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**Environmental Contamination by Trace Metals at the Inactive
Pacific Mine Site, American Fork Canyon:
Legal and Remediation Implications**

By

Phyllis Ann Bustamante

A project submitted in partial fulfillment of the requirements for the degree of
Master of Science
from the
Department of Landscape Architecture and Environmental Planning

UTAH STATE UNIVERSITY
Logan, Utah
1996

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PURPOSE

Environmental Planners are often involved in decision making processes concerning the future use and treatment of the environment. In order to make educated decisions on land uses, planners must be familiar with concepts and applications from a wide variety of disciplines. The purpose of this project is to gain knowledge of three specific aspects in environmental planning as they apply to hardrock mining. While there are many factors involved in environmental planning, the following have been considered for this project: 1) environmental laws and regulations, 2) data and conditions indicating a degrading environment, and 3) remediation measures for areas that have been disturbed. While there are a multitude of land uses and environmental conditions, it would be beyond the scope of a masters project to cover legal, quantitative and remediation issues for more than one land use. For this project, hardrock mining and trace metal contamination from an inactive mine site in Utah have been selected as the focus.

INTRODUCTION

Over the next decade, the environment will be the most important issue facing the minerals industry (Ary, 1990). Mining and its associated activities can be a source of considerable environmental damage (Table 1). Mining activities can cause surface and ground water pollution, damage the land and subsurface strata, and destroy terrestrial and aquatic habitat. While mining contributes to environmental degradation, it may be considered a necessary activity because it provides natural resources with a variety of commercial uses.

Table 1. Potential Effects of Mining Activities on the Environment (UNEP, 1991).

	Surface Water Pollution	Ground Water Pollution	Air Pollution	Solid Waste	Excavation	Noise and Vibration	Remarks
Human Health and Activity	soluble contaminants in domestic or agri. use deposition on agri. lands industrial water use	soluble contaminants in wells, springs water source depletion from drop in water table	dust blown on inhabited areas and agricultural lands	hazards related to lack of stability of waste deposits		effects on human health damage to buildings from blasting	impacts on ground water - depends on hydrogeology
Fauna	alteration of aquatic fauna destruction of fish species, accumulation of toxic elements				loss of habitat	disturbance habitat	address - migration corridors, watering areas, endangered species
Flora	alteration of aquatic flora		accumulation in plants of toxic elements carried by dust		loss of habitat		consider - species with limited geographical extent
Land Use	sand deposition in river channels, sea shallow zones			land disturbance loss of agri. land	land disturbance land subsidence		

The hardrock mining industry has existed in the United States as a major industry for more than one hundred years (Lyon et al. , 1993). The main piece of Federal legislation regulating hardrock mining operations is the Mining Law of 1872 which simply

opens Federal lands to exploration and purchase for mining purposes. The Mining Law of 1872 does not mandate prevention of or mitigation for environmental impacts created by mining operations. In the past, hardrock mining facilities were operated solely under the 1872 law. Today, there are state programs regulating active hardrock mines, but a Federal oversight program does not exist, and abandoned and inactive hardrock mines are not regulated under some state hardrock programs. In the past many miners extracted minerals and left behind unreclaimed spoil and tailings piles leading to the degradation of environmental quality both on and off the mine site.

Today there are an estimated 557,650 abandoned hardrock mine sites nationwide (Lyon et al., 1993), and an estimated 17,000 in Utah. Hardrock mine sites may contain trace metals and toxic contaminants such as arsenic, asbestos, cadmium, copper, cyanide, lead, mercury, sulfur, and zinc. Through migration, mine wastes have contributed to the pollution of approximately 12,000 miles of water ways, and 180,000 acres of polluted lakes and reservoirs in the United States (Lyon et al. , 1993).

While some nationwide approximations of pollution from abandoned and inactive mine sites exist, the extent of environmental damage in Utah is unclear because assessment of abandoned mines often focuses on public safety and health threats such as adits, shafts, and other immediate physical dangers, and neglects environmental quality aspects such as water, soil, and sediment contamination. In Utah, some environmental quality data are available for American Fork Canyon and the Cottonwood Canyons, but information is limited to water quality. Existing reports do not address potential trace metal contamination of soils or stream sediments which may negatively effect aquatic environments. A water quality study conducted by Lidstone and Anderson (1993) revealed elevated levels of trace metals in solution from abandoned and inactive mine sites in American Fork Canyon. The study documented water quality data, but did not assess trace metals in sediments or soil around the mine site. Because the report does not contain data on metal levels in soils

and sediments, knowledge on environmental quality on and around the mine sites is limited.

With trace metals, only a small proportion remains in the soluble fraction. The major portion exchanges or precipitates out of solution and becomes associated with suspended or bottom sediments (Luoma, 1988). Sediments constitute a concentrated pool of metals in aquatic environments and, under certain conditions, this pool of metals may become bioavailable and potentially toxic to aquatic life (Luoma, 1989). Because trace metals often reside in sediments, it is important to determine what levels of metals exist in the various fractions. A fraction refers to an analytes (elements being analyzed for) position in soil or sediment - bound to organic matter, adsorbed on to clay or other particles.

At mine sites and other areas impacted by human activities, trace metals can be found in one or more fraction: 1) dissolved in solution, 2) occupying inorganic exchange sites, 3) specifically adsorbed on inorganic soil constituents, 4) complexed with organic matter, and 5) precipitated as pure or mixed solids (McLean, and Bledsoe, 1992). The fraction in which a metal exists largely determines its mobility and bioavailability (Luoma, 1989; Ramos et al., 1994). The more mobile and bioavailable metals are, the more they are considered an environmental threat.

Unlike organic compounds, trace metals persist in the environment because they may exist in elemental form and are not susceptible to further decomposition. While trace metals occur naturally in relatively small amounts in biological systems and some metals are essential to life, at sufficient concentrations all metals become toxic to plant and animal life (Luoma 1988, Förstner and Wittman 1979; Adriano 1986).

Because of the potential toxicity to biological systems, it is important to recognize levels of metals in sediment fractions that can potentially remobilize. The current approach to assessing abandoned hardrock mines does not consider metal contamination at such sites. In order to gain further knowledge of trace metal contamination and the

bioavailability of trace metals from an inactive mine site in Utah, the Pacific Mine in American Fork Canyon has been investigated for this project.

OBJECTIVES

The objectives of this project were: 1) to review the legal requirements (Federal and State of Utah) for reclamation and pollution prevention from abandoned and active hardrock mines, 2) to obtain concentrations of metals in sediments at the Pacific Mine site and determine if the metals are remobilizing, or bioavailable, and 3) to catalogue remediation and reclamation techniques for hardrock mines where metal contamination has occurred.

RECLAMATION REQUIREMENTS FOR HARDROCK MINING

With increasing awareness of environmental degradation from mine sites, some regulative measures have been taken to avoid further environmental deterioration from the hardrock mining industry. Many of the regulations have only been promulgated within the past few decades, and they only apply to active mine sites. Because of this, abandoned or inactive sites may not be regulated under any act or regulation other than the Mining Law of 1872. This section will review the legal requirements (Federal and State of Utah) for reclamation and pollution prevention from abandoned and inactive, and active hardrock mines.

General Mining Law of 1872

Traditionally, the hardrock mining industry operated under the Mining Law of 1872. Because environmental awareness has only increased within the last half of this century, the Mining Law of 1872 was passed without making any stipulations or mention of environmental requirements for mining operations. The Forty-Second Congress. Sess. II. Ch. 152 of May 10, 1872 titled "An Act to promote the Development of the mining Resources of the United States" states;

... all valuable mineral deposits in lands belonging to the United States, both surveyed and unsurveyed, are hereby declared to be free and open to exploration and purchase, and the lands in which they are found to occupation and purchase, by citizens of the United States...

Chapter 152 continues with specifications and procedures for making mining claims and filing for patents, but within sixteen chapters, fails to include remediation or reclamation requirements for hardrock mining facilities.

Because environmental quality was not addressed in the Mining Law of 1872, actions have been taken to reform the act. Two bills were introduced during the 103rd Congress in 1994 to reform the 1872 Mining Law. The reforms would include reclamation requirements, and would establish abandoned hardrock mine cleanup programs. The

program for the cleanup of abandoned mine lands introduced in the House of Representatives, Bill HR.322, would be funded by royalties from active hardrock mining operations. The Senate Bill, S.775, presented a program for abandoned mine cleanup, but did not provide a means of funding. The 103rd Congress was not able to reach a compromise, so neither of the reform bills nor a compromise bill have been passed in the House or Senate (Cravits, 1994). Consequently, a Federal Act to standardize state regulations directed at cleaning up abandoned hardrock mine sites, or requiring active hardrock mine operators to reclaim disturbed land and minimize environmental degradation does not exist.

While a Federal oversight program which directly regulates the reclamation of, and pollution prevention from abandoned, inactive, or active mine sites does not exist, active operations do have to comply with other acts and regulations mandating the prevention of environmental deterioration. The Federal and state acts and codes regulating environmental issues associated with hardrock mine sites to be discussed in this paper are: the National Forests Service Minerals Codes, the Bureau of Land Management Hardrock Minerals Leasing Codes, the National Environmental Policy Act (NEPA), the State of Utah Minerals Reclamation Program, The Clean Water Act and Utah's Surface water program, the Resource Conservation and Recovery Act (RCRA) (even though it excludes mining), and the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA). Because the Pacific Mine has not been in production for approximately forty years, most of the laws and regulations discussed in this section do not apply to the study site for this project. They have been included to show what is currently done to prevent future environmental degradation from hardrock mining operations.

Federal Land Management - National Forest Service and the Bureau of Land Management

The two land agencies which control the majority of western Federal lands, the National Forest Service (NFS) and the Bureau of Land Management (BLM), have adopted regulations which address environmental issues pertaining to hardrock mining.

The National Forest Service states the purpose of 36 C.F.R. CH. 11 Subpart A § 228.1; 1994

... to set forth rules and procedures through which use of the surface of National Forest System lands is in connection with operations authorized by the United States mining laws, which confer a statutory right to enter upon the public lands to search for mineral, shall be conducted so as to minimize adverse environmental impacts on National Forest System surface resources. It is not the purpose of these regulations to provide for the management of mineral resources; the responsibility for managing such resources is in the Secretary of the Interior.

While the NFS does not intend on managing mineral resources, it does require claimants to file a notice of intent to mine with the local district ranger before commencing operations that will cause surface disturbance. If the district office determines that mining will significantly disturb surface resources, the miner must file a plan of operations which is to include the measures needed to meet the requirements for environmental protection.

The requirements for environmental protection on NFS land are directed at minimal damage to surface resources. Requirements include air quality, water quality, solid wastes, scenic values, fisheries and wildlife habitat, roads, and reclamation. All of these requirements are to meet standards set by other Federal and state acts and regulations. In addition to conforming to environmental laws, the final operating plan must include a bond requirement to cover the costs of damage, or unfinished reclamation. While provisions have been set for mining operations on NFS land, regulations cannot extend beyond preserving and protecting the National Forests from needless damage by miners. It is recognized that miners have statutory right to mine Federal land (36 C.F.R. CH. 11 Part § 228.1-§ 228.15). The Forest Service's reclamation regulations apply only to mining disturbances created after 1974 (GAO, 1988).

Mining activities on BLM lands follow regulations similar to those regulating NFS land mining operations. One of the main differences in regulations is that BLM regulations do not require a plan of operations to be submitted for approval for mines disturbing less than five acres of ground per year. Another difference is that the BLM does not require mine operators to file a reclamation bond for all operations. Mining operations disturbing less than five acres must simply provide a notice of intent to mine to the BLM (43 C.F.R. CH. 11 § 3809). The sub-five acre cut off exempts approximately 2000 mining operations each year (Coggins et al. 1993). The Bureau of Land Management's reclamation requirements apply to mining disturbances created in 1981 or after (GAO,1988).

While regulations mandating reclamation of sites disturbed by hardrock mining on Federal lands do exist, there is still a problem of unreclaimed mine lands on BLM and NFS land. The General Accounting Office (GAO) estimates that 424,049 acres of Federal land in 11 western states disturbed by hardrock mining operations are unreclaimed. Of this amount, 281,581 acres are abandoned, suspended, or unauthorized mining operations, and 142,468 acres are from active operations (GAO, 1988). Of the 281,581 acres of unreclaimed land, 60,054 acres are either undergoing reclamation or did not require reclamation at the time of the GAO review. Table 2 provides a summary of the types of reclamation needed at abandoned, suspended, and unauthorized hardrock mining sites on Federal land.

The GAO (1988) has projected the total cost to reclaim the 281,581 acres of disturbed Federal land to be approximately \$284 million. The estimate is broken down as:

\$196 million to reclaim 161,314 acres of land disturbed prior to the date reclamation regulations took effect,

\$49 million to reclaim 48,874 acres of land disturbed after the date reclamation regulation took effect

\$39 million to reclaim 71,393 acres of land that the date of disturbance is unknown to Federal officials (GAO,1988).

At the time of the GAO report (1988), Utah had spent \$13,389 in state funds for work on NFS land, and \$20,667 in funds from title iv (Abandoned Mine Lands) of the Surface

Mining Control and Reclamation Act of 1977 (SMCRA). SMCRA is the Federal oversight Act ensuring reclamation on coal sites. SMCRA money was primarily used to close mine shafts on BLM land (GAO, 1988).

Table 2. Types of Reclamation Needed For Federal Land from Abandoned, Suspended, and Unauthorized hardrock mines (GAO, 1988).

<u>Reclamation Action Needed</u>	<u>Acres</u>
Reshaping or recontouring of surface	162,911
Reseeding	157,322
Top Soil Replacement	68,040
Vegetation Replacement	57,179
Mine waste, harmful materials, or other litter requiring removal or disposal	24,916
Measures to control erosion, landslides, and water runoff	74,236
Closing of mine shafts and openings	26,034
Capping or plugging drill holes	5,271
Removal of structures, equipment, and materials	23,159
Other actions needed to restore land or ensure public safety	6,548

The National Environmental Policy Act

The National Environmental Policy Act of 1969 (NEPA), 42 U.S.C.A. 4331-4335 enacted by Congress requires all Federal agencies to investigate all potential adverse environmental effects caused by major actions that are anticipated on the lands that they manage. Section 102 (2)(C) states;

all agencies of the Federal Government shall: include in every recommendation or report on proposals for legislation and other major Federal actions significantly affecting the quality of the human environment a detailed statement by the responsible official on -

- (i) The environmental impact of the proposed action,
- (ii) Any adverse environmental effects which cannot be avoided should the proposal be implemented,
- (iii) Alternatives to the proposed action,
- (iv) The relationship between local short-term uses of man's environment and the maintenance and enhancement of long term productivity, and

- (v) Any irreversible and irretrievable commitments of resources which would be involved in the proposed action should it be implemented.

In order to conform with the above codes, the NEPA process generally consists of the preparation of an Environmental Assessment (EA), and an Environmental Impact Statement (EIS). An Environmental Assessment is a document including the environmental impacts of a proposed action and alternative actions on Federal lands in order to determine if an EIS is needed. An EIS is a decision making tool designed to assess significant environmental impacts, and to identify reasonable alternatives which would avoid or minimize adverse impacts to the human environment (40 C.F.R § 1502.1). Through the EIS process, Federal agencies are able to determine if a proposed action would cause too large of a threat to the human environment, and agencies are able to base their land use decisions on the acquired information.

Because Federal lands are open to mining activities, it should follow that an EA and possibly an EIS would be required prior to commencing mining operations on Federal land (§ 102 (2)(C)(i)). In a 1980 court case, (*State of South Dakota v. Andrus*, United States Court of Appeals, Eighth Circuit), the State of South Dakota filed an action against the Department of Interior and Pittsburgh Mining arguing that an EIS must be prepared prior to the issuance of a mineral patent on NFS land. The Judge ruled that an EIS was not required because the issuance of a patent is not a "major action" taken by the government agency, and the issuance of a patent is not considered to enable a party to begin mining operations (Coggins et al. 1993).

In a case involving BLM land, (*Sierra Club v. Penfold*, United States Court of Appeals, Ninth Circuit, 1988), the Sierra Club filed action stating that a placer mining operation in Alaska should be required to prepare an EA for a "Notice" mine, a mining operation causing a cumulative surface disturbance of five acres or less per year (43 C.F.R. Ch. 11 § 3809.1). The ruling went against the Sierra Club because a Notice mine was not considered a major Federal action (Coggins et al. 1993).

Even though the previous two mining cases have been identified as exempt from the NEPA process, an EA or and EIS may be required prior to commencing mining operations depending on the area of land to be disturbed, and the anticipated extent of damage to the surrounding environment. BLM does require an EA for operations that will exceed five acres or for sites proposed on environmentally sensitive areas (43 C.F.R. Ch. 11. § 3809).

In the State of Utah, operations which will disturb more than five surface acres are required under R647-4-109 of the Utah regulations to prepare an impact statement describing the potential impacts to: ground and surface water, endangered species and their habitats, impacts to soil resources, slope stability, erosion control, air quality and public health and safety.

State of Utah Minerals Reclamation Program

The Utah Mined Land Reclamation Act of 1975 Title 40 Chapter 8 paragraph 2 states:

"The Utah Legislature finds that : (1) A mining industry is essential to the economic and physical well-being of the State of Utah and the nation. (2) It is necessary to alter the surface of the earth to extract minerals required by our society, but this should be done in such a way as to minimize undesirable effects on the surroundings. (3) Mined land should be done in such a way as to prevent conditions detrimental to the general safety and welfare of the citizens of this state and to provide for the subsequent use of the lands affected. Reclamation requirements must be adapted to the diversity of topographic, chemical, biologic, economic and social conditions in the areas in which mining takes place."

In accordance with the legislature, the State of Utah Department of Natural Resources, Division of Oil, Gas and Mining (DOGGM) established the Minerals Reclamation Program R647-1 through R647-5. The regulations specify reclamation and pollution prevention for three types of mining activities: exploration, small mining operations (operations affecting less than five surface acres), and large mining operations. Even though the regulations are broken down by type of activity, the requirements are similar for all three.

The general "Reclamation Practices" (R647-2-109 (exploration), 3-109 (small operations), and 4-107-111 (large operations)), require operators to minimize hazards

which pose physical dangers to public safety and welfare. The sections also contain regulations for: drainages, erosion control, deleterious materials, future land use, slopes, highwalls, roads and pads, dams and impoundments, trenches and pits, structures and equipment, topsoil redistribution, and revegetation. Even though regulations exist to prevent environmental deterioration from mining operations, following each section of Reclamation Practices, there is a section on "Variance" which allows operators to file for a variance against the regulations requiring reclamation.

A measure used by DOGM to insure the reclamation of mined sites is bonding. Operations that will result in the disturbance of more than five acres, are required to pay a surety [R647-2-111 (exploration), and 4-113 large mining operations]. If an operation fails to reclaim a site, the surety is used by the state for reclamation, and legal fees for taking action against the mining operation.

The Clean Water Act and Utah Administrative Code - Standards of Quality For Waters of the State

In 1972 Congress passed the Water Pollution Control Act Amendments (PL 92 - 500) (originally passed in 1948). The objective of the act was to "restore and maintain the chemical, physical, and biological integrity of the nation's waters". The act also specified a national goal of providing "water quality which provides for the protection and propagation of fish, shellfish, and wildlife and provides for recreation in and on the water", and prohibited the discharge of toxic pollutants into waterways. In 1977, the Water Pollution Control Act was amended and retitled The Clean Water Act (CWA) (PL 95 - 217).

The main impact the CWA has had on mining operations is from the National Pollutant Discharge Elimination System (NPDES). The CWA requires a NPDES permit for pollutant discharges into navigable waters. Because the CWA is a primacy act, EPA has delegated administration of the NPDES program to the Utah Department of Environmental Quality, Division of Water Quality. The Division of Water Quality is responsible for permitting waste water discharges into Utah waters.

In accordance with the CWA, The Utah Division of Water Quality devised "Standards of Quality for Waters of the State of Utah" R317-2, Utah Administrative Code. The standards are arranged by a classification system that applies to all waters within the state of Utah. The first category designations, High Quality Waters, are under R317-2-3 "Antidegradation Policy". High quality waters have been given 2 categories. Categories 1 and 2 are similar and refer to surface waters with "exceptional recreational or ecological significance or have been determined to be a State or National resource requiring protection". The difference between 1 and 2 is that new point source discharges of treated or untreated waste waters are prohibited from entering into waters after they have been given category 1 status. Category 2 waters do accept permitted discharges if they do not threaten the existing water quality (R317-2-3.2 and 3.3). Waters that have been designated category 1 are "all surface waters geographically located within the outer boundaries of U.S. National Forests whether on public or private lands..." (R317-2-12).

Another classification system within the State of Utah Codes are "Use Designations" R317-2-6. This classification system categorizes waters into classes by use. The range of classes goes from 1 to 6 with 1-3 having subclasses. The class determines the quality of water allowed to enter into receiving waters. Generally, class 1 designates water for domestic uses, class 2 for recreation, class 3 for wildlife, class 4 for agricultural, and classes 5 and 6 for varied uses.

The North Fork River, adjacent to the Pacific Mine and receiver of portal discharge, is grouped with the American Fork Creek and its tributaries (R317-2-13.5.c.). This group of surface waters are classed:

- 2B protected for secondary contact recreation such as boating, wading, or similar uses;
- 3A protected for cold water species of game fish and other cold water aquatic life, including the necessary aquatic organisms in their food chain;
- 4 protected for agricultural uses including irrigation of crops and stock watering.

Because the North Fork is located within National Forest Service boundaries, it is also designated as High Quality Water category 1.

The Resource Conservation and Recovery Act (RCRA)

The Resources Conservation and Recovery Act of 1976 (RCRA) (PL - 94 -580).
was established as an act to:

provide technical and financial assistance for the development of management plans and facilities for the recovery of energy and other resources from discarded material and for the safe disposal of discarded materials and to regulate the management of hazardous materials.

Through its designation, RCRA has become the principal Federal statute regulating hazardous wastes. Hazardous wastes as defined under § 1004 of the act are:

a solid waste, or combination of solid wastes, which because of its quantity, concentration, or physical, chemical, or infectious characteristics may (a) cause or increase in mortality or an increase in serious irreversible, or incapacitating reversible illness (b) pose a substantial present or potential hazard to human health or the environment when improperly treated, stored, transported, or disposed of, or otherwise managed.

The hardrock mining industry does produce solid wastes from extraction and milling procedures of non-ferrous metal compounds. Because base metals (commercial metals such as copper, lead or zinc, and precious metals, gold and silver), occur in association with other metals, it is likely that solid wastes produced from the mining industry will contain cadmium and lead (UNEP, 1991) which have both been listed under RCRA as hazardous substances (40 C.F.R. § 261.24). It should follow that under RCRA, mining wastes, like other solid wastes, are subject to classification as hazardous wastes, and are subject to regulation under RCRA if they contain elevated levels of Cd and lead.

In 40 C.F.R. § 261.4 (b)(3), "mining overburden returned to a mine site", and § 261.4 (b)(7) "solid waste from the extraction, beneficiation, and processing of ores and minerals", have all been excluded from RCRA authority. These exclusions were made because of the Beville Amendment enacted in 1980. The Beville amendment exempted mining wastes from RCRA's hazardous waste regulation pending studies by the EPA to

determine if mining wastes warranted regulation under RCRA (Dwyer, 1992). The EPA study concluded that mining wastes should not be regulated under RCRA.

A mining waste study conducted by the University of California at Berkeley (1988) stated EPA's reasons for excluding mining wastes from RCRA were:

- (1) most mines are relatively remote from human populations and sources of drinking water;
- (2) most hazardous waste problems come from abandoned mines, and retrofitting those sites to meet current RCRA standards would be both prohibitively expensive and of dubious value (because the methods are largely untested);
- (3) only about 8 percent of mining wastes generated annually would be defined as hazardous under RCRA;
- (4) the cost of regulating existing mining operations under RCRA, although it would vary substantially from site to site, is potentially huge.

The UC Berkeley (1988) study attained this information from EPA's *Report to Congress: Wastes from the Extraction and Benefication of Metallic Ores, Phosphate Rock, Asbestos, Overburden from Uranium Mining, and Oil Shale* 1-9 (1985). The UC report noted that the EPA report addressed the regulation of wastes resulting from extraction and benefication, but not the processing of ores and minerals, or the wastes from inactive or abandoned sites, so the conclusions of EPA were derived from incomplete information on mining wastes.

Currently, action has been taken to reauthorize RCRA to include regulations for mining wastes. Representative Al Swift of the House Transportation and Hazardous Materials Subcommittee was expected to introduce major mine waste regulation during the 103rd Congressional Session (Dwyer, 1993). It is expected that the amendments or reauthorization of RCRA will die at the end of the 103rd Congress, but will be pursued during the next session (Morgan, 1994; Cravits, 1994).

Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA)

The Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA), Public Law 96-510 of Dec. 11, 1980 was established as a Federal Act to;

provide for liability, compensation, cleanup, and emergency response for hazardous substances released into the environment and the cleanup of inactive hazardous waste disposal sites.

Depending on the type of ore extracted, composition of overburden, and the mineral extraction procedure, mine wastes may contain substances considered hazardous by CERCLA. Although mine sites may fit the general description of CERCLA sites, often they are not remediated under the Act.

In order for a site to become a Superfund site (eligible for CERCLA dollars), it must be placed on the National Priorities List (NPL) of sites to be reclaimed under CERCLA. Before a site gets on the NPL, it must get into CERCLIS. CERCLIS is the Information System or EPA database containing sites that have been "discovered" and sites that are in various stages of review prior to entering the NPL. The review stages are a "Preliminary Assessment" and a "Site Inspection".

The Preliminary Assessment determines if the site has the potential of scoring high enough on the Hazard Ranking Scoring System (HRS), a risk assessment model giving sites a score of 0 -100 for comparison of other potential sites. The Preliminary Assessment considers any existing environmental quality data, site history, location, hydrogeology, potential risk a site poses to human health, and other available material related to the site. If the reviewer suspects the site will rank high on the HRS, the site is recommended for a Site Inspection.

The Site Inspection is an investigation of the environmental quality of a site. Some of the procedures at the inspection stage are preparation of a sampling plan, a quality assurance plan, and sampling of a site. All of the data accumulated is then used for the HRS model. The score from the HRS is compared with other site's scores, and the highest ranking sites make the NPL. Sites with scores below 28.5 receive no further action.

Because many abandoned hardrock mine sites do not rank high enough on the NPL or never reach NPL status, they are not eligible for cleanup under Superfund.

Presently, the Pacific Mine in American Fork Canyon has been given site discovery status (January 24, 1992), which places it in CERCLIS. The National Forest Service is in the process of preparing a Preliminary Assessment which was due in August of 1994. The Forest Service will ask EPA for an extension on the Preliminary Assessment (Buntrock, 1994).

While some Federal and state regulations do exist to protect the environment from adverse affects caused by the hardrock mining industry, many of the regulations only apply to active mine sites. Table 3 summarizes the notices and permits required in Utah for hardrock mining. This may serve as a preventive measure for future environmental problems, but today there is still the issue of mine sites operating prior to any environmental regulations.

As mentioned in the introduction, inoperative mine sites are responsible for the pollution of approximately 12,000 miles of water ways, and 180,000 acres of polluted lakes and reservoirs in the United States (Lyon et al. , 1993). The environmental quality data available for Utah sites is solely for water.

Metal contamination problems in streams and around mine sites would be more evident in sediments because metals become associated with sediment constituents. In order to gain further knowledge of trace metal contamination and the bioavailability of trace metals from an inactive mine site in Utah, the Pacific Mine in American Fork Canyon has been studied for this project.

Table 3. Summary of Required Notices and Permits to Mine in Utah (Division of Oil Gas and Mining, Revised September 1991).

<u>Agency</u>	<u>S/F</u> [†]	<u>P/N</u> [*]	<u>Comments</u>
Div. of Oil Gas and Mining	S	N	>5 acres of disturbance requires bond, all mining requires a notice of intent
Div. State Lands and Forestry	S		lease and plan required, bond usually required
Div. of Water Rights	S	P	water rights, structures, wells and stream alterations require permits
Div. of Water Quality	S	P	potential of pollution or degradation to ground or surface water requires permits
Div. of Air Quality	S		large mines require approval
Div. of Drinking Water	S	P	drinking water for ≥ 25 people, permits for septic systems, leach fields, and sanitary from local DEQ
Div. Env. Response and Remediation.	S	P	permit required for underground storage tanks
Div. of Radiation Control	S	P	permit required for radio active material
Div. of Solid and Hazardous. Waste	S	P	permit required for any type of landfill
Mine Safety and Health Adm.	F	N	notice prior to start-up and shutdown, must comply with MSHA
Army Corp of Engineers	F	P	permit required near navigable water, and discharge into streams or wetlands
Bureau of Land Management	F	N	notice for < 5 acres of disturbance, operation plan for > 5 acres or environmentally sensitive areas, bond may be required
United States Forest Service	F		plan of operation and bond usually required

† State (S) or Federal (F) Government

* Requirement of Permit from an agency (P) or Notice to an agency (N)

THE PACIFIC MINE AMERICAN FORK CANYON

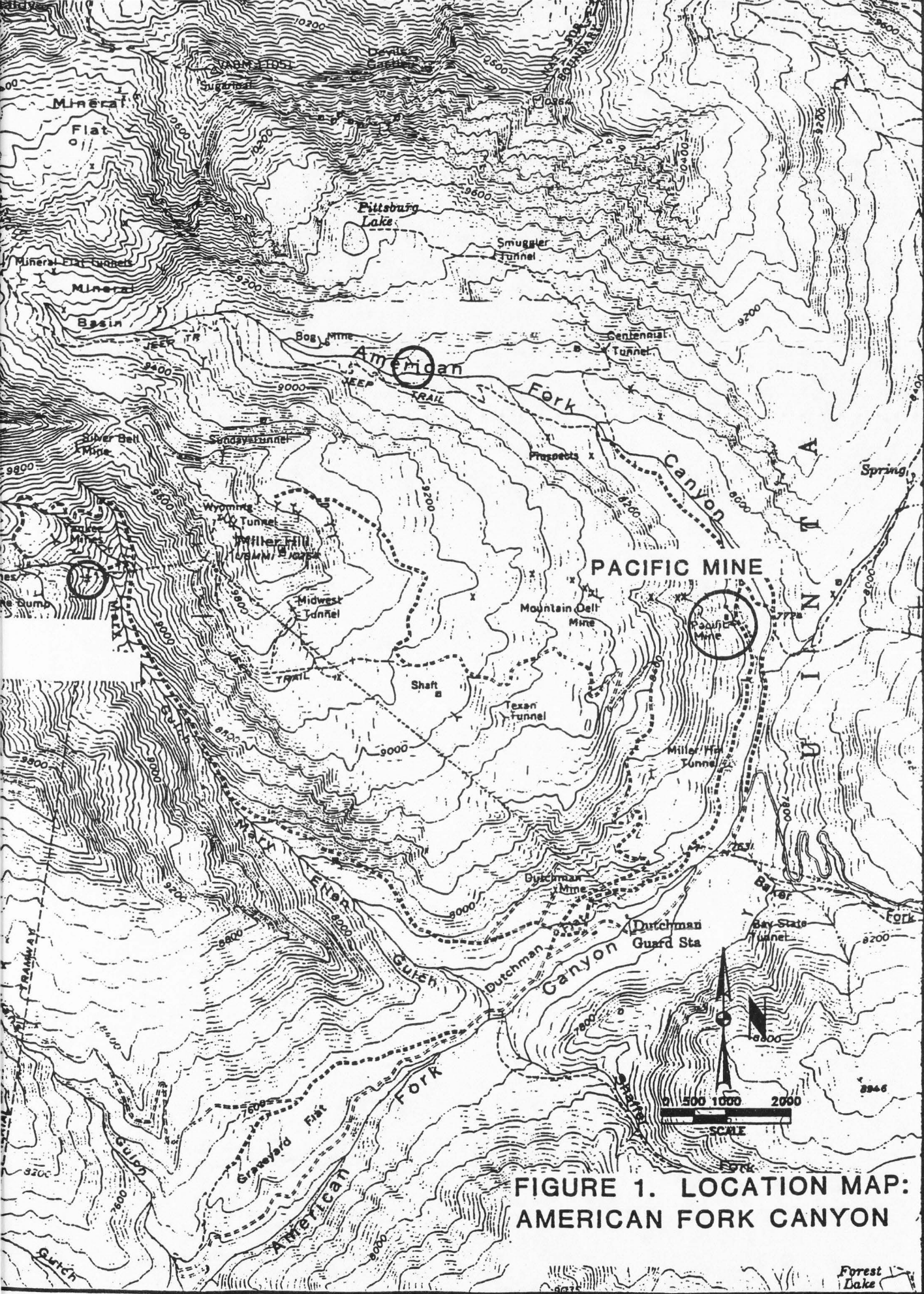
The second objective of this project was to obtain environmental quality data (levels of metals) that would indicate sources of metal contamination in sediments of the North Fork of the American Fork River and to determine if the metals are remobilizing, and bioavailable. This objective was selected to gain knowledge of environmental conditions that threaten the environmental quality of the Pacific mine site and the surrounding area.

The Pacific Mine (7800 feet above sea level and approximately 3 acres in area), located in Mineral Flats of the American Fork Canyon (Figure 1), was selected for this project because of the availability of background data (water quality), the existence of tailings, its proximity to a river, and its accessibility for recreationalists, and sampling.

Site History

Mining in American Fork Canyon was initiated in 1870 when a miner discovered silver and lead bearing ore in the southern canyon (Ercanbrack, 1970). As a result, the American Fork Mining District was organized on July 21, 1870 (Keetch, 1978; Ercanbrack, 1970; Crosland and Thompson, 1994). At a miners meeting on July 21, 1873, the American Fork District decided to adopt the United States Mining Law of 1872. During this period, the early 1870s, many miners made claims in the mining district (Ercanbrack, 1970). One of the claims was called Blue Rock which was acquired by the Pacific Gold Milling and Mining Company in 1904, and in 1912 the company patented the claims (Crosland and Thompson, 1994).

The Pacific Mine, one of the mines purchased by the company, was one of the largest and most successful mines in American Fork Canyon (Crosland and Thompson, 1994). In December of 1916, a mill was completed at the Pacific mine site. The cost of the mill was \$20,000, and was justified because in the same year a rich vein up to twenty-six feet wide was discovered (Keetch, 1978). By the end of



**FIGURE 1. LOCATION MAP:
AMERICAN FORK CANYON**

1919 and into 1920, most of the top grade ore had been extracted and the boom period for the mining district had ended (Ercanbrack, 1970). The Pacific Mine has not been actively mined since the 1950's (Proctor, 1994).

The current patent holder of the inactive Pacific Mine does not intend to extract ore in the near future (Proctor, 1994). In a recent study conducted by the Uinta National Forest, the Pacific Mine was found eligible for the National Register of Historic Places because the site contains structural features and potentially buried deposits, and the mill played a major role in the American Fork Mining District's history. This designation recommends that closure of mine openings be done without disturbing the historical integrity of the mine site (Crosland and Thompson, 1994).

The Pacific Mine portal is currently discharging water which eventually enters the North Fork River. There are tailings heaps that may contain high levels of metals. These two sources may be introducing levels of metals that adversely impact the aquatic life of the North Fork. In order to determine if metal concentrations leaving the mine site are high enough to threaten the environment, the following study has been conducted.

Methods and Materials

Sampling

A sampling scheme was devised to give a preliminary assessment of metals at the Pacific mine site and to determine where metals may enter the North Fork. Because of various constraints (economic and temporal), this is only a preliminary assessment. A full assessment should include a more complete sampling scheme that would better represent temporal and spatial aspects of the site. Temporal monitoring at the site would be required to differentiate seasonal variations in metal levels. Samples of variable depths should be included in a full assessment of metals at the mine site to show vertical distributions of metals.

The preliminary assessment involved eight sites at the Pacific Mine which were sampled in August of 1994 (Figure 2). Water samples were collected and stored in high density polyethylene containers. At the time of sampling, pH was measured for water samples. Sediment and soil samples were collected at a depth of 2 inches with a coated hand shovel and stored in plastic bags. The shovel method was also used by Schintu et al, 1991, but would not be recommended because of loss of particles. Other studies used grab samplers (Förstner and Salomons, 1980) and core samplers (Ramos et al., 1994; Karlsson et al., 1988) which might be a preferred method. A trip blank (water) was also taken to the site on the day of sampling to detect any contamination during the sampling or transport.

Site # 1 is the mine portal where only a water sample was collected to estimate the concentrations of metals coming out of the mine. At site # 2, a soil sample was collected for comparison of metal levels with those in the tailings pile.

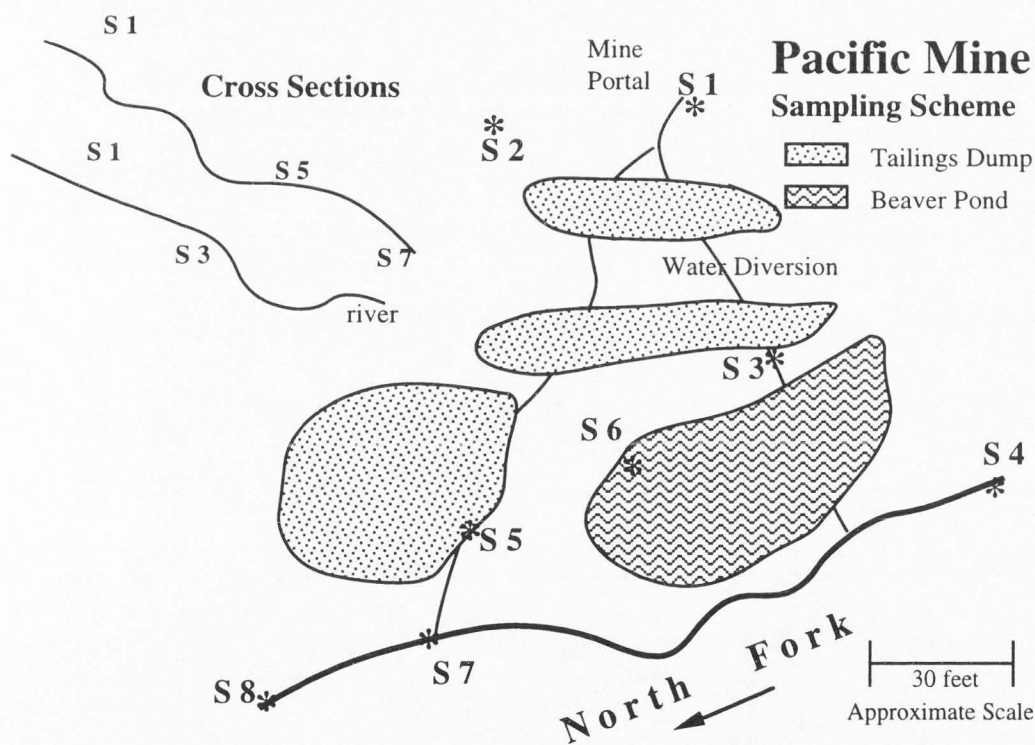


Figure 2. Conceptual Map of the Sampling Scheme at the Pacific Mine.

Site # 2 cannot be considered a background sample because it could have received metal contamination from the overall site disturbances. At site # 3, sediment and water samples were collected. This site is at the end of the diverted channel which carries the portal water into the beaver pond (Figure 3). Site # 4 is approximately twenty feet above the Pacific Mine on the North Fork River. This site will indicate background levels of metals in water and sediment. At site # 5, a water and a tailings sample were taken (Figure 4). Data from this site will indicated what levels of trace metals exist on the site from tailings.



Figure 3. Diversion channel for water from the Pacific Mine portal leading into the Beaver Pond.



Figure 4. Mine discharge flowing over tailings before entering the North Fork River.

Site # 6 is at the southeast end of the beaver pond. If metals remain suspended through the diverted channel, accumulation in the pond is possible (Figure 5). Site # 7 is on the North Fork River directly below (topographically) the tailings pile. This site is susceptible to metal contamination from erosion of the tailings heap. At site # 8, sediment and water samples were collected approximately thirty feet below the Pacific Mine to determine if trace metals are migrating down stream, or if they are remaining relatively close to the mine site. Table 4 is a summary of the sampling sites and reasons for sampling at each site.



Figure 5. Beaver Pond used as a settling pond for runoff from the mine portal.

Table 4. Summary Table Describing Sampling Locations.

<u>Site</u>	<u>Matrix</u>	<u>Location</u>	<u>Rationale</u>
1	water	mine portal	check for metals from mine waters
2	soil	off of tailings dump	check for metals on site (not tailings)
3	water/sediment	end of diversion	metals being transported through diversion
4	water/sediment	upstream in N. F.	background water and sediment
5	water/sediment	large tailings pile	metal check in water above tailings, and tailings
6	water/sediment	beaver pond	metal levels in sediment pond
7	water/sediment	below tailings in N.F.	metal check from erosion from tailings into N.F.
8	water/sediment	down stream in N.F.	migration of metals down stream

Laboratory Analysis

Theory

An experimental approach commonly used for assessing mobility and bioavailability of metals in sediments and soils involve sequential extraction procedures to determine what fraction trace metals exist in. (Luoma, 1988, Ramos et al. 1994, Schintu et al. 1991, Levy et al. 1992, and Rauret et al. 1989). A number of extraction methods have been developed for categorizing metals into operationally defined geochemical fractions. Generally, extraction procedures consist of reacting sediment or soil samples with chemical solutions of increasing strengths (McLean and Bledsoe, 1992; Ramos et al., 1994).

The fractionation procedure for metal analysis selected for this segment of the project is a modified version of the Tessier et al. (1979) method. This method was selected because it is widely used in partitioning studies (Rauret et al., 1989; Levy et al., 1992; Karlsson et al., 1988; and Pardo et al., 1990; Ure et al., 1993) and has been applied to studies specific to mining spoils and wastes (Schintu et al., 1991; and Ramos et al., 1994, Prairie and McKee, 1994). The Tessier method consists of a five step extraction procedure which partitions metals into 1) exchangeable, 2) carbonate bound, 3) iron or manganese oxide bound, 4) organic matter and sulfide bound, and 5) residuals.

For this study, the first two fractions were extracted. These two were selected because metals in these fractions are considered the most likely to remobilize and become bioavailable through solution (Luoma, 1988). The first fraction (exchangeable), may remobilize with an influx of other cations (positive ions) into the system which can exchange cations on the sediment surface. Physical processes such as diffusion, compaction, erosion and bioturbation (disturbance of sediments from biological organisms) may also cause a release of trace metals into solution (Förstner and Salomons, 1980). Metals in the exchange fraction are the most likely pool to remobilize (McLean and Bledsoe, 1992) and become bioavailable (Förstner, 1993). Metals associated with the carbonate fraction may remobilize with changes in pH.

The remaining three fractions were not specifically extracted. They were determined and grouped by subtracting the exchangeable and carbonate bound fractions from the total digestion. The environmental conditions (low pH, changes in redox conditions) required to remobilize or alter these fractions (Förstner, 1993) are unlikely to occur at the Pacific Mine Site. Acid mine drainage or acid conditions are not an issue at this site because of the buffering capacity of the carbonate rich geochemistry. The host rock at the Pacific Mine is limestone (CaCO_3) (Lidstone and Anderson, 1993) which contributes to the acid neutralizing capacity of the system.

Procedure

Sediment and soil samples were taken to the lab, oven dried at 95°C , and sifted through a 2mm mesh sieve in preparation for extractions. Exchangeable metals were extracted by placing 2.0 grams of dry sample into polyethylene centrifuge tubes. Sixteen milliliters of 1M MgCl_2 (pH 7.0) were added to each tube. The tubes were weighed and placed in an agitator for 1 hour at room temperature followed by centrifugation for 30 minutes at 10,000 rpm. Solutions were filtered through Whatman no. 42 filter paper into acid washed high density polyethylene containers and stored until analysis. The residue was weighed in order to subtract the amount of solution and concentration left from the MgCl_2 extraction from the second extraction.

Carbonate bound metals were extracted by adding 16ml of 1M NaOAc (pH 5.0 with HOAc) to the residue from the first extraction. The tubes were continuously agitated for 5 hours at room temperature and then centrifuged for 30 minutes at 10,000 rpm. Solutions were filtered through Whatman no. 42 filter paper into acid washed high density polyethylene bottles for storage until analysis.

Total digestions (chemical method to desorb cations) were performed using a modified version of Standard Method 3050A (EPA, 1986). One gram samples were digested with 10 ml of 1:1 HNO_3 : deionized distilled water (DDW) H_2O in covered

beakers at 95° C on a hot plate for 15 minutes. The samples were cooled and an additional 5 ml of concentrated HNO₃ were added. The slurries were heated for 30 minutes and the step was repeated. The solution was evaporated to 5 ml and cooled. Two milliliters of water and 3 ml of 30% H₂O₂ were added to the slurries and the covered beakers were returned to the hot plate. The samples were heated until effervescence subsided and were cooled. Until reaction was minimal, 30% H₂O₂ in 1 ml aliquots (portion of samples) were added with warming after each addition. After cooling, the samples were diluted with water to 100 ml and were filtered through Whatman no. 42 filter paper into acid washed polyethylene bottles.

The water samples were filtered through a 0.45 μ filter and acidified to a pH < 2 after they were returned to the lab. Blanks, controls, and spiked samples were processed with all of the extraction procedures and water samples. All samples were analyzed with an Inductively Coupled Argon Plasma Emission Spectrometer (ICP). Sodium acetate (NaOAc) extractions were also analyzed with Atomic Absorption Spectrometry because of matrix interferences with the ICP.

Because of low levels of Pb in the water samples, the selection of ICP for analysis of Pb for the water samples was inadequate. A detection limit, "the lowest concentration of an analyte that the analytical procedure can reliably detect" (McLean, 1993), was not specifically established for this procedure. Because a detection limit was not determined, the lowest standard used for dissolved Pb analysis (160 μg/L) is considered the lowest level of reliable detection. While the detection limit may be below the 160 μg/L value for this procedure, Pb concentrations below this level were not determined in this study. In order to determine if dissolved Pb at the Pacific Mine Site is above water quality criteria, (50 μg/L -domestic source standard from Standards of Quality for Waters of the State, Division of Water Quality February 16, 1984), a graphite furnace attachment to a flame A.A. which has lower detection limits should have been used.

Results and Discussion

Site 1

Discharge from the mine portal had a field pH 6.6. This measurement was comparable to the 6.5 value taken in July of 1992 by Lidstone and Anderson Inc. (1992). The pH indicates that acid mine drainage from the portal does not exist. Acid mine drainage would be a possibility because the ore at the Pacific Mine is sulphide ore which is notorious for producing acid conditions (see appendix for acid mine drainage). Limestone (CaCO_3) contributes to the alkalinity of the system and maintains non-acid conditions. Acid conditions are responsible for bringing metals into solution. Because of the near neutral pH (7.0 being neutral), metal levels exiting the portal are lower than they would be under low pH conditions. Table 5 summarizes metal concentrations exiting the portal.

Table 5. Water quality data for water exiting the mine portal (site 1).

<u>pH</u>	<u>Cadmium</u> (Cd) $\mu\text{g/L}$	<u>Copper</u> (Cu) $\mu\text{g/L}$	<u>Lead</u> (Pb) $\mu\text{g/L}$	<u>Zinc</u> (Zn) $\mu\text{g/L}$
6.6	12.2	5.3	< 160	1.9×10^3

The level of Cd, 12.2 $\mu\text{g/L}$, exiting the mine portal is relatively high compared to the level usually found in most fresh waters, < 1 $\mu\text{g/L}$ (Callahan, 1980). The presence of Cd in solution is consistent with mobility literature on Cd. Cadmium is mobile relative to other trace metals (Callahan, 1980; Alloway, 1990; McLean and Bledsoe, 1992) and is likely to remain in solution (Callahan, 1980). If Cd is not retained enroute to the North Fork, mine portal discharge may contribute Cd to the stream.

Copper has a strong affinity for hydrous iron and manganese oxides, clays, and carbonate minerals, and is more likely to be in the solid phase (Callahan, 1980). Because Cu is specifically adsorbed or fixed, it is one of the least mobile trace metals (Baker, 1990). Copper concentrations for unpolluted surface waters in the United States are between 1 - 10 $\mu\text{g/L}$ (EPA, 1984 a.). The Cu level, 5.3 $\mu\text{g/L}$, is considerably lower than the concentration (47 $\mu\text{g/L}$) determined by Lidstone and Anderson (1992). The decrease may

be from internal chemical changes of the mine such as Cu content or increased retention by limestone.

A level of Pb comparable to state water quality standards was not assessed at this site. The Pb concentration at this site was below 160 µg/L. A water quality study conducted by Lidstone and Anderson Inc. (1992) recorded a dissolved Pb concentration of 15 µg/L. The dissolved Pb level at this site is consistent with information provided in mobility literature. Lead rarely remains dissolved and generally precipitates or adsorbs to bed sediments in close proximity to its source (Callahan, 1990). At a pH > 6, Pb is adsorbed on clay or forms Pb carbonate and is retained by soils (McLean and Bledsoe, 1992) or sediments. Lead is one of the least mobile metals and seldom remains in solution (Callahan, 1980; Alloway, 1990; McLean and Bledsoe, 1992). Lead exiting the portal may not enter the North Fork.

The Zn concentration at the mine portal (1.9×10^3 µg/L), is similar to the 1.8×10^3 µg/L level found in 1992 (Lidstone and Anderson, 1992) and is above the fresh water range. Concentrations of Zn in uncontaminated fresh water generally range from 0.5 - 10 µg/L (EPA, 1987). This level reflects the geochemistry of the mine shaft. With a proper source, sulfide ores, high zinc concentrations would be expected. Precipitation is not a major mechanism in immobilizing Zn under most conditions (Callahan, 1980), and Zn compounds have relatively high solubilities (McLean, and Bledsoe, 1992; EPA, 1987). The mine portal is contributing to Zn loadings to the Pacific Mine site.

Because of the pH, metal levels exiting the mine portal are possibly lower than they would be under acid conditions. The concentration of Cd is higher than the criteria for domestic, and agricultural uses for the State of Utah (10 µg/L). This is the only measured metal concentration that exceeds state standards for domestic, recreational and agricultural uses (Pb would need further analysis). Under 40 C.F.R. 440 (J), EPA provides criteria for water discharges using Best Practicable Technology (BPT - average of the best existing performance by well operated facilities within each industrial category). Table 6 provides

the EPA criteria and the concentrations of dissolved metals exiting the portal. Zinc is the only metal that is above mine discharge criteria. Because the portal is not under a discharge permit, action to control metal levels is not required by the patent owner.

Table 6. One day average criteria for mine drainage using BPT and Pacific Mine portal concentrations.

	<u>Cd $\bar{\mu}\text{g/L}$</u>	<u>Cu $\bar{\mu}\text{g/L}$</u>	<u>Pb $\bar{\mu}\text{g/L}$</u>	<u>Zn $\bar{\mu}\text{g/L}$</u>
Portal	12.2	5.3	< 160	1870
EPA Criteria	-	300	600	1500

Site 2

This site was selected as a representative soil sample without tailings. The site cannot be considered a background sample (uncontaminated) because the entire area in and around the mine site is badly disturbed and may have been affected by mining activities in the past. As noted in the sampling section of this chapter, more samples and replicates would be needed for a more accurate estimate of metal content in soil around the mine site.

Site # 2 is topographically situated above the tailings so tailings erosion would not contribute metals to the site. The data obtained from the soil sample is provided in Table 7.

Table 7. Partitioned metal concentrations and their percentages from site 2 at the Pacific Mine.

<u>Analyte</u>	<u>Exchangeable mg/kg</u>	<u>Carbonate mg/kg</u>	<u>Residual mg/kg</u>	<u>Total mg/kg</u>	<u>% of total Exchangeable</u>	<u>% of total Carbonate</u>	<u>% of total Residual</u>
Cd	8.8	8.13	19.1	36	24.4	22.6	53
Cu	1.6	102	400	504	0.32	20.3	79.4
Pb	126	4.4 x 10 ³	2.6 x 10 ³	7.2 x 10 ³	1.76	61.7	36.5
Zn	181	1.1x 10 ³	3.1 x 10 ³	4.4 x 10 ³	4.15	24.8	71

Site # 2 has extremely elevated levels of metals. Table 8 gives the content of metals in unpolluted soils.

Table 8. Ranges and Averages for trace metals in unpolluted soils (From Alloway, 1990 - various chapters; McLean and Bledsoe, 1992).

<u>Metal</u>	<u>Average mg/kg</u>	<u>Ranges mg/kg</u>	<u>Site 2 Totals mg/kg</u>
Cd	0.27	0.005-2.4	36
Cu	30	2-100	504
Pb	10	2-200	7152
Zn	50	10-300	4358

The Cd content at this site is mostly associated with the residual fractions (53%). Because these were not specifically extracted, and total organic matter was not assessed, the relative position of Cd in these fractions is unknown. These are, however, considered the least mobile fractions. Exchangeable Cd (8.8 mg/kg) constitutes 24.4 % of the total Cd. Cd in this fraction may become mobile if desorption (detachment of chemical from other solids) arises from ionic exchange. If the exchangeable Cd were to remobilize, it could either move vertically through the soil profile and potentially into groundwater, or it could move horizontally towards the tailings pile. Cadmium at very high concentrations is likely to form precipitates with carbonates and phosphates (Alloway, 1990). This may account for the Cd in the carbonate bound fraction (22.6%).

Copper concentrations at site # 2 were mainly associated with the residual fractions (79.4%). This analysis is consistent with Cu geochemistry. Baker (1990) states that the position of Cu in soil usually ranks as: organic > Fe/Mn oxides >>> clay minerals. The remainder Cu at this site was associated with the carbonate fraction (20.3 %). The strong affinity of Cu for carbonate minerals is well documented (Callahan, 1980; McLean and Bledsoe, 1992). While the total Cu concentration (504 mg/kg) is relatively high, it remains in the least mobile fractions. Because Cu mobilization is highly pH dependent (Callahan, 1980), an extreme drop in pH would be needed for Cu remobilization and migration through solution.

At site # 2, the Pb content was largely associated with the carbonate bound fraction (61.7 %). McLean and Bledsoe (1992) state that at pH > 6, Pb is adsorbed on clay or

forms lead carbonate. While pH was not taken at this site, it is likely above 6 because of the limestone geology of the area. The limestone serves as a source for carbonate. The remainder of Pb was associated with the residual fractions (36.5 %) which are relatively immobile. Total Pb at site # 2 was extremely high (7.1×10^3 mg/kg). Davies (1990) reported general Pb levels between 1000 - 10,000 mg/kg at various non-agricultural polluted sites. In an extreme case, a Pb concentration was reported at 30,090 mg/kg (Davies, 1990). Because of the low solubility of Pb, it is likely to accumulate if a source is present. Current activities at the mine site do not indicate a perpetual Pb source for this sampling site. Lead at this site was probably deposited during an active mining period.

Zinc concentrations at site # 2 were mostly associated with residual fractions (71 %). Zinc readily associates with Fe and Mn oxides which may account for the high percentage in the residual fraction. Zinc may hydrolyze with a pH >7.7 to form Zn(OH) which strongly sorbes to solids or it may precipitate as Zn(OH)₂ (McLean and Bledsoe, 1992). Because pH was not taken at this site, hydrolyzed Zn may not be the Zn species at this site. If the pH is high enough, it is a possibility. Zinc was also found in the carbonate fraction (24.8 %) with the exchangeable fraction being minimal. Even though Zn was mostly associated with the residual fraction, migration is possible. Zinc compounds have relatively high solubilities (McLean and Bledsoe, 1992).

All four metals exist in very high concentrations at site # 2. Currently, mine drainage does not flow in the immediate area which could either increase metal levels by deposition, or transport existing metals via chemical and physical processes. The site is not part of a tailings dump nor is it topographically situated to receive eroded tailings. Metals at this site were most likely deposited during active mining and milling. In order to gain a more complete understanding of metal behavior at this site, additional information on the organic content of the soil, pH, geochemistry of the soil, plant uptake, and other physical and chemical factors would be needed. As mentioned earlier, this may not be a

representative soil sample of the mine area. More samples in replicate would be needed to better assess metal levels in soils at the Pacific Mine site.

Site 3

Site # 3 is located at the end of the diversion channel from the mine portal. Metal concentrations in drainage water and bottom sediments at this site may indicate levels of metals remaining in solution through the diversion, the quality of water entering the beaver pond, and levels of metals existing in sediments above the beaver pond. Table 9 summarizes the data obtained from the water and sediment samples taken at site # 3.

Cadmium in solution at this location (10.9 $\mu\text{g/L}$) is similar to the level exiting the mine portal (12.2 $\mu\text{g/L}$). Cadmium seems to remain in solution through the diversion, which suggests that the diversion does not serve as a filter for removal of Cd. Cadmium in sediments is largely associated with the exchangeable phase ($\approx 93\%$). Because the exchangeable fraction is considered most likely to remobilize, fluctuations of ions could increase the soluble Cd at this site. This site indicates a consistent Cd source for the beaver pond.

Table 9. Partitioned metal concentrations and their percentages from site 3 at the Pacific mine (pH of water 6.8).

<u>Analyte</u>	<u>Water</u> <u>$\mu\text{g/L}$</u>	<u>Exchangeable</u> <u>mg/kg</u>	<u>Carbonate</u> <u>mg/kg</u>	<u>Residual</u> <u>mg/kg</u>	<u>Total</u> <u>mg/kg</u>	<u>% of total</u> <u>Exchangeable</u>	<u>% of total</u> <u>Carbonate</u>	<u>% of total</u> <u>Residual</u>
Cd	10.9	5.6*	1.2*	-	6	93	20	-
Cu	9.4	0.8	0.6	38.6	40	2	1.5	96.5
Pb	< 160	3.2	112	296	411	0.78	27.2	72
Zn	1×10^3	110	424	6.0×10^3	1.1×10^3	9.8	37.5	52.7

* Experimental error caused the concentrations from the extractions to exceed the total concentration. The emphasis for this procedure is placed in the fraction the metal exists in, not the exact value.

The soluble Cu (9.4 $\mu\text{g/L}$) at site # 3 is higher than the concentration of 5.3 $\mu\text{g/L}$ at the mine portal. The diversion channel is not filtering Cu out. Instead, it seems to contribute slightly. Additional analysis would be needed to confirm an increase of Cu from

the diversion channel. While an increase is shown, the concentration at site # 3 remains in the general unpolluted range for fresh waters of 1 - 10 $\mu\text{g/L}$ (EPA, 1984a.). Copper in sediment at site # 3 is mostly associated with the relatively immobile residual fractions (96.5 %). The total concentration of Cu in sediments (40 mg/kg) is not considerably higher than average soil concentrations (30 mg/kg) (This comparison is made because average sediment concentrations were unobtainable). Water and sediment entering the beaver pond from this site are not contributing excessive levels of Cu.

Dissolved Pb was not detected at levels less than 160 $\mu\text{g/L}$. The Pb concentration in sediment at site # 3 was mainly attributed to the residual fractions (72 %), with the remainder in the carbonate bound fraction (27.2 %). The pH of the water at the site measured 6.8. At a pH > 6, Pb is adsorbed to clay particles and forms lead carbonate (McLean and Bledsoe, 1992) which serves as a solubility control (Callahan, 1980). Because Pb is not likely to dissolve under the current conditions, Pb contribution to the beaver pond from the diversion channel may result from physical rather than chemical processes.

Zinc in solution at site # 3 remains fairly high. The mine discharge level (1.9×10^3 $\mu\text{g/L}$) is comparable to the water concentration at site # 3 (1×10^3 $\mu\text{g/L}$). The diversion channel is not retaining all of the Zn exiting the portal. Complexes of Zn with common ligands (ions or neutral molecules attached to a metal ion) are usually soluble in neutral and acidic conditions (Callahan, 1980) which might explain the persistence of the dissolved Zn concentration. In sediments, the majority of Zn is in the residual fraction (52.7 %) followed by carbonate bound (37.5 %) and exchangeable (9.8 %). Zinc concentrations were more evenly distributed among the various fractions than the other analytes. This may support the notion that Zn is generally more mobile than Pb and Cu, and less mobile than Cd (McLean and Bledsoe, 1992). Because of high concentrations of soluble Zn and potential contributions from the exchangeable fraction, site # 3 is a substantial source of Zn for the beaver pond.

Dissolved concentrations at site # 3 indicate that water from the mine portal, flowing through the diversion channel, is contributing metals to the beaver pond. The main contributions are zinc, Cd, and some Cu. The diversion channel does not seem to act as a sink for metals exiting the mine portal.

Site 4

This site is considered the background sample site for water and sediment in the North Fork of the American Fork. Compared to the other sampling sites for this project, this site has the lowest total metals. Because it is in the drainage of mine sites up stream, it should not be considered in its natural condition. The pH at this site is higher than the pH's of other sites.

Site # 4 is located approximately twenty feet above the Pacific mine. The values in Table 10 will be used for comparison with values for site # 7. Site # 7 is adjacent to the tailings heap and below the confluence of the beaver dam and the North Fork.

Table 10. Partitioned metal concentrations and their percentages from site 4 at the Pacific mine (pH of water 8.3).

<u>Analyte</u>	<u>Water µg/L</u>	<u>Exchangeable mg/kg</u>	<u>Carbonate mg/kg</u>	<u>Residual mg/kg</u>	<u>Total mg/kg</u>	<u>% of total Exchangeable</u>	<u>% of total Carbonate</u>	<u>% of total Residual</u>
Cd	2.5	0.8*	1.6*	-	2	40	80	-
Cu	3.8	< 1	1	17	18	< 1	5.6	94.4
Pb	< 160	< 1	24	66	90	< 1	26.7	73.3
Zn	80	1.6	102	302	405	0.4	25.1	74.5

* Experimental error caused the concentrations from the extractions to exceed the total concentration. The emphasis for this procedure is placed in the fraction the metal exists in, not the exact value.

Site 5

Site # 5 is located at the end of the water channel running over the tailings. This site indicates metal levels in tailings at the Pacific Mine site as detected in the water and tailings samples that were analyzed. Because water is running over the sediment, some of

the metals in sediment may have been transported by water. This site is significant because it is topographically just above the North Fork. Metals from this site may either be carried into the river by chemical dissolution, or eroded material. Table 11 summarizes the data obtained from site # 5.

Table 11. Partitioned metal concentrations and their percentages from site 5 at the Pacific mine (pH of water 7.8).

<u>Analyte</u>	<u>Water µg/L</u>	<u>Exchangeable mg/kg</u>	<u>Carbonate mg/kg</u>	<u>Residual mg/kg</u>	<u>Total mg/kg</u>	<u>% of total Exchangeable</u>	<u>% of total Carbonate</u>	<u>% of total Residual</u>
Cd	13.6	4	2	22	28	14.29	7.14	78.6
Cu	3.02	1.6	20.2	132	154	1.04	13.1	85.8
Pb	< 160	518	5x 10 ³	1.8x10 ³	7.3x10 ³	7.1	68.6	24.3
Zn	1220	114	310	3.3x10 ³	3.7x10 ³	3.05	8.27	88.7

Dissolved Cd at site # 5 (13.6 µg/L) is similar to the concentrations in drainage from the mine portal (12.2 µg/L) and in water at the end of the diversion channel (10.9 µg/L). The exchangeable Cd in the tailings (4 mg/kg, 14 %) is low relative to the total concentration (28 mg/kg). This exchangeable fraction is the most likely to remobilize. Because of the low exchangeable Cd in the tailings, the dissolved Cd at this site may not be associated with the tailings, but rather with water emanating from the portal.

The association of Cd with the residual fractions in the tailings (78.6 %) differs considerably from its association in soil fractions. In soil, Cd is usually found in high percentages in the exchangeable phase (McLean and Bledsoe, 1992). A complete analysis of the tailings would be needed to determine what mechanism are immobilizing Cd at this site. Because Cd is largely associated with the residual fraction and its mobility may be impeded, a large percentage of total Cd concentration is less likely to enter the North Fork through chemical processes.

The dissolved Cu concentration from this sampling point (3.02 µg/L) is similar to that in the portal discharge (5.3 µg/L). The soluble Cu concentration at this site may have been transported from the portal. The largest percentage of Cu in tailings is associated with

the residual fractions (85.8 %) indicating a decreased mobility. Data from this site suggests that a large percentage of Cu from the tailings may not enter the North Fork unless erosion occurs.

Dissolved Pb at site # 5 was below 160 $\mu\text{g/L}$. The water quality study conducted by Lidstone and Anderson Inc. (1992) reported a dissolved Pb concentration of 130 $\mu\text{g/L}$ at a site near sample site # 5. The dissolved Pb they reported from the portal is 15 $\mu\text{g/L}$. Further analysis is required to determine the dissolved Pb concentration at this site.

The total Pb concentration in tailings (7.3×10^3 mg/kg) at this site is the highest of all sampling sites. A large percentage (68.6) of the total Pb is in the carbonate bound fraction. Lead carbonate is considered one of the most bioavailable and toxic Pb species to human health from a mine site. The EPA considers 500 mg/kg total metal concentration a safe threshold for Pb in soil (Nelson, 1984). The total Pb content at this site is considerably above the EPA threshold and exists in a form that is harmful to human health. The tailings at the Pacific Mine may be the most prevalent source for Pb at the mine site. Because portal discharge did not contain great quantities of Pb, water channels will not contribute considerable levels of Pb to the beaver pond or the North Fork. Lead entering the North Fork will most likely be from erosion of tailings.

Dissolved Zn at site # 5 (1×10^3 $\mu\text{g/L}$) is comparable to the concentrations at the mine portal (1.9×10^3 $\mu\text{g/L}$) and the end of the diversion channel (1.2×10^3 $\mu\text{g/L}$). The percentage of Zn from the tailings in the exchange fraction (3 %) is low relative to the residual percentage (88.7%). The exchangeable fraction is the most likely to remobilize. Because of the low exchangeable Zn, Zn in solution is probably from the portal and not the tailings.

The total level of metals analyzed for site # 5 indicates high concentrations of metals in the tailings. Lead at this site may pose a threat to human health if ingested by children. Even though most of the metals exist in the less mobile fractions, they may still enter the North Fork by erosion of tailings. Mine portal water entering the North Fork just below this sample site is also carrying significant concentrations of metals.

Site 6

The beaver pond at the Pacific Mine serves as a settling and collection pond for metals from the mine portal flowing through the diversion. The data from the sample taken at this site (Table 12) indicate that some metals are accumulating in pond solids.

Table 12. Partitioned metal concentrations and their percentages from site 6 at the Pacific Mine (pH of water 8.3).

Analyte	Water µg/L	Exchangeable mg/kg	Carbonate mg/kg	Residual mg/kg	Total mg/kg	% of total Exchangeable	% of total Carbonate	% of total Residual
Cd	10.5	56.8	98.6	24.6	180	31.6	54.8	13.7
Cu	7.2	4	35.9	1.6x10 ³	1.7x10 ³	0.24	2.16	97.6
Pb	< 160	1.6	791.8	2.6x10 ³	3.4x10 ³	< 1	23	77
Zn	1110	305.6	6x10 ³	1.9x10 ⁴	2.5x10 ⁴	1.2	25.3	73.5

The dissolved Cd concentration in the pond (10.5 µg/L) is very similar to the concentration at site # 3 (10.9 µg/L) just before water enters the pond. While this comparison suggests that Cd is not being assimilated into sediments, the total sediment concentration of Cd (180 mg/kg) indicates an accumulation in pond sediments. The sample from the pond was at a location near the tailings. Through erosion, tailings may contribute to the Cd content at this site. A difference in the Cd from the tailings at site # 5 and in the pond is the fractions Cd is associated with. In the beaver pond, 54.8 % of the total Cd was associated with the carbonate fraction and 13.7 % with the residual fraction. At site # 5, only 7.14 % was carbonate bound with the majority in the residual fraction. If the tailings are the source of the elevated levels of Cd in the pond, Cd is shifting its position in sediments. Further chemical analysis would be needed to determine the source of Cd at this site. Callahan (1980) states Cd found in association with carbonate minerals are less likely to remobilize than Cd adsorbed on clay or organic material. The majority of Cd at this site is relatively stable, with 31 % more likely to remobilize by exchange.

The dissolved Cu in the beaver pond (7.2 µg/L) is consistent with the concentration entering through the diversion channel (9.4 µg/L). While the dissolved concentration

remains similar, there is an increase in sediments in the beaver pond (total 1.7×10^3 mg/kg) compared with sites three (40 mg/kg) and five (154 mg/kg). This increase might support the idea of tailings eroding into the pond. The Cu in the residual fraction represents 97.6%. The Cu at this site under current conditions is unlikely to remobilize.

The total Pb concentration in sediments of the beaver pond (3.4×10^3 mg/kg) is less than the total at site # 5 (7.3×10^3 mg/kg). This contradicts the consistent increases in total sediment concentrations in the pond seen in the other metals. This difference may have occurred because Pb may not be introduced through solution at substantial levels from the diversion channel (Pb is generally sorbed on bed sediments close to the source (Callahan, 1980)), or that the other metals (usually considered more mobile, especially Cd and Zn) were transported into the pond from the tailings through solution and assimilated back into sediments. Further sampling and analysis would be needed to determine what factors are contributing to the differences in concentrations from site # 5 to # 6.

The dissolved Zn concentration (1.1×10^3 $\mu\text{g/L}$) is relatively close to the concentration entering from the diversion channel (1.2×10^3 $\mu\text{g/L}$). Because Zn remains in solution in the beaver pond, it creates a problem for the receiving aquatic system, the North Fork. Mangum (1988) states that Zn appeared to be one of the most potentially damaging heavy metals from mining effluents along the North Fork. The beaver pond effluent, even though diluted at the confluence, seems to have an impact on benthic (bottom dwelling) taxa. Mangum (1988) reported a 61% reduction in macroinvertebrate biomass and an 81% reduction in the numbers of organisms between a sampling site above the Pacific Mine and a sampling station just below the mine.

The total Zn concentration in sediments (2.5×10^4 mg/kg) indicates that Zn constitutes 2.5 % of the total sediment sample. While the concentration of Zn is relatively higher than the other metals, the majority is in least mobile fractions (73.5 % residual, and 25.3 % carbonate bound). The pH of the water at this site is 8.3. As stated earlier, at a pH greater than 7.7 Zn hydrolyzes and strongly sorbs to soils (McLean and Bledsoe, 1992;

Callahan, 1980). While this fraction may remain immobile, once entering the North Fork, it is available for sediment feeding macroinvertebrates.

The sediment sample taken from the pond shows an increased level of metals compared with all other sample sites (except Pb at site # 5). This suggests that the beaver pond is accumulating metals. However, the concentrations in solution are not significantly lower than the concentrations from the portal. Because of this, water exiting the pond may be as high in metal concentration as the portal or metals could be released from bottom sediments and organics. The pond may not be serving as an adequate filter. This may be due to saturation of sediments with metals, or other chemical factors that are not allowing Cd, Zn, and Cu to settle or be adsorbed. Because the beaver pond discharges into the North Fork, it introduces metals through solution and sediment. Further chemical analysis of the beaver pond would be needed in order to understand differences in concentrations from tailings.

Site 7

Sample site # 7 is located in the North Fork just below the tailings heap and is the approximate confluence for water coming off of the tailings. This site is also below the confluence of the beaver pond and the North Fork. The source for metals at this site is mainly from the tailings and associated water and a dilution of water and sediments from the beaver pond. Table 13 summarizes the data obtained from sample site # 7.

Table 13. Partitioned metal concentrations and their percentages from site 7 at the Pacific mine (pH of water 8.7).

<u>Analyte</u>	<u>Water µg/L</u>	<u>Exchangeable mg/kg</u>	<u>Carbonate mg/kg</u>	<u>Residual mg/kg</u>	<u>Total mg/kg</u>	<u>% of total Exchangeable</u>	<u>% of total Carbonate</u>	<u>% of total Residual</u>
Cd	0.5	1.6	1.5	23.9	27	5.4	5.6	88.5
Cu	< .01	0.8	18.1	98.1	117	0.68	15.5	83.9
Pb	< 160	258	3.6x10 ³	763	4.6x10 ³	5.6	77.7	16.7
Zn	10	79.2	420.5	3.6x10 ³	4.1x10 ³	2	10	88

Comparing metal levels at sites # 4, 5, 6, and 7 (Table 14) indicates that metals in the North Fork at site # 7 can be attributed to tailings erosion and sediment migration from the beaver pond. Site # 4 would not contribute high enough levels of metals to explain the concentrations at site # 7. The Pb concentration in sediments at site # 7 is considerably lower than the tailings (7.3×10^3 mg/kg vs. 4.6×10^3 mg/kg). Because the Pb concentration at site # 7 is lower than the tailings concentration, and Pb is readily retained by sediment (Callahan, 1980), tailings concentrations seem to be diluted at site # 7.

Table 14. Total sediment concentrations at sites five, six and seven.

<u>Site</u>	<u>Total Cd mg/kg</u>	<u>Total Cu mg/kg</u>	<u>Total Pb mg/kg</u>	<u>Total Zn mg/kg</u>
4 (upstream)	2	18	90	405
5 (tailings)	28	154	7.3×10^3	3.7×10^3
6 (pond)	180	1.7×10^3	3.4×10^3	2.6×10^4
7 (river)	27	117	4.6×10^3	4.1×10^3

The Zn, Cd, and Cu concentrations at site # 7 are similar to the tailings. This indicates that another source for Zn, Cd, and Cu must exist if tailings are diluted at this site. Concentrations of Zn, Cd, and Cu, are all higher in the pond than in tailings, while Pb is lower in the beaver pond. Sediment from the beaver pond would contribute proportionately higher levels of Zn, Cd, and Cu to the North Fork. This may explain why Zn, Cd, and Cu concentrations persist from site # 5 to site # 7, while total Pb decreases at site # 7. The exceedingly high Zn concentration in the beaver pond may account for the increase in Zn at site # 7 over site # 5. Figure 6 illustrates the similarities of total metal levels at sites five and seven, and the difference in the beaver pond.

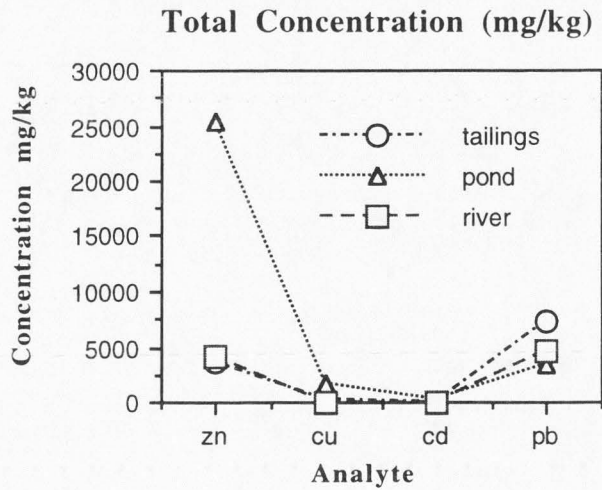


Figure 6. Total sediment concentrations at sites 5, 6, and 7.

The tailings seem to account for more of the metals at site # 7 than the beaver pond. While total concentrations suggest this, this assumption is also based on the fractions in which the metals exist at the three sites (# 5, 6, and 7). Graphs of the percentages of concentrations of each analyte at the 3 sites show that the tailings and the river sediments have stronger similarities (Figures 7 and 8).

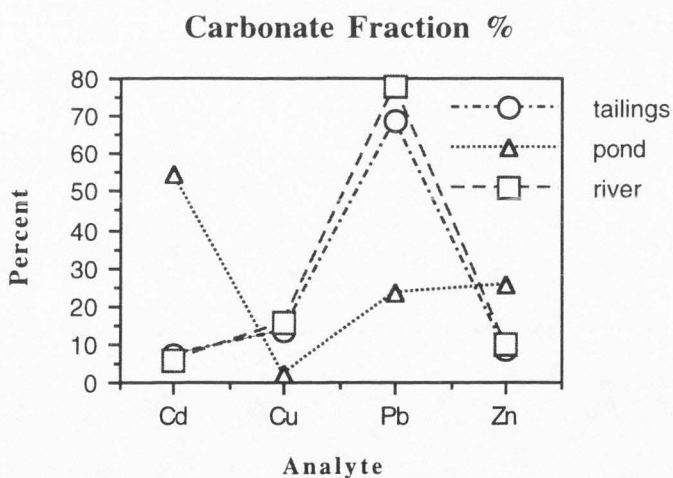


Figure 7. Carbonate bound sediment percentages at sites 5, 6, and 7.

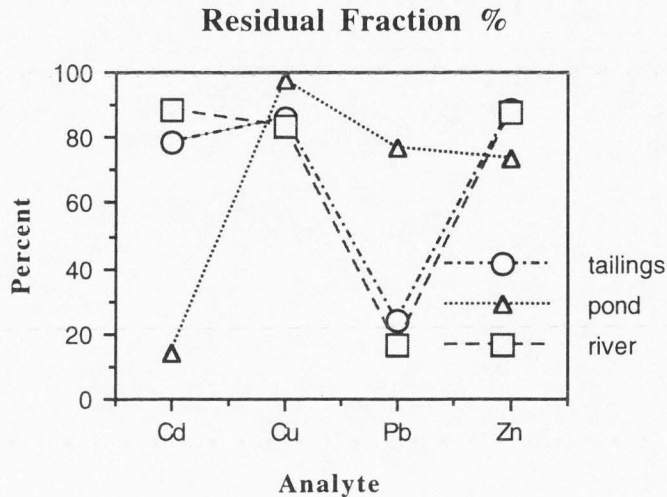


Figure 8. Residual fraction percentages at sites 5, 6, and 7.

The dissolved Cd in the river at site # 7 ($0.5 \mu\text{g/L}$) is lower than the effluent coming from the tailings ($13.6 \mu\text{g/L}$). The dilution by unpolluted streamflow occurring at this site is adequately reducing the Cd concentration in the North Fork according to water quality criteria. The National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses for Cd is $2.0 \mu\text{g/L}$ (at hardness 200 mg/L as CaCO_3 - four-day average - see appendix for hardness factors and exposure) (EPA, 1984c). The exchangeable fraction accounts for 5.93 % of the Cd in sediment. An alteration of ionic composition could potentially remobilize some Cd, but the majority of Cd is considered relatively immobile because it mainly exists in the residual fraction (27 mg/kg - 88.5 %).

Dissolved Cu in the water sample taken at this site was below detection limits. The Cu in solution from the tailings pile runoff ($3.02 \mu\text{g/L}$) and the beaver pond ($7.2 \mu\text{g/L}$) must be diluted by upstream discharge prior to reaching this site. The national criteria for Cu concentration at a hardness of 200 mg/L as CaCO_3 is $21 \mu\text{g/L}$ (EPA, 1984a). Dissolved Cu may not be an environmental concern at this site.

The Cu in sediment at site # 7 (117 mg/kg) is mostly in the residual fraction (83.85 %) with the remainder in the carbonate fraction (15.6 %). While soluble Cu does not seem to be an environmental concern, it may be impacting the mortality of benthic invertebrates feeding off sediments. Luoma (1988) states that deposit and suspension eating organisms are exposed to all metal compounds in sediments. Because Cu and Zn (Zn 10 µg/L at this site) are synergistic, low concentrations of either (in combination) may compound negative effects on benthic organisms (Mangum, 1994). Further investigation would be needed to confirm the effects of sediment bound Cu on benthic invertebrates at this site.

Soluble Pb was less than 160 µg/L at site # 7. In order to determine if dissolved Pb is higher than the criteria for aquatic organism protection (7.7 µg/L at a hardness of 200 mg/L CaCO₃) (EPA, 1994b), additional samples should be analyzed with a more sensitive instrument. Sediment bound Pb may have a negative impact on the aquatic system through ingestion of particles by deposit feeders. This has not been confirmed at this site and would require further study.

The soluble Zn concentration (10 µg/L) is well below the EPA (1987) concentration for Zn (190 µg/L at a hardness of 200 mg/L as CaCO₃) at site # 7. As stated earlier, Zn and Cu are synergistic and only a low concentration of either in combination with the other will amplify negative effects to aquatic life (Mangum, 1994). Zinc bound to sediments at this site and other North Fork locations adjacent to the mine site may also impact aquatic life. The total concentration of sediment Zn is 4.1×10^3 mg/Kg. The largest percentage (87.7 %) is in the residual fraction. Because deposit feeders are not able to discriminate against metal latent particles (Luoma, 1988), Zn in sediment may adversely affect the benthic population at this site.

Site 8

Site # 8 was included to determine if metals are migrating down the North Fork away from the Pacific Mine. Comparing the total metal levels in sediment at sites # 4 (up stream) and # 7 (below tailings) with site # 8 (down stream), indicates that metals from the mine are migrating in the stream. Concentrations at site # 8 are well above the concentrations at four, but are lower than seven, Figure 9.

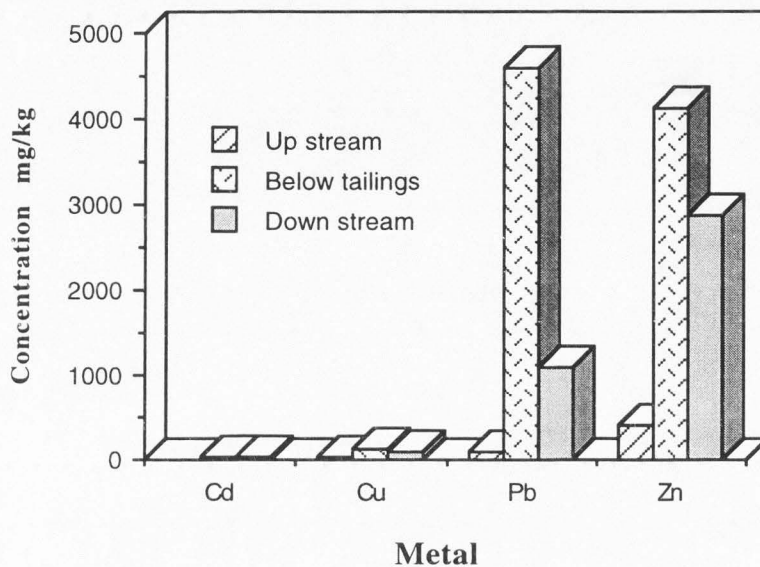


Figure 9. Total sediment concentrations at sites 4, 7, and 8.

The site upstream (# 4) had low levels of metals in bottom sediment. Thus, the levels of metals at site # 8 must be attributed to the mine, however, they are substantially attenuated compared to levels at site # 7 just 30 feet upstream (by the tailings). A summary of metal concentrations at site # 8 is provided in Table 15.

Metals in sediments at site # 8 are between 4 times lower for Pb and approximately 1.3 times lower for Cu, Cd, and Zn higher than at site # 7. The discrepancy may be from the fraction in which the metals are found. At site # 7, Cd, Cu, and Zn were mainly

associated with the residual fractions. Almost 78 % of the total Pb at site # 7 was in the carbonate bound fraction. The associations in fractions may account for the low mobility of Pb over the other metals.

Table 15. Partitioned metal concentrations and their percentages from site 8 at the Pacific mine (pH of water 8.7).

<u>Analyte</u>	<u>Water µg/L</u>	<u>Exchangeable mg/kg</u>	<u>Carbonate mg/kg</u>	<u>Residual mg/kg</u>	<u>Total mg/kg</u>	<u>% of total Exchangeable</u>	<u>% of total Carbonate</u>	<u>% of total Residual</u>
Cd	1.8	8.8*	11.7*	-	19	46	62	-
Cu	1.18	1.6	7.3	81.1	90	2	8	90
Pb	< 160	4	423.5	662.5	1.1x10 ³	0.37	38.9	60.8
Zn	< DL	67.2	1.2x10 ³	1.6x10 ³	2.9x10 ³	2.34	41	56.7

* Experimental error caused the concentrations from the extractions to exceed the total concentration. The emphasis for this procedure is placed in the fraction the metal exists in, not only the total value.

The levels of metals, in particular Zn and Pb are high at site # 8 and may negatively impact aquatic life. Further investigation of invertebrate and vertebrate communities would be needed to determine the effects of sediments at this site.

Dissolved metals at site # 8 contradict the increase in sediment metals from four to eight. Instead of an increase over site # 4, there are lower levels of dissolved metals at site # 8. Figure 10 illustrates the decrease in dissolved metals. The reduction in dissolved metals may be caused by adsorption of metals with sediments, precipitating onto limestone, or the water sample was not a good representative of the site. Additional sampling and analysis would be needed to determine if the concentrations vary consistently. In any case, the levels of metals in the water sample from this site are under the water quality criteria for aquatic protection.

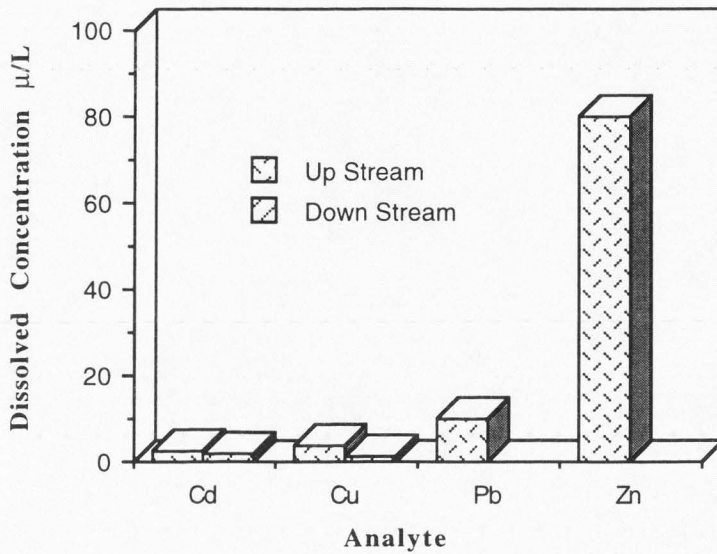


Figure 10. Dissolved metal concentrations at sites 4 (up stream) and 8 (down stream).

Conclusions - Environmental Quality Impacts to the North Fork of the American Fork

Mine portal water contains Zn at a higher concentration (1870 µg/L) than the water quality criteria (40 C.F.R. 440 (J) - BPT) for mine drainage (1500 µg/L). The other metals are below this criteria. EPA criteria suggest a remedy for this excess in concentration, but the mine drainage is not under a discharge permit. Water from the portal is flowing over tailings and into the North Fork, and through a diversion channel into a pond to the North Fork. Dissolved metals do not seem to attenuate enroute to the North Fork.

Site # 2 (soil) and # 5 (tailings) present the highest risk to human health through solids. Both of these sites contain Pb in the carbonate fraction well above EPA's safety level (500 mg/kg). Erosion of tailings from site # 5 seems to contribute metals to the beaver pond and the North Fork. Metals in sediment may be responsible for decreased numbers in benthic life and diversity.

The sample taken at site # 7 reveals that metals from the Pacific Mine site are being introduced through solution from the mine portal and by sediment via the beaver pond and

tailings. While the dissolved metal levels were not above EPA criteria (Table 16) at this site (Pb needs further analysis), metals may be affecting aquatic life. It should be noted that the EPA criteria are derived from many studies on various invertebrate and vertebrate species. Individual species in this section of the North Fork may have lower or higher tolerances for dissolved metals. Isom (1992) states that EPA criteria is limited in that it is based on laboratory studies. It is suggested that criteria should be site specific for protection of the aquatic environment (Isom, 1992).

Table 16. Summary table of water criteria and water quality at site # 7.

<u>Analyte</u>	<u>Concentration site 7 $\mu\text{g/L}$</u>	<u>Domestic source $\mu\text{g/L}$*</u>	<u>Agriculture $\mu\text{g/L}$*</u>	<u>Aquatic protection/uses $\mu\text{g/L}$[^]</u>
Cd	0.5	10	10	2
Cu	< DL	-	200	21
Pb	< 160	50	100	7.7
Zn	10	-	-	190

* Source - Standards of Quality For Waters Of The State R317-2, Utah Administrative Code, Effective Date of Last Revision - February 16, 1994.

[^] EPA (1984a, b, c, and 1987). and Standards of Quality For Waters Of The State R317-2, Utah Administrative Code, Effective Date of Last Revision - February 16, 1994. Assuming a hardness factor of 200 mg/L CaCO₃ four day average (see appendix for calculation).

While dissolved metal concentrations are low and would not seem to affect aquatic life, sediments add to the metal load of the stream and may negatively impact aquatic life. A site just below sample site # 7 reported a 61% reduction in macroinvertebrate biomass and an 81% reduction in the numbers of organisms at a sampling station just below the Pacific Mine. Numerous studies have indicated that benthic macroinvertebrate communities reflect the health of an aquatic system (Clements, 1994; Woodward et al., 1994; Lynch et al., 1988; Kiffney et al., 1993). The reduction in macroinvertebrates suggests that aquatic life in the North Fork was under stress during the time of the study (Mangum, 1988). Updated information would be needed to determine if macroinvertebrate

communities are still stressed from mine wastes. Additional information to be included in future studies would be the impact of metal sediments on benthic organisms and fish.

Dallinger and Kautzky (1985) have stated that water quality just below legislative limits may still impact aquatic communities. Additional pollution in sediments and invertebrates may present significant contamination risk to fish (Dallinger and Kautzky, 1985).

The Pacific Mine is only one site that contributes metals through solution to the North Fork of the American fork. In order to gain a comprehensive understanding of the water quality of the North Fork, other abandoned mine sites in the watershed would require investigation.

While water quality criteria is the main measure for determining stream quality, sediments play a key role. Recently in Ontario, sediment quality guidelines were issued by the Ontario Ministry of the Environment and Energy. The guidelines were developed to protect aquatic communities including benthos (Prarie and McKee, 1994). While no such criteria exists at the national level in the United States, it could be developed to increase protection of aquatic communities. By using sequential extraction procedures, the fraction in which a metal exists can be determined. This type of information can be used to determine possible bioavailability from desorption of metals on exchange sites or if metals are in less mobile fractions that are likely to remain near the source and effect local populations. In this study, sequential extractions provided data for determining which sampling sites were contributing metals in various fractions to the North Fork of the American Fork. This information can be used to determine what type of remediation techniques might be appropriate for the site.

REMEDIATION TECHNIQUES

Abandoned and inactive mines are responsible for the pollution of approximately 12,000 miles of water ways, and 180,000 acres of polluted lakes and reservoirs in the United States (Lyon et al., 1993). Currently there is not a Federal mechanism to insure the reduction of pollution from non-active mine sites, or the funds available for a reclamation program. Because of this, reclamation of abandoned and inactive mine sites is minimal. While funds are not available to reclaim many mine sites, the technology does exist.

This chapter reviews methods of prevention and control of pollution that could be implemented at hardrock mining sites. Because tailings and mine drainage exist at the Pacific Mine, remediation for these sources of pollution will be the focus. Although acid mine drainage does not seem to be a threat at the Pacific Mine, it will be discussed because it is a highly publicized problem at abandoned mine sites nation wide.

Acid Mine Drainage

Acid mine drainage (AMD) is an environmental problem associated with mining activities that has received a considerable amount of attention. AMD is the production of sulfuric acid from sulfide ores (see appendix for discussion). This type of drainage is capable of leaching residual metals from tailings material (Dave et al., 1985) and overburden (UNEP, 1991). Through AMD, metals may be transported through solution to groundwater, surface water, and soils.

Many methods have been developed to eliminate AMD. Some of the methods are preventive and others are response oriented. AMD requires three ingredients: sulfide ore, water, and an oxidizing agent (usually oxygen). In 1928, an observation was made that caved-in mines were producing less AMD than other mines (Kim et al., 1982). The assumption was that a reduction in air and water entering mine shafts was limiting the rate and formation of AMD. Through this discovery, mine sealing has become a standard for

reducing acid conditions (Kim et al., 1992; UC Berkeley, 1988). Another well established method used to eliminate acid production is by flooding shafts to eliminate air which decreases oxidation and acid formation (Kim et al., 1992).

Acid formation is enhanced by bacteria called *Thiobacillus ferrooxidans*. By the use of bactericidal materials, *T. ferrooxidans* are not able to promote acid generation. It has been shown that the use of anionic surfactants (cleaning detergents) reduces the activity of *T. ferrooxidans* which decreases acid production (Erickson et al., 1985).

The most common method of treating AMD is acid neutralization and chemical precipitation. In this process, an alkaline reagent such as hydrated lime (Bosman, 1983), quick lime, limestone and other alkaline materials (UC Berkeley, 1988) are added to the acidic solution. The increased alkalinity reduces the acidity and raises the pH. As the pH is raised, metals precipitate from the solution becoming less mobile (Hustwik and Sykes, 1994).

Because of the host rock at the Pacific Mine, limestone, AMD is not a process that increases the metal load from the mine portal. The metals exist in solution under non acid conditions. While AMD does not exist, metals are exiting the portal at levels that might adversely affect the aquatic environment of the North Fork.

Constructed Wetlands

Constructed wetlands are based on biological and geochemical processes that accumulate and remove metals from influent water (Bates and Brown, 1993). The general processes that act to immobilize metals in wetlands are the same processes that determine that mobilize metals outside of wetlands. The main mechanisms responsible for the removal of metals in wetlands are : 1) adsorption and complexation of metals by organic substrates, 2) microbial sulfate reduction followed by precipitation of metals as sulfides, 3) precipitation of ferric and manganese oxides, 4) adsorption of metals by ferric

hydroxides, 5) metal uptake by plants, and 6) filtration of suspended and colloidal materials (Cohen and Staub, 1992).

Various studies have been conducted to determine the success of wetlands for metal removal. UC Berkeley (1988) compiled information suggesting that marsh type wetlands using *Typha* (cattails), *Sphagnum* (moss), and *Equisetum* (horsetails) are efficient at reducing metal levels in mine discharge. They suggest that wetlands using vegetation should not be deeper than four inches for metal retention. This is based on efficiency of cattails and mortality of wetland vegetation at depths greater than two feet. While vegetation may initially reduce dissolved metal, vegetation may not serve as a perpetual means for metal removal from a system. Once a plant reaches its equilibrium metal concentration, it no longer takes up metals.

Egar et al.(1994), reported 50 - 90 % of metal removal at a constructed wetland in Minnesota. Although metal concentrations increased in vegetation (cattail, grass, and sedge), vegetation was not responsible for the majority of metal removal. Metal removal was attributed to the peat substrate (over 99%). Peat is able to accumulate metals because it contains a complex mixture of organic compounds with various functional groups that are able to bind metals (Egar et al., 1994).

Cohen and Staub (1992) provide further evidence that plants account for as little as 1-5 % of metal accumulation in wetlands. The claim is that an anaerobic system with plants and other attenuating mechanisms eventually becomes saturated so metals remain in solution. Cohen and Staub (1992) state that although plants may not accumulate metals at high rates, they do promote metal retention by generating microzones of oxidizing and reducing conditions which may aid in metal precipitation. This process may only account for 2 % of the metal removal in a system (Cohen and Staub, 1992). Vegetation also contributes to the biomass or peat substrate which provides binding sites for metals.

The constructed wetland design recommended by Cohen and Staub (1992) uses sulfate reducing bacteria (SRB). SRB are anaerobes that decompose organic compounds

and use sulfate as the electron acceptor in a resultant redox reaction (Cohen and Staub, 1992). The result is the production of sulfide which reacts with metals to form metal sulfide precipitates. The organic matter used by Cohen and Staub was cow manure and hay. Using this substrate, 98-100 % of the metals were removed. This type of wetland system also differs from the traditional use of wetland plants in that it is dependent on anaerobic conditions. Because of this, a tall and narrow tower design is recommended over a large horizontal area with greater surface area for exposure to atmospheric oxygen (Cohen and Staub, 1992).

In a report by Lidstone and Anderson (1992), it was recommended that a wetland system should be constructed for treatment of portal water at the Pacific Mine. The suggestion was to use organic matter (humus, manure, and soils borrowed from the beaver pond area). This was probably recommended as a means to provide organic binding sites for metals. This recommendation may not suffice because of current conditions at the Pacific Mine site. The sediment and water samples taken at site # 6, the beaver pond, indicate that high levels of metals already exist in the organic matter in the pond. The metal concentrations in the water sample did not indicate metal reduction from the mine portal to the pond. It seems that the immobilization potential of organic material in the beaver pond is not adequate for reducing dissolved metals. The pH at this site (8.3) should not be a major factor in maintaining dissolved metal levels.

In order for a wetland system to maintain continued metal removal, maintenance would be required. This would be needed for a wetland based on metal reduction by peat, or precipitation of metal sulfides. Either method would require a perpetual source of organic matter. Because of the location of the Pacific Mine, a wetland system may be more of an expense than a device to deter metals from entering the North Fork. The dissolved levels of metals in the beaver pond are persisting. This may eventually occur in a constructed wetland if it is not maintained properly. In addition to adding organic matter with sites for binding metal, removal of saturated organic matter may be necessary.

Another solution to metal concentrations from the mine portal may be to divert water from the North Fork onsite for dilution of metals from the portal. Instead of sending the water into a holding pond, the diluted water would be sent directly back to the stream. While metals would not be attenuated on site by this method, the current ponding approach does not seem to be effective and a constructed wetland may not suffice after its retention capacity is used. The diversion would simply dilute metal concentrations before entering the North Fork. Further studies would be needed to determine if loss of water from the diversion would adversely affect the aquatic environment of the North Fork more than dissolved metals.

Tailings

Portal water continues to flow over the tailings pile even though a diversion has been built to redirect the water. Proper drainage construction needs to be installed to insure that all portal water is diverted away from the tailings.

The tailings pile at the Pacific Mine poses at least two potential threats to life. The level of Pb in the sample taken from the tailings is well above the EPA threshold for human safety. If ingested, Pb poisoning may occur. Tailings are also eroding into the North Fork. The tailings contain high levels of Cd, Cu, Pb, and Zn. In an aquatic study, it was found that the macroinvertebrate community near the Pacific mine was stressed (Mangum, 1988). Because of the threats to human and aquatic health, some remediation of tailings is needed.

The EPA (1993) provides a reference guide of remedies for soils, sediments, and sludges. Some of the remedies may be applicable to tailings (see appendix for screening matrix for remediation technologies). The reference guide is divided into *in situ* and *ex situ* processes. EPA (1993) gives an average excavation cost of \$ 50/ton. The standard for soil of 1 acre 10 feet deep is 20,000 tons. The Pacific tailings dump is probably over 1 acre at 10 feet deep. While Pb poisoning may occur and eroded tailings may limit the

health of the North Fork, *ex situ* remediation of tailings may not be practical because exposure of Pb from tailings at this site is not known and excavation costs may be over \$ 1 million (this would not include treatment). Further information would be needed to warrant the excavation of tailings.

The tailings at this site serve as a reminder of the mining activities that occurred at the turn of the century. As mentioned previously, this site has been listed as eligible for the National Register of Historic Places. Because the tailings have historical value, they should remain on site. If this area is to be visited by historians and recreationalists, signs should be posted informing people of the potential health hazards of the tailings. Measures should be taken to keep off-road vehicles off of the tailings in order to reduce erosion potential.

The tailings directly adjacent to the North Fork should be removed and taken to a location, on site, where they do not pose an immediate threat to the stream. Structural erosion control measures such as riprap are needed to keep tailings out of the stream. The stream bank could also be stabilized by using silt fencing, and or straw bails. As a buffer from the remaining tailings, green plants may be used to decontaminate and stabilize soils and remaining tailings that are polluted with heavy metals (Berti and Cunningham, 1993). Currently, a research program at USU is working to develop plants with increased metal uptake. These plants may have potential use at the Pacific Mine.

By reducing the levels of metals entering the North Fork from the Pacific Mine, the macroinvertebrate community may recover. A study conducted by Holland et al., (1994), on the South Fork and mainstem of the Coeur d'Alene River in Idaho showed that decreased levels of heavy metals allowed the benthic community to recover gradually. Taxonomic richness and species diversity increased. By reducing the concentration of metals from the mine portal and the erosion of tailings into the North Fork, health to this section of the North Fork may be restored.

CONCLUSIONS

Hardrock mining has traditionally operated under the Mining Law of 1872 which does not mandate remediation or reclamation for damages created by mine operators. Today there are state programs and other regulations to insure that some reclamation occurs at active mine sites. To date, a Federal program to insure the clean up of inactive sites has not been passed in both the House and the Senate. Consequently, Federal funds are not available for reclamation of abandoned and inactive sites. Because many operations existed prior to any reclamation regulations, abandoned and inactive hardrock mines contribute pollution to soil and water ways around the country.

In Utah, mining districts including the American Fork contain abandoned and inactive mine sites that contribute metals to streams. The current approach to assessing abandoned mine sites does not always consider environmental affects from abandoned sites. Abandoned mines receive remediation and reclamation attention primarily if they have open shafts, steep high walls, or other physical dangers. Eliminating pollution is not the priority in abandoned mine reclamation in Utah. This study has confirmed that high levels of metals from tailings and portal water from the Pacific mine enter the North Fork of the American Fork. The Pacific Mine continues to pollute the receiving waterway.

An earlier study indicated aquatic invertebrate life was stressed from metal levels in the North Fork. Numerous studies have indicated that macroinvertebrate communities reflect the health of aquatic systems (Clements, 1994; Woodward et al., 1994; Lynch et al., 1988; Kiffney et al., 1993). In 1988, an inventory of macroinvertebrates reported a 61% reduction in macroinvertebrate biomass and an 81% reduction in the numbers of organisms at a sampling station just below the Pacific Mine (Mangum, 1988). The decline in population was primarily attributed to dissolved metals from the Pacific Mine. Because heavy metals exist in sediments, metals associated with solids from tailings may contribute to the mortality of the macroinvertebrate community. Further studies would be needed to

determine if metals from sediment or dissolved metals are responsible for the decline in the aquatic system of the North Fork near the Pacific mine.

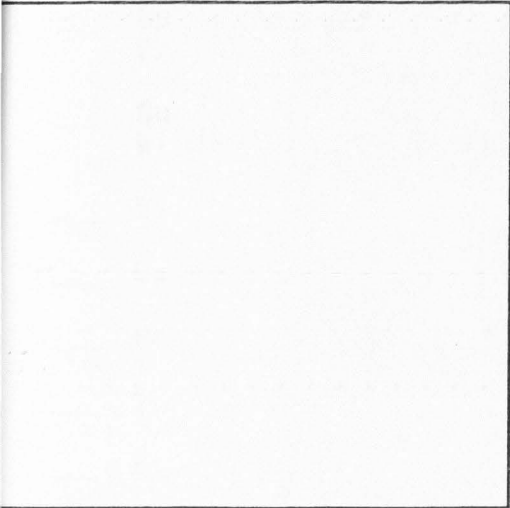
Metals in portal water and tailings may be a threat to human health. Lead in tailings exceeds EPA standards and is mainly in the carbonate bound fraction. Lead carbonate is considered a bioavailable Pb species and is harmful to human health. Dissolved Cd from portal water is higher than drinking water standards. In order to determine if metals from the Pacific mine are hazardous to human health, further information on human exposure to metals at this site would be needed.

Because of the adverse effects of metals to aquatic life and the potential threat to human health, additional reclamation at the Pacific Mine is needed. Technology to remediate abandoned mine sites does exist. The remediation needed at the Pacific does not need to be highly technical or expensive (advanced technology may not be practical because human exposure to metals at the site is unknown). Because the North Fork seems to be the only immediate environment that suffers from metals, stopping metals from entering the North Fork should be the priority in any reclamation plan for this site. A constructed wetland may serve as a sink for metals exiting the portal, but would require perpetual maintenance. Tailings adjacent to the North Fork should be removed and relocated away from the stream or retained with structural erosion control measures. Vegetation should be planted next to the stream to serve as a buffer. Other suggestions would be warning signs to stop people from playing in the beaver pond, eating tailings, and drinking mine portal water.

Reclamation is needed at abandoned and inactive mine sites. Unfortunately, funds are not always available. Until some type of legislation is passed, abandoned mine sites may not receive remediation attention. This will perpetuate pollution problems associated with abandoned and inactive mine sites.

APPENDIX A
Abandoned Mine Assessment Sheets

RIDGETOP PROJECT INVENTORY FORM



TAG NUMBER _____

QUAD NAME _____

CRIB NUMBER _____

SITE NAME _____

CLAIM NAME _____

WATERSHED _____

TOWNSHIP 1 2 3 4 SOUTH

RANGE 1 2 3 4 EAST

SECTION _____ _____ † _____ † _____ †

COUNTY _____ ELEVATION _____ FEET

ASPECT _____ NATURAL SLOPE _____ %

DESCRIPTION OF SITE CONDITIONS

ACCESS TO SITE AND OPENING

SITE	OPENING	ACCESS TYPE
_____	_____	By foot only
_____	_____	2-wheel drive
_____	_____	2-wheel access possible with substantial regrading
_____	_____	4-wheel drive access
_____	_____	ATV/Motorcycle access
_____	_____	Rubber-tired backhoe
_____	_____	Crawler (track) equipment

ENGINEER OBSERVATION OF BAT HABITAT

_____ Bats _____ None _____ Unknown

ANIMAL USAGE

_____ Yes _____ None _____ Unknown

TYPE OF MINE OPENING

_____ Adit (Horizontal) _____ Shaft

_____ Adit (Inclined) _____ Subsidence

OPENING DESCRIPTION

_____ Drainage _____ Consolidated

_____ No Drainage _____ Unconsolidated

STATE OF UTAH
DEPARTMENT OF NATURAL RESOURCES
DIVISION OF OIL, GAS AND MINING

INVENTORIED BY SPECTRUM ENGINEERING
BILLINGS, MONTANA DURING 1994
BY BILL MAEHL AND DAVE MURJA

RIDGETOP PROJECT INVENTORY FORM

RECOMMENDED CLOSURE ACTION

- Bulkhead Only, With or Without Drainage Provisions
- Bulkhead & Backfill, With or Without Drainage Provisions
- Backfill, With Drainage Provisions
- Backfill, Without Drain Provisions
- Probe for Mine Opening
- Standard Gate Closure Method
- Bat Gate Closure Method
- Locking Adit Door
- Local Site Clean-up & Regrading
- Structure Demolition
- Drainage Control
- Cultural Features _____

CONDITION OF ENTRANCE

- Completely collapsed, no access
- Partially collapsed or backfilled at portal, mine visible but not accessible
- Partially collapsed or backfilled at portal, mine workings accessible
- Portal open for access with little or no backfill or collapse
- Obstruction at adit makes evaluation of conditions impossible
- Shaft is open
- Existing wall _____, fence _____, or grate _____ exists which is:
 - intact and acceptable intact and unacceptable
 - damaged, mine accessible
- Other _____

CLOSURE METHOD

- Hand Equipment Blast
- Hand & Equipment Other

REVEGETATION AREA

- _____ Sq.Ft. (_____ Ft. x _____ Ft.)
Over the reclaimed entrance
- _____ Sq.Ft. of the dump & other areas

BACKFILL MATERIAL SOURCE (see Field Description Form)

- Mine Dump
- Other Material Source
- _____
- _____
- _____
- No Backfill Material Present on Site

- _____ Feet, Depth of placement inby from brow
- _____ Cuyd, Estimated backfill volume needed
- _____ Volume calc.

STATE OF UTAH
DEPARTMENT OF NATURAL RESOURCES
DIVISION OF OIL, GAS AND MINING

INVENTORIED BY SPECTRUM ENGINEERING

TAG NO. _____

RIDGETOP PROJECT INVENTORY FORM

FOR MINE SITES REQUIRING NO FURTHER ACTION AT THIS TIME

TAG NUMBER _____

QUAD NAME _____

TOWNSHIP 1 2 3 4 SOUTH

RANGE 1 2 3 4 EAST

SECTION _____

COUNTY SALT LAKE SUMMIT WASATCH UTAH

TYPE OF MINE OPENING

_____ Adit (Horizontal)

_____ Adit (Inclined)

_____ Shaft

_____ Exploration Pit

_____ Subsidence

CONDITION OF OPENING

_____ Completely collapsed, no access.
No closure is currently required.

_____ Access locked or secured, no action.

_____ Open but permission denied by owner.

_____ Drainage _____ Consolidated

_____ No Drainage _____ Unconsolidated

ACCESS TO SITE AND OPENING

SITE	OPENING	ACCESS TYPE
_____	_____	By foot only
_____	_____	2-wheel drive
_____	_____	2-wheel access possible with substantial regrading
_____	_____	4-wheel drive access
_____	_____	ATV/Motorcycle access
_____	_____	Rubber-tired backhoe
_____	_____	Crawler (track) equipment

MATERIAL SOURCE

_____ Small Sized Mine Dump

_____ Medium Sized Mine Dump

_____ Large Sized Mine Dump

_____ Other Material Source on Site

_____ No Backfill Material Present on Site

CULTURAL FEATURES

_____ Yes _____ No

_____ If Yes, Describe

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APPENDIX B

Lab Analysis - Quality Control Data

Table 17. Quality Control Data.

Cd	Water	MgCl	NaOAc	Total
Spike 1 % R	93	80	100	92
Spike 2 % R	118	80	100	98
RPD	15.96	0	0	4.9
Cu				
Spike 1 % R	187	100	78	61.3
Spike 2 % R	184	100	74	61.5
RPD	1.6	0	1.5	2.89
Pb				
Spike 1 % R	131	-300	44.5	49
Spike 2 % R	125	-300	38	31
RPD	4.44	0	.9	24.5
Zn				
Spike 1 % R	140	64	14	96.7
Spike 2 % R	144	60	12	95.5
RPD	1	1.4	.25	.25

A spiked sample is a field sample that has a know amount of analyte added to it. The concentrations of the unspiked and spiked samples are compared to determine if the laboratory method was adequate. Two samples are spiked in order to calculate the relative percent difference (RPD) which indicates precision. While many of the spiked samples are not near the ideal 100 % recovery, the data obtained was accepted for this study because the samples were run multiple times with consistent concentration returns.

APPENDIX C

Acid Mine Drainage

Acid Mine Drainage From Hardrock Mining

Non-ferrous metal compounds occur in associations with other metals. Some common associations are: lead - zinc, lead - zinc - copper, lead - silver, zinc - cadmium, copper - molybdenum, copper - gold, nickel - copper, and nickel - cobalt (UNEP, 1991). Base metals, commercial metals such as copper, lead or zinc which are not considered precious, are usually found as compounds of other elements in metallic ore (Table 17). Except for nickel, base metals usually occur as sulfides, or oxidized ore (oxidized less frequent than sulfides) (UNEP, 1991).

Table 18. Cu, Ni, Pb, and Zn Ore Minerals (UNEP, 1991)

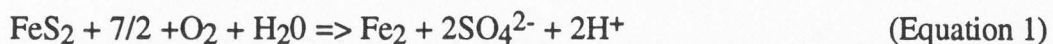
Metal	Sulfides	Oxidized Minerals
Copper	chalcopyrite	CuFeS ₂
	bornite	Cu ₅ FeS ₄
	chalcocite	Cu ₂ S
	tetrahedrite	(Cu,Fe,Zn,Ag) ₁₂ (Sb,As) ₄ S ₁₃
		malachite azurite cuprite
		Cu ₂ (OH) ₂ CO ₃ Cu ₂ O Cu ₃ (OH) ₂ (CO ₃) ₂
Nickel	pentlandite	(Fe, Ni) ₉ S ₁₃
	millerite	NiS
		garnierite
		laterite nickel ores
Lead	galena	PbS
		cerussite anglesite pyromorphite
		PbCO ₃ PbSO ₄ Pb ₅ (PO ₄ , (AsO ₄) ₃) ₃
Zinc	sphalerite	ZnS
		smithsonite calamine zincite willemite franklinite
		ZnCO ₃ SiO ₂ , 2 ZnO, H ₂ O ZnO ZnSiO ₄ (Fe, Mn, Zn)O (Fe, Mn) ₂ O ₃

Ore is composed of metals, soil and rock. The unwanted material from ore is termed gangue. Common gangue minerals are silica, silicates, carbonates and sulfides (UNEP, 1991). Sulfide ores may contain high percentages of sulfide minerals as gangue

material (UNEP, 1991). Sulfide minerals are crystalline substances containing sulfur and a metal (e.g. iron) or semi-metal (e.g., arsenic) without oxygen (Sengupta, 1993). Some sulfide minerals found in mine overburdens are pyrite, pyrrhotite, marcasite, calcopyrite, and sphalerite (Sobek et al., 1987). Sulfide minerals are responsible for the formation of acid mine drainage or acid rock drainage. Acid mine drainage is the most significant source of liquid waste in the non-ferrous metal mining industry (UNEP, 1991).

Acid mine drainage may be defined as "the inorganic chemical water pollution resulting from the oxidation of sulfide-containing minerals, mainly pyrite (FeS₂) and pyrrhotite (Fe_{1-x}S)" (UNEP, 1991). Water pollution (acidic waters) resulting from the oxidation of sulfide minerals requires three ingredients: sulfide minerals, water or a humid atmosphere, and an oxidant - particularly oxygen. Without moisture or an oxidant, acid formation will cease (Sengupta, 1993). Throughout the literature (Sengupta, 1993; UNEP, 1991; Kempe, 1983; Wallace and Wallace, 1992; Sobek et al., 1987; Lovell, 1983; and Kim, 1982) oxidation reactions for the formation of acid mine drainage are found. The most common reaction seen is the oxidation of pyrite where pyrite, oxygen, and water form sulfuric acid.

Acid formation begins with iron disulfides reacting with water and oxygen to form ferrous sulfate which releases 2 moles of hydrogen ions (Equation 1). At this stage the rate of oxidation is relatively slow because the pH is near neutral (UNEP, 1991) and *Thiobacillus ferroxidans* (iron oxidizing bacteria) are not in their most active pH range (≈ 2.0 - 5.0) (Sobek et al., 1987).



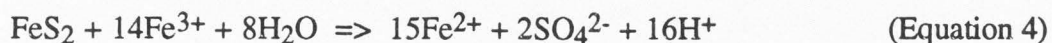
The ferrous iron will eventually use 1 mole of H⁺ and oxidize to the more stable ferric form. (Equation 2).



As the buffering capacity of the environment decreases, acid begins to accumulate and pH decreases. With decreasing pH, the oxidation rate of *T. ferrooxidans* increases causing additional acid production (Kim, et al., 1982). At approximately pH values of 2.3 to 4.0, the ferric ion will hydrolyze and precipitate leaving little Fe^{3+} in solution and freeing 3 moles of H^+ which lowers the pH further (Equation 3).



As the pH falls below 3, $\text{Fe}(\text{OH})_3$ precipitation decreases, and the solubility of iron increases. At this point, ferrous is oxidized by *T. ferrooxidans* and the ferric ion in turn oxidizes the pyrite (Equation 4) (Kim, et al., 1982).



In the last stage, acid production is high, and is only limited by ferric ions (Kim, et al., 1982).

The above example is only one pathway for the production of acid from pyrite. There are several types of sulfide ores and minerals which have various ratios of metal:sulfide and metals other than iron. Because of differing ore and mineral composition, sulfide minerals have different reaction pathways, stoichiometries, and rates. Sengupta (1993), lists pH, temperature, oxygen content of the gas phase if saturation is <100%, oxygen concentration in the water phase, degree of saturation with water, chemical activity of Fe^{3+} , surface area of exposed metal sulfide, and chemical activation energy required to initiate acid generation as the primary chemical factors determining the rate of acid generation. Similar factors were discussed in Wallace and Wallace (1992), and Kelley and Tuovinen (1988).

APPENDIX D

Ambient Water Quality Criteria for the Protection of Aquatic Life and thier Uses - Hardness Factors

Hardness is a recognized characteristic which reduces metal toxicity to aquatic life (Davies et al., 1993). As hardness increase, toxicity decreases. EPA (1984a, b, c, and 1987) considers hardness factors for deriving numerical "National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses". The following equations are used by EPA and the State of Utah class 3A water quality programs to determine metal concentrations for four day averages that should not be exceeded more than once every three years.

$$\text{Cd concentration in } \mu\text{/L} = e^{(0.7852[\ln(\text{hardness})]-3.490)} \quad (\text{EPA 1984c}).$$

$$\text{Cu concentration in } \mu\text{/L} = e^{(0.8545[\ln(\text{hardness})]-1.465)} \quad (\text{EPA 1984a}).$$

$$\text{Pb concentration in } \mu\text{/L} = e^{(1.237[\ln(\text{hardness})]-4.705)} \quad (\text{EPA 1984b}).$$

$$\text{Zn concentration in } \mu\text{/L} = e^{(0.8473[\ln(\text{hardness})]+0.7614)} \quad (\text{EPA 1987}).$$

The hardness factor used for the table 13 considered a harness of 200. Hardness was not determined from the water samples taken on site. Two hundred was selected because of the limestone geology that would contribute Ca^{2+} to the system. Hardness would need to be determined to establish water quality criteria for this area. It should also be noted that the water quality criteria derived from the above equations is generic, not site specific. The aquatic species in the North Fork may be more or less tolerant than the equations predict. Water quality criteria for the protection of the environment should be site specific due to local conditions (Isom, 1992).

APPENDIX E

Remediation Techniques

The U.S. EPA's (1993) document "Remediation Technologies Screening Matrix and Reference Guide" contains forty-eight technologies for *in situ* and *ex situ* biological, thermal, and physical/chemical processes for the remediation of soils, sediments and sludges, ground water, and air. Of the forty-eight, six are appropriate for the remediation of solid substrates contaminated by metals. These six will briefly be discussed, and a table will provide information on the applicability of each method.

In Situ

Soil Flushing is performed by injecting water or water containing an additive for enhanced contaminant solubility into ground water to raise the water table into the contaminated zone. Water with or without additives can also be applied to soil to raise the water table. Once contaminants are leached into the groundwater, the groundwater is extracted and treated.

Solidification/Stabilization is used to demobilize contaminants. Solidification is physically binding or enclosing contaminants within a stabilized mass. Stabilization requires inducing of chemical reactions between a stabilizing agent and contaminants.

Vitrification is also used to limit the mobility of contaminants. Electrodes are applied for electricity, or joule heating is used to melt contaminated soils. A glass is produced and the resultant crystalline has reduced mobility.

Ex Situ

Soil Washing is performed by adding an aqueous solution to a contaminated matrix to encourage desorption. Heavy metals are usually removed by augmenting wash water with chelating agents, and adjusting the pH.

Solidification/Stabilization can also be performed by removing the contaminated material and performing similar procedures as mentioned above.

Chemical Reduction/oxidation aims to chemically convert hazardous contaminants to non-hazardous or less toxic compounds. The resultant compounds are less toxic, more stable, less mobile, and inert. Ozone, hydrogen peroxide, hypochlorites, chlorine, and chlorine dioxide are agents most commonly used to alter oxidation states of contaminants.

Vitrification is also when the contaminated material is removed from site. This process is similar to *in situ* in that it melts the material to form glass and crystalline structures with low leaching characteristics.

Table 18 is a modified version of the Remediation Technologies Screening Matrix in EPA (1993). The table or guide does not provide costs of remediation, just relative (to other methods) information.

Table 19. Screening Matrix for Remediation Techniques. (EPA, 1993).

Technique	1	2	3	4	5	6	7	8	9	10
<i>In Situ</i>										
Soil Flushing	I	O	B	N	L	V	Y	W	A	W
Solidification/Stabilization	B	C	B	N	S	M	I	B	B	A
Vitrification	W	B	W	N	L	M	Y	B	W	W
<i>Ex Situ</i>										
Soil Washing	A	B	A	Y	B	V	Y	B	A	A
Solidification/Stabilization	B	C	B	N	S	M	I	B	B	A
Chemical Reduction/Oxidation	A	N	B	Y	S	T,M	I	B	B	A
Vitrification	W	B	A	N	L	M	Y	A	A	W

1. Overall Cost*
2. Capital (C) or Operational/Maintenance (O) intensive, Both (B), Or Neither (N)
3. Commercial Availability
4. Part of Treatment Train Yes (Y), No (N)
5. Residuals Produced - Solids (S), Liquid (L), Or Both (B)
6. Addresses Toxicity (T), Mobility, or Volume (V)
7. Long term effectiveness and Permanence Yes (Y), No (N), or Inadequate Info. (I)
8. Time to Complete Cleanup*
9. System Reliability/Maintainability*
10. Regulatory/Permitting Acceptability*

* Better (B), Average (A), Worse (W), Inadequate Information (I), and Not Applicable (NA)

Appendix F

Summary of Data

Zinc Site #	Water (mg/L)	MgCl (mg/Kg)	NaOAC (mg/Kg)	Total (mg/Kg)	Difference
1	1.87				
2		180.8	1082.3	4358	3094.9
3	1.19	110.4	423.8	1129	594.8
4	0.08	1.6	101.5	405	301.9
5	1.22	114.4	309.6	3745	3321
6	1.11	305.6	6442.8	25448	18699.6
7	0.01	79.2	420.5	4125	3625.3
8	-0.03	67.2	1178.4	2874	1628.4

Copper Site #	Water (mg/L)	MgCl (mg/Kg)	NaOAC (mg/Kg)	Total (mg/Kg)	Difference
1	0.0053				
2		1.6	102.3	504	400.1
3	0.0094	0.8	0.6	40	38.6
4	0.0038	0	1	18	17
5	0.00302	1.6	20.2	154	132.2
6	0.0072	4	35.9	1660	1620.1
7	-0.0014	0.8	18.1	117	98.1
8	0.00118	1.6	7.3	90	81.1

Cadmium Site #	Water (mg/L)	MgCl (mg/Kg)	NaOAC (mg/Kg)	Total (mg/Kg)	Difference
1	0.0122				
2		8.8	8.13	36	19.07
3	0.0109	5.6	1.2	6	-0.8
4	0.0025	0.8	1.6	2	-0.4
5	0.0136	4	2	28	22
6	0.0105	56.8	98.6	180	24.6
7	0.0005	1.6	1.5	27	23.9
8	0.0018	8.8	11.7	19	-1.5

Lead Site #	Water (mg/L)	MgCl (mg/Kg)	NaOAC (mg/Kg)	Total (mg/Kg)	Difference
1	-0.01				
2		125.6	4414.5	7152	2611.9
3	-0.03	3.2	111.7	411	296.1
4	0.01	0	24	90	66
5	0.02	518.4	5009	7305	1777.6
6	-0.02	1.6	791.8	3418	2624.6
7	-0.04	257.6	3559.4	4580	763
8	0	4	423.5	1090	662.5

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