $kg^{-1}$ and 337 mg As $kg^{-1}$. The mean Zn concentration of Alta waste from 3 samples taken in 1993 was 300 mg $kg^{-1}$ (Pioneer 1994), which was actually lower than mean Zn concentrations found in 30 soil pits in summer of 2006. Concentrations of total metals and As in Alta mine surface soils were not significantly different than concentrations at depth (Table 9). It is possible that metals have either migrated from mine impacted subsoils into the more recently applied topsoil by capillarity or that topsoil cover has been removed by erosion.

Table 8. Heavy metals and As (mg kg $^{-1}$) in Alta mine soils.

| | Heavy Metals and As | | | |
|---|---|---|---|---|
| Interval | As | Pb | Zn | Total |
| 0 - 15 cm | 316.8 ± 473.8 a | 355.7 ± 304.9 a | 391.9 ± 120.6 a | 1,064.4 ± 756 a |
| 60 - 75 cm | 357 ± 336.3 a | 390.6 ± 398.3 a | 440.3 ± 234 a | 1187.9 ± 614 a |

Values followed by same letter within columns are not significantly different ($p<0.1$).

Target removal concentrations of Pb and As used in the 1999 reclamation effort were 235 mg $kg^{-1}$ and 323 mg $kg^{-1}$, respectively (PRC-EMI 1997). This means that the waste rock pile was removed until surface concentrations were routinely at or below targets. Once target levels were reached, removal ceased. Current soil As and Pb concentrations in the 0 – 15 cm interval were tested against reclamation targets, assuming a lognormal distribution and a Z* test statistic. The null hypothesis of the test was that mean soil concentrations were equal to target concentrations; conversely, the alternative hypothesis was that surface soil concentrations were higher than targets. Though Pb and As levels in surface soils were indicative of mine impacted materials, neither element was significantly ($p<0.1$) concentrated above target levels (Table 10).

Table 9. Recreational clean up levels and current Pb and As in Alta mine soils.

| | Comparison to Reclamation Targets | | | |
|---|---|---|---|---|
| Element | Target mg $kg^{-1}$ | mean ± standard deviation mg $kg^{-1}$ | Z* | p |
| As | 323 | 316.8 ± 473.8 | -3.474 | 0.9997 |
| Pb | 235 | 355.7 ± 304.9 | 0.694 | 0.25 |

Soluble Metals and Arsenic. Water extractable Cu, As, and Zn concentrations were determined for a random subset of soils in the 0 – 15 cm interval, as this was the primary rooting depth observed during soil pit excavation. The soluble fraction of these constituents is not only indicative of bioavailability but also of the amount of potentially toxic trace elements that may migrate from soil to surface and ground water. Concentrations were variable but higher for Cu and Zn than for As. Concentrations ranged from 0.07 – 18.7 mg $L^{-1}$ and 0.01 – 57.1 mg $L^{-1}$ for Cu and Zn, respectively. Water soluble As concentrations ranged from non detectable by ICPMS to 0.299 mg $L^{-1}$.

## Acid-Base Potential

ABP was measured in the 0 – 15 cm interval for all 30 soil pits and in the 60 – 75 cm interval for 7 soil pits. The Wilcoxon Rank Sum Test was used to draw comparison between intervals. There was not a significant difference ($p<0.1$) between ABP in surface soils and ABP at depth. In the 0 -15 cm depth interval, ABP ranged from net acid neutralizing (13 T-CaCO3 1kT-soil-$1^{-1}$) to net acid producing (-57 T-CaCO3 1kT-soil$^{-1}$), with a mean value of -4.23 T-$CaCO_3$ kT-soil$^{-1}$. ABP from 8 T-$CaCO_3$ kT-soil$^{-1}$ to -45 T-$CaCO_3$ kT-soil$^{-1}$ were found in the 60 – 75 cm soil interval. Prior to waste rock removal in 1999, the top 75 cm of waste rock had ABP ranging from -12 T-$CaCO_3$ 1kT-soil-$1^{-1}$ to -196 T-$CaCO_3$ 1kT-soil-$1^{-1}$ (PRC EMI 1997). Though soils at the Alta mine appeared to have less acid generating potential than extreme values present in pre-reclamation substrates, net acid producing values of ABP indicated that low soil pH at the Alta mine may yet be the result of active sulfide mineral weathering.

## Vegetation Survey

Vegetative cover on the Alta reclaimed site was generally sparse. Mean vegetative cover determined from all sample frames was 11.7 %; yet, it ranged from 0 – 67.5 %. On the slope of north aspect, mean vegetative cover was 21.6 %, as compared to only 6.1 % on south facing slopes. Bare ground, rocks, and litter were abundant on site. Together, they accounted for a minimum of 40 % or 45 % of the total area in any given sample frame on the north or south facing slope, respectively. Much of the sampled area on the south facing slope was 95 – 100 % bare. Despite having minimal vegetative cover, 31 different plant species were identified on site (Table 11).

The Alta reclaimed area, with relatively high species richness and low vegetative cover, does not correspond with patterns of vegetative establishment on 3 reclaimed mines nearby. Tafi (2006) found only 12 species on a reclaimed site with 38.1 % cover and 37 species in an area with 70.9 % cover. Of the 31 species found at Alta, 20 were native and only 12 were seeded during the 1999 reclamation effort. Complete results of the vegetation survey are tabulated in Appendix B.

Table 10. Plant Species Found 7 Years After Reclamation of Alta Mine.

| Common Name | Scientific Name | Native Y/N | 1999 Seed Mix Y/N |
|---|---|---|---|
| Western Yarrow | *Achillea millifolium* | Y | Y |
| Thickspike wheatgrass | *Agropyron dasystachyum* | Y | Y |
| Intermdiate wheatgrass | *Agropyron intermedium* | N | N |
| Western wheatgrass | *Agropyron smithii* | Y | N |
| Slender wheatgrass | *Agropyron trachycaulum* | Y | N |
| Redtop | *Agrostis alba* | N | Y |
| Ticklegrass | *Agrostis scrabra* | N | N |
| Fringed sagewort | *Artemesia frigeria* | Y | N |
| Aster | *Aster spp* | Y | N |
| Cheatgrass | *Bromus tectorum* | N | N |
| Wavyleaf thistle | *Cirsium undulatum* | Y | N |
| Idaho fescue | *Festuca idahoensis* | Y | Y |
| Sheep fescue | *Festuca ovina* | Y | N |
| Rough fescue | *Festuca scabrella* | Y | Y |
| Hawk's beard | *Hierceulum* | Y | N |
| Dalmation toadflax | *Linaria dalmatica* | N | N |
| Lewis flax | *Linaria lewisii* | Y | Y |
| Annual ryegrass | *Lolium multiflorum* | N | Y |
| Silky Lupine | *Lupinus sericeus* | Y | N |
| Alfalfa | *Medicago sativa* | N | Y |
| Canada bluegrass | *Poa compressa* | N | Y |
| Kentucky bluegrass | *Poa pratensis* | N | N |
| Sandberg bluegrass | *Poa sandbergii* | Y | Y |
| American bistort | *Polygonum bistortoides* | Y | N |
| Quaking aspen | *Populus tremuloides* | Y | N |
| Bluebunch wheatgrass | *Pseudoroegneria spicatum* | Y | Y |
| Raspberry | *Rubus* | Y | N |
| Columbia needlegrass | *Stipa columbiana* | Y | Y |
| Green needlegrass | *Stipa viridula* | Y | N |
| Western salsify | *Tragopogon dubius* | N | N |
| Mullen | *Verbascum thapsus* | N | N |

## Correlation between Soil and Vegetative Cover

Univariate and multifactor linear regression models were constructed in R 2.5.1 to determine correlation between soil physical and chemical properties and vegetative cover. Results of all linear regression models, including correlation coefficients, p-values, and diagnostics plots are included in Appendix B.

Univariate Models

Single variable models found significantly correlated with vegetative cover were pH (0 – 15 cm), EC (60 – 75 cm), P (60 – 75 cm), soluble As (0 – 15 cm), and aspect (Table 12). EC and aspect were negatively correlated; while, pH, P, and As were positively correlated with vegetative cover.

Table 11. Significant Models for Vegetative Cover on Lower Alta Reclaimed Site.

| | Results | | | |
|---|---|---|---|---|
| Model | Coefficient | Intercept | r | p |
| Cover ~ As, soluble (0-15cm interval) | 108.193 | 6.145 | 0.84 | 0.00058 |
| Cover ~ Aspect | -9.36 | 16.071 | -0.49 | 0.05434 |
| Cover ~ EC (60-75cm interval) | -0.01183 | 18.587 | -0.54 | 0.02987 |
| Cover ~ P (60-75cm interval) | 0.41313 | 0.057 | 0.6 | 0.0694 |
| Cover ~ pH (0-15cm interval) | 3.775 | -6.948 | 0.48 | 0.0594 |

The most unexpected of the single variable regression models was the strong positive correlation between vegetative cover and soluble As. Plant yield reduction is expected when levels of bioavailable As are high, as it is a metabolic inhibitor (Kabata-Penias 2001). At low levels of soluble As, such as those found at Alta, As may not have any damaging effects on plant growth. Abedin and Meharg (2002) found rice seed germination to be unaffected by As(V) and As(III) at concentrations up to 2 mg $L^{-1}$ in soil solution. Further, root growth was only inhibited by 17% when arsenic was added to soil solution at 0.5 mg $L^{-1}$. Recall that concentrations of As in Alta soil solution ranged from non-detectable to 0.299 mg $L^{-1}$. A possible explanation for the positive correlation between As and vegetative cover is competition for soil sorption sites between As (V) and phosphate. As (V) was found to be more strongly sorbed than phosphate over a range of pH from 3 – 10 when the anions were tested at equimolar concentrations in a ferrihydrite suspension (Jain and Loeppert 2000). If As (V) has a competitive advantage for adsorption sites over phosphate, more phosphate may be available to plants. There was a positive correlation between vegetative cover and P (phosphate), though it was with P in the 60 – 75 cm depth interval. An equally plausible explanation for the strong positive correlation between As and vegetative cover is the presence of a single influential point. The influential point (soil pit 16) had the highest observed vegetative cover (38.2%), near the highest soil pH (pH=6.46), and the greatest soluble As concentration (0.299 mg As $L^{-1}$). Near neutral pH at the influential point may have resulted in both higher vegetative cover and soluble As. As noted, however, the enriched level of soluble As was likely not phytotoxic.

EC (dissolved solids) in the 60 – 75 cm interval were likely not available to plants growing at the Alta mine; however, EC was negatively correlated with vegetative cover. As noted above, plant growth would be most heavily dependent on EC when potentially toxic trace elements account for a portion of the dissolved salts. The relationship of EC at depth and vegetative cover was considered further in multivariate models.

Vegetative cover was significantly correlated with aspect, as a binomial factor. Slopes were considered as either north (1) or south (0) of the Alta tributary. Slopes south of the Alta tributary (north aspect) were shaded by trees, held increased snow in the winter, and were likely exposed to less solar radiation in summer. Increased vegetative cover on the north facing slope is apparent in aerial photography of the lower Alta mine (Figure 5). Slope aspect control on vegetation in this study is consistent with Martinez-Ruiz (2005), who found that vegetative succession at a reclaimed uranium mine in an arid climate was expedited on slopes of north aspect.

Soil pH in the rooting zone at the reclaimed Alta mine was significantly correlated ($p<0.1$) with vegetative cover. This result was not surprising, given the low pH values found on site. Many of the variables tested, however, were not correlated with vegetative cover. These included nutrients (N, P, and K), OM, ABP, total trace elements (As, Pb and Zn), and soluble trace elements (Cu and Zn). Though not found to be correlated with vegetative cover, many of these parameters were within ranges found to be limiting to plant growth by other researchers. For example, total As, Pb and Zn concentrations were well above the 10% yield reduction toxicity threshold for assumed transfer coefficients (Munshower 1994, Alloway 1995, Kabata-Pendias 2001). Factors that were insignificant alone were applied to multivariate models so that cumulative factor effects could be identified.

## Multivariate Models

Cumulative factor effects on vegetative cover were considered in multifactor linear regression models. Then, by ANOVA, restricted models (models which accounted for fewer variables) were compared to unrestricted models (models that included more variables). Similar to univariate regression models, unrestricted models found to be significant ($p<0.1$) by the ANOVA approach to linear regression included pH (0 -15 cm interval) and aspect as variables (Table 13). ABP (0-15 cm interval) was also found to be significant when considered with pH or pH and aspect.

The model that considered both pH and ABP was significantly stronger ($p<0.1$) than the model with pH alone. Intuitively, pH is likely to be low in substrates with net acid generating potential. Schippers (2000) examined vegetation death on tailings that ranged in pH and acid generating potential. Mortality was resoundingly higher on tailings with low pH and net acid generating potential. However, these tailings had mean concentrations of water extractable Cu and Zn (mg $L^{-1}$) that were twice as high as Alta concentrations and mean water extractable As concentrations (mg $L^{-1}$) that were nearly 200 times greater than those found at Alta. The pH + ABP model indicates that acid generating potential and pH significantly control vegetative succession even in the absence of enriched levels of phytotoxic elements.

Table 12. Significant Multivariate Regression and ANOVA Models for Vegetative Cover on Lower Alta Reclaimed Site.

| *Regression Models* | r | p |
|---|---|---|
| pH | 0.23 | 0.059 |
| Aspect | 0.24 | 0.054 |
| pH + Aspect | 0.39 | 0.041 |

| | | |
|---|---|---|
| pH + ABP | 0.49 | 0.013 |
| pH + ABP + Aspect | 0.57 | 0.014 |
| *ANOVA, (model 1, model 2)* | F* | p |
| pH, (pH + ABP) | 6.540 | 0.024 |
| pH, (pH + Aspect) | 3.360 | 0.090 |
| pH, (pH + ABP + Aspect) | 4.780 | 0.030 |
| Aspect, (pH, Aspect, ABP) | 4.67 | 0.032 |
| (pH + Aspect), (pH + ABP + Aspect) | 5.142 | 0.043 |
| (pH + ABP), (pH + ABP + Aspect) | 2.350 | 0.150 |

Aspect was correlated with vegetative cover when paired with pH and ABP or pH alone. This statistical outcome indicated that harsher environmental conditions present on the south facing slope and soil acidity had a cumulative effect on vegetative cover. The model that included ABP, pH, and aspect, however, did not significantly ($p<0.1$) account for more variability in cover than a model restricted to just pH and ABP as independent variables.

Several other multivariate models were found to be significant by linear regression; however, they did not account for more variability in vegetative cover than restricted models. As an example, regression models that considered EC (60 – 75 cm interval) and several other variables were significant; but ANOVA comparisons between these and restricted models were not.

## Soil Amendments and Revegetation

### Treated Soils

Changes in soil productivity with respect to pH, OM, and heavy metal and As availability were variably induced by lime and compost amendments. Each of the three treatments, lime alone, compost alone, and lime and compost, resulted in at least an apparent positive effect on 1 or all of the soil parameters analyzed.

The pH of soils collected from all 3 treatments was ostensibly greater than that of control soils; but the only statistically significant increases in soil pH were induced by compost alone or by compost and lime amendment (Table 14). Soil pH ranged from 6.15 – 8.07 over both the compost alone and compost and lime plots. Plots treated with lime alone had pH ranging from 5.03 – 6.2. At the applied rate, lime alone was not enough to raise pH to the target level of 6.5. Though this was the case, Ryan (1986) has shown plant growth was significantly less inhibited at pH of 4.8 than at 3.0. All treatments raised pH from the level of acute toxicity (3.5 – 4.0) published by Rengel (2002). Soil pH of control plots ranged from 3.43 – 3.92.

All plots that received compost were within the targeted range of 2 – 6% OM at the end of the growing season. Control and lime plots, which received no compost, were below or at the lower end of the targeted range. Soil OM for these plots ranged from 1.8 – 3.0%. The resulting increase in OM was significant for compost and lime treatments but not for the compost alone treatment; although OM in soils of the compost treatment was noticeably higher than non-compost plots (Table 14).

Table 13. Soil pH and OM in Treated Soils

| | Soil pH and OM expressed as mean of 4 replicates | | |
|---|---|---|---|
| Treatment | No. of treatments | pH | OM (%) |
| Control | 4 | 3.68 b | 2.15 b |
| Lime | 4 | 5.73 b | 2.38 b |
| Compost | 4 | 7.08 a | 4.28 a,b |
| Compost & Lime | 4 | 7.27 a | 5.1 a |

Values followed by same letter within columns are not significantly different ($p<0.1$).

Water extractable metal levels in soils of both the compost and lime and compost plots were not detectable by 10:1 extraction and ICPMS (Table 15). Pb was not detectable for all soils tested. Zn in the control soil and lime amended soils appeared higher than Zn in compost and lime and composted soils; but significant statistical differences were not found due to non detected (ND) values. Lime treatment resulted in a significant ($p<0.1$) difference in extractable As concentration. Relatively low values of each element were likely not significant inhibitors to plant growth.

Table 14. Water Extractable As and Heavy Metals (mg kg$^{-1}$) in Treated Soils.

| | Extractable Heavy Metals and As | | | | |
|---|---|---|---|---|---|
| Treatment | As | Cd | Cu | Pb | Zn |
| Control | 0.22 ± 0.29 a | ND - 2.0 | 13 ± 12.7 | ND | 84.25 ± 77.3 |
| Lime | 0.035 ± 0.01 b | ND - 1.0 | ND - 12 | ND | 38 ± 65.8 |
| Compost | 0.74 ± 0.72 a | ND | ND | ND | ND |
| Compost & Lime | 0.32 ± 0.037 a | ND | ND | ND | ND |

Values followed by same letter within columns are not significantly different.($p<0.1$)

## Vegetative Cover

Despite apparent and statistically significant improvements in the relative soil productivity of treated soils, canopy cover estimates performed at the end of the growing season indicated that none of the treatments resulted in vegetation establishment indicative of rehabilitated lands. Mean canopy cover was 1.48%, 3.38%, 5.53%, and 8.65% on control, lime, compost, and lime and compost treatments, correspondingly. Neuman (2005) achieved a median canopy cover of 62.5% on plots treated with lime and compost; and Tafi (2006) classified cover less than 25% as poor in her assessment of 3 abandoned mines near Alta. Emergence of plant varieties in the seed mix was minimal on treated plots. Of the grasses seeded, only Bluebunch wheatgrass (*Psedoroegnaria spicatum*) was counted in Daubenmire frames on any of the 3 treatments; however, it was present on plots of all 3 treatments and the control. The lone seeded forb, Lewis flax (*Linaria lewisii*), was counted on 1 lime plot and on 1 compost plot but on none of the control or lime and compost plots. None of the seeded grasses, forbs, or cover crops accounted for more than 1% of the total canopy cover for a given treatment type.

A total of 11 species were counted on the treated area of the lower Alta site (Table 16). Seven (7) volunteer plant varieties were counted in addition to 4 seeded species. With only 1 exception, the total cover of any given species for all treatments was <1%; thus, vegetation has not been broken down into treatment type. The notable exception was the presence of Kochia on treatments amended with compost. Kochia canopy cover was 1.92% and 2.27% on compost alone and lime and compost treatments, respectively. It was suspected that Kochia seeds were present in some element of the compost and not destroyed during the composting process.

Table 15. Plant Species on Treated Plots.

| Common Name | Scientific Name | Seeded (Y/N) |
|---|---|---|
| Western wheatgrass | *Agropyron smithii* | N |
| Wavyleaf thistle | *Cirsium undulatum* | N |
| forb | *Forb spp.* | N |
| Kochia | *Kochia spp.* | N |
| Lewis flax | *Linaria lewisii* | Y |
| Alfalfa | *Medicago sativa* | Y |
| Sandberg bluegrass | *Poa secunda* | N |
| Bluebunch wheatgrass | *Pseudoroegneria spicatum* | Y |
| Russian thistle | *Salsola kali* | N |
| Green needlegrass | *Stipa viridula* | N |
| Spring wheat | *Triticum spp.* | Y |

Because of the apparent gross introduction of a weedy species, no significant statistical analyses were performed in regard to treatment effectiveness at increasing vegetative cover. The low emergence of alfalfa and other seeded species also negated any relevant comparison between the alfalfa block and the trees block.

Aspen planted below treatment plots near the Alta tributary had a 100% mortality rate. The absolute toxicity of Alta mine soils to aspen saplings may explain the apparent lack of acid mine revegetation trials with this species (Pulford and Watson 2003).

Douglas fir and Limber pine mortality, however, was only 40% at the end of the first growing season. Though coniferous tree mortality was higher than that found by Ryan et al. (1986) in a controlled laboratory, results were consistent with transplanted shrub mortality observed on steep mine waste dumps (Leavitt et al. 2000). If the pattern of mortality observed by Leavitt et al. (2000) is representative of Alta mine conditions, transplant survivorship may be as low as 25% in 3 years. In which case, more than 12 of the original 50 coniferous trees will remain alive on site. Results of both the soil productivity analyses and end of growing season vegetation survey are included in Appendix C.

Downslope Migration of Seed Bank

Soil erosion was expected ensuing treatment plot construction; thus, silt fence was installed below plots as a best management practice. The silt fence collected a noticeable but inestimable amount of soil by the end of summer 2007. Soil loss by erosion, and the subsequent mobilization of the seed bed, was a likely cause for vegetation failure. Though our experimental design did not include a direct measure of either erosion or seed bank removal, a spatial trend in vegetative cover was recognized during analysis (Figure 21).

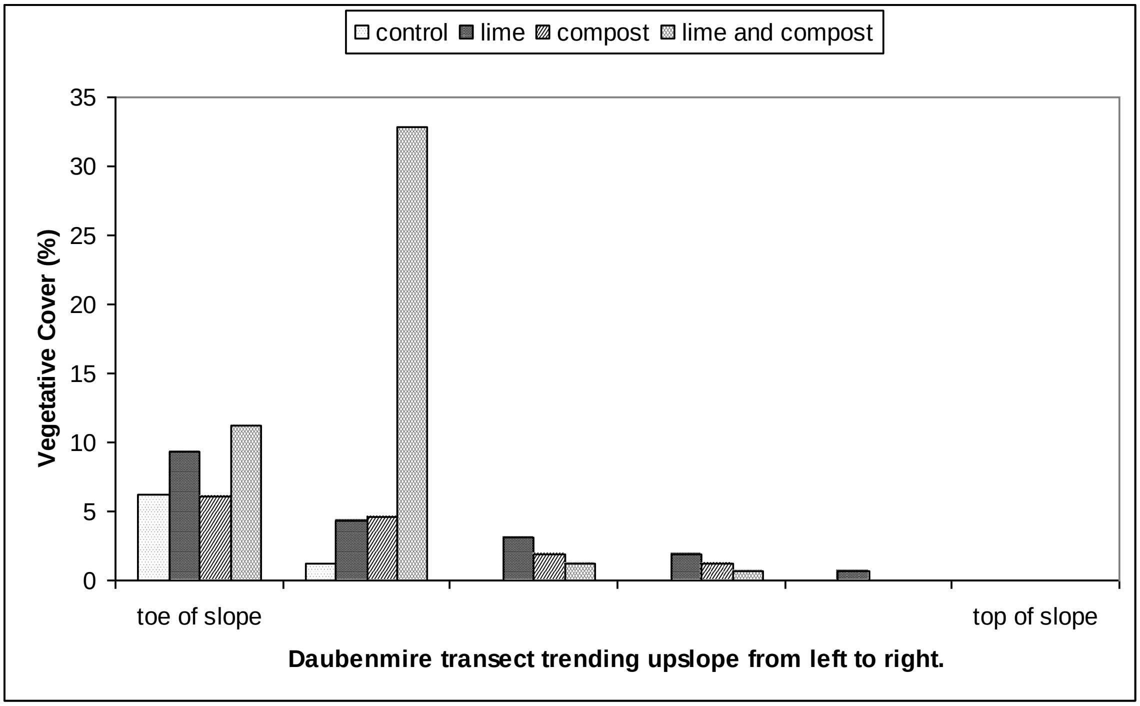

Figure 18. Apparent Trend in Vegetative Cover and Slope on Experimental Treatment Plots.

A greater vegetative cover was established at or near the toe of each treatment plot. Soil, soil amendments, and seed were likely displaced by erosion to the toe of the slope; there, seeds that were not deeply buried had the greatest success of emergence.

Soil loss (sediment yield) is controlled by the collateral relationship between vegetative cover and precipitation (Langbein and Schumm 1958). Sediment yield reaches a maximum when annual effective precipitation is between 30 and 35 cm. Precipitation greater than 30 cm stimulates the growth of vegetation; thus, less erosion occurs. Precipitation <30 cm does not provide enough erosive energy to move soil down slope. Annual precipitation at the Alta mine site is 28.7 cm (29.3 in 2006), which is in the range that limits vegetation and practically in the range suggested to yield increased erosion. In unamended soils, the amount of soil loss is compounded by other controlling variables like acidity, which has already been shown to effect plant growth.

# CONCLUSIONS AND RECOMMENDATIONS

Three (3) hypotheses were tested in 2 years (2006 – 2007) of study at the abandoned reclaimed Alta mine complex: 1) Waters that drain the site are the significant source of impairment to downstream water quality in terms of total maximum As, Cd, Cu, Pb, Zn, and sediment load; 2) Alta mine soils are still impacted by chemical or physical influences that preclude vegetative cover; and 3) Lime and compost amendments applied to experimental plots would improve soil productivity and vegetative cover. Each hypothesis was accepted or rejected based on results of field measurements and monitoring activities, laboratory analyses, and statistical inference.

## Water Quality

Results of the 2006 – 2007 monitoring effort indicate that abandoned underground mine workings, shallow subsurface flow, and soil leachate were all sources of As and heavy metals in the Alta tributary. The Alta mine sub-watershed is not unlike numerous other mined watersheds that have been found to discharge As and heavy metals from various sources and media (Brooks et al. 2001, Sullivan et al. 2001, Kimball et al. 2002). The primary source of individual elements varied; but the source with the highest overall discharge of As and heavy metals was the 180-m deep #8 shaft. The significant contribution of AMD discharge from point source abandoned mine features such as adits, shafts, and portals has been resoundingly confirmed by other researchers (Runkel and Kimball 2002, Kimball et al. 2002, Caruso 2003, and Herr et al. 2003).

Patterns of As and heavy metal discharge along the longitudinal profile of the Alta tributary downstream from the #8 shaft revealed that the Alta sub-watershed stores a portion of the As and heavy metals loads; and solutes that do reach Corbin Creek likely undergo chemical transformation and attenuate in the mixing zone. Decreases of metal and As load are common in mixing zones with neutral streams (Runkel and Kimball 2002, Butler II 2006). Nonetheless, loads that remain mobile in Corbin Creek are attributable to significant water quality impairment and hindrance to TMDL attainment. During periods of low flow, the Alta tributary is the only source of heavy metals and As in Corbin Creek. It was observed that flow above the Alta tributary had stopped by June 9, 2007. During flow regimes where background loads are contributed from Corbin Creek above the Alta tributary, they equal less than 40% of the total load in Corbin Creek below the confluence. In contrast, Alta tributary loads ranged from 286% to 1,492% of the total As, Cd, Cu, Pb, and Zn in Corbin Creek. These results support the hypothesis that the Alta tributary is the major source of As, Cd, Cu, Pb, and Zn in Corbin Creek.

Significantly lower loads in Corbin Creek above the Alta tributary suggest that reclamation activities performed in the Corbin Creek watershed, namely the removal of the Bertha tailings pile, were effective at mitigating water quality impairments. The pH of Corbin Creek above the Alta tributary was neutral, as compared to Alta tributary pH that ranged from 2.06 to 4.63. Removal of the Bertha tailings pile had positive effects on water quality because no additional source of AMD, like the #8 shaft, was present. However, AMD products were not the only indices of impaired water quality in the Alta tributary.

The abundance of bare ground and uncovered waste rock still present in the Alta sub-watershed has the potential to cause water quality impacts due to sedimentation. Sediment accumulation in both the Alta tributary and Corbin Creek channels was evident during the course of the study. It is likely that the sediment is carried by debris flows during low frequency, high intensity rainfall events. These events were not monitored during the summer of 2007; thus, further sediment samples taken during thundershowers, such as those measured by our rain gage in August 2007, are needed to accurately quantify the Alta tributary sediment load. For this reason, the hypothesis that the Alta tributary is the major source of sediment in Corbin Creek could neither be accepted nor rejected. Ideally, however, sediment load reductions could be achieved by revegetation.

## Limitations on Revegetation

The physical character of soils at the reclaimed Alta mine was consistent with soils found naturally around the Boulder Batholith (Deckler 1982, and Veseth and Montagne 1980); however, the chemical characteristics, in terms of elevated heavy metals, ABP, and low pH, were consistent with mine waste rock (Pioneer 1994). Acid base accounting performed on Alta soils proved that much of the substrate was moderately to highly net acid generating in 2006, seven years after the 1999 reclamation effort. Soil pH was also indicative of acidic conditions. Vegetative cover was significantly ($p<0.1$) controlled by net acid generating potential and low pH in soils. Cover was also significantly ($p< 0.1$) influenced by slope aspect. Dry conditions compounded acid toxicity, especially on the south facing slope. The south facing slope at the reclaimed Alta mine was generally devoid of vegetation and had numerous eroding rills. The hypothesis that chemical (soil pH and ABP) and physical (aspect) variables precluded vegetation on the reclaimed Alta mine site was accepted based on statistical significance determined by the ANOVA approach to linear regression. Other suspected variables (i.e. heavy metals and As) were not found to significantly impact vegetative cover.

Lime and compost amendments used to increase soil productivity on treated plots on the south facing slope of the Alta mine had marginal success. Lime treatment alone did not significantly ($p<0.1$) raise soil pH above control levels or, to the target of 6.5. Compost alone and compost and lime treatments both resulted in soil pH that was significantly ($p<0.1$) better than controls. OM content was also significantly ($p<0.1$) greater in compost treated plots, with or without lime, than in control plots. Despite increased soil productivity, none of the treated plots resulted in vegetative cover that was indicative of successful land rehabilitation. The hypothesis that lime and compost amendments would raise both soil productivity and vegetative cover was only partially accepted. Lime and compost improved soil quality; but vegetation establishment at the Alta mine was still limited.

The precipitation regime in southwest Montana is consistent with a level of effective annual precipitation that does not support enough vegetation to overcome the erosion that it creates. Sandy soils on steep south facing slopes were particularly susceptible to erosion at the Alta mine; thus, experimental plots built during the current revegetation effort, on the approximate original hill contour, were subject to seed loss. Seed loss due to unstable topsoil has been the unfortunate outcome of previous attempts to revegetate steep slopes (Leavitt 2000). Slopes at the Alta site must be significantly stabilized, by means other than terraces, before vegetation can be established. It cannot be assumed that a vegetative cover alone will stabilize the slopes.

Bluebunch wheatgrass (*Psedoroegnaria spicatum*) was the lone seeded grass species that had any apparent success during the revegetation trial. It appeared in all of the treatment plots and on the unamended controls. Aspen (*Populus tremuloides*) establishment was entirely unsuccessful in the acidic soils of the Alta mine, despite the tree's presence nearby. Douglas fir (*Pseutosuga menziesii*) and limber pine (*Pinus flexilis*) exhibited some tolerance to acidic conditions, as predicated by Ryan (1986); but 40% mortality of conifers still occurred.

## Abandoned Mines and Watershed Restoration

This study successfully linked abandoned, acid-producing, heavy metal mines to major downstream water quality impairments. The Alta mine, specifically, is the major contributing source of contaminants in Corbin Creek. Other mines of similar size, which are abundant in southwest Montana, may need to be considered for their impact on surface water quality and TMDL attainment status. Ineffective land rehabilitation may have a substantial impact on water quality impairments through sediment or As and heavy metals leached from soils. This may be true of Cu at the Alta mine, since soil-water extractable concentrations were higher than concentrations in #8 shaft discharge. However, deep underground mine workings were the most abundant source of other heavy metals and As discharge at the study site. Consequently, any level of successful land treatment will have little effect on Corbin Creek heavy metal and As TMDL attainment status.

## Recommendations

Sufficient amounts of waste rock were removed by MT DEQ in 1999 in order to obtain surface soils at the Alta mine with As and Pb concentrations below a recreational risk-based standard. These standards appeared to have been met; however, minimal top soil and fertilizer application did not prevent acid generating substrates from inhibiting plant growth. It is recommended that more emphasis be placed on soil productivity than on arbitrary or risk-based soil metal action levels if successful revegetation is an expected outcome of further reclamation activities at the Alta Mine. In this study, soil productivity was increased on the south-facing slopes of the Alta reclaimed area with modest amounts of lime and compost; yet vegetation failure resulted because of unstable slopes. It is recommended that the slopes be stabilized with tackifier or erosion control fabric in addition to soil amendment. .

It is not expected that more waste will be removed from the Alta sub-watershed; however, residential real-estate development in the immediate vicinity may provide the impetus for aesthetic enhancement of the landscape. In the event of further waste removal, it is recommended that the effort be focused on sediment accumulation in the Alta tributary channel. This research has shown the channel to be a sink for As and heavy metals that are suspected to have coprecipitated with Fe-hydroxides in the stream bed. The stored sediment and heavy metals have the potential to result in downstream exceedances of aquatic life standards, TMDLs, and other water quality standards if they are flushed during high magnitude storm events.

It is entirely meaningless to remove the heavy metal and As laden sediment if the source of heavy metals and As is not addressed. This research identified the #8 shaft as the most significant source of heavy metals and As in the Alta tributary; thus, the most progressive action that could be taken at the Alta mine would be the direct treatment of AMD discharged from the shaft. It was beyond the scope of this research to address feasible treatment options for shaft discharge. Further research is needed to identify applicable and cost effective water treatment options.

# REFERENCES CITED

Abedin, M.J. Meharg, A.A. 2002. Relative toxicity of arsenite and arsenate on germination and early seedling growth of rice (*Oryza sativa* L.). Plant and Soil. 243:57 – 66.

Adu-Wusu, C. Yanful, E.K. 2006. Performance of engineered test covers on acid-generating waste rock at Whistle mine, Ontario. Canadian Geotechnical Journal. 43:1 – 18.

Alloway, B.J. 1995. The origin of heavy metals in soils. *In Heavy Metals in Soils2nd edition* [Alloway, B.J. editor]. Glasgow, UK. Blackie academic and professional. 368 p.

American Public Health Association (APHA). 2001. Standard methods for the examination of water and wastewater: including bottom sediment.

American Society of Agronomy, Inc., Soil Science Society of America, Inc. (ASA, SSSA). 1982. Methods of soil analysis part 2-chemical and microbiological properties, 2nd edition. Madison, WI.

Baker, A.J.M. 1987. Metal Tolerance. New Phytologist. 106:93 – 111.

Ballen, K.G. Graham, P.H.  2002. Regulation of microbial processes by soil pH. *In Handbook of plant growth: pH the master variable* [Rengel, Z. editor]. New York, NY. Marcel Dekker, Inc. 446 p.

Basta, N.T. Ryan, J.A. Chaney, R.L. 2005. Trace element chemistry in residual-treated soils: key concepts and metal bioavailability. Journal of Environmental Quality. 34:49- 63.

Becraft, G.E. Pinckney, D.M. Rosenblum, S. 1963. Geology and mineral deposits of the Jefferson City quadrangle, Jefferson and Lewis and Clark Counties, Montana. USGS Professional Paper 428.

Benda, L. 1990. Influence of debris flows on channels and valley floors in the Oregon Coast Range, U.S.A. Earth Surface Processes and Landforms. 15(5):457 – 466.

Bonta, J.V. 2000. Impact of coal surface mining and reclamation on suspended sediment in three Ohio watersheds. Journal of the American Water Resources Association. 36(4):869 – 887.

Borgegard, S.O. Rydin, H. 1989. Biomass, root penetration and heavy metal uptake in birch in a soil cover over copper tailings. The Journal of Applied Ecology. 26: 585 - 595.

Brooks, P.D. McKnight, D.M. Bencala, K.E. 2001. Annual maxima in Zn concentrations during spring snowmelt in streams impacted by mine drainage. Environmental Geology. 40:1447 – 1454.

Broshears, R.E. 1996. Reactive solute transport in acidic streams. Water, Air and Soil Pollution. 90:195 – 204.

Butler II, T.W. 2006. Geochemical and biological controls on trace metal transport. Environmental Geochemistry and Health. 28:231 – 241.

Cardon, G.E. Davis, J.G. Bauder, T.A. Waskom, R.M. 2003. Salt-affected soils. Colorado State University Extension Publication no. 0.503.

Caruso, B.S. 2003. Water quality simulation for planning restoration of mined watersheds. Water, Air, and Soil Pollution. 150:359 -382.

Cela, S. Sumner, M.E. 2002a. Critical concentrations of copper, nickel, lead, and cadmium in soils based on nitrification. Communications in Soil Science and Plant Analysis. 33(1):19 – 30.

Cela, S. Sumner, M.E. 2002b. Soil zinc fractions determine inhibition of nitrification. Water, Air, and Soil Pollution. 141:91 - 104.

Christensen, T.H. 1984. Cadmium soil sorption at low concentrations: I. effect of time, cadmium load, pH, and calcium. Water, Air, and Soil Pollution. 21:105 – 114.

Colbeck, S., Akitaya, E., Armstrong, R., Gubler, H., Lafeuille, J, Lied, K., McClung, D., Morris, E. 1990. The international classification for seasonal snow on the ground: International Association of Scientific Hydrology. 23 p.

Crowley, D.E. Alvey, S.A. 2002. Regulation of microbial processes by soil pH. *In Handbook of plant growth: pH the master variable* [Rengel, Z. editor]. New York, NY. Marcel Dekker, Inc. 446 p.

Datta, R. Sarkar, D. 2004. Effective integration of soil chemistry and plant molecular biology in phytoremediation of metals: an overview. Environmental Geosciences. 11: 53 – 63.

Daubenmire, R. 1959. A canopy coverage method of vegetational analysis. Northwest Science. 33:43-64.

Deckler, J.H. 1982. Characterization of four inactive hard rock mines in western Montana. M.S. thesis, Land Rehabilitation. Montana State University. Bozeman, MT.

Devore, J. Farnum, N. 2005. Applied statistics for engineers and scientists 2nd edition. Belmont, CA: Thomson Brooks/Cole. 605p.

Edwards, T.K. Glysson, D.G. 1999. Field Methods for Measurement of fluvial sediment. U.S. Geological Survey Techniques of Water Resources Investigations, Book 3, Chapter C2.

Evangelou, V.P. 1995. Pyrite oxidation and its control. Boca Raton, FL: CRC Press. 293 p.

Foth, H.D. Ellis, B.G. 1997. Soil Fertility 2nd Edition. Boca Raton, FL: CRC Press. 290 p.

Friesl, W. Lombi, E. Horak, O. Wenzel, W.W. 2003. Immobilization of heavy metals in soils using inorganic amendments in a greenhouse study. Journal of Plant Nutrition Soil Science. 166:191 – 196.

Gallagher, L.M. 2001. Clean water act. *In Enivironmental law handbook* [Sullivan T.F.P. editor]. Rockville, MD. Government Institutes,Inc. 811p.

Gilbert, G.K. 1917. Hydraulic mining debris in the Sierra Nevada. USGS Professional Paper 105. Wasington D.C. Government Printing Office. 154 p.

Goldberg, S. 2002. Competitive adsorption of arsenate and arsenite on oxides and clay minerals. Soil Science Society of America Journal. 66:413 – 421.

Henry, T.B. Irwin, E.R. Grizzle, J.M. Wildhaber, M.L. Brunbaugh, W.G. 1999. Acute toxicity of an acid mine drainage mixing zone to juvenile bluegill and largemouth bass. Transactions of the American Fisheries Society. 128: 919-928.

Herr, J.W. Chen, C.W. Goldstein, R.A. Herd, R. Brown, M.J. 2003. Modeling acid mine drainage on a watershed scale for TMDL calculations. Journal of the American Water Resources Association. 39(2):289 – 300.

Inskeep, W.P. McDermott, T.R. Fendorf, S. 2002. Arsenic cycling in soils and natural waters: chemical and microbiological processes. *In Environmental Chemistry of Arsenic* [Frankenberger Jr., W.T. editor]. New York, NY. Marcel Dekker, Inc. 391p.

Jain, A. Raven, K.P. Loeppert, R.H. 1999. Arsenite and arsenate adsorption on ferrihydrite: surface charge reduction and net $OH^-$ release stoichiometry. Environmental Science and Technology. 33(8):1179 – 1184.

Jain, A. Loeppert, R.H. 2000. Effect of competing anions on the adsorption of arsenate and arsenite by ferrihydrite. Journal of Environmental Quality. 29:1422 – 1430.

Jennings, S.R., Dollhopf, D.J. 1995. Geochemical characterization of sulfide mineral weathering for remediation of acid producing mine wastes. Reclamation Research Unit, Montana State University, Bozeman, MT. Reclamation Research Publ. No. 9502.

Jiang, Q.Q. Singh, B.R. 2004. Effect of different forms and sources of arsenic on crop yield and arsenic concentration. Water, Air, and Soil Pollution. 74 (3-4):321 – 343.

Jones, C.A. Inskeep, W.P. Neuman, D.R. 1997. Arsenic transport in contaminated tailings following liming. Journal of Environmental Quality. 26: 433 – 439.

Kabata-Pendias, A. 2001. Trace elements in soils and plants 3rd edition. Boca Raton,FL: CRC Press. 413p.

Khan, D.H. Frankland, B. 1983. Plant and Soil. Effects of cadmium and lead on radish plants with particular reference to movement of metals through soil profile and plant. Plant and Soil. 70:335 – 345.

Kimball, B.A. Runkel, R.L. Walton-Day, K. Bencala, K.E. 2002. Assessment of metal loads in watersheds affected by acid mine drainage by using tracer injection and synoptic sampling: Cement Creek, Colorado, USA. Applied Geochemistry. 17:1183 – 1207.

Kramer, P.A. Zabowski, D. Scherer, G. Everett, G.L. 2000. Native plant restoration of copper mine tailings: I. substrate effect on growth and nutritional status in a greenhouse study. Journal of Environmental Quality. 29:1762 – 1769.

Langbein, W.B. Schumm, S.A. 1958. Yield of sediment in relation to mean annual precipitation. Transactions of the American Geophysical Union.

Leavitt, K.J. Fernandez, G.C.J. Nowak, R.S. 2000. Plant establishment on angle of repose mine waste dumps. Journal of Range Management. 53:442 – 452.

Lefebvre, R. Hockley, D. Smolensky, J. Gelinas, P. 2001. Multiphase transfer processes in waste rock piles producing acid mine drainage 1: conceptual model and system characteristics. Journal of Contaminant Hydrology. 52:1-4, 137 – 164.

Ma, L.Q. Rao, G.N. 1997. Chemical fractionation of cadmium, copper, nickel, and zinc in contaminated soils. Journal of Environmental Quality. 26:259 – 264.

Macdonald, B. 1986. Practical woody plant propagation for nursery growers. Portland, OR. Timber Press. 669p.

Macdonald, B.C.T. White, I. Astrom, M.E. Keene, A.F. Melville, M.D. Reynolds, J.K. 2007. Discharge of weathering products from acid sulfate soils after a rainfall event, Tweed River, eastern Australia. Applied Geochemistry. 22: 2695 – 2705,

Manning, B.A. Goldberg, S. 1997. Arsenic(III) and arsenic(V) adsorption on three California soils. Soil Science. 162(12):886 – 895.

Manning, B.A. Suarez, D.L. 2000. Modeling arsenic(III) adsorption and heterogenous oxidation kinetics in soils. Soil Science Society of America Journal. 64:128 – 137.

Marin, A.R. Masscheyleyn, P.H. Patrick Jr., W.H. 1993. Soil redox-pH stability of arsenic species and its uptake on arsenic uptake by rice. Plant and Soil. 152(2):245 - 253

Martinez, C.E. Motto, H.L. 2000. Solubility of lead, zinc, and copper added to mineral soils. Environmental Pollution. 107: 153 – 158.

Marcus, A.W. Meyer, G.A. Nimmo, D.R. 2001. Geomorphic control of persistent mine impacts in a Yellowstone Park stream and implications for the recovery of a fluvial system. Geology. 29(4):355- 358.

Montana Department of Environmental Quality (MT DEQ). 2005. A guide to abandoned mine reclamation. Published online at: http://www.deq.state.mt.us/abandonedmines/bluebook.asp. Accessed January 2008.

Montana Department of Environmental Quality (MT DEQ). 2006. Montana numeric water quality standards. Circular DEQ-7. Helena, MT. 40p.

Montana Department of Environmental Quality (MT DEQ). 2008. Priority mine sites. Published online at: http://www.deq.state.mt.us/abandonedmines/priority.asp. Accessed January 2008.

Nelson, D.W. Sommers, L.E. 1996. Total carbon, organic carbon, and organic matter. *In Methods of soil analysis part 3 – chemical methods* [Sparks, D.L. editor]. Madison, WI. Soil Science Society of America. 1390p.

Neuman, D. R. Vandeberg, G.S. Blicker, P.B. Jennings, S.R., Ford, K. 2005. Phytostabilization of acid metalliferous mine tailings at the Keating sing in Montana. Presented at the National Meeting of the American Society of Mining and Reclamation. ASMR.

Niyogi, D.K. Lewis, Jr., W.M. McKnight, D.M. 2001. Litter breakdown in mountain streams affected by mine drainage: biotic mediation of abiotic controls. Ecological Applications. 11: 506 – 516.

Niyogi, D.K. Lewis, Jr., W.M. McKnight, D.M. 2002. Effects of stress from mine drainage on diversity, biomass, and function of primary producers in mountain streams. Ecosystems. 5:554 – 567.

O’Dell, R. Silk, W. Green, P. Claassen, V. 2007. Compost amendment of Cu-Zn minespoil reduces toxic bioavailable heavy metal concentrations and promotes establishment and biomass production of *Bromus carinatus* (Hook and Arn.). Environmental Pollution. 148:115 – 124.

Ohio State University Research Foundation. 1971. Acid mine drainage formation and abatement. EPA 14010 FPR 04/71.

O’Neill, P. 1995. Arsenic. *In Heavy metals in soils 2nd edition* [Alloway, B.J. editor]. Glasgow, UK. Blackie academic and professional. 368 p.

Peppas, A. Komnitsas, K. Halikia, I. 2000. Use of organic covers for acid mine drainage control. Minerals Engineering. 13:563 – 574.

Pioneer Technical Services, Inc. 1994. Abandoned hardrock mines priority sites: summary report. Montana Department of State Lands-Abandoned Mine Reclamation Bureau.

PRC Environmental Management, Inc. (PRC EMI). 1997. Expanded engineering evaluation/cost analysis for the Alta Mine Jefferson County, MT. Prepared for Montana DEQ.

Pulford, I.D. Watson, C. 2003. Phytoremediation of heavy metal-contaminated land by trees—a review. Environmental Pollution. 29:529 – 540.

R version 2.5.1. (2007). The R foundation for statistical computing. Copyright © 2007.

Reclamation Research Unit. Bitteroot Restoration, Inc. (RRU-BRI). 2004. Clark Fork River riparian evaluation system a remedial design tool. Prepared for US EPA, region 8. Montana office.

Reddy, K.J. Wang, L. Gloss, S.P. 1995. Solubility and mobility of copper, zinc, and lead in acidic environments. Plant and Soil. 171:53 – 58.

Rengel, Z. 2002. Role of pH in availability of ions in soil. *In Handbook of plant growth: pH the master variable* [Rengel, Z. editor]. New York, NY. Marcel Dekker, Inc. 446 p.

Runkel, R.T. Kimball, B.A. 2002. Evaluating remedial alternatives for an acid drainage stream: application of a reactive transport model. Environmental Science and Technology. 36:1093 – 1101.

Ryan, P.J. Gessel, S.P. Zasoski, R.J. 1986. Acid tolerance of pacific northwest conifers in solution culture. Plant and Soil. 96:239 – 257.

Schippers, A. Jozsa, P.G. Sand, W. Kovacs, Z.M. Marian, M. 2000. Microbiological pyrite oxidation in a mine tailings heap and its relevance to the death of vegetation. Geomicrobiology Journal. 17:2, 151 – 162.

Schroth, A.W. 2001. Geochemical pathways and processes in an acid mine drainage system, Alta mine, MT. M.S. thesis, Geology. Northern Arizona University. Flagstaff, AZ.

Schroth, A.W. Parnell Jr., R.A. 2005. Trace metal retention through the schwertmannite to goethite transformation as observed in a field setting, Alta mine, MT. Applied Geochemistry. 20:907 – 917.

Sobek, A.A. Schuller, W.A. Freeman, J.R. Smith, R.M. 1978. Field and laboratory methods applicable to overburdens and minesoils. U.S. Environmental Protection Agency Environmental Protection Technology. Cincinnati, OH: EPA-600/2-78-054.

Soucek, D.J. Cherry, D.S. Currie, R.J. Latimer, H.A. Trent, C.G. 2000a. Laboratory to field validation in an integrative assessment of an acid mine drainage-impacted watershed. Environmental Technology and Chemistry. 19: 1036 -1043.

Soucek, D.J. Cherry, D.S. Trent, C.G. 2000b. Relative acute toxicity of acid mine drainage water column and sediments to *Daphnia magna* in Puckett’s Creek watershed, Virginia, USA. Archives of Envirionmental Contamination and Toxicology. 38: 305 – 310.

Sracek, O. Choquett, M. Gelinas, P. Lefebvre, R. Nicholson, R.V. 2004. Geochemical characterization of acid mine drainage from a waste rock pile, Mine Doyon, Quebec, Canada. Journal of Contaminant Hydrology. 69: 45 – 71.

Sullivan, A.B. Drever, J.I. 2001. Spatiotemporal variablilty in stream chemistry in a high-elevation catchment affected by mine drainage. Journal of Hydrology. 252: 237 – 250.

Tafi, T.C. 2006. Reclamation effectiveness at three reclaimed abandoned mine sites in Jefferson County, Montana. M.S. Thesis, Land Rehabilitation. Montana State University. Bozeman, MT.

Tan, K.H. 2005. Soil sampling, preparation, and analysis 2nd edition. Boca Raton, FL: CRC Press. 623 p.

Temminghoff, E.J.M. Van Der Zee, S.E.A.T.M. De Haan, F.A.M. 1997. Copper mobility in a copper contaminated sandy soil as affected by pH and solid and dissolved organic matter. Environmental Science and Technology. 31:1109 – 1115.

Tetra Tech 1997. Hydrogeologic evaluation of the Alta mine site, Jefferson County, MT. Prepared for MTDEQ.

Tetra Tech. 2002. Construction report for the Alta lower waste rock dump reclamation project, Jefferson County, MT. Prepared for MT-DEQ.

Thomann, R.V. Mueller, J.A. 1987. Principles of surface water quality modeling and control. New York, NY. Harper Collins Publishers. 644p.

Tordoff, G.M. Baker, A.J.M. Willis, A.J. 2000. Current approaches to the revegetation of metalliferous mine wastes. Chemosphere. 41, 219 – 228.

U.S. Department of Agriculture, Natural Resources Conservation Service (USDA, NRCS). 2007. National Soil Survey Handbook, title 430-VI. [Online] Available: http://soils.usda.gov/technical/handbook/

U.S. EPA. 1995. In test methods for evaluating solid waste, 3rd edition, 3rd update. Washington, D.C.

U.S. EPA. 2006. Framework water quality restoration plan and total maximum daily loads (TMDLs) for the Lake Helena Watershed Planning Area: volume II final report. Prepared for the MT-DEQ.

U.S. Environmental Protection Agency National Research Laboratory (US EPA, NERL). 1993. Methods for the determination of inorganic substances in environmental samples. EPA/600/R-93/100.

Weggler-Beaton, K. McLaughlin, M.J. Graham, R.D. 2000. Salinity increases cadmium uptake by wheat and Swiss chard from soil amended with biosolids. Australian Journal of Soil Research. 38(1): 37 – 47.

Whitney, D.A. Lamond, R.E. 1993. Liming acid soils. Kansas State University Agricultural Experiment Station and Cooperative Extension Service. MF-1065.

Wu, L. Antonovics, J. 1975. Zinc and copper uptake by *Agrostis stolinifera,* tolerant to both zinc and copper. New Phytologist. 75:231 – 237.

Xian, X. 1989. Effect of chemical forms of cadmium, zinc, and lead in polluted soils on their uptake by cabbage plants. Plant and Soil. 113: 257 – 264.

Ye, Z.H. Wong, J.W.C. Wong, M.H. 2000. Vegetation response to lime and manure compost amendments on lead/zinc mine tailings: a greenhouse study. Restoration Ecology. 8:3, 289 – 295.

Zhixun, L. Herbert Jr., R.B. 1996. Heavy metal retention in secondary precipitates from a mine rock dump and underlying soil, Dalarna, Sweden. Environmental Geology. 33:1 – 12.