Acid Mine/Rock Drainage Effects On Water Quality, Sediments, Invertebrates and Fish Located in Uncle Sam Gulch, Cataract Creek and the Boulder River, Northern Jefferson County, Montana

By

Douglas H. Martin
B.S., Montana State University, 1986

Presented to the Department of Environmental Engineering,
Montana College of Mineral Science and Technology
in Partial fulfillment of requirements for the degree of

Master of Science in Environmental Engineering
Montana College of Mineral Science and Technology
Butte, MT 59701
May, 1992

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ABSTRACT

Abandoned mines from gold mining operations that produce Acid Mine/Rock Drainage (AMD) can affect water quality and stream ecosystems downriver. This thesis reports on copper, cadmium and zinc being emitted from the Crystal Mine near Basin, MT. Water quality, sediment, invertebrate and fish populations were analyzed to determine if the Crystal Mine AMD was the main source of pollution effecting Uncle Sam Gulch, Cataract Creek and the Boulder River.

Results of this study indicate that the Crystal Mine is the main source of pollution impacting downriver water quality and aquatic ecosystem. Remediation of mine tailings, disturbed areas and mine adit effluent are needed to decrease the AMDs impact.

Keywords: AMD, copper, cadmium, zinc, remediation

ACKNOWLEDGEMENTS

This thesis was completed with the help and guidance of many. I would like to give special thanks to my committee members for their guidance. Dr. Elmer Gless, who gave me the opportunity to work on this project. Dr. Rod James, Dr. Tom Finch and Peter Nathanson thanks for your direction and instructions. Also, thanks to Dr. Rick Appleman for answering many a questions. Thanks to Steve McGrath and all the people at the Bureau of Mines Lab. I would also like to thank all who helped me along the way with my sampling and writing. Mike Risher and Kelly Packerd, special thanks for all their help during the sampling. Thanks also to everyone else how helped me collect my October samples. I believed we all learned something about invertebrates.

A very special thanks needs to go out to my parents, Joe and Joan, my Grandmother Hane and Grandfather Martin who made this all possible with their support and help. Last but definitely not least thanks to my fiancee, Carrie for her encouragement along the way when things got tough.

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INTRODUCTION

Acid Mine Drainage(AMD) or Acid Rock Drainage (ARD) occur as a legacy of waste rock, mine tailings and drainage tunnels left behind by the mining industry. There are thousands of sources of AMD in the U.S.; 66,500 sources exist in the Appalachia region, from active and inactive coal mines which have polluted approximately 10,500 miles of rivers and streams (Cohen, 1991), Colorado has approximately 1,300 miles of streams effected by AMD (Meyer, 1991), Montana has approximately 200 to 300 abandoned mines (Cherry, 1991). Remediation of AMD and water quality effected by AMD has been researched recently in the West.

There is a demand today for high quality water. Streams with poor water quality are a wasted resource. Currently, Montana does not have a severe water shortage problem, however, recurrent droughts in the Rocky Mountain region are drastically depleting water levels. As public demand for higher water quality rises, levels of water quality standards will continue to increase. This demand for good "clean" mountain water will require remediation of AMD.

Acid Mine Drainage from mines into streams is a pollution problem that has been adversely affecting water quality and stream ecosystems throughout the country. Hard rock mining exposes sulfide minerals to oxidation, resulting in the discharge of strong acids and metals which are detrimental to water quality and aquatic life (Platts, 1979). AMD is considered a non-point source of pollution because it can be generated over a large area. Also, AMD is considered the actual effluent from the mine adits and the areas of waste rock and tailings left from the mining operation (Cohen, 1991). Montana's abandoned mines, which are associated with small gold claims, are not large by themselves. Although, within a few square miles, many mines may exist.

Acid Mine Drainage results from the oxidation of pyrite and other sulfides when the refuse is exposed to oxygen and water. Oxidation of pyrite produces hydrogen ions, causing the lowering of pH levels in the mine water. Micro-organisms are closely associated with the overall process, which involves several reactions.

Reaction 1: Oxidation of pyrite

$$2FeS_2 + 2H_2O + 7O_2 \rightarrow 4H^+ + 4SO_4 + 2Fe^{2+}$$

Reaction 2: Oxidation of ferrous ion (Fe^{2+}) to ferric ion (Fe^{3+}).

$$4Fe^{2+} + O_2 + 4H^+ \rightarrow 4Fe^{3+} + 2H_2O$$

Reaction 3: Ferric ion dissolves pyrite.

$$FeS_2 + 14\ Fe^{3+} + 8H_2O \rightarrow 15Fe^{2+} + 2SO^{2-}{}_4 + 16\ H^+$$

Reactions 2 and 3 complete the cycle for the dissolution of pyrite. Reaction 2 is driven by the iron bacterium *Thiobacillus ferrooxidans* at pH levels below 3.5 and catalyzed by a variety of *Metallgenium* at pHs between 3.4-4.5. These reactions also produce sulfuric acid which is the most toxic component within these reactions (Manahan, 1990).

$Fe(OH)_3$ is a acidic ion and at higher pH values will precipitate out as hydrated iron (III) oxide.

$$Fe^{3+} + 3H_2O <-> Fe(OH)_3 + 3H^+$$

This precipitation produces the condition, called "yellow boy", associated with stream beds effected by AMD.

Objective

Acid Mine Drainage produced by the Crystal Mine near Basin, MT exhibits the same characteristics as other AMD problems in the west. This thesis attempts to determine whether, or not, AMD from the Crystal Mine has detrimental effects on the water quality, sediment, invertebrates and the fisheries located downstream of the mine. Results are presented from two sampling sessions, June and October, 1991. Results are analyzed to determine both the fate and effects of the AMD as it is transported downstream.

Purpose of Research

Concentrations of copper, cadmium and zinc affect the water quality of the stream, and at certain levels, can cause acute or chronic toxicity to invertebrates and fish. Sediment concentrations of these heavy metals can affect the invertebrate and fish populations by eliminating habitat and interfering with the natural progression of nutrients in the food chain.

Dr. Elmer Gless of Montana Tech, Glen Phillips and Curt Hill of the Montana Fish, Wildlife and Parks Department and Ken Knudson participated in previous studies that examined water quality in the Boulder River. These studies suggested that the Crystal Mine produced AMD which degraded water quality and limited the fisheries populations in Cataract Creek and the Boulder River. This thesis attempts to identify the Crystal Mine AMD as the pollution source effecting water quality and the fisheries population by taking baseline samples upstream of confluences downstream of the Crystal Mine.

Experimental Plan

This study tracks copper, cadmium and zinc in the flow of the AMD, originating from the Crystal Mine, as the AMD enters into Uncle Sam Gulch, Cataract Creek and the Boulder River. Samples of the water, sediment, invertebrate populations and fish tissue were analyzed to determine if the Crystal Mine is having a detrimental effect on the stream ecosystem. Concentrations of these metals were analyzed at various sites along the water course. After determining the fate of the AMD from the Crystal Mine, remediation alternatives are examined.

Study Area

The study area is shown on Figure 1. Figure 2 details the sampling sites, tributaries, mining activities and mine tailings which are adjacent to the streams. Sampling sites

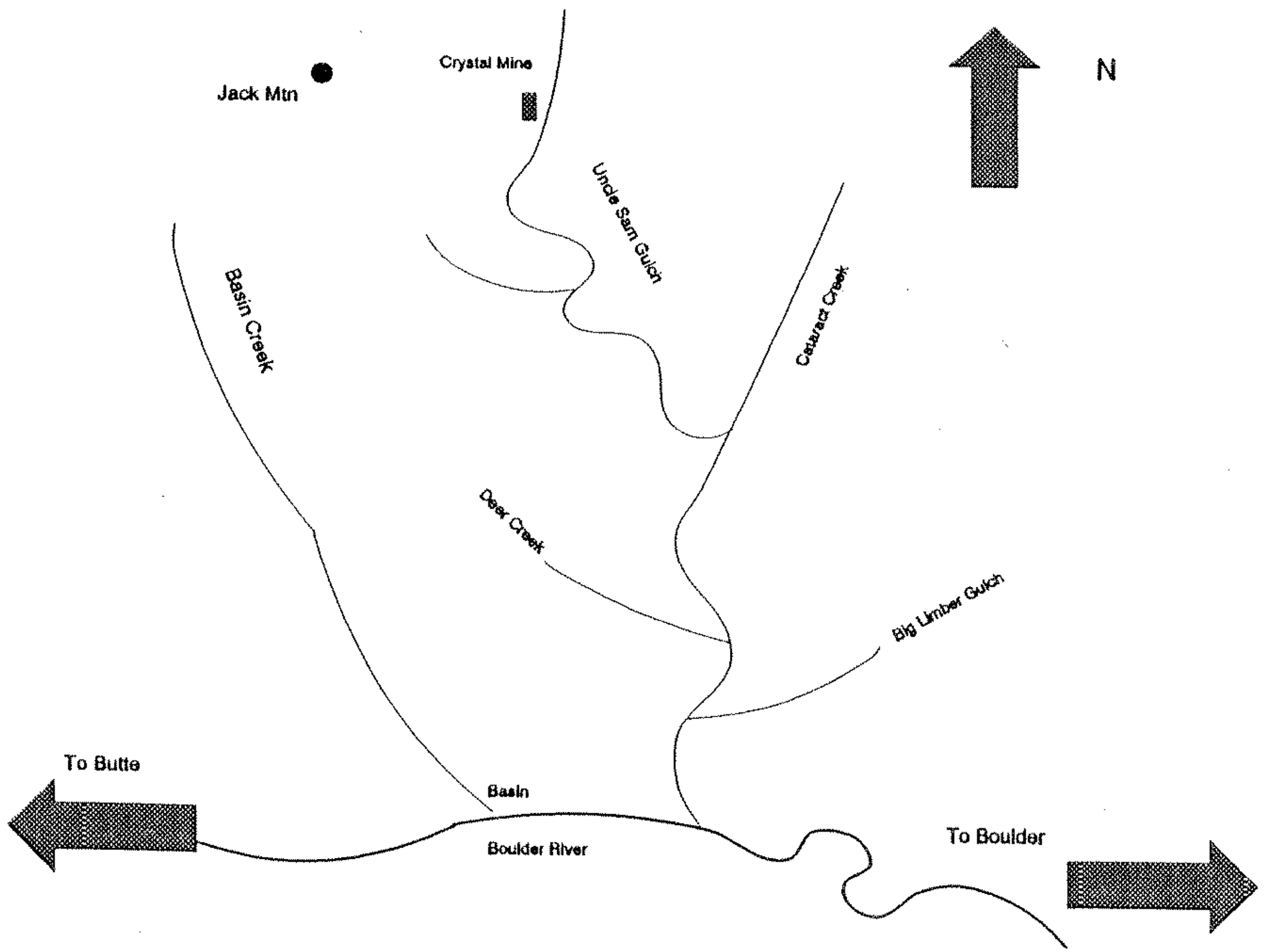
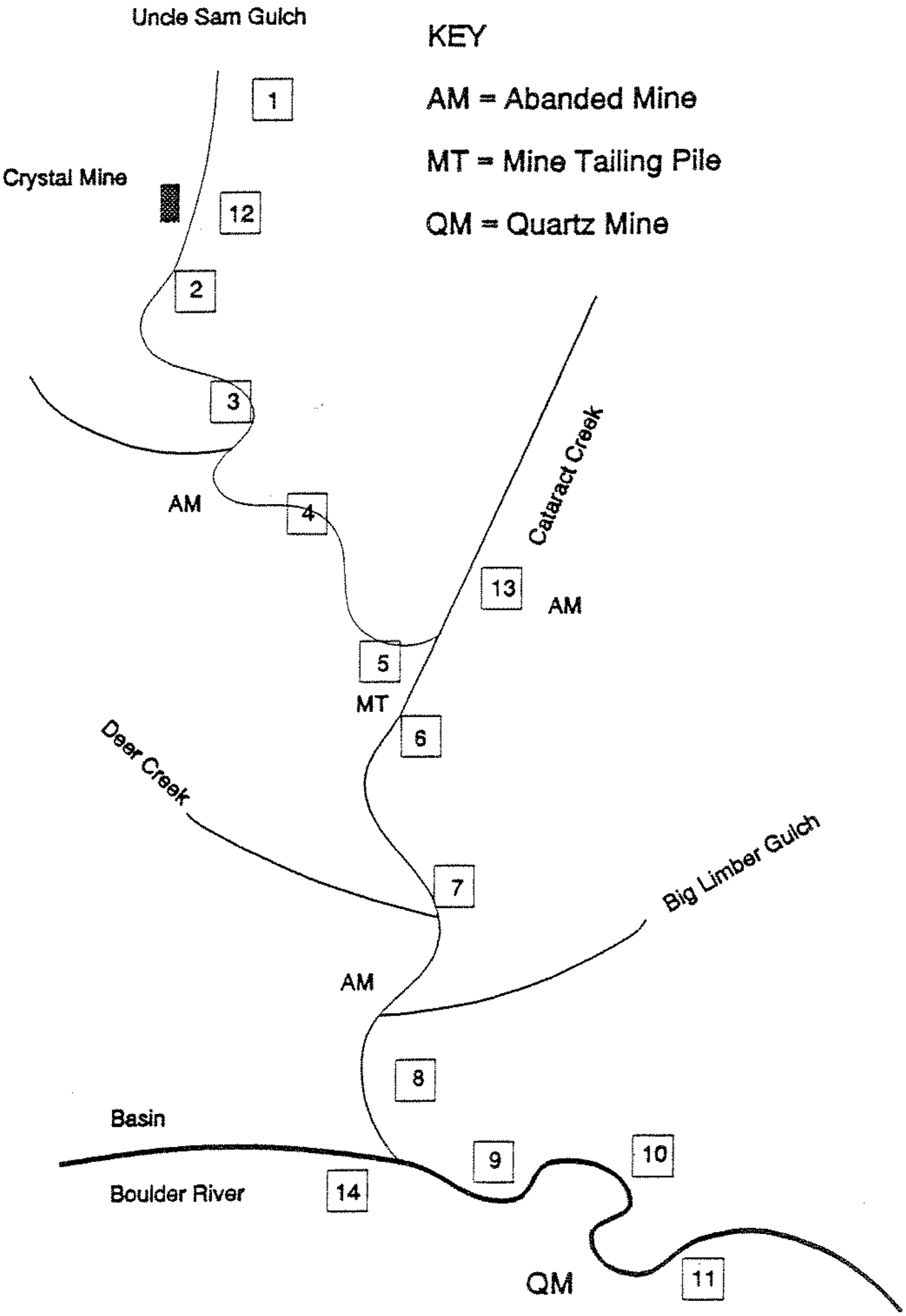


Figure 1. Study Area

Figure 2. Study Area



include background Sites 1, 13 and 14 which are located above the Crystal Mine, upstream of the confluence of Uncle Sam Gulch and Cataract Creek and upstream of the confluence of Cataract Creek and the Boulder River, respectively. Four sampling sites; 2, 3, 4 and 5 are located from the Crystal Mine downstream along Uncle Sam Gulch, 3.2 miles, to Cataract Creek. Three sampling sites; 6, 7 and 8 are located along Cataract Creek, 3.6 miles downstream to the Boulder River. Another three sampling sites; 9, 10 and 11 are located on the Boulder River, 1.4 mile downstream. The Crystal Mine adit effluent is Site 12.

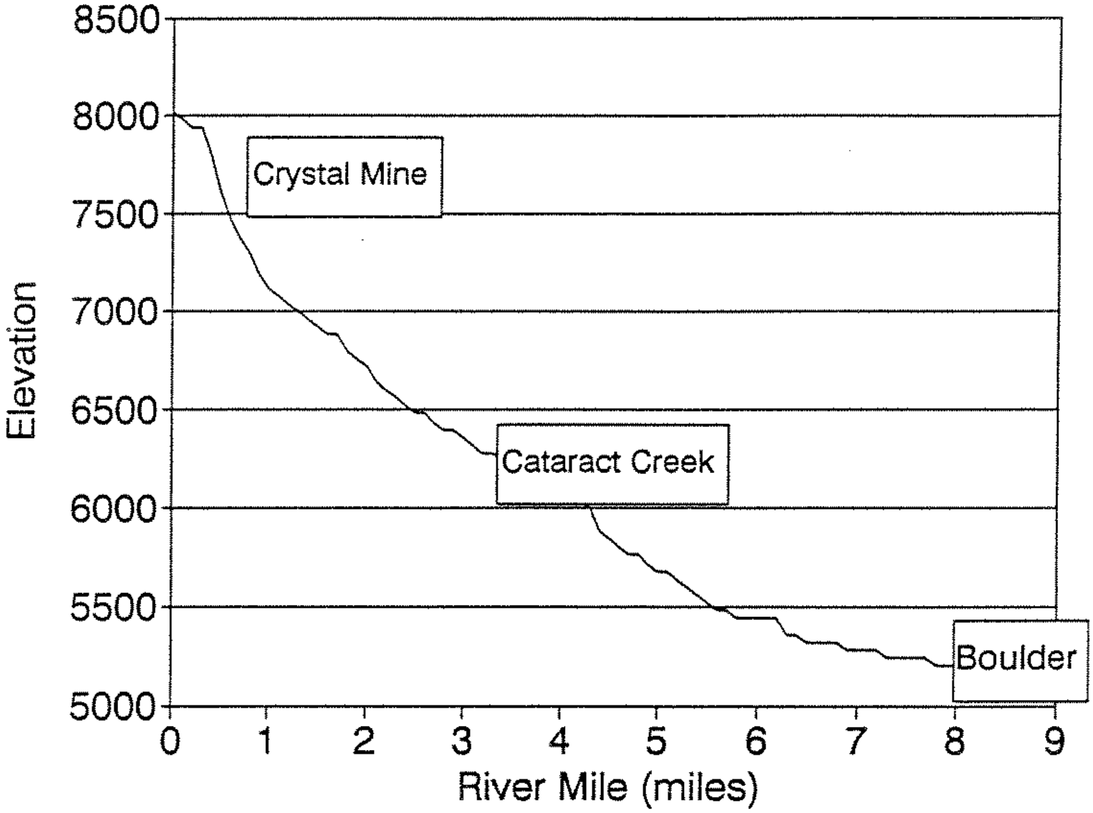
Figure 3 illustrates the elevation profile of the stream course through the study area.

LITERATURE REVIEW

AMD is a pollution problem which has been affecting water quality since mining began in the 1800's. And, the problem is not limited to the U.S.; European nations have recognized AMD as a major source of water pollution and have conducted extensive research on the effects on the water quality, sediment, invertebrates and fish. Most of the research completed in the U.S. has dealt with AMD effects on water quality. Invertebrate and fish populations have been researched by biologists observing the effects of heavy metal concentrations.

Figure 3

Elevation vs Rivermile



Water Quality

Reviewed literature agrees that AMD has adverse effects on water quality and the stream ecosystems. Heavy metals such as copper, cadmium and zinc can degrade water quality and be chronically or acutely toxic to aquatic organisms (Chambers, 1991, Kelly, 1988, Letterman, 1978, Mason, 1991, Meyer, 1991, Vandenberg, 1974). Vandenberg, 1974, has suggested that the elimination of AMD in Galena Creek, located in central Montana, should be the principle consideration for the rehabilitation of the creek. Phillips and Hill, 1985, suggest that concentrations of copper and zinc from Cataract Creek are the limiting factors affecting the trout population in the Boulder River near Basin, Montana.

Copper concentrations are reported to range from a low of 1 μg/l to a high of 280 μg/l with a mean of 15 μg/l in clean waters of the U.S., according to Stoker, 1976. Water quality surveys completed by the Washington Power Company, 1989, reported copper levels from unpolluted natural waters of 0.0 to 2.2 μg/l. Gless, 1990, reported copper levels from Jefferson County waters near and in the study area. These levels ranged from 4 μg/l to 198 μg/l. Boulder River copper concentrations were below the detection limits except for samples from the mouth of Cataract Creek which ranged from 10 μg/l to 50 μg/l (Phillips, 1985).

AMD research conducted in the U. S. has focused upon the coal mining regions the eastern portion of the country.

Prediction of AMD has been evaluated by testing the overburden and estimating the potential for AMD formation in the coal mining region. Overburden is analyzed on an acid-base basis; low acid neutralizes samples having a potential for AMD formation (diPretoro, 1988, Erickson, 1988, Miller, 1988). This procedure has been tested and rejected for the pre-mining prediction of AMD from coal mining areas. Miller, 1988, sites the use of this technique to predict AMD from hard-rock and precious metal mining operations. The Net Acid Producing Potential (NAPP) is calculated from the total sulfur content and the inherent Acid Neutralizing Capacity (ANC) measured as $CaCO^3$.

$$\text{NAPP} = \%\text{S} \times 3.125 - \text{ANC (Miller, 1988)}$$

Low pH which is a result of AMD, causes the dissolution of heavy metals which then affects water quality and organisms downstream of the AMD production site. Knudson, 1984, and Phillips and Hill, 1985, predict effects of the copper and zinc from the Crystal Mine, located 6.8 miles from the Boulder River, are the limiting factors for the growth of the fish population. Chambers, 1991, found adverse impacts from AMD near Cooke City, Montana, approximately 2 kilometers downstream of the AMD source. The effects were observed in plant life adjacent to the stream affected. Effects of the AMD were seen 18.5 kilometers downstream of the source in Galena Creek, located in central Montana (Vandenberg, 1974). AMD has affected water quality, and, thus eliminated fish from approximately 10 miles of river downstream of an abandoned mine in Idaho (Platts, 1979). In Europe effects of AMD have been observed in water, sediment and invertebrate biomass ranging from 2 kilometers to 8 kilometers, although maximum distance affected was not measured (Brown, 1977, Burrows, 1983).

Acid Mine Drainage causes unnaturally high levels of copper, cadmium and zinc. Evolution has provided organisms with metabolic pathways which cope with naturally occurring toxic compounds. The synthesis and degradation of toxic compounds provides a balance resulting in natural levels (Wood, 1973). Weathering and dissolution of natural copper and zinc, which usually occurs as sulfides and oxides, results in the background levels in natural waters (E.P.A. 1,2, 1980). Aquatic ecosystems are particularly sensitive to cadmium pollution because: 1) cadmium is biologically nonessential and non-beneficial, 2) very low levels of cadmium are present in natural waters, and 3) cadmium has a strong tendency to bioaccumulate in the food chain (Nriagu, 1987). A summary of heavy metals in waters of the U.S. by Stoker, 1976, showed copper, cadmium and zinc with a mean concentrations of 15 μg/l, 9.5 μg/l and 64 μg/l respectively.

The affects of copper, cadmium and zinc on an aquatic ecosystem depends on various factors. Temperature increases in water tend to increase the toxicity of metals (Mason, 1991). The effects of the pH on AMD is associated with dissolved metal precipitation and bio-availability of metals. Precipitation of dissolved metals increases as the pH of the water rises and the availability of hydrogen ions is reduced (Kelly, 1988, Letterman, 1978). Mason, 1991, sites the lowering of the number of hydrogen ions as the pH rises to an increase in the bio-available metals.

Acid Mine Drainage is associated with a wide variety of heavy metals which, individually, are toxic to the aquatic ecosystem. Combinations of metals can have additive, non-interaction or antagonistic effects. The combination of copper, cadmium and zinc has an additive or more than additive effect (Mason, 1991, Nriagu, 1987).

Sediments

Sediments associated with the stream bottom are affected by AMD heavy metal concentrations, and are a pathway for metals into the aquatic foodchain. Fine sand and clay particles adsorb a large portion of the heavy metal ions (Chand, 1981). Metal ions are lost from solution by precipitation or adsorption, especially when iron oxides and aluminum floc are present. The extent of adsorption is dependent on the pH level. As the pH raises the ratio of bound ions to free ions, copper, cadmium and zinc concentrations increase, leading to higher concentrations of metals in the sediment (Kelly, 1988). Brown, 1977, and DeGroot, 1973, showed sediment concentrations to be functions of pH, river flow, sediment load and water hardness. Brown, 1977, observed lower metal concentrations in sediment in conjunction with lower pH values. DeGroot, 1973, associated a major portion of metals in sediment downstream from mining activities with the suspended material which was transported downstream. Water to sediment concentrations from the Rhine River were 1:1.8, 1:0.6 and 1:0.8 for copper, cadmium and zinc, respectively (DeGroot, 1973). Concentrations of copper increased markedly during low water sampling. This increase in metal concentration was probably due to low flowrates depositing "particulate" copper suspended in the water (Brown, 1977). Brown, 1977, also observed that zinc had higher dissolved water concentrations then copper, and copper had higher sediment concentrations, although zinc concentrations in the sediment remained constant.

Invertebrates

There is general agreement that AMD drastically reduces biomass of invertebrates present in the affected stream (Letterman,1978, Mason, 1991, Platts, 1979, Roembild, 1991, Vandenberg, 1974). Vandenberg, 1974 found depressed numbers of: individuals, taxa and diversity 18.3 Km downstream of the AMD source on Galena Creek in central Montana. The depressed numbers coincided with an increase of the more tolerant species, such as *Tendipedidae*, due to lack of competitors and predators (Kelly, 1988, Vandenberg, 1974). Platts, 1979, reports that Panther Creek, which is effected by AMD in central Idaho, has a healthy invertebrate population with "good" biomass above the AMD. Below the AMD entrance, invertebrate biomass was "fair to good" with the invertebrate population dominated by drift-through species. Diversity indices, which indicate invertebrate population reactions to pollution, are used more in Europe than in the U. S. Diversity indices are more useful for assessing organic pollution than metal-related pollutants (Letterman, 1988, Mason, 1991). Biomass of invertebrates may be a better indicator of AMD effects than diversity indices, because populations may adjust and adapt to the AMD over time (Letterman, 1998, Mason, 1991). AMD associated with mountain streams may have inaccurate invertebrate indices due to invertebrate drift caused by high flows (Letterman, 1978, Platts, 1989, Roembild, 1991).

Acid Mine Drainage affects invertebrate populations by altering the bottom characteristics of the stream. Precipitation of Fe(III) and other metals leads to the elimination of invertebrate habitat by hardening the bottom material (Kelly, 1988, Letterman, 1978, Mason, 1991). pH which plays a role in the precipitation of Fe(III) is effected by the bicarbonate buffering system of the water. Magnitude shifts of pH, which can be detrimental to invertebrate populations, are controlled by the bicarbonate buffering system (Kelly, 1988).

Chironomidae are frequently associated with AMD and are often found in areas where Fe(III) has been deposited (Kelly, 1988, Letterman, 1978, Mason, 1991).

Heavy metal concentrations in invertebrates are associated with tolerance and feeding habits (Brown, 1977, Burrows, 1983, Smock, 1983). Burrowing species (i.e. Ephemeridae, Chironomidae), which randomly ingest sediment along with detritus material, have the highest metal concentrations. Filter feeders (i.e. Hydropsychidae) have the second highest metal levels. Species such as Ephemeroptera, Plecoptera, and Tricoptera, which selectively ingest detritus and periphyton, have the third highest metal levels. Carnivorous species such as Gerridae have the lowest metal concentrations (Brown, 1977, Burrows, 1983,Smock, 1993).

The relationship between metal concentrations in sediments and invertebrates seem to be metal and species dependant (Anderson, 1977, Brown, 1977, Burrows, 1983, Smock, 1983). As stated earlier, the feeding habits play a role in metal accumulation. A general relationship between sediment and invertebrates does not always exist throughout the invertebrate population and their locations above or below AMD. Anderson, 1977, sampled the Fox River in Illinois where pollution is mostly storm water runoff and industrial effluent. Comparing 35 genera of invertebrates showed there were no significant trends of accumulations between classes and orders.

Fish

Fisheries are adversely affected by AMD due to loss of invertebrate populations and the toxicity of heavy metal pollutants. Metal concentrations in water demonstrated to cause avoidance by trout are: 2 μg/l for copper and 5 μg/l for zinc. A combination of copper and zinc will lead to lower concentrations required for avoidance (Sprague, 1964).

In Pennsylvania, stream systems affected by AMD, large differences in the fish biomass were observed. Upstream of the AMD, 10 species and 22,812 kilogram per hectare (kg/ha) of fish were present, whereas downstream of the AMD, 6 species and 11 kg/ha of fish were found (Mason, 1991).

Knudson, 1984, states that the limiting factor in the Boulder River is the metal toxicity produced by the mining activities in the surrounding areas. The fishery above the town of Basin, in the Boulder River, is considered a healthy population. Below the town of Basin the fish population was reduced by 56 percent; 1135 trout/mile above Basin and 635 trout/mile below the confluence of Cataract Creek. Nelson, 1976, found a correlation between metal contamination and fish population on the Boulder River. Effects of metals were observed 30 miles downstream from the mining area (Nelson, 1976).

Along Panther Creek, located in central Idaho, fish populations have been eliminated by AMD (Platts, 1979). Elecrofishing produced a large number of species and biomass above the AMD and no fish were found downstream of the AMD entrance. Platts suggests that few fish indicates avoidance of undesirable conditions.

Fingerling survival tests in Panther Creek, revealed 86 percent mortality below the AMD entrance and 48 percent mortality 5 miles downstream. Studies by the Idaho Bureau of Mines and Geology showed that increased metal concentrations, during high water, could cause acute toxicity. Platts ,1979, concludes that the high concentrations of heavy metals lead to toxic conditions, behavioral changes, avoidance, and long-term chronic toxicity to fish. These factors could be responsible for the reduction and/or elimination of fish populations.

Copper, cadmium and zinc all effect fish populations (E.P.A. 1, 1978 ,E.P.A. 2, 1980, E.P.A. 3, 1980, Mason, 1991). Tolerance to pollutants is species-specific and affects each life stage of a species differently (Mason, 1991). AMD affects fish by increasing the metal concentrations in the water. High copper and zinc concentrations in the water affect the fish's ability to transport oxygen across gill tissue (E.P.A. 1, 1978). Cadmium is chronically damaging to fish at levels of 3.4 μg/l (E.P.A.1, 1978). It is also reported in E.P.A. 1, 1978, that fish reach a plateau zinc concentration within 30 days in waters exceeding 47 μg/l zinc and adverse effects, reduced growth, are observed in water concentrations of 51 μg/l. Metals from AMD also accumulate in liver, kidneys and intestines of fish although no correlations have been accurately been made (E.P.A.1, 1978, Carnes, 1984).

Metal concentration toxicity levels are dependent on pH and hardness of the water (E.P.A.1, 1978, E.P.A. 2, 3, 1980). Tolerance to copper, cadmium and zinc has been shown in rainbow trout when levels are at low concentrations (Mason, 1991). Seven day lethal concentrations of 50 percent mortality (LC_{50}) are 44 μg/l, 7 μg/l, 560 μg/l with a water hardness of 40 mg/l $CaCO_3$, for copper, cadmium and zinc, respectively (E.P.A. 2, 1980, E.P.A. 3, 1980, Mckim, 1975).

METHODS AND MATERIALS

Data collection for water, sediment, invertebrates and fish consisted of two collection periods; the first beginning in June, 1991 and the second in October, 1991. During the first sampling period, June 10 through June 26, 1991, a collection from all sampling sites was conducted. The second sampling period, October 19 through October 21, 1991, consisted of the collection of data from specific sites. The second sampling period was conducted due to the laboratory loss of invertebrates from the first sampling period.

Sampling locations were selected along Uncle Sam Gulch, Cataract Creek and the Boulder River. Figure 2 shows the location of the sampling sites. Legal description (section, township, range) of the sites is reported in appendix. Five sites were located along Uncle Sam Gulch, four sites on Cataract Creek, four sites on the Boulder River and one at the Crystal Mine adit, Site 12. Background or baseline samples were collected from Sites 1, 13 and 14. Site 1, located above the Crystal Mine, was used as the baseline stream due to the lack of any mining activity above the mouth in the Uncle Sam Gulch drainage. Sites 13 and 14 were sampled to determine the copper, zinc and cadmium levels present in the aquatic system before contamination from the Crystal Mine enters into the stream system. Sampling sites were selected by observational selection. Sites were located downstream of any tributary and below the mixing zone of the tributary.

Water

Water samples were collected from all sites during the June sampling period and from Sites 5, 6, 8, 9, 12, 13, 14 during the October sampling. Water was collected in a 1000 ml beaker from the streams and transferred into bottles supplied by the Bureau of Mines. The collection of the water was by representative grab sample from each site. Four types of samples were collected for analysis; raw unfiltered, raw acidified, filtered raw and filtered acidified. McGrath, 1992; raw unfiltered samples were collected for alkalinity, specific conductivity and lab pH, raw acidified samples were for biologically-available and total trace metals, filtered raw samples were for anion analysis and filtered acidified samples were for analysis of major cations and trace metals.

Filtered samples were field filtered with MicronSep membrane filters. Acidified samples were acidified with HNO_3 to 5% by volume. Samples were stored in a cooler in the field and transferred to a refrigerator until submitted to the Bureau of Mines for analysis.

Temperature and pH were measured in the field at the time of the water sample collection. A Fisher Scientific Acument 1003 pH meter with a automatic temperature probe was used to measure the temperature and pH. Total suspended solids (TSS) samples were collected at all sampling sites and at the same time as the water samples were collected. TSS measurements were measured from 500 ml samples using Standard Methods procedure 209 C Total Suspended Solids Dried at 103-105 Celsius (Standard Methods, 1985). The water samples were submitted to the Bureau of Mines Analytical Lab for a complete analysis which included metals scan, $CaCO_3$ hardness, chemical oxygen demand, alkalinity, phosphate and nitrate levels with the Inductively Coupled Plasma-Atomic Emission Spectrometry (I.C.P.) unit. E.P.A. method 200.7 was used by Bureau of Mines for water analysis. Procedures are reported in Appendix. Chemical oxygen demand was conducted at Sites 6, 7 and 8 only. To establish if the reduction of invertebrates at these sites was oxygen related.

Sediment

Sediment samples were collected at each site except Site 12, the Crystal Mine adit. Samples were collected from the locations and areas where invertebrates were found. Fine sediment from slow moving waters were best sampled using a small lid as a scoop. Sediment and water were sieved through a 67 micron sieve until a liter of sediment and water were collected. Sediment samples were placed in glass jars until the water appeared clear. The water was discarded and jars were placed in a drying oven at 70 Celsius until the sediment dried. Digestion of the sediment, for metal analyses, followed U.S. Bureau of Mine method 1-5485-85 "Extraction Procedure, Bottom Material." This method was modified by replacing double distilled HCl with HNO_3. This modification was used due to the lack of quality double-distilled HCl. The modification with HCl was used in the original method to prevent the iron oxides from precipitating. Since analysis did not include iron levels, the substitution from HCl to HNO_3 was suggested. Other adjustments included; the amount of sediment used for each sample was measured to the nearest 1/100th of a gram, using approximately 1 gram for each analysis and dilution of sample with distilled water was decreased. Method 1-5485-85 is listed in Appendix.

Invertebrates

Invertebrates were collected from Sites 1, 6, 7, 8, 9, 10, 11, 13 and 14 during the June sampling and from Sites 6, 9, 13 and 14 during the October sampling. Six square feet were sampled at Site 1 using a Surber Sampler. Invertebrates were collected from the other sites with a 3 foot by 4 foot screen. Collections were made by displacing rocks located approximately 4 feet upstream of the screen. Four collections at each site, totaling 48 square feet, were made across the representative habitats at each site. All invertebrates collected at each site were placed in glass jars and preserved in formaldehyde. Identification was completed by Dr. George Roembild of Bozeman, MT. Digestion of the invertebrates for analysis followed the E.P.A. procedure 200.3 "Sample preparation procedure for spectrochemical determination of total recoverable elements in biological tissue." Modifications of procedure 200.3 included the substitution of HCL with HNO_3 because of the absence of quality HCL and volumes used in digestion were reduced by half because the sample size was smaller then procedures suggested. The procedure is listed the Appendix.

Fish

Fish were collected from Sites 6, 7, 8, 9, 10, 11, 13 and 14 during the June sampling and from Sites 8, 9, 13 and 14 during the October sampling. Fish were not collected at other sites during the June sampling because there were no fish present. Collection of fish was by Curt Hill of the Montana Fish, Wildlife and Parks Department. A backpack electrical shocker was used to collect fish at sites indicated. The fish were frozen and the kidneys and livers removed and refrozen. Each sample of kidney and liver was placed in a zip-lock bag and freeze-dried in the chemistry building, on Montana Tech campus. The digestion of the fish kidney and liver tissues in preparation for metal analyses was the same E.P.A. method 200.3 indicated previously for the invertebrate digestion.

Table 1. Water Quality Parameters Measured
(Flow, pH, Temperature, T.S.S., Hardness as $CaCO_3$)

Site	Flow(cfs)	pH	Temp(C)	TSS(mg/l)	$CaCO_3$
1	0.5	6.54	6.6	0	7.39
2	1.3	3.65	7.7	18	37.33
3	3.5	4.66	5.7	6.4	30.95
4		6.38	5.7	2.6	24.15
5a 5b	5.1 1.24	6.91 6.51	5.1 0.6	1.8	23.99 68.56
6a 6b	43.2 8.0	7.36 7.82	8.3 0.9	0.8	24.49 52.39
7	43.2	7.55	7.8	0.4	24.08
8a 8b	43.8 7.79	7.57 8.3	7.9 0.1	1.2	25.49 54.12
9a 9b	204.0 33.62	7.63 8.1	8.1 2.7	3.2	29.38 54.45
10	204.0	7.61	8.0	3.2	28.97
11	204.0	7.93	8.4	4.0	29.38
12a 12b	39(gpm) 30(gpm)	2.86 3.12	4.7 3.3	43.4	206.44 185.84
13a 13b	38.6 6.75	7.73 8.03	8.2 1.1	0.8	23.99 48.32
14a 14b	260.0 25.33	7.76 8.26	8.8 3.3	2.6	28.22 51.13
B Limber	0.2	8.07	8.8	0	103.21
DeerCr	0.5	7.92	7.7	0	50.54

a = June samples b = October samples

Metal concentrations for water samples are listed in Table 2. These concentrations are the total available concentrations or bio-available concentrations. Concentrations less then 6 μg/l were below the detection limits of the ICP analyzer. Thus, results which are less than 6 μg/l will be treated as no concentrations present.

Table 2. Metal Analysis of Water Samples
(Copper, Cadmium, Zinc in μg/l)

Site	Copper(μg/l)	Cadmium(μg/l)	Zinc(μg/l)
1	9	<6	<6
2	2850	115	9550
3	1520	60	5250
4	235	22	1630
5a 5b	183 312	12 59	1450 5160
6a 6b	43 61	<6 8	185 802
7	25	<6	178
8a 8b	29 47	<6 <6	165 528
9a 9b	17 27	<6 <6	51 134
10	8	<6	49
11	13	<6	48
12a 12b	2520 12000	939 872	7650 64160
13a 13b	14 <6	<6 <6	34 77
14a 14b	<6 17	<6 <6	25 70
Big Limber	<6	<6	<6
Deer Creek	<6	<6	40

a: June samples b: October samples

Figure 4. Water Copper Concentration

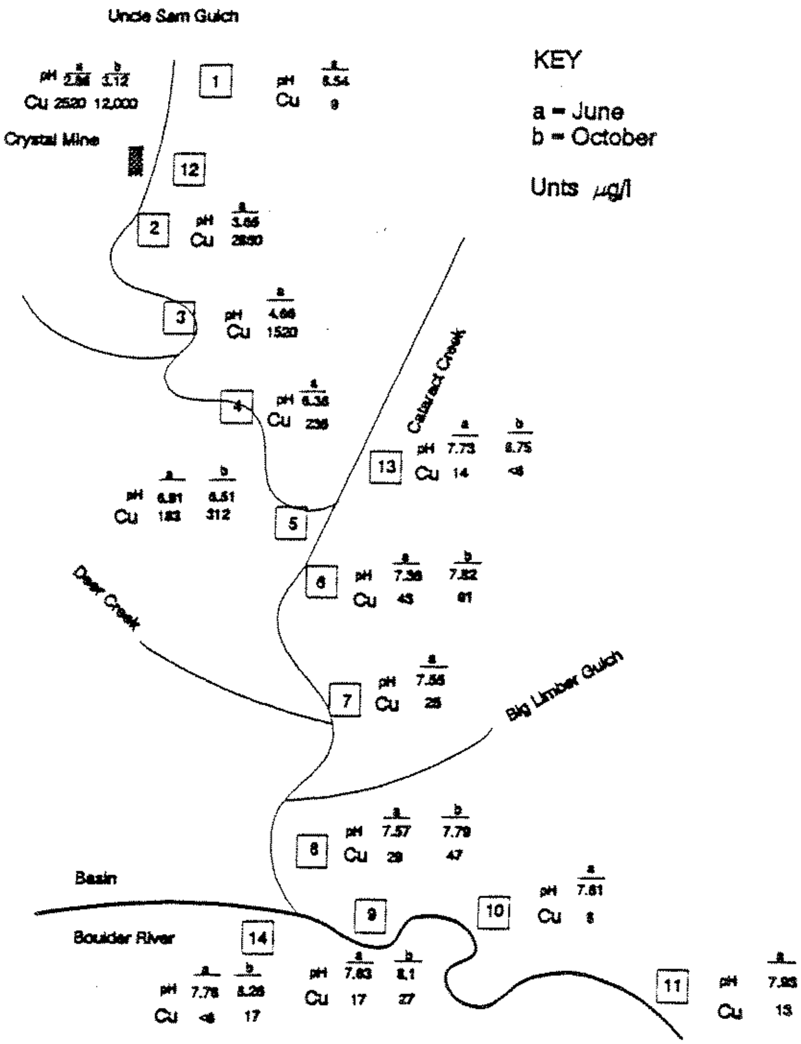


Figure 5. Water Cadmium Concentration

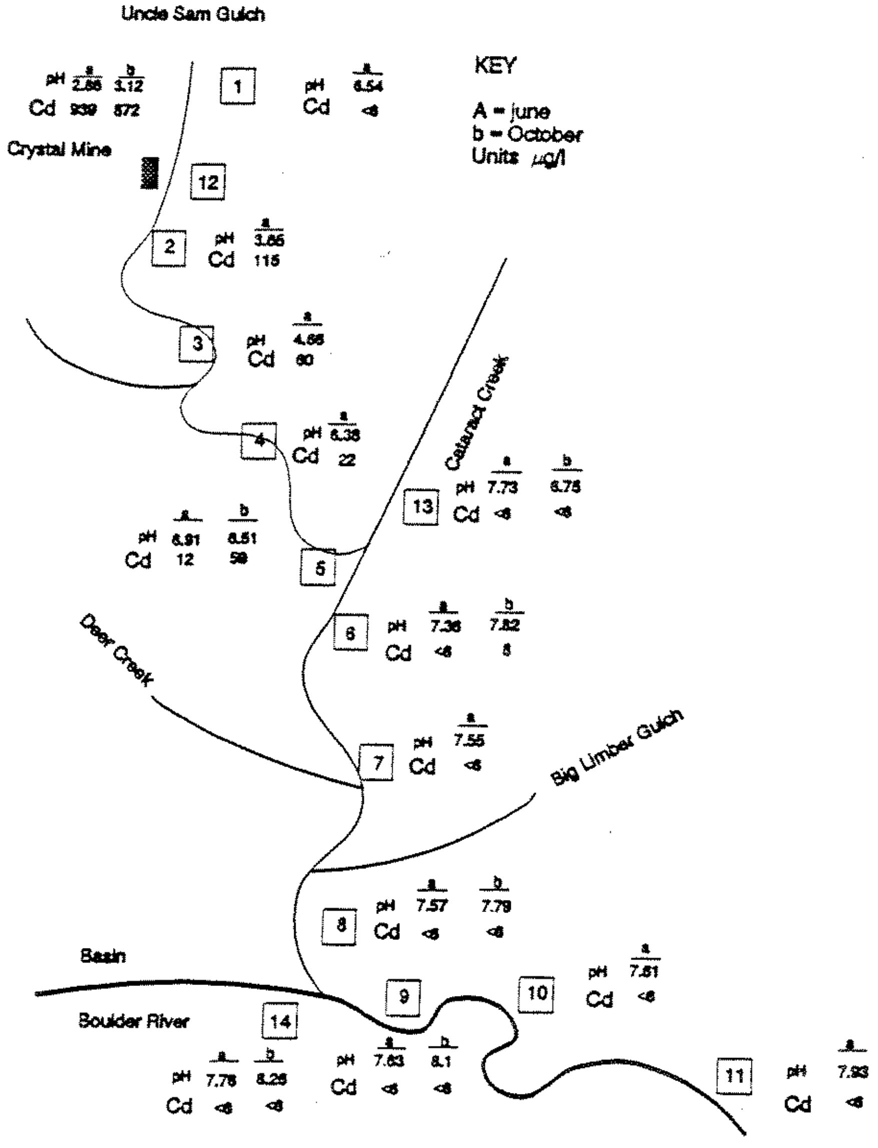
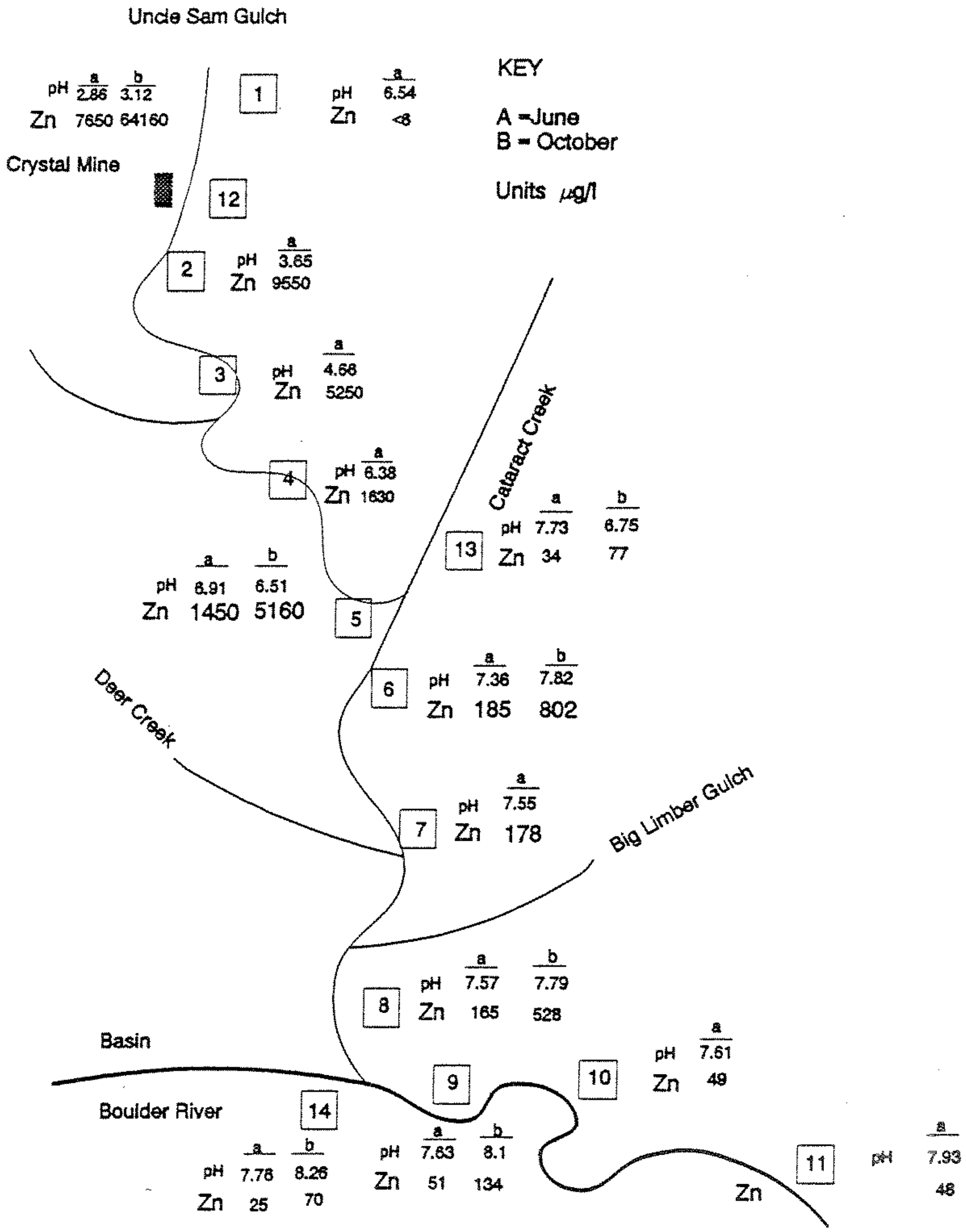


Figure 6. Water Zinc Concentration



RESULTS AND DISCUSSION

Water

June water samples were collected at all sites and October water samples were collected at Sites 5, 6, 8, 9, 12, 13 and 14. Results reported in Table 1 include the site-location number and the parameters measured at each site. These parameters include flowrate, field pH, field temperature, total suspended solids (TSS) and water hardness($CaCO_3$). The data is presented for each site during the June and October sampling periods.

Field-measured water temperatures ranged from a low of 4.7 degrees Celsius at the Crystal Mine adit to a high of 8.8 degrees Celsius at Sites 13 and 14 in June, and 0.1 degree Celsius at Site 8 to 3.3 degrees Celsius at the Crystal Mine adit in October. The pH levels ranged from a low at the Crystal Mine of 2.86 to a high at Site 11 of 7.93 in June. In October the low pH was at the Crystal Mine adit, 3.12 and the high of 8.26 was recorded at Site 14. TSS levels measured in June ranged from highs at Site 2 and Site 12 of 18 mg/l and 43.4 mg/l, respectively, to zero TSS levels recorded at site 1. June water hardness ranged from 206.44 to 7.39, measured as $CaCO_3$, at the Crystal Mine adit and Site 1 respectively. Water hardness measured in October ranged from 185.84 to 48.32 $CaCO_3$, at the Crystal Mine adit and Site 13, respectively.

Table 1. Water Quality Parameters Measured
(Flow, pH, Temperature, T.S.S., Hardness as $CaCO_3$)

Site	Flow(cfs)	pH	Temp(C)	TSS(mg/l)	$CaCO_3$
1	0.5	6.54	6.6	0	7.39
2	1.3	3.65	7.7	18	37.33
3	3.5	4.66	5.7	6.4	30.95
4		6.38	5.7	2.6	24.15
5a	5.1	6.91	5.1	1.8	23.99
5b	1.24	6.51	0.6		68.56
6a	43.2	7.36	8.3	0.8	24.49
6b	8.0	7.82	0.9		52.39
7	43.2	7.55	7.8	0.4	24.08
8a	43.8	7.57	7.9	1.2	25.49
8b	7.79	8.3	0.1		54.12
9a	204.0	7.63	8.1	3.2	29.38
9b	33.62	8.1	2.7		54.45
10	204.0	7.61	8.0	3.2	28.97
11	204.0	7.93	8.4	4.0	29.38
12a	39(gpm)	2.86	4.7	43.4	206.44
12b	30(gpm)	3.12	3.3		185.84
13a	38.6	7.73	8.2	0.8	23.99
13b	6.75	8.03	1.1		48.32
14a	260.0	7.76	8.8	2.6	28.22
14b	25.33	8.26	3.3		51.13
B Limber	0.2	8.07	8.8	0	103.21
DeerCr	0.5	7.92	7.7	0	50.54

a = June samples b = October samples

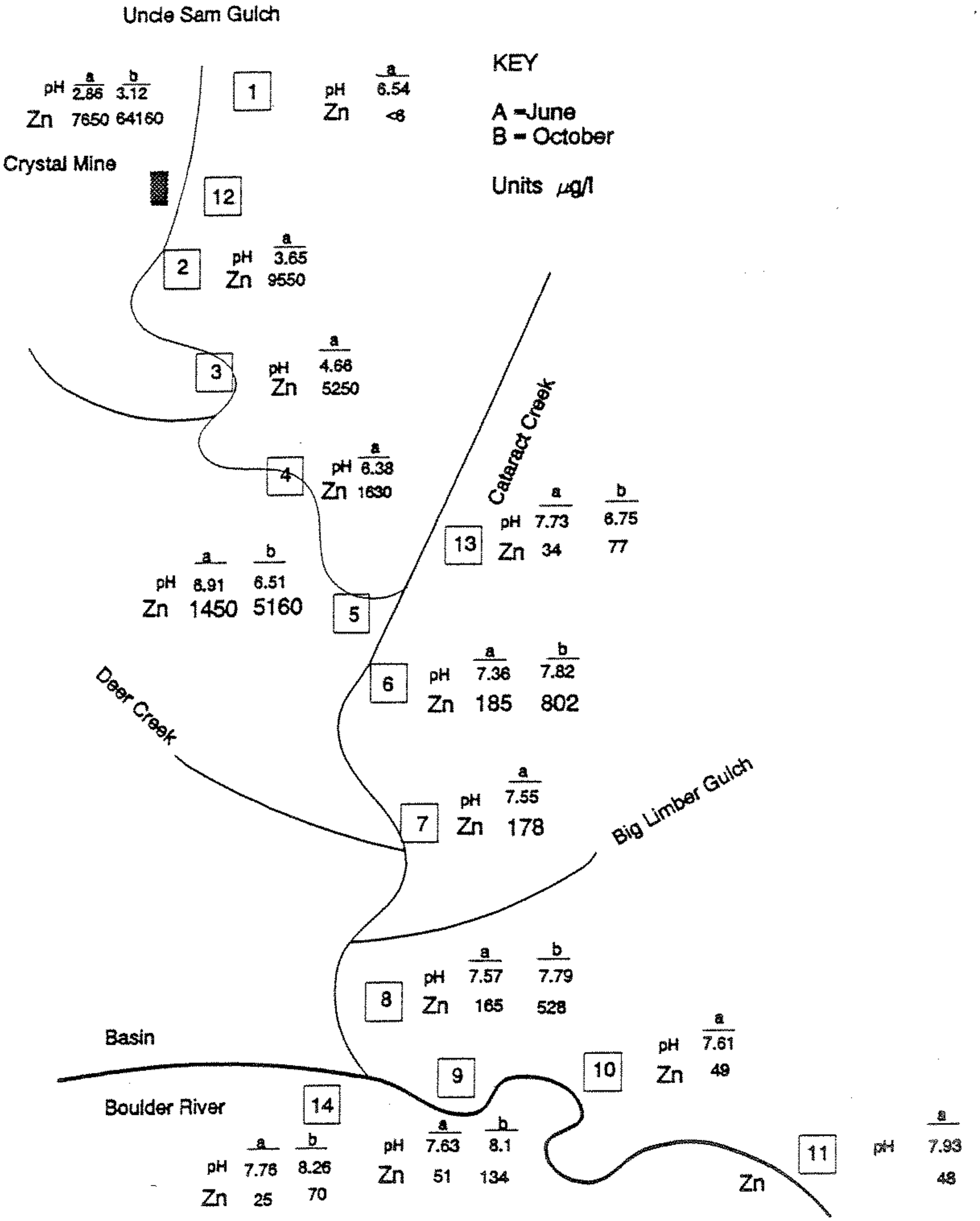
Metal concentrations for water samples are listed in Table 2. These concentrations are the total available concentrations or bio-available concentrations. Concentrations less then 6 μg/l were below the detection limits of the ICP analyzer. Thus, results which are less than 6 μg/l will be treated as no concentrations present.

Table 2. Metal Analysis of Water Samples
(Copper, Cadmium, Zinc in μg/l)

Site	Copper(μg/l)	Cadmium(μg/l)	Zinc(μg/l)
1	9	<6	<6
2	2850	115	9550
3	1520	60	5250
4	235	22	1630
5a	183	12	1450
5b	312	59	5160
6a	43	<6	185
6b	61	8	802
7	25	<6	178
8a	29	<6	165
8b	47	<6	528
9a	17	<6	51
9b	27	<6	134
10	8	<6	49
11	13	<6	48
12a	2520	939	7650
12b	12000	872	64160
13a	14	<6	34
13b	<6	<6	77
14a	<6	<6	25
14b	17	<6	70
Big Limber	<6	<6	<6
Deer Creek	<6	<6	40

a: June samples b: October samples

Figure 6. Water Zinc Concentration



Copper concentrations of 2850 µg/l at Site 2 in June, and 12000 µg/l at the Crystal Mine in October were the highest copper levels measured. Cadmium levels were the highest at the Crystal Mine adit, reaching 939 µg/l during June and 872 µg/l in October. Zinc levels range from 9550 µg/l at Site 2 to 25 µg/l at Site 14 in June and 64,160 µg/l at the mine adit to 70 µg/l at Site 14, in October. Figure 4 illustrates the concentration of copper as it is transported downstream. Figure 5 shows the levels of cadmium; and Figure 6 shows the zinc concentrations as they were transported downstream. Bureau of Mines Analytical Lab data sheets, which include complete analysis of all metals, are included in Appendix.

Water quality results for Uncle Sam Gulch, Cataract Creek and Boulder River show levels above "The Gold Book" (E.P.A., 4) standards at various sites. Gold Book concentrations are Montana and Federal standards. Table 3 shows the E.P.A. water quality criteria for the protection of freshwater organisms, acute and chronic levels.

Environmental Protection Agency acute water quality criteria for copper and cadmium are corrected for water hardness, maximums to be exceeded only once every 3 years (E.P.A., 4). Zinc acute concentration maximum is 47 µg/l, 1-hour average, to be exceeded only once every 3 years (E.P.A., 4). E.P.A. chronic water quality criteria standards for copper, cadmium and zinc are corrected for water hardness as $CaCO_3$. Correction for chronic concentrations as referenced in E.P.A. Documents 2 and 3, and Forba, 1992, are:

$e^{(0.94[\ln CaCO3]-1.23)}$ for copper

$e^{(0.83[\ln CaCO3]+1.95)}$ for zinc

$e^{(0.7852[\ln CaCO3]-3.49)}$ for cadmium

Chronic criteria standards are the maximum 4-day average to be exceeded only once every 3 years (E.P.A., 4).

Copper E.P.A. acute water quality criteria standards were exceeded at all sites in June and October. Cadmium exceeds the acute criteria on Uncle Sam Gulch only, during June and October. Acute criteria for zinc were surpassed at all sites, except background Sites 13 and 14 in June. October samples were all above zinc standards.

Copper concentrations exceed E.P.A. chronic criteria at all locations, except background Sites 13 and 14, during June sampling. October copper concentrations were above E.P.A. criteria at Sites 5, 6, 8, 9 and 14. Site 13 was the only location under criteria limits for copper. Cadmium was above the E.P.A. chronic criteria at all sites in June and October, when concentrations were detectable. Zinc surpasses E.P.A. chronic criteria concentrations along Uncle Sam Gulch and Cataract Creek. Zinc levels at the background Sites, 1, 13, 14, and on the Boulder River were below the corrected limits.

Table 3. WATER QUALITY: Copper, Cadmium and Zinc Measured Concentrations and Regulated Metal Levels

COPPER: June and October

Site	Metal Level (μg/l)	E.P.A. Acute Criteria (μg/l)	E.P.A. Chronic Criteria (μg/l)
1	9	1.523	1.580
2	2850	7.004	6.305
3	1520	5.871	5.372
4	235	4.647	4.345
5a 5b	183 312	4.618 12.42	4.321 10.60
6a 6b	43 61	4.709 9.640	4.398 8.422
7	25	4.634	4.335
8a 8b	29 47	4.890 9.939	4.551 8.659
9a 9b	17 27	5.590 9.997	5.138 8.704
10	8	5.516	5.077
11	13	5.590	5.138
12a 12b	2520 12000	35.09 31.78	27.185 24.849
13a 13b	<6 <6	4.618 8.932	4.321 7.860
14a 14b	<6 17	5.382 9.421	4.964 8.249

a: June sample b: October sample

Table 3. cont
CADMIUM: June and October

Site	Metal Level (μg/l)	E.P.A. Acute Criteria (μg/l)	E.P.A. Chronic Criteria (μg/l)
1	<6	0.208	0.310
2	115	1.291	1.107
3	56	1.045	0.956
4	22	0.789	0.7866
5a	12	0.784	0.7825
5b	59	2.562	1.7850
6a	<6	0.802	1.445
6b	8	1.892	1.445
7	<6	0.787	
8a	<6	0.839	0.820
8b	<6	1.962	1.492
9a	<6	0.985	0.917
9b	<6	1.976	1.489
10	<6	0.970	
11	<6	0.955	
12a	939	8.884	4.240
12b	872	7.851	3.905
13a	<6	0.784	0.782
13b	<6	1.727	1.356
14a	<6	0.941	0.889
14b	<6	1.840	1.416

a: June samples b: October samples

Table 3. cont
ZINC: June and October

Site	Metal Level (μg/l)	E.P.A. Acute Criteria (μg/l)	E.P.A. Chronic Criteria (μg/l)
1	<6	47	36.97
2	9550	47	141.80
3	5250	47	121.37
4	1630	47	98.78
5a 5b	1450 5160	47	98.24 168.59
6a 6b	185 802	47	99.93 134.86
7	178	47	98.54
8a 8b	165 528	47	103.31 138.56
9a 9b	51 134	47	116.24 139.25
10	49	47	114.89
11	48	47	116.24
12a 12b	7650 64160	47	586.34 385.74
13a 13b	34 77	47	98.24 126.10
14a 14b	25 70	47	112.42 132.16

a: June samples b: October samples

June concentrations of copper, cadmium and zinc decrease with distance downstream from the Crystal Mine. Table 1 shows and Figure 7 compares the log of metal concentration with the water pH for copper, cadmium and zinc at Sites 2 through 5. This relationship suggests that pH is a major factor controlling the concentration of metals dissolved in the water. Regression coefficients of .9883 for copper, .9892 for cadmium and .9916 for zinc illustrates that results approach theoretical saturation levels for copper, cadmium and zinc. Since results do not match theoretical saturation, this system does not reach equilibrium and must be influenced by other factors, also.

Theoretical saturation equation:(Handbook of Physics and Chemistry, 1987).

$$[Cu^{+2}][OH^{-}]^{2} = K_{sp} \quad \text{yields}$$

$$\log[Cu^{+2}] = 8 - 2pH$$

Figure 8 compares metal concentrations to pH levels for copper and zinc from Site 2 downstream to Site 11. Figure 8 implies the pH level, also, governs metal concentrations between Sites 2 and 11. Regression coefficients for copper over this area is .9099 and .8310 for zinc. These results do not match theoretical values as well as the results from Sites 2 through 5 due to the influence from the metal concentrations which enter into the streams from upstream of the confluences. (Uncle Sam Gulch entering Cataract Creek and Cataract Creek entering the Boulder River).

Sites 7 and 10 had lower copper concentrations, 25 and 8 μg/l, than Sites 8 and 11, 29 μg/l and 13 μg/l, respectively. Sites 8 and 11 are located downstream of Sites 7 and 10. The higher downstream concentrations could result from a non-point source of dissolved copper from a tailings pile along Cataract Creek located upstream of Site 8. Higher copper levels at Site 8 could also be a result of copper redissolving from suspended material or bottom material, since Site 8 also contains high sediment copper concentrations. The chemical oxygen demand was also higher at Site 8 then at Site 7. Similar situations could be causing higher copper levels at Site 11 compared to Site 10. A quartz mine is located above the Boulder River between Sites 10 and 11. Sampling error could also be the cause of these fluctuations in the data.

June copper and zinc concentrations for Sites 5 through 9 are presented in Figure 9. These concentrations are compared to the results for October samples from the same sites, Figure 10. Figure 9 illustrates a theoretical relationship between the pH and the metal concentration. Regression coefficients for copper reduction is .9790 and .9165 for zinc. October samples illustrated a similar relationship in Figure 10, with regression coefficients of .9726 and .8833 for copper and zinc, respectively. Copper and zinc concentration equations for each sampling period are different. The equation for June copper concentrations is $Y = -1.341X + 3.722$. October copper concentrations result in a line equal to $Y = -0.634X -1.17$. Zinc concentrations showed line equations of $Y = -1.773X + 7.594$ in June and $Y = -0.867X + 1.60$ in October. Differences in slopes of the lines may indicate higher October concentrations than June.

pH levels do not decrease or increase proportionally, site to site from June to October. Overall, copper, cadmium and zinc concentrations in Uncle Sam Gulch, Cataract Creek and the Boulder River appear to be a function of pH levels.

Figure 9. Metal Concentration(Cu,Zn)
Controlled by pH, sites 5,6,8,9, June

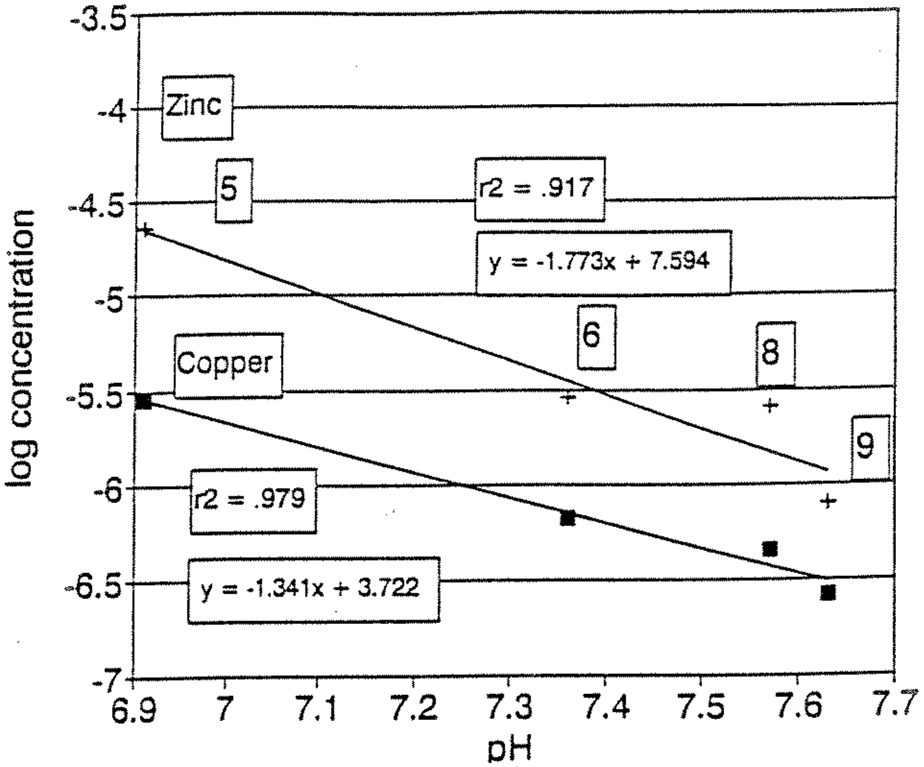


Figure 10. Metal Concentration(Cu,Zn)
Controlled by pH, Sites 5-9, October

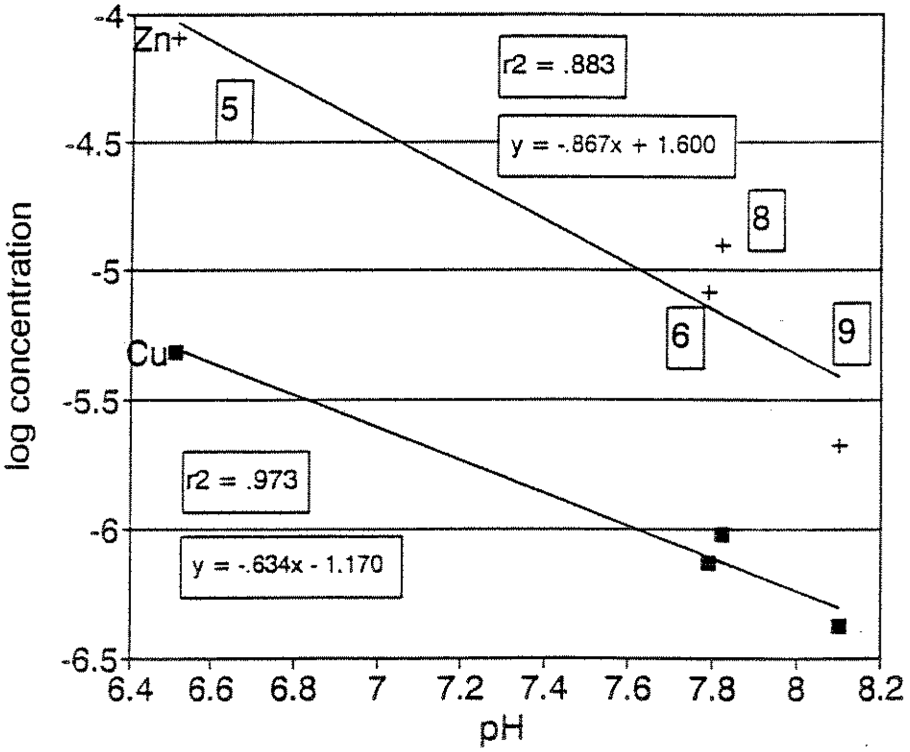


Figure 7. Metal Concentration
Controlled by pH, Sites 2-5, June

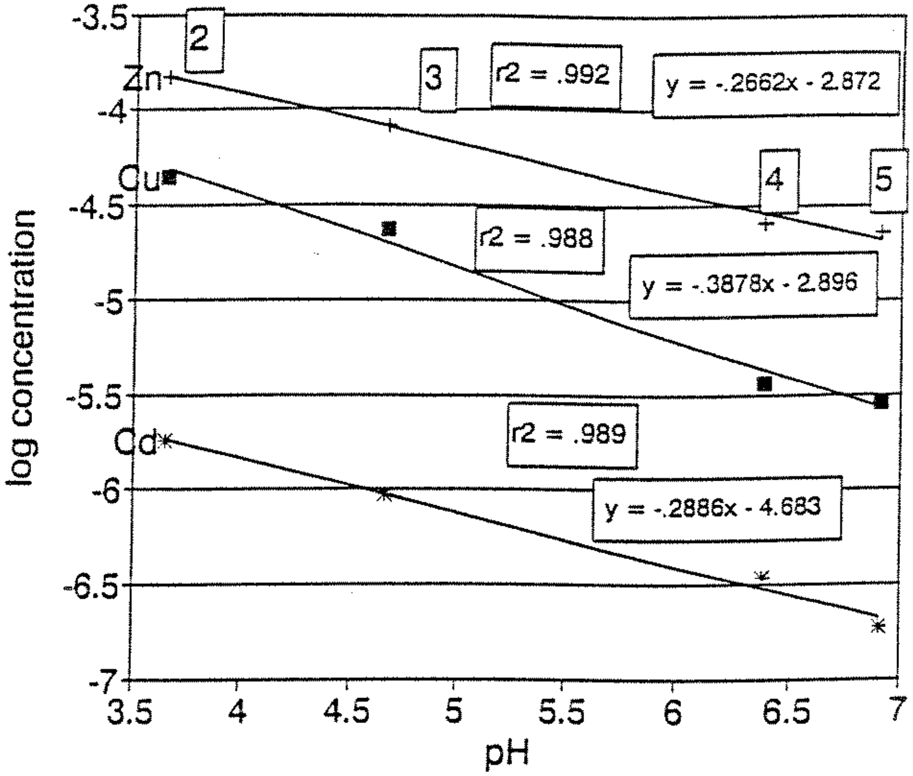
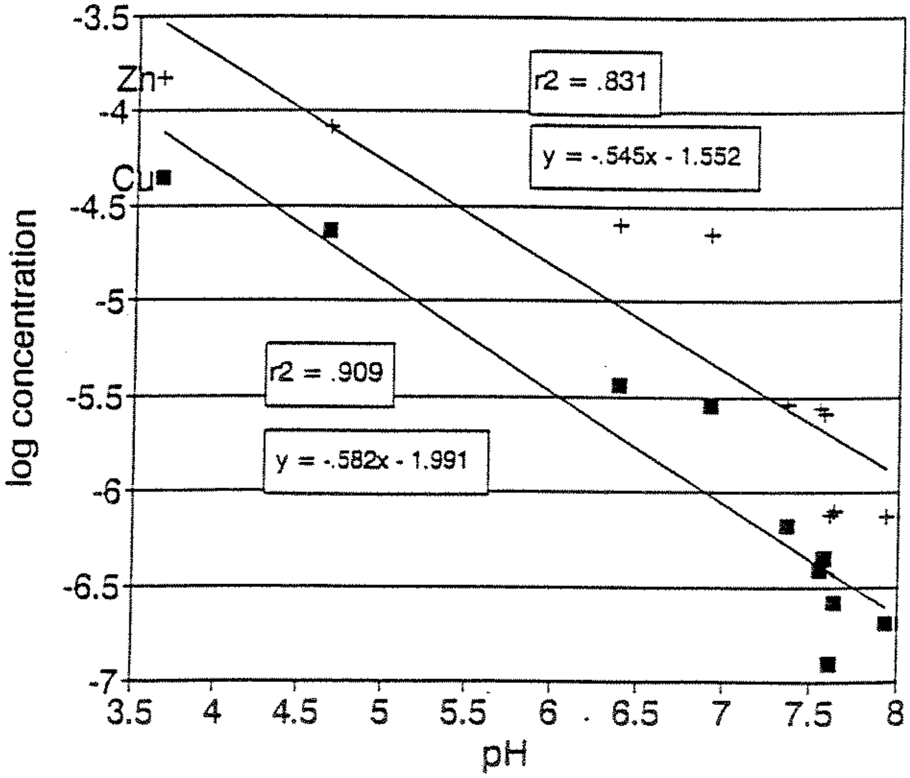


Figure 8. Metal Concentration
Controlled by pH, Sites 2-11, June



Ratios of the June to October copper, cadmium and zinc concentrations are not equivalent at Site 12, the adit, and Site 5. June copper, Site 12 to Site 5 ratio is 0.587, cadmium is 0.203 and zinc is 0.281. October, Site 12 to Site 5 ratios for copper, cadmium and zinc are 0.21, 1.077 and 0.119, respectively. These unequal ratios at Site 5 and Site 12 indicate the concentrations of copper, cadmium and zinc are not equal throughout the year and that the influence of snow runoff and spring rains may be an important influence in metal concentrations entering Uncle Sam Gulch.

Water quality data indicate that Uncle Sam Gulch, Cataract Creek and the Boulder River water quality is adversely effected by the Crystal Mine AMD. Standards were exceeded at all sites during sampling, while, background sites were below standards, except Site 14 which exceeded E.P.A. chronic criteria for copper in the October sampling.

Mass Loading

June and October mass loading rates of copper, cadmium and zinc are listed in Table 4. Figures 11, 12 and 13 map mass loading at each site for copper, cadmium and zinc, respectively. Mass load of metal concentration was calculated using flowrate, in cubic feet per second, water metal concentration and a conversion factor, 5.39 (lb/day/ cfs-mg/l) (Thomann, 1987).

$$\text{Mass load} = (\text{cfs})(\text{concentration mg/l})(5.39)$$

Table 4. MASS LOADING OF COPPER, CADMIUM, ZINC (lb/day)

Site	COPPER June October (lb/day)	CADMIUM June October (lb/day)	ZINC June October (lb/day)
1	0.024	0	0
2	19.97	1.35	71.31
3	28.67	1.132	99.04
5	5.03 2.09	0.33 0.394	31.45 34.49
6	10.13 2.63	0 0.345	43.58 34.58
7	5.89	0	41.93
8	6.92 2.1	0	39.4 23.62
9	18.69 4.89	0	56.08 24.28
10	8.8	0	53.88
11	14.29	0	52.78
12	1.448 4.38	0.5396 0.319	4.4 23.44
13	2.91 0	0	7.07 2.80
14	0 2.32	0	27.49 9.56

Figure 12. Cadmium Mass Loading Rate

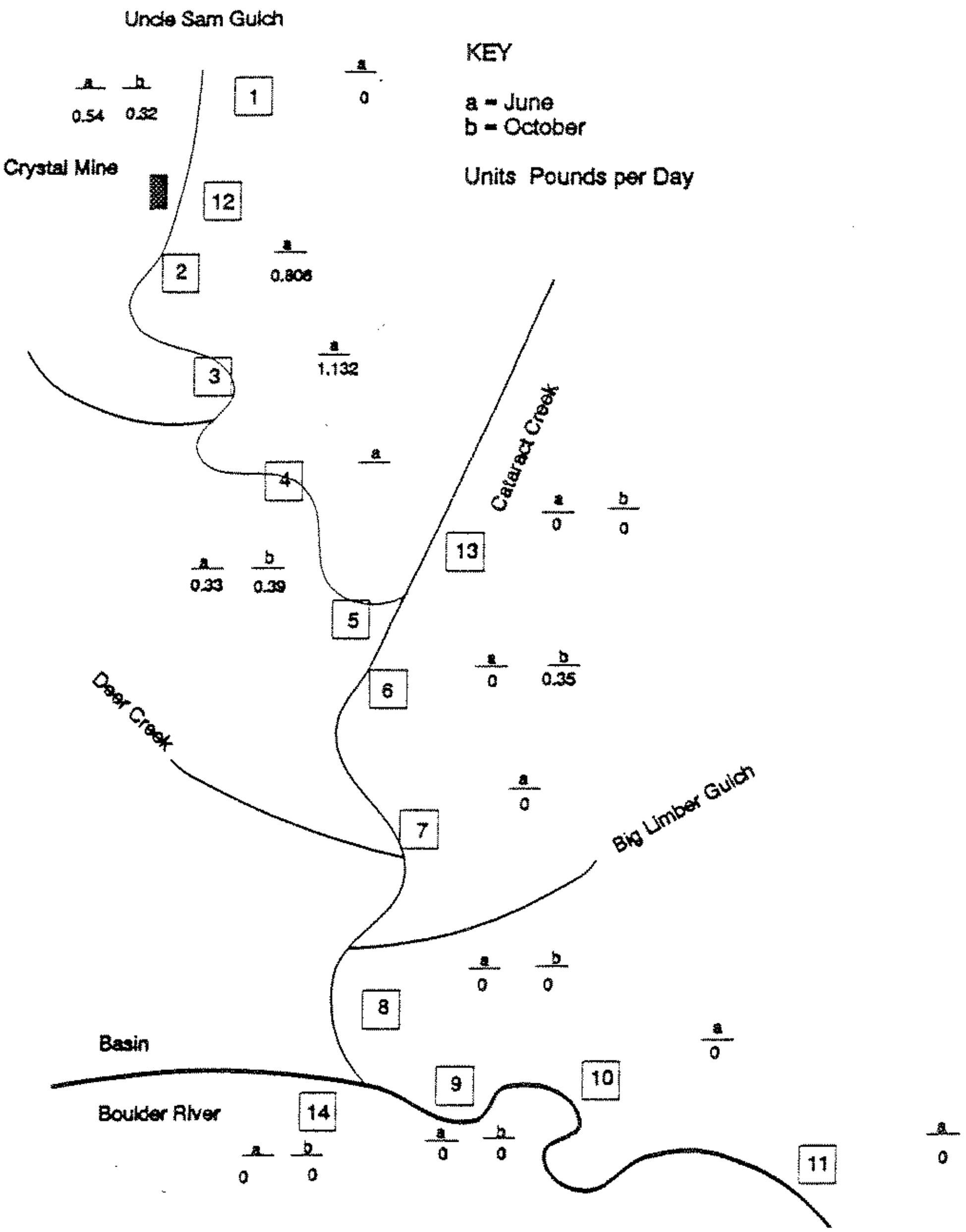


Figure 11. Copper Mass Loading Rate

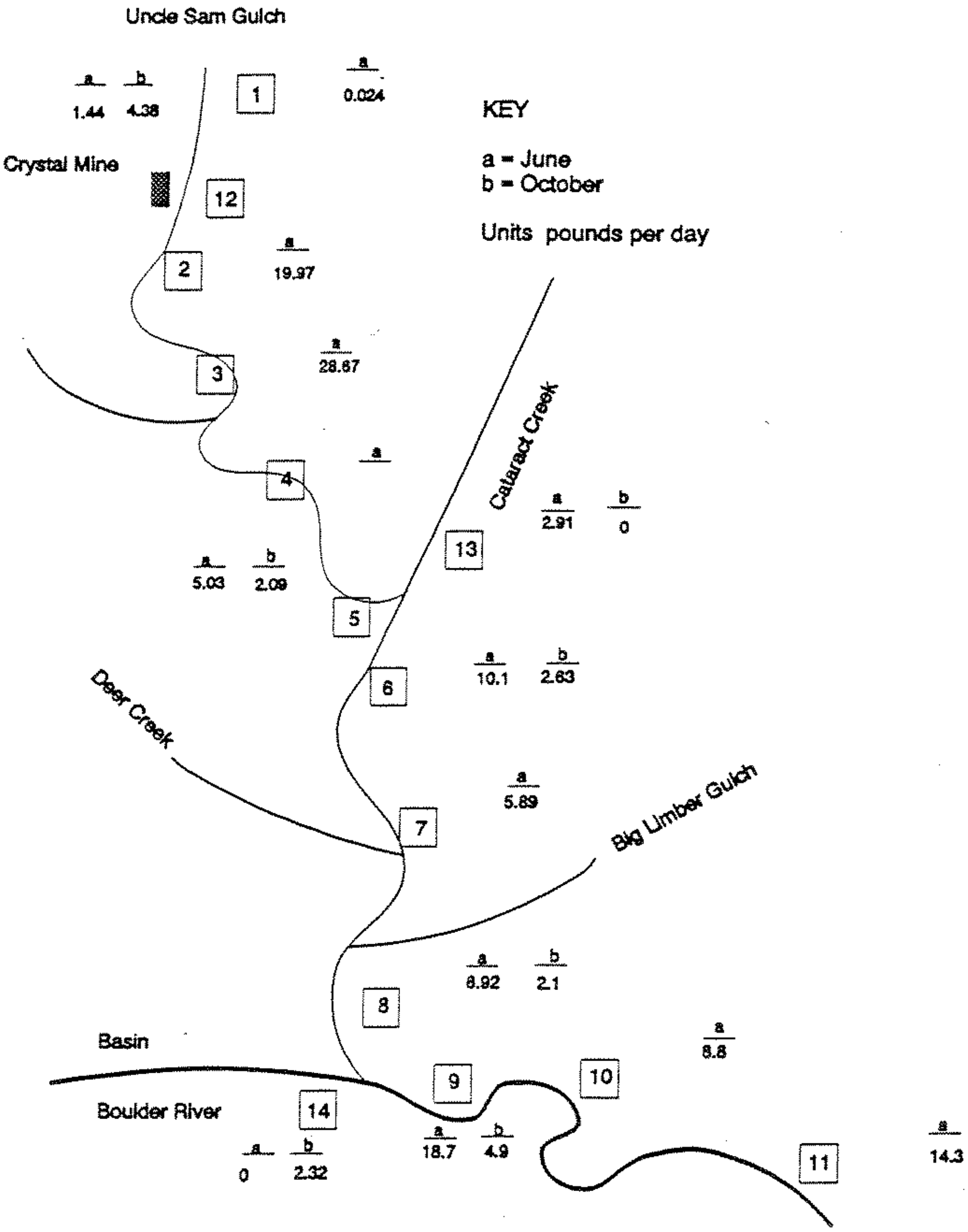
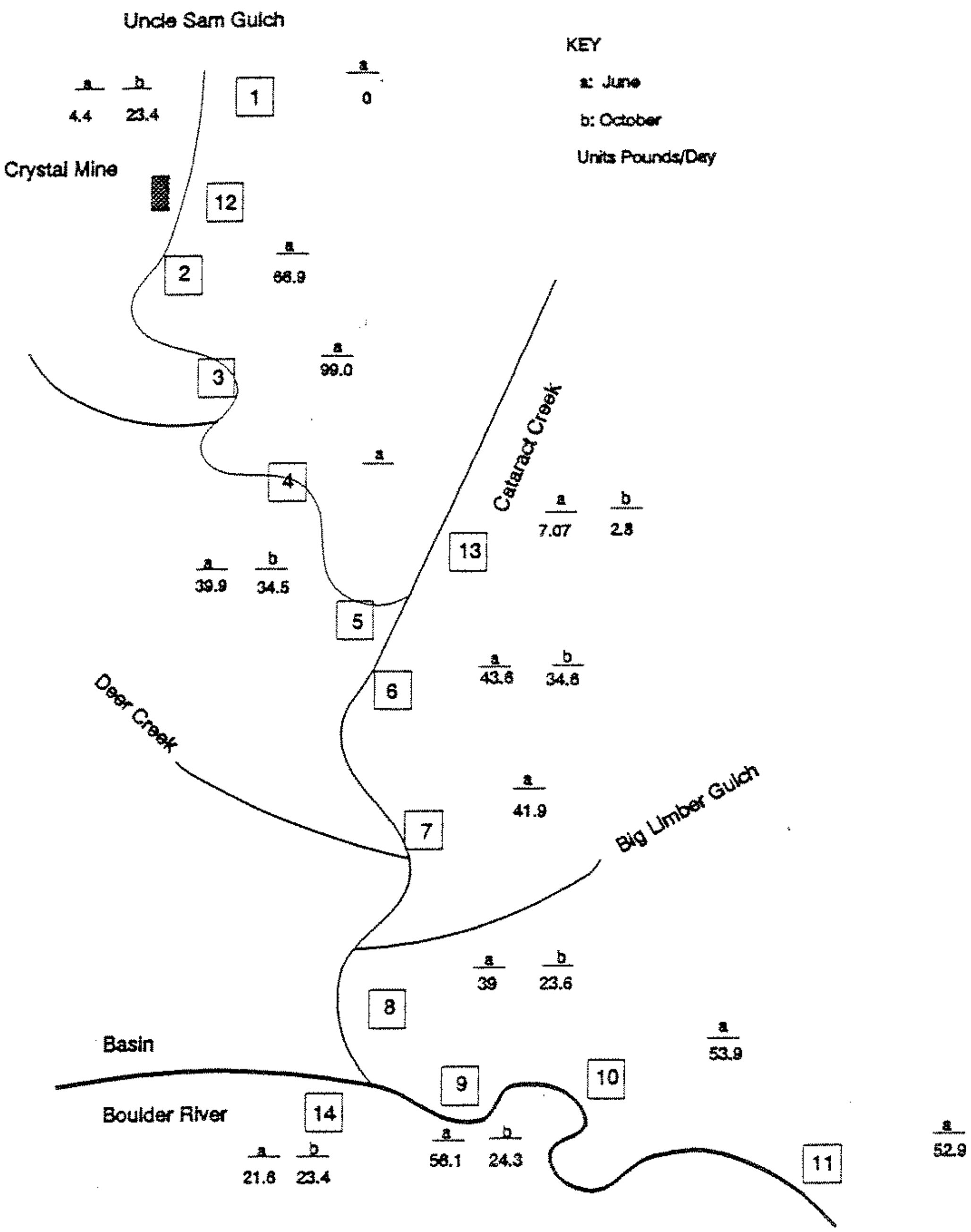


Figure 13. Zinc Mass Loading Rate



Mass loading of copper is shown on Figure 11. Mass balances at each site, mass entering and mass leaving were used to determine deposition. Summations of copper mass loading around the entire study area reveals a deposition or precipitation rate of 22.9 pounds per day (lb/day) during June and 1.81 lb/day in October. Mass loading imbalances occur at Sites 3, 6, 8, 9 and 11. At each of these sites mass loading of copper leaving the site was higher than the summation of the mass loading above the site.

A copper mass load difference of 9 lb/day was observed between Sites 2 and 3. This difference may be caused by the reabsorption of copper into the water from suspended material and sediments found in Uncle Sam Gulch. Suspended material and sediments enter Uncle Sam Gulch from the large tailings piles and unreclaimed areas above the mine. The pH is 4.66 at Site 3 compared to 3.65 at Site 2. Mass loading at Site 2 may not represent the reabsorption of copper from suspended material due to the short time that the material is influenced by the pH at Site 2. Copper hydroxide precipitates completely at a pH of 7.2 (Kelly, 1988). If this is true, then the reabsorption of the copper from suspended material is possible because the pH above Site 2 at Site 1 is 6.54. There is also a correlation between the sediment concentrations and the mass loading. Measured sediment concentrations of copper at Site 3 are lower then those measured at Site 2, 244.5 mg/kg and 298 mg/kg, respectively. This difference could be caused by a dissolving copper, indicated by the increased mass loading at Site 3. There could possibly be a non-point source of copper in this region, either from a spring or ground water.

The same type of events may be occurring at Sites 6 and 9, the influence of the upstream flows and a lower pH may affect the mass loading downstream of the confluences by redissolving a portion of the copper in the suspended material. The same mass loading differences occurred in June and October at Sites 6 and 9. June copper mass loading increase was 2 lb/day at Site 6 and 12 lb/day at Site 9. October copper mass loading increases at Sites 6 and 9 were 0.6 lb/day and 0.5 lb/day, respectively. Site 6 has two large tailings piles, one located upstream at the Morning Glory Mine and one adjacent to the sampling site. These tailings piles could be supplying suspended material into Cataract Creek, and the influence of the lower pH between Site 13 and Site 6 could redissolve some copper thus increasing the mass loading at Site 6. The difference in mass loading at Site 9 could also be attributed to the fact that copper concentration at Site 14 was below the detection limits in June, thus giving the impression that no copper is entering the system from upstream, when 0 μg/l to 6 μg/l could be present. A copper concentration of 17 μg/l in October suggests that there is some background copper present.

Site 8 has a higher concentration of dissolved copper, T.S.S. and a larger flowrate than Site 7 which may account for the difference in the mass loading. The difference in concentration could result from a non-point source, tailing piles between sites or a higher chemical oxygen demand at Site 8. Higher mass loading at Site 11 compared to Site 10 may be caused by the quartz mine between the sites or from another non-point source of copper.

Cadmium mass loading levels were measured only on Uncle Sam Gulch in June due to the low cadmium levels downstream of Uncle Sam Gulch. Cadmium mass loading values are displayed in Figure 12. Cadmium deposition during June sampling was 0.5 lb/day. Cadmium exhibits the same results as copper and zinc between Sites 2 and 3. Kelly, 1988, states that the cadmium hydroxides will precipitate completely at pH of 9.7. Again, the pH at Sites 2 and 3 is far below this level which may be causing the dissolving of cadmium from suspended material entering the stream system from the Crystal Mine tailing piles above Site 2. There is no mass loading data for October because of low cadmium water concentrations.

Zinc mass loading values are reported in Figure 13. Zinc depositions of 47.2 lb/day during June, and 11.5 lb/day during October. A mass balance at each of the sites illustrates lower loads of zinc leaving than entering each site, except for Site 3. The difference at Site 3 could be the same for zinc as copper. The precipitation limit for zinc hydroxides is at a pH of 8.4 (Kelly, 1988). The suspended material could be acting as a sink for the zinc to redissolve between Sites 2 and 3, causing an increase in mass loading.

Sediment

Copper, cadmium and zinc concentrations accumulated in the sediment are presented in Table 5.

Table 5. COPPER, CADMIUM, ZINC CONCENTRATIONS IN SEDIMENTS
Concentration (mg of metal/kg of dry sediment)
Mean concentration present

Site	COPPER (mg/kg)	CADMIUM (mg/kg)	ZINC (mg/kg)
1	65.67	2.05	86.5
2	298	3.99	404.9
3	244.5	3.02	348.2
5	2700	38.77	2955
6a	1080	20.85	1496
6b	4020	70.54	8370
7	512.5	12.09	1023
8a	843.2	27.93	1873
8b	1374	36.61	4107
9a	170.2	4.74	520.2
9b	698.7	22.11	2922
10	133.3	3.77	426.0
11	342.8	12.13	900.8
13a	139.0	4.13	435.7
13b	233.0	6.79	1131
14a	89.74	2.14	283.1
14b	576.13	6.09	1291

a: June samples   b: October samples

Figure 14 displays the concentrations of copper at each site sampled during June and October. June copper levels ranged from a high of 2700 mg/kg at site 5 to a low of 65.67 mg/kg at Site 1. June and October cadmium sediment levels are shown with respective sites in Figure 15, with concentrations ranging from a high of 38.77 mg/kg at Site 5 to a low of 2.14 mg/kg at Site 14. Figure 16 displays June and October sediment

Figure 15. Sediment Cadmium Conc.

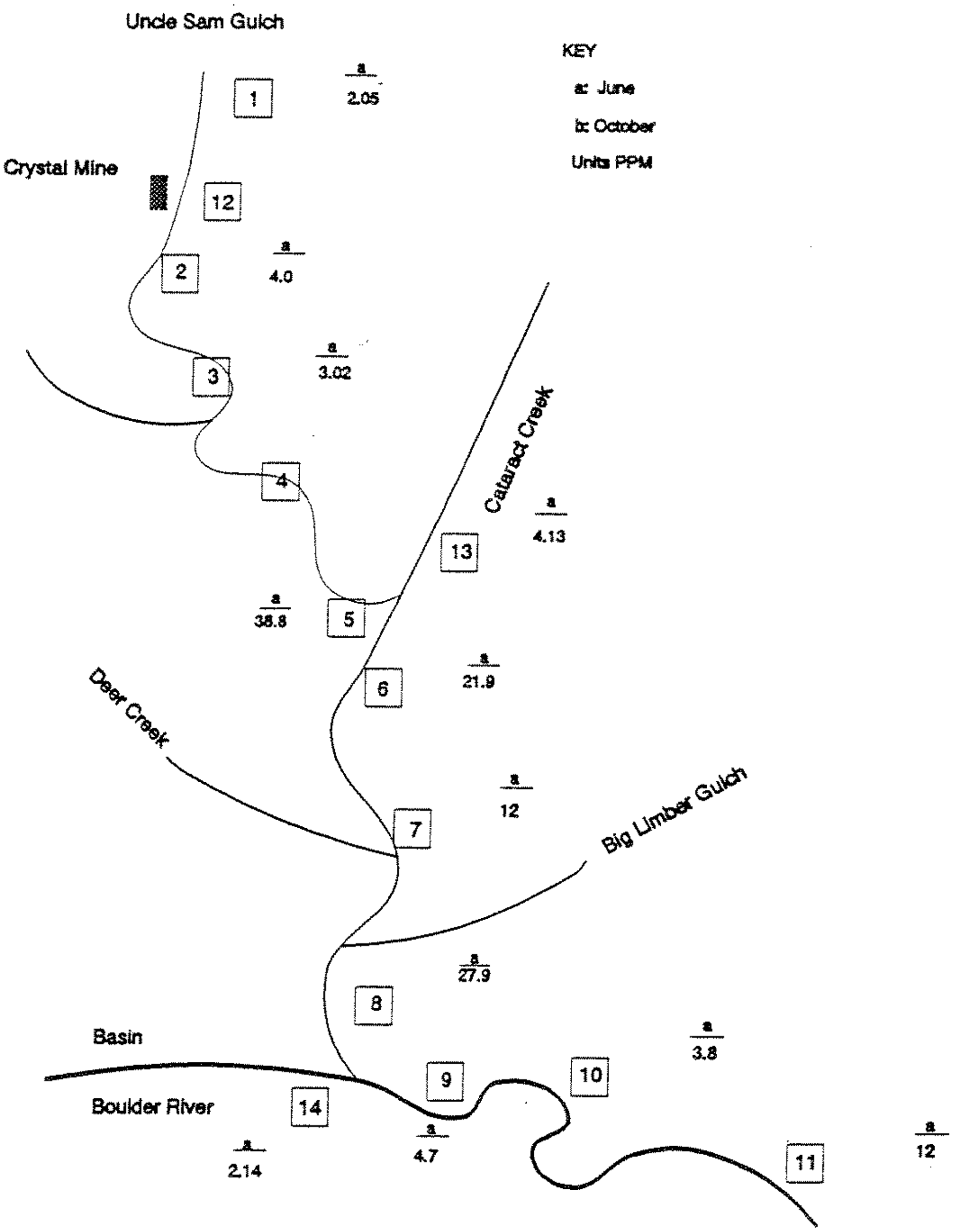


Figure 14. Sediment Copper Conc.

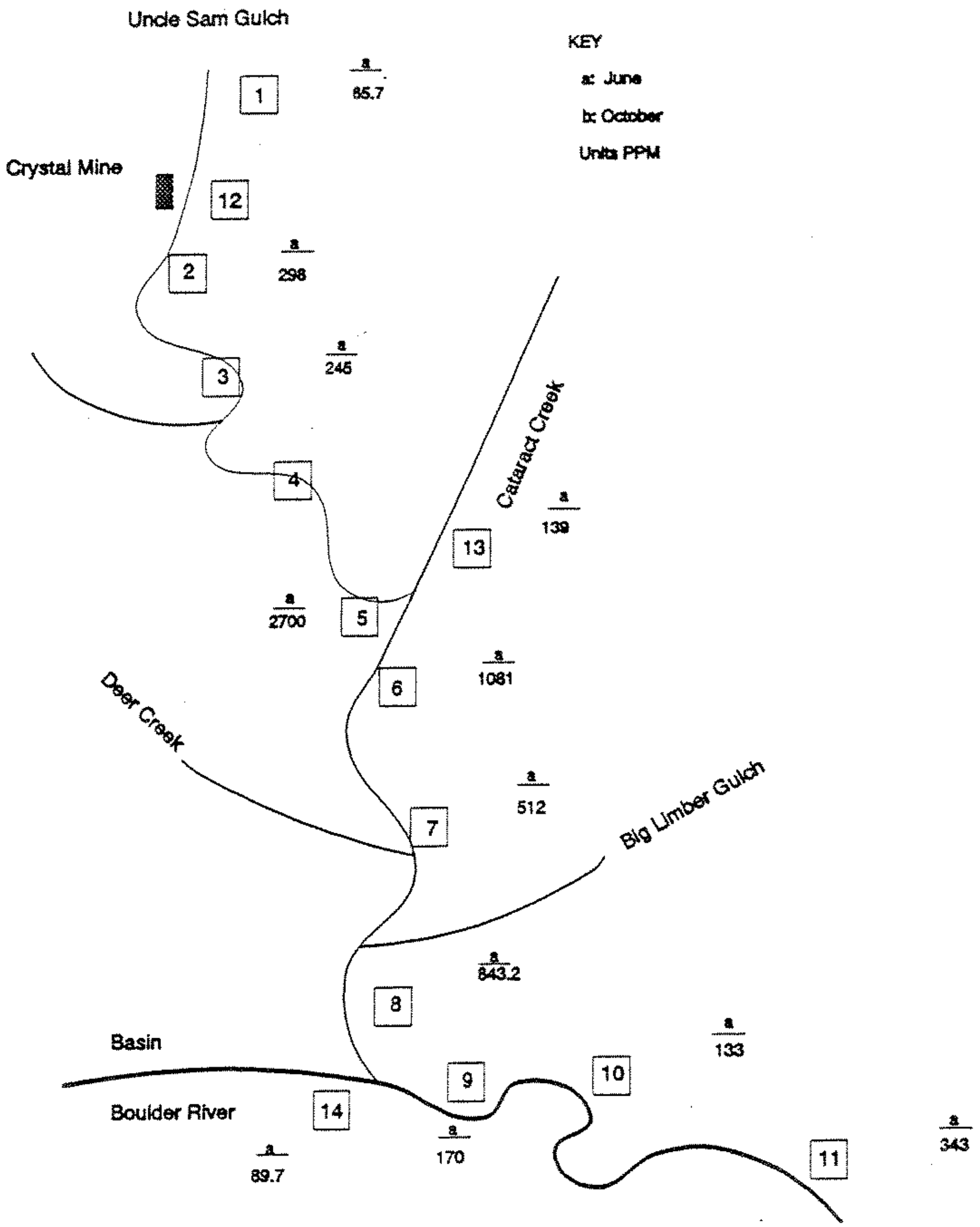
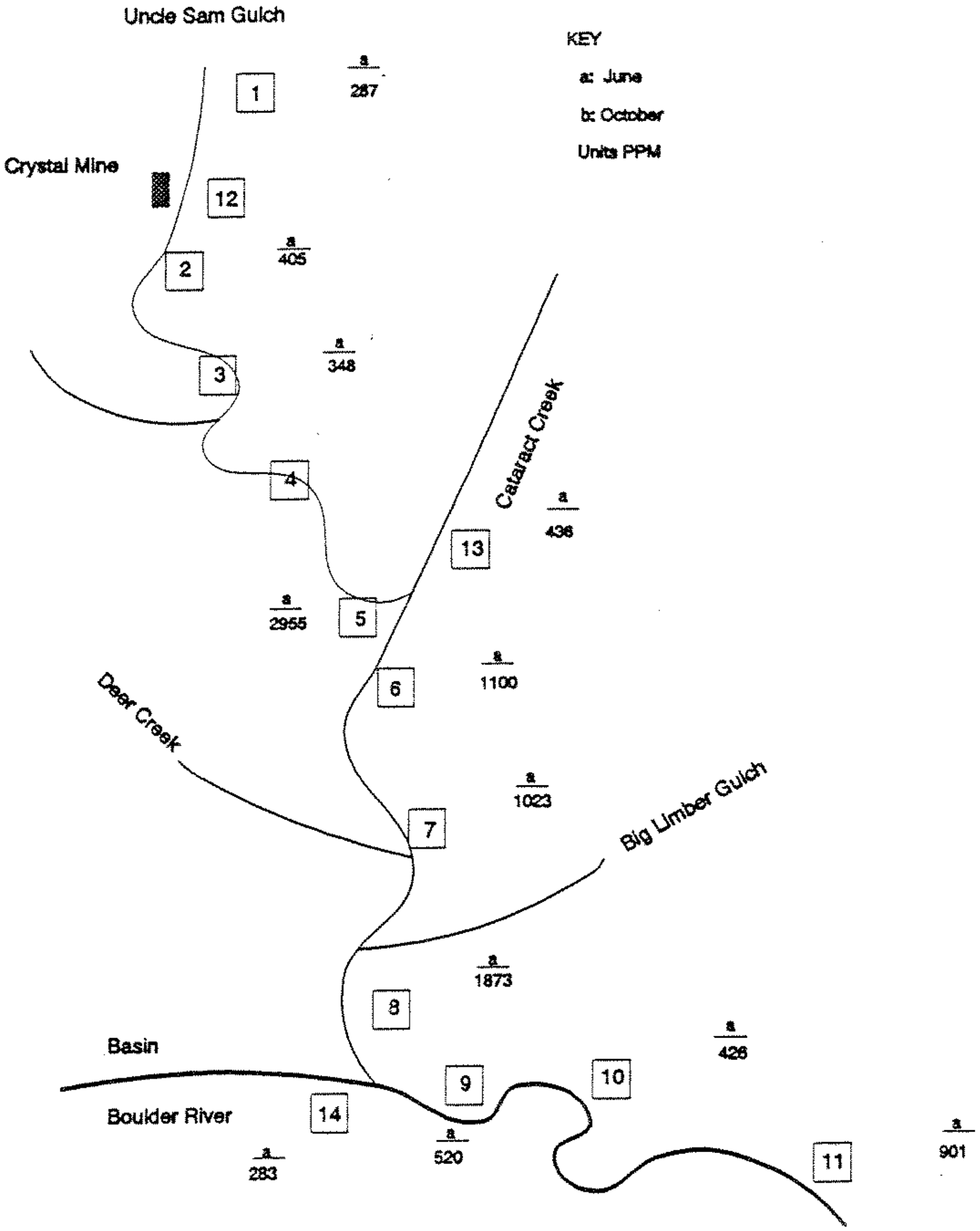


Figure 16. Sediment Zinc Conc.



zinc concentrations. Zinc concentrations range from a high of mg/kg at Site 5 to a low of 283.1 mg/kg at Site 14 October.

Sediment metal concentrations were highest in Uncle Sam Gulch. Site 5 had the highest copper, cadmium and zinc concentration with levels of 2700 mg/kg, 38.77 mg/kg and 2955 mg/kg, respectively. High sediment concentrations at Site 5 correspond to lower water concentrations at this site. Precipitation of copper, cadmium and zinc into the sediment appear to correlate with decreasing metal concentrations in the water.

June sediment concentrations at Sites 3, 7, 8, 10 and 11 exhibit highs and lows corresponding to mass loading rates. The low concentration at Site 3 may result from redissolving of copper, cadmium and zinc into solution or a hydraulic effect from scouring. Higher levels at Sites 8 and 11 probably result from erosion of material into Cataract Creek and the Boulder River from mine tailing piles and a quartz mine located above these sites.

Sediment metal concentrations were significantly higher at all sites sampled in October compared to June concentrations. At Site 6 copper concentrations were 3.7 times higher in October compared to June, and cadmium, and zinc levels were 3.4 and 5.4 times higher. Site 8 metal concentrations increased by 1.6, 1.3 and 2.2 times, respectively, for copper, cadmium and zinc. Copper, cadmium and zinc concentrations increased at Site 9 by 4.1, 4.7 and 5.6. Background Sites 13 and 14 also exhibit sediment concentrations increases of 1.7 and 6.4, respectively, for copper, 3.17 and 2.8 for cadmium and 2.6 and 4.6 for zinc. Metal concentrations in October probably result from sampling at low water levels compared to high water levels during June sampling. October samples were collected during low water in the region of the streams which are affected by the precipitation of metals throughout the entire year. Higher October metal precipation could also be associated with lower hydraulic effects during low water levels. Brown, 1977, reported similar effects of increased sediment concentrations during low water.

High levels of sediment concentrations not only affect the physical aspect of the streams by causing hardening of sediments, but also contaminate sediment for burrowing invertebrates. It appears this impact, on the stream beds, is occurring along Uncle Sam Gulch and Cataract Creek. Sediment levels at Site 8, compared to Site 9, are 5 times higher in copper concentrations, 6 times higher for cadmium and 3.6 times higher for zinc. The higher metal concentrations, along with hardening of the stream bottoms, has either eliminated or severely limited habitat for aquatic invertebrates. Kelly, 1988, stated that the observed numbers of invertebrate species was greater in AMD affected waters where the pH levels were lower. Higher precipitation rates of Fe(III) at higher pH levels result in the loss of invertebrate habitat. This same scenario appears to be occurring in Uncle Sam Gulch and Cataract Creek in regard to sediments.

Invertebrates

Table 6 shows invertebrate populations at sites where collections were made during June and October. This table list the Family and Genus of the invertebrates collected. Some _species_ were identified when possible. Invertebrates were collected to test population biomass, diversity and accumulation of copper, cadmium and zinc into the food-chain. Table 7 lists the total number of invertebrates collected and the diversity index at each site.

Table 7. INVERTEBRATE DIVERSITY
Per Site: Diversity Index
a: June samples b: October samples

Site	# Taxa	Diversity Index
1	44	5.476
2	0	0
3	0	0
5	0	0
6a	185	4.852
6b	4	4.983
7	379	4.654
8	68	6.548
9a	580	5.790
9b	141	6.979
10	377	6.599
11	284	8.152
13a	301	5.648
13b	203	5.200
14a	576	7.608
14b	537	5.495

TABLE. 6 AQUATIC INVERTEBRATE INDENTIFICATION

SITE	Family	Genus	Number
1	Ephemeroptera	Paraleptophlebia sp.	9
	Plecoptera	Chloroperlidae	4
		Megarcys sp	2
		Podmosta sp	2
		Yoroperla brevis	5
	Trichoptera	Ecclisomyia sp	13
		Lepidostoma sp	1
	Diptera	Hexatoma sp	5
		Prosimulium sp	2
	Pelecypoda	Pisisdium sp	2
7	Ephemeroptera	Ameletus sp	37
		Baetis biccaudatus	201
		Cinygmuia sp	6
		Epeorus albertae	3
	Plecoptera	Chloroperlidae	24
		Cultus sp	10
		Ecclisomyia sp	1
		Isoperla quinquepunctata	32
		Kogotus sp	30
		Megarcys sp	1
		Taenionema sp	25
	Diptera	Hexatoma sp	8
		Prosimulium sp	1

Site	Genus		Number
6	Ephemeroptera	Ameletus sp	58
		Baetis biccaudatus	63
		Ci nygmula sp	14
		Drunella doddsi	7
		Heptagenia sp	4
	Plecoptera	Chloroperlidae	7
		Hesperoperla pacificum	10
		Kogotus sp	3
		Megarcys sp	8
	Trichoptera	Rhyacophila	8
	Diptera	Hexatoma sp	2
8	Ephemeroptera	Ameletus sp	1
		Baetis bicaudatus	22
		Drunella grandis	1
		Epeorus albertae	1
		Epeorus grandis	2
		Ephemerella infrequens	2
	Plecoptera	Chloroperlidae	4
		Doroneuria theodora	1
		Kogotus sp	7
		Podmosta sp	5
	Trichoptera	Rhyacophila	19
		Micrasema sp	1
	Diptera	Agathon sp	2

Table 6. Continued

Site	Genus	Species	Number	Site	Genus	Species	Number
Spring 9	Ephemeroptera	Baetis tricaudatus	3	10	Ephemeroptera	Druella grandis	11
		Drunella grandis	22			Drunella doddsi	6
		Drunella coloradensis	1			Drunella coloradensis	2
		Epeorus longimanus	3			Ephemerella infrequens	129
		Ephemerella infrequens	166				
					Plecoptera	Chloroperlidae	3
	Plecoptera	Chloroperlidae	1			Diura knowletoni	7
		Diura knowletoni	3			Isoperla fusca	22
		Isoperla fusca	2			Claassenia sabulosa	6
						Pteronarcella badia	5
						Pteronarcys californica	6
	Trichoptera	Apatania sp	1				
		Brachycentrus sp	239		Trichoptera	Hydropsyche sp	3
		Lepidostama sp	5			Brachycentrus sp	79
		Micrasema sp	63			Lepidostama sp	4
						Micrasema sp	58
	Diptera	Atherix pachypus	31			Dicosmoecus sp	1
		Hexatoma sp	2				
		Tipula sp	1		Diptera	Atherix pachypus	30
						Dicranota sp	3
	Coleoptera	Laccophilus maculosus	36			Tipula sp	2
	Pisces	Cottus sp	1				
11	Ephemeroptera	Drunella grandis	20				
		Drunella coloradensis	5				
		Epeorus grandis	1				
		Epeorus longimanus	3				
		Paraleptophlebia sp	1				
		Ephemerella infrequens	46				
	Plecoptera	Chloroperlidae	3				
		Diura knowletoni	17				
		Isoperla fusca	15				
		Claassenia sabulosa	1				
		Hesperoperla pacificum	3				
		Pteronarcys californica	6				
		Kogotus sp	1				
		Magarcys sp	2				
	Trichoptera	Hydropsyche sp	3				
		Brachycentrus sp	70				
		Micrasema sp	18				
	Diptera	Atherix pachypus	65				
		Tipula sp	1				
	Pelecypoda	Sphaerium sp	1				
	Pisces	Cottus sp	2				

Table 6. Continued

Site	Genus	Species	Number
Spring 13	Ephemeroptera	Ameletus sp	90
		Drunella doddsi	1
		Baitis bicaudatus	67
		Epeorus grandis	6
		Cinygmula sp	19
		Rhithrogena sp	3
	Plecoptera	Chloroperlidae	2
		Cultus sp	3
		Isoperla fusca	8
		Claassenia sabulosa	40
		Hesperoperla pacificum	16
		Doroneuria theodora	15
	Trichoptera	Hydropsyche sp	9
		Rhyacophila sp	21
	Diptera	Agathon sp	1

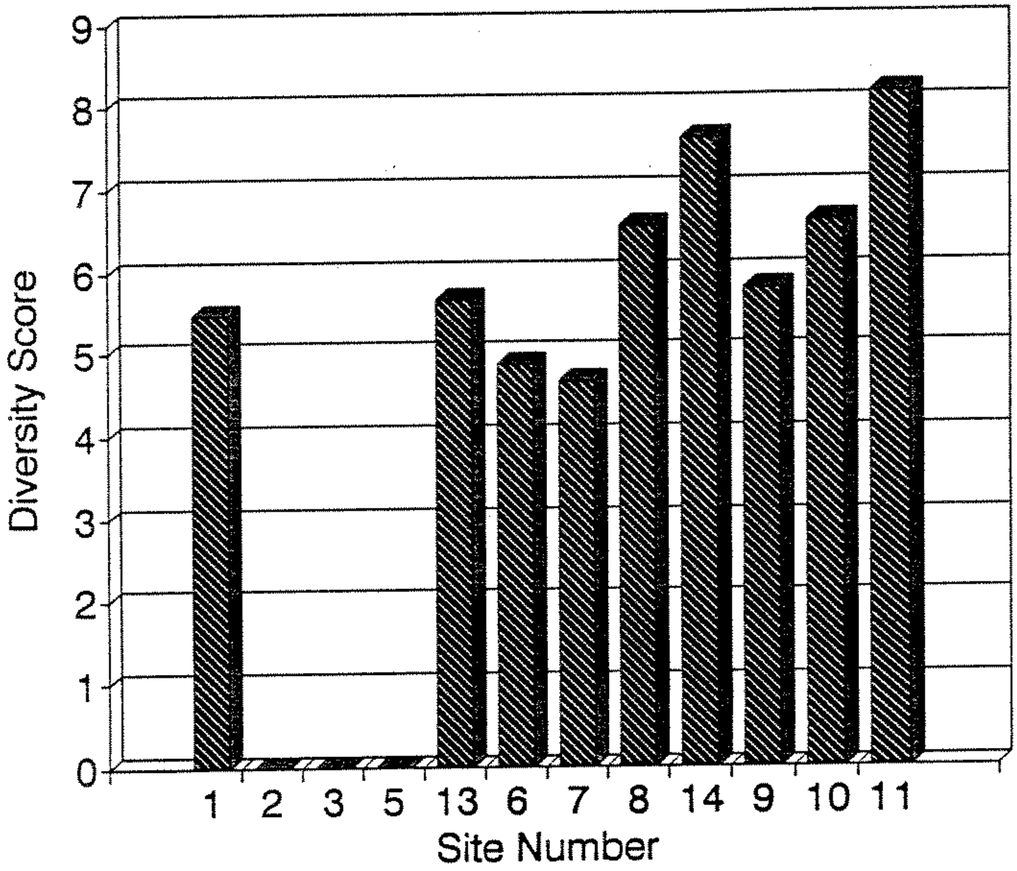
Site	Genus	Species	Number
Spring 14	Ephemeroptera	Baetis tricaudatus	25
		Cinygmula sp	13
		Drunella grandis	8
		Drunella flavilinea	79
		Drunella doddsi	8
		Epeorus longimanus	37
		Ephemerella infrequens	107
	Plecoptera	Cultus sp	1
		Isoperla fusca	9
		Claassenia sabulosa	2
		Hesperoperla pacificum	4
	Trichoptera	Amiocentrus sp	47
		Arctopsyche grandis	1
		Hydropsyche sp	4
		Brachycentrus sp	191
		Dicosmiecus sp	2
		Lepidostoma sp	1
		Onocosmoecus sp	1
	Diptera	Atherix pachypus	33
		Hexatoma sp	1
		Ormosia sp	1
	Pisces	Cottus sp	1

Table 6. Continued

Site	Genus	Species	Number
Fall 9	Ephemeroptera	Baetis tricaudatus	2
		Drunella grandis	8
	Plecoptera	Chloroperlidae	27
		Cultus sp	13
		Isoperla isoperla	6
		Pteronarcella badis	4
	Trichoptera	Brachycentrus sp	1
		Hydropsyche sp	1
		Micrasema sp	4
	Diptera	Atherix pachypus	65
		Chi ronomidae	1
		Pedicia sp	3
		Tipula sp	1
	Coleoptera	Dytiscidae	2
		Oreodytes rivulus	2
	Pisces	Cottus sp	
Fall 13	Ephemeroptera	Ameletus sp	76
		Drunella grandis	2
		Baitis bicaudatus	22
		Cinygmula sp	1
	Plecoptera	Amphinemoura sp	39
		Chloroperlidae	32
		Claassenia sabulosa	7
		Hesperoperla pacificum	11
	Trichoptera	Arctopsyche grandis	5
		Polycentropus sp	1
	Diptera	Empidiae sp	1
		Hexatoma sp	2
		Tipula sp	4
Fall 6	Plecoptera	Hesperoperla pacifica	1
		Pteronarcella badia	1
	Hemiptera	Gerris sp	1
Fall 14	Ephemeroptera	Drunella grandis	63
		Drunella doddsi	11
	Plecoptera	Chloroperlidae	11
		Cultus sp	68
		Isoperla fulva	11
		Isoperla sobria	27
		Hesperoperla pacificum	6
		Pteronarcys californica	4
		Pteronarcella badia	42
	Trichoptera	Hydropsyche sp	61
		Brachycentrus sp	9
		Micrasema sp	9
		Rhyacophila sp	5
	Diptera	Atherix pachypus	205
		Pedecia sp	2
	Coleoptera	Oreodytes rivulus	3

Figure 17. Invertebrate Diversity

Diversity Index Test



Biomass of invertebrates sampled from areas affected by AMD, are a good measure of the impact to the invertebrate population (Letterman, 1978). In this study the total number of invertebrates collected will be used as a measure of biomass. Separate evaluation of Uncle Sam Gulch, Cataract Creek and the Boulder River invertebrate biomass is done to eliminate habitat differences between streams.

Total numbers of invertebrates collected in Uncle Sam Gulch indicate that below the Crystal Mine invertebrate populations do not exist. Only Site 1 is capable of sustaining a invertebrate population probably due to water and sediment toxicity and the reduction of invertebrate habitat below the Crystal Mine. Total biomass numbers found in Cataract Creek indicate that background Site 13 and Site 7 are similar. June biomass totals at Site 6 and 8 are 39 percent and 87 percent reduced when compared to Site 13. Biomass numbers in the Boulder River show that Sites 9 and background Site 14 are very similar in June and Site 9 is 85 percent less then Site 14 in October. Sites 10 and 11 are 35 percent and 51 percent less then Site 14 in June.

Biomass measurements indicate that stream sections affected by AMD have reduced invertebrate numbers. Invertebrate drift, during high water, in June may have increased numbers at Site 6, 7 and 9. Sampling procedures may also have influenced biomass numbers. As sampling progressed technique improved which could increase numbers at Sites 7, 8, 9, 10, 11 and 14.

Invertebrate samples collected in June and October were tested for diversity using Margalf's diversity index suggested by Warren, 1971. Figure 17 represents scores calculated for each site. Diversity indices were derived to assess invertebrate populations reactions to waters with organic pollutants and results of diversity indices used for metal-polluted waters is uncertain (Letterman, 1978). Diversity indices rank sites by giving a high number to more diverse populations.

June samples assessed with Margalef's diversity index indicated that Sites 6 and 7 along Cataract Creek were the least diverse and Sites 11 and 14 were the most diverse. Site 1, which is located above the Crystal Mine at a elevation of 8000 feet, could have a low diversity index (5.476) due to its limited habitat. Site 8 has a higher diversity index, possibly due to the larger number of square feet sampled. A larger sampling area was used to gather a sample of significant size. October invertebrate samples indicate that Site 6 is, again, the least diverse and Site 9 is the most diverse.

Sampling technique and environmental factors which influence collection can have large effects on diversity indices. Diversity indices do not consider habitat differences, sampling errors or variations, environmental factors-high stream flow causing invertebrate drift which may influence rankings. No aquatic insects were found in Uncle Sam Gulch, except above the Crystal Mine. Sites along Cataract Creek below Uncle Sam Gulch are less diverse than

Figure 18. Diversity Index
versus pH

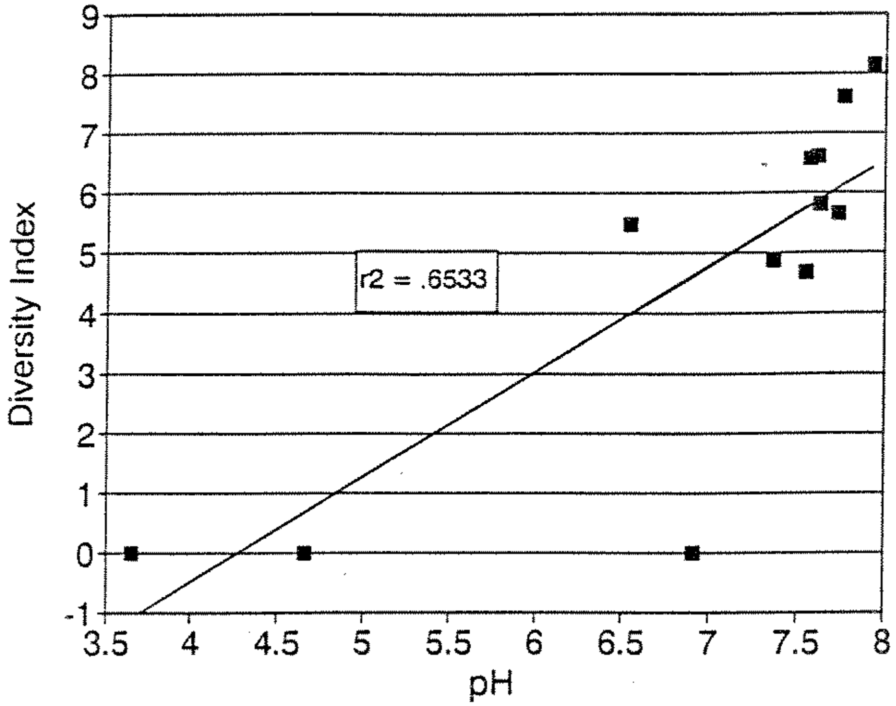
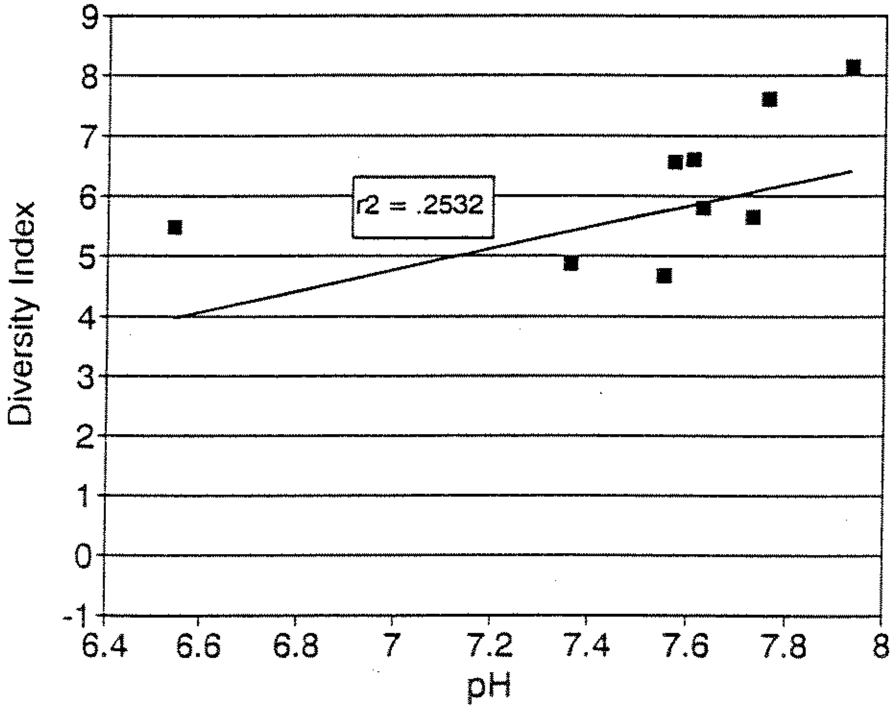


Figure 19. Diversity Index
versus pH (selected sites)



sites along the Boulder River. Site 6 may have received an inordinately higher score because of invertebrates drifting downstream from above the confluence of Uncle Sam Gulch and Cataract Creek.

Regression analyses of June data, using multiple stepwise regressions, was used to test for controlling factors which could influence the invertebrate diversity index. These statistics attempt to test this diversity index and the prediction that metal polluted waters will have lower invertebrate diversities. Regressions were conducted between the diversity index, as the dependent variable, and pH, water copper concentration, water cadmium concentration, water zinc concentration, copper sediment concentration, cadmium sediment concentration and zinc sediment concentration as the independent variables. Results are displayed in Table 8.

Table 8.
Correlation Coefficients
Individual Significance

CuW	-.72	<.05
CdW	-.74	<.05
ZnW	-.76	<.05
pH	.80	<.05
CuSd	-.45	<.05
CdSd	-.21	<.05
ZnSd	-.27	<.05

Multiple Regressions

pH		62.9%
CuSd	26.6%	89.5%

Cu, Cd, ZnSd = copper, cadmium, zinc sediment concentration
Cu, Cd, ZnW = copper, cadmium, zinc water concentration

Data from multiple regressions suggest, with 95% confidence interval, that pH may have the most impact on population diversity, being 62.9 percent of the controlling additive factor. Copper sediment concentrations and pH together control approximately 89.5 percent of the invertebrate diversity, according to this regression analyses. Correlation coefficients show copper, cadmium and zinc water concentrations and pH may have the most significant individual effects. Results imply that invertebrate diversity improves with increasing pH and decreasing water and sediment concentrations. Figure 18 shows the distribution of the diversity indices versus pH. The regression coefficient for the best fit line is .0.629, for pH individually and 0.9486 for all independent variables. Diversity indices are clustered at high diversity sites and zero diversity sites. Best fit line between the two clusters results in the high regression. Invertebrate limiting factor, probably pH, inhibits population diversity somewhere along this best fit line. Since no data is available between the two clusters of plots this line could fall to zero at any location below the upper cluster of plots.

Figure 19 illustrates diversity index versus pH at sites where a diversity greater then zero was recorded. Regression coefficient for this relationship is .2532. Multiple regression using diversity as dependent variable and pH, water and sediment metal concentrations as independent variables results in .9612 regression. Although cadmium water concentrations had to be eliminated for the regression to run.

Copper, cadmium and zinc metal concentrations found in invertebrates are reported in Table 10. Figure 20, 21 and 22 illustrate the relationship between metal concentrations found in invertebrates and the sediment metal concentration.

Table 9. METAL CONCENTRATIONS IN INVERTEBRATES
October sampling sites 9, 13, 14

Site	Diversity Index	COPPER (μg/l)	CADMIUM (μg/l)	ZINC (μg/l)
9	6.979	73.59	3.58	201.7
13	5.200	21.29	2.83	173.35
14	5.495	38	1.22	188.41

In invertebrates, copper, cadmium and zinc concentrations increase with higher metal concentrations in sediment. There appears to be a correlation, but the small sample size drastically limits conclusions. Regression coefficients of 0.8954, .5926 and .7870 for copper, cadmium and zinc represent each line, respectively. Correlation coefficients, 95% confidence, for copper and zinc indicate a significant relationship between invertebrate metal concentrations and sediment concentrations. No, significant correlation is found between cadmium in sediment and invertebrate metal concentrations. Although, further and larger sampling may result in a better correlation between sediments and invertebrates.

The highest concentrations found in invertebrates were at Site 9 which has the largest population of sediment-associated or sediment-dependent invertebrates and the highest sediment metal concentrations. Sites 13 and 14 have approximately the

Figure 20: Invertebrates vs Sediment

Copper sites 14, 13, 9 October

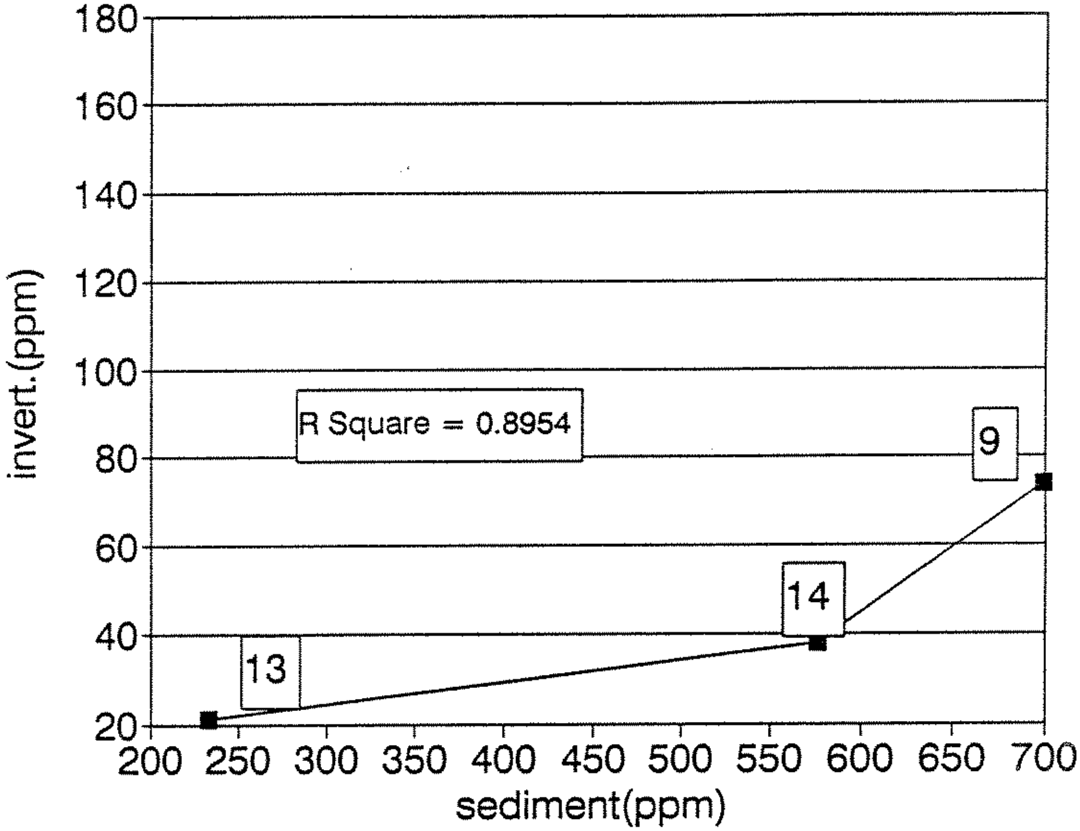


Figure 21. Invertebrates vs Sediment
Cadmium sites 14, 13, 9 October

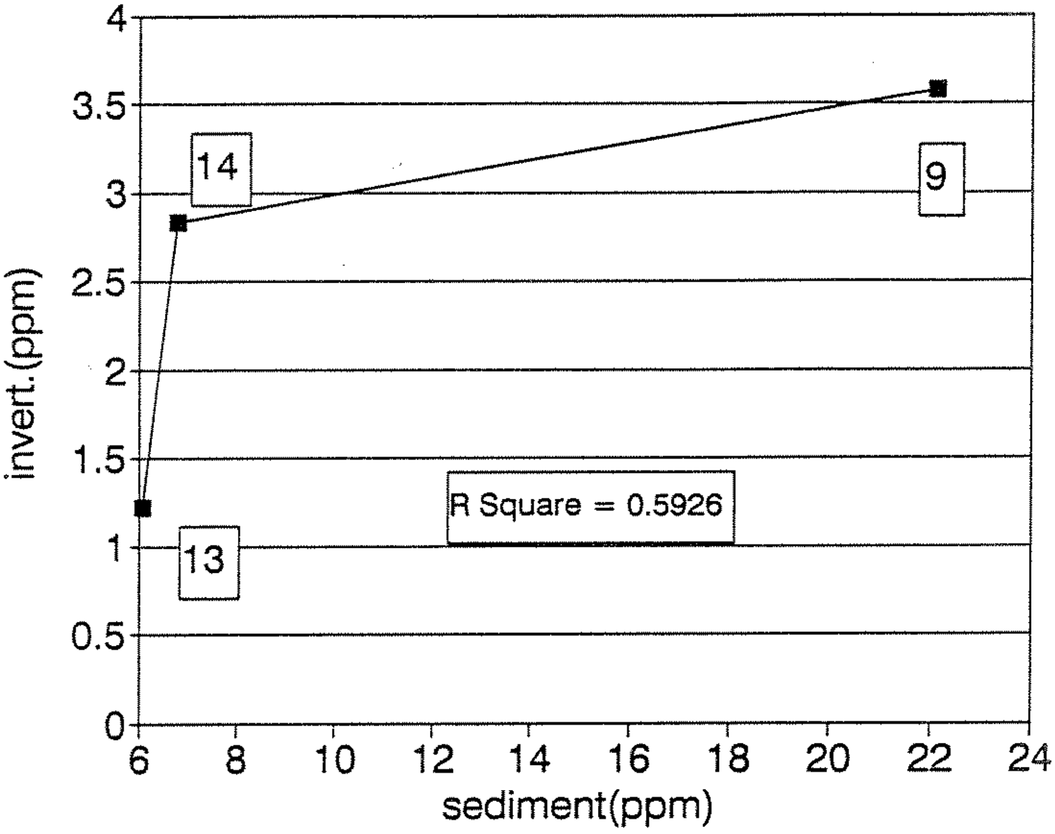
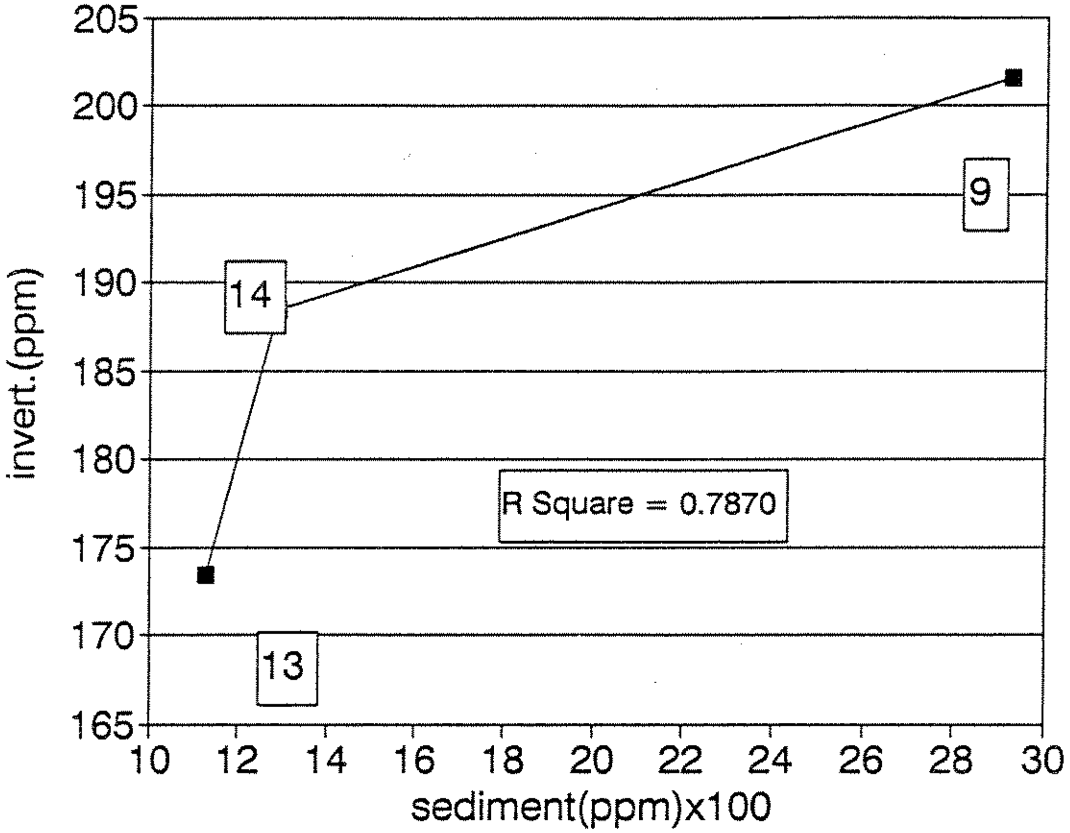


Figure 22. Invertebrates vs Sediment

Zinc sites 14, 13, 9 October



same types of; sediment-associated, carnivorous and filter-feeding invertebrates. Burrows, 1983, found, in almost all cases, that elevated concentrations of metals in water and sediments were followed by higher concentrations in animals. In all cases, metal concentrations in surrounding waters exceeded concentrations in invertebrates and sediment. However, exact relationships between the components differed with types of metals and species of invertebrates considered. Cadmium and zinc concentrations measured by Burrows, 1983, associated with individual species of invertebrates, showed higher concentrations at the same sediment concentrations as observed during this studies October sampling.

The invertebrate diversity index scores for this study do not reflex metal concentrations found in invertebrates. Regression analyses conducted on diversity index greater then zero also showed a reduced relationship between metal pollution and the invertebrate diversity index calculated. This observation was also reported by Letterman, 1978.

Fish

Results from the fish collection and analysis are presented in Table 9. Site, species, number of specimens and mean metal concentrations are presented.

Table 10. FISH COLLECTION DATA AND METAL CONCENTRATIONS

Site	Species	#Species	Cu(μg/l)	Cd(μg/l)	Zn(μg/l)
6a	CT	6	118.7	13.06	160.93
6b	none	none			
7	none	none			
8a	RB	4	195.2	51.05	213.9
8b	RB	3	209.7	30.76	88.81
9a	RB	3	71.06	4.39	76.92
9b	RB BT	2 1	85.29 62.52	15.05 6.03	89.6 83.94
10	RB BT WF	1 2 1	108.6 58.19 4.32	24.13 13.10 8.61	132.22 84.35 31.61
11	RB	3	163.4	20.01	113.5
13a	CT	7	224.2	15.26	100.5
13b	CT	3	427.9	39.92	144.3
14a	RB	3	236.0	5.71	129.1
14b	RB	3	164.7	7.53	90.31

a= June sample, b= October sample, RB= rainbow trout, BT= brook trout, WF= whitefish, CT= cutthroat trout

October samples yield relatively higher concentrations of metals than June samples, although numerous factors, which could alter the data, are not accounted for: species age, size and ability of fish to move from one site area to another. Fish collected at Site 6 in June were probably transported downstream from Site 13 due to high spring flows (Hill, 1991). These samples will not be considered because there is no way to evaluate the length of time these fish had spent in the area of Site 6. Fish collected from Site 14 could have been transported from below the confluence of Cataract Creek. Three Sites 8, 11 and 13 probably represent fish samples native to the sampling area.

Using Site 14 as the baseline for metal concentrations in rainbow trout (*Salmo gairderi*) kidney and livers for this area, indicates that copper metal concentrations are lower in waters effected by Crystal Mine AMD during the June sampling period. October samples show that Site 8 copper concentrations are 1.3 times greater then concentrations at Site 14. Site 9 October samples are again lower then the background samples from Site 14. E.P.A. 1, 1978, reports baseline copper concentrations for rainbow trout kidneys to be 12.9 μg/g.

Cadmium concentrations increase downstream of the background site during both June and October sampling periods. In June, rainbow trout collected at Site 11 have 3.5 times the cadmium concentration as samples from Site 14. Fish samples from site 9 had similar cadmium levels as Site 14 during June and were 2 times greater in October. At Site 8 rainbow trout had 8.9 and 4.1 times the cadmium concentrations in June and October, respectively when compared to Site 14 samples.

Rainbow trout zinc concentrations from sites affected by AMD vary only slightly from background levels. Sites 9 and 10 have lower concentrations then Site 14 and Site 11 has 3 μg/l more zinc then Site 14. Site 8 rainbow trout have 1.66 times the zinc levels as compared to Site 14. Baseline concentrations of zinc in rainbow trout kidney and liver, as reported by E.P.A. 1, 1978, are 125 and 150 μg/g, respectively.

Comparing fish samples from Site 8 and 11 indicates rainbow trout in these areas are accumulating copper, cadmium and zinc relative to the metal concentrations in the sediment. Since the kidneys and livers perform digestion and blood cleansing functions, metals found in these organs represent metals from food sources (E.P.A. 1, 1978).

Samples collected at Site 13 are of a different species, cutthroat trout (*Salmo clarki*), and little information is available on correlations between rainbow trout and cutthroat trout. Thus, using cutthroat trout from Site 13 to obtain background metal levels is not acceptable because rainbow trout were the only other species of fish found in Cataract Creek.

Results of this study indicate that rainbow trout at Site 8 are accumulating copper and cadmium. At other sites affected by AMD cadmium appears to be the only metal which is accumulating in fish kidney and liver tissue, when compared to background Site 14.

Fish biomass was not determined during this study, however, Nelson, 1974, found a 51 percent decrease in the fish population, below the confluence of Cataract Creek and the Boulder River, compared to populations above the town of Basin. Biomass differences in this study were evident along Cataract Creek. Cutthroat trout found at Site 13 were native trout. Hill, 1991, believed that the cutthroat trout collected at Sites 6 and 7, in June were washed downstream in the high water. No fish samples were found at Sites 6 or 7 in October. Fish collected at Site 8 in June and October were small in size and difficult to locate. Hill, 1991, also mentioned that it has and is easier to collect fish above the confluence of Cataract Creek and the Boulder River then below the confluence. Absence or difficultly in collecting fish from Sites 6, 7, 8, 9, 10 and 11 may be a function of higher metal levels. E.P.A. 1, 1978 and Sprague, 1964, report that increased copper and zinc levels will cause avoidance behavior by trout. Thus, increased copper, cadmium and zinc levels downstream of Uncle Sam Gulch may displace fish.

Lethal Concentrations of 50 percent mortality (LC_{50}) of copper for rainbow trout are; 7 days 44 μg/l and 96 hour 309 μg/l (E.P.A. 2, 1980). LC_{50} for 7 days 44 μg/l may explain the absence of fish from Uncle Sam Gulch and Cataract Creek, also. Copper concentrations along Cataract Creek are above the LC_{50} 7 day levels during low water and at Site 6 during high water.

Cadmium LC_{50}'s for 48 hours is 91 μg/l and for 10 days is 7 μg/l (Calamari, 1980, Mckim, 1975). Cadmium levels in this study were under detection limits at most sites. Where cadmium was detected levels exceeded the 10 day LC_{50}, which could limit fish growth and migration into Site 6 and Uncle Sam Gulch.

E.P.A., 2, reports zinc LC_{50} of 2800 μg/l for 24 hours in water with hardness, as $CaCO_3$, 40 mg/l, which is near or above

$CaCO_3$ levels in the study area. The 7 day LC_{50}, at the same hardness, is 560 μg/l for rainbow trout. A 14 day LC_{50} for cutthroat trout as reported by E.P.A., 2, is 670 μg/l. These zinc concentrations would adversely affect fish in Uncle Sam Gulch and Cataract Creek near Site 6.

There was a absence of fish where a absence of or deduction of invertebrates was found. Since invertebrates would be a fish food source a lack of invertebrates would limit the fish population growth.

CONCLUSION

Results of data collected in June and October, 1991, indicate that the AMD produced by the Crystal Mine is adversely effecting Uncle Sam Gulch, Cataract Creek and the Boulder River. Water quality is severely altered along Uncle Sam Gulch. Copper concentrations in stream water were increased 316 times below the Crystal Mine. Cadmium levels were increased 115 times. Zinc concentrations were 9550 times higher below the mine compared to background zinc levels. Water quality in Cataract Creek, below the confluence with Uncle Sam Gulch, was also affected; increases of 43, 0 and 5.4 times were recorded for copper, cadmium and zinc, respectively, compared to upstream background concentrations. The Boulder River water quality was also impacted with concentrations increases of 17, 0 and 2 times the background levels.

Sediments have also been adversely affected by precipitation of metals resulting from the AMD. Sampling sites along Uncle Sam Gulch, Cataract Creek and the Boulder River which have been effected by AMD of the Crystal Mine have increased copper, cadmium and zinc concentrations compared to the background sampling sites.

Acid Mine Drainage resulting from the Crystal Mine has had a detrimental impact on the living organisms in Uncle Sam Gulch, Cataract Creek and the Boulder River. Currently, Uncle Sam Gulch only supports a limited population of bacteria (Anderson, 1991). Invertebrate and fish populations in Cataract Creek were depressed. A large portion of invertebrates present in Cataract Creek were drift species, originating from sources above Uncle Sam Gulch. Invertebrate biomass was reduced 40, 0 and 77 percent in June at Sites 6, 7 and 8, respectively, compared to Site 13. In October no invertebrates were found at these sites and 203 invertebrates were found at Site 13. Fish are only present in the lower portion of the Cataract Creek. Previous reports have suggested that the fish population in the Boulder River above Cataract Creek is healthier then the fish population below Cataract Creek (Knudson, 1984). Results presented here suggest that metal concentrations are high enough to cause avoidance behavior in trout (Platts, 1987).

Proper reclamation of the Crystal Mine, including the tailing piles, open pits and adit effluent, would improve water quality in all water below the mine. Stream bed impacts will only improve with time. Metal deposition along the stream course over the past 100 years will take time to naturally reclaim. As water and sediment quality improve, invertebrate and fish populations will restock Uncle Sam Gulch and Cataract Creek. The Boulder River fish population should benefit from improved water quality.

REMEDIATION

Elimination of AMD from the Crystal Mine will consist of at least two major steps: reclamation of barren ground, tailings and open pits, and treatment of effluent from the mine adit. Both steps are vital for the improvement of water quality in Uncle Sam Gulch, Cataract Creek and the Boulder

River. During spring runoff and snow melt, mass loading from the barren land and exposed tailings represented approximately 18.5, .81 and 66.9 lb/day of copper, cadmium and zinc respectively. Adit drainage during October was 4.38, .319 and 23.44 lb/day of copper, cadmium and zinc respectively. Open pits above the adits store and supply additional water which in turn drain into the mine tunnels.

Acid Mine Drainage formation requires 4 essential elements: 1) oxygen, 2) water, 3) mineral source, 4) *Thoibacillus ferrooxidans* (Wildeman, 1991). Removal of any one of these variables will control the production of AMD. Reclamation procedures considered should focus on controlling or eliminating at least one of these elements.

Reclamation

Reclamation of the disturbed area is the only solution to control runoff AMD production potentials. Approximately, 50 acres of unreclaimed land is currently exposed to the four elements essential for AMD. Large open pits provide storage of snow and rain water. Reclamation would eliminate the storage of water and its contact with minerals which produce AMD.

Water storage in the open pits can be reduced by the backfilling the open pits. Excess fill is available from tailing piles and earth piles created from the mining operations. Compaction and covering of material used as backfill will be needed to help control infiltration from surface water.

Land reclamation practices for control of erosion and infiltration will limit the production AMD. Since acid forming properties are probably present in the existing soil material, capping of the area with top soil and a limestone barrier between the topsoil and contaminated soil may be essential.

Proper revegetation and contouring will be required to control erosion of top soil and water infiltration. Native vegetation tested in Colorado was easiest to establish at higher elevations (Colbert, 1991). Contouring will require reduction of steep slopes and directing water flow away from the open pit areas where infiltration would be the most detrimental.

Colbert, 1991, found the use of organic fertilizer as the most effective. Organic supplements improve soil properties which is beneficial for long term soil nutrients. Artificial fertilizers supply nutrients directly to the plants and do not help improve soil properties.

Proper reclamation could help reduce adit effluent flowrates; possibly up to 50 percent of the current rate (Sonderegger, 1991). Flowrates from 1991 and 1992 presently being recorded are approximately 25 gal/min for mid-winter flows while spring maximum flowrate was close to 50 gal/min. Reduction of AMD flowrates would simplify effluent treatment.

Effluent Control

Treatment of AMD, produced by the effluent, flowing from the adit could be achieved by a number of alternatives. The location of the mine is a limiting factor for some types of remediation. Environmental factors such as temperature, snowpack and spring rainfalls as well as lack of electrical power will also limit some remediation alternatives.

One alternative to eliminate the adit effluent is to eliminate oxygen from the mine tunnels. This method could be accomplished by the use of explosives to collapse mine tunnels. This method would be successful only if oxygen was completely eliminated. This may be accomplished by filling of open pits with sand or other small diameter sediments. Sand could fill in spaces which otherwise might allow the passage of air into the ground water stream. This procedure might be the simplest and most cost effective; however, complete elimination of AMD would be difficult to achieve. There is also the chance that a larger opening into the adit could be created.

Another alternative to control the presence of oxygen without building external reactors would be to purge or flood mine tunnels with nitrogen (Worcester, 1992). Purging nitrogen into the tunnels would eliminate oxygen and eliminate activity of aerobic bacteria. This method would require secure sealing of the tunnel entrances and a continuous supply of nitrogen. This method would be rather impractical when considering the remoteness of the mine and difficultly of containing the nitrogen within the tunnels.

Recently, the use of natural and manmade wetlands has been used successfully to treat AMD in coal mining and hard rock mining operations. Removal of metals in wetlands, through microbial activity, takes place both aerobically and anaerobically. The majority of metal pollution reduction and pH control takes place at the anaerobic level, located in wetland bottom material (Wildeman, 1991). Construction of man-made wetlands is a low cost, long-term treatment. A list of the removal processes active in a wetland as reported by Wildeman, 1991, follows:

1) Exchange of metals by an organic-rich substrate, i.e. peat.
2) Sulfate reduction with precipitation of iron and other sulfides.
3) Precipitation of ferric and manganese hydroxides.
4) Adsorption of metals by ferric hydroxides.
5) Metal uptake by living plants.
6) Filtering of suspended and colloidal material.
7) Neutralization of water.
8) Adsorption or exchange of metals onto algal materials.

Wetland treatment of metal contaminated waters has been used successfully in some coal mining regions of the U.S. The treatment of AMD with wetlands for high elevation abandoned mines is being researched. The Tracy Wetlands near Great Falls, MT, were used for a short time but under sizing and poor flow direction made them ineffective (Hiel, 1988). Researchers in Colorado have experimented with wetland treatment process with some success (Wildeman, 1991).

The two most significant limiting factors involving the use of a wetland at the Crystal Mine site would be the length and temperature of the winter months, and the limited amount of workable space. Wildeman, 1991, found that successful winter operation required mine water to be 12 - 15 degrees C and a portion of the wetlands to be exposed to the sun during the day. Construction of the system to inhibit shortcircuiting, i.e. freezing, during the winter is a design requirement. Results from Colorado also determined a loading factor of 1000 ft^2 of wetlands per gallon/minute of influent (Wildeman, 1991). Maximum flowrate measured in 1991 from the Crystal Mine was 50 gal/min. This would require a site or sites totaling 5000 ft^2 and 5 feet deep. A site requiring 5000 ft^2 would be difficult to construct at the Crystal Mine site.

Anaerobic treatment processes are also being considered for the remediation of AMD. Dvorak, 1991, reports systems utilizing anaerobic bacteria that oxidize simple organic compounds with sulfate to generate hydrogen sulfide and bicarbonate ions.

$$SO_2^{-2} + 2CH_2O + 2H^+ \rightarrow H_2S + 2H_2O + CO_2$$

Hydrogen sulfide reacts with the metal polluted water to create insoluble metal sulfides. The production of the bicarbonate consumes available protons and raises the pH. Anaerobic reactors using mushroom compost, manure and other waste organic matter have reduced metal concentrations significantly. The use of other composts and increasing holding times will increase efficiency (Dvorak, 1991).

Remediation of AMD utilizing an ion-exchange process is another alternative. There are various techniques which operate with similar methods. Electroplating or electrolytic cells remove metals from solution through oxidation-reduction reactions. These systems require electrical power, a limiting factor at the Crystal Mine. Solar power generators could possibly produce enough power depending on the system design. Ion-exchange treatments also provide the alterative of recovering metals which may be marketable.

Chemical treatment of effluent with lime or other alkaline substances would also reduce metal content and raise the pH. Chemical treatment is an active treatment process with high initial cost and regular operating costs (Schaffer, 1989). These costs would be increased at the Crystal Mine due to the remoteness and accessibility problems during the winter months.

Bactericides which inhibit the production of AMD by interfering with the oxidation of Fe^{2+} to Fe^{3+} or introduction of bacteria which produce organic acids that adversely affect _Thoibacillus_ are other alternative treatments. These technologies are experimental and the longevity and success are unknown (Schaffer, 1989, Shuttleworth, 1988). Placement of such treatment would be a problem at the Crystal Mine because of the large amount of underground workings.

Alternatives involving water treatment; wetlands, anaerobic cells, ion-exchange and chemical treatment, may require sludge disposal plans. Analysis of sludge from the Big Five Wetlands in Colorado indicated that after one year the substrate would be defined as Reactive under the Resource Conservation and Recovery Act (RCRA) due to the possible production of hydrogen sulfide gas (Wildeman, 1991).

Wetland treatment appears to be the most practical alternative treatment. Considering the factors associated with the Crystal Mine; location, lack of electrical power and winter temperatures. The wetland treatment would possibly be the easiest alternative to construct and maintain. Although, proper reclamation of tailing piles, barren ground and open pits should be a priority with any remediation alternatives.

Monitoring should continue to identify other potential AMD sources which could adversely affect receiving waters. Further research should continue toward remediation alternatives. The Crystal Mine and surrounding mines offer unique opportunities to test various technologies. Higher elevation, remoteness and environmental factors create location-specific challenges for remediation of AMD.

The most essential aspect for additional research is that of treatment alternatives. Testing of various techniques would be required to determine the most efficient alternatives. Since the Crystal Mine is located in a unique location, cost analyses of alternatives would also be essential.

Further research in the area of AMD prediction and location would also be beneficial. Use and development of diversity indices or other ranking formulas to assess aquatic populations could help locate AMD problems. Prediction of potential AMD sites is a concern in pre-mining plans and further development of prediction methods could help prevent AMD problems.

References:

Anderson, Deanna. 1991. Personnel Communication. Conversation. Montana Tech. Biology Dept. Butte, MT.

Anderson, Richard, V. 1977. "Concentration of Cadmium, Copper, Lead and Zinc in 35 Genera of Freshwater Macro-Invertebrates from the Fox River in Illinois and Wisconsin." Bulletin of Environmental Contamination and Toxicology. vol. 18.

Brown, Barbara. 1977. "Effects of Mine Drainage on the River Hayle, Cornwall A) Factors Affecting Concentrations of Copper, Zinc and Iron in Water, Sediment and Dominant Invertebrate Fauna." Hydrobiologia. vol 52, 2-3, pg. 221-233.

Burrows, I.G., B.A. Whitton. 1983. "Heavy Metals in Water, Sediment and Invertebrates from Metal-Contaminated River Free of Organic Pollution." Hydrobiologia. vol. 106.

Cairns, Victor W., P.V. Hodson, J.O. Nriagu. 1984. Contaminant Effects on Fisheries. Wiley Series vol.16.

Calamari, D., et. al. 1980. "Influence of Water Hardness on Cadmium Toxicity to Salmo gairdneri." Richland Water Research. 14: 1421-1426.

Chambers, J.C., M.C. Amacher, R.W. Brown. 1991. "Effects of AMD on a Riparian Area: Element Concentrations in Soils and Plants." Proceedings of the 1991 National Meeting of the American Society of Surface Mining and Reclamation, vol. 1.

Chand, Fateh. 1981. A Manual of Geochemical Exploration Methods. Ministry of Primary Industries Malaysia.

Cherry, John., E. Gless. 1991. "Crystal Mine Acid Mine Drainage." Unpubl. report.

Cohen, Ronald R.H., J. Gorman. 1991. "Mining-Related Nonpoint-Source Pollution." Water Environment and Technology. vol. 3 No. 6.

Cushnie, G.C. 1985. Electroplating Wastewater Pollution Control Technology. Pollution Technology Review No. 115. Noyes Publication.

Degroot, A.J., E. Allersma. 1983. "Field Observations on the Transport of Heavy Metals in Sediment." Heavy Metals in the Aquatic Environment.

References continued

E.P.A. 1. 1978. Metal Bioaccumulation in Fishes and Aquatic Invertebrates; A Literature Review. E.P.A.-600/3-78-103.

E.P.A. 2. Ambient Water Quality Criteria for Copper. 1980. E.P.A. Document 440/5-80-036.

E.P.A. 3. Ambient Water Quality Criteria for Zinc. 1980. E.P.A. Document 440/5-80-079.

E.P.A. 4. Quality Criteria For Water. (The Gold Book). 1986. E.P.A. Document 440/5-86-001.

Forba, Russ. 1992. Personal communication, Phone Conversation. E.P.A. Helena, MT. Office.

Kelly, Martyn. 1988. Mining and the Freshwater Environment. Elsevier Science Publ. Ltd. New York, N.Y.

Klaverkamp., W.A. Macdonald, D.A. Duncan, R. Wageman. 1986. Metallothionein and Acculimation to Heavy Metals in Fish: A Review.

Knudson, Ken. 1984. "A review of the Impacts to the Trout Fishery and Initial Reclamation Recommendations." Consulting Biologist.

Gless, E.E. 1990. Biological and Chemical Baseline Studies Along the Boulder River and Its Tributaries in Jefferson County, Montana. Montana College of Mineral Science and Technology, Butte, MT.

Handbook of Chemistry and Physics. 1979. 60th Edition. CRS Press, Inc. Boca Raton, FL.

Hill, Curt. 1991. Personal Communication. Conversation. Montana Fish, Wildlife and Parks Dept., Helena, MT.

Manahan, Stanley E. 1990. Environmental Chemistry. 4th edition. Lewis Publishers. Chelsea, MI.

Mason, C.S. 1991, Biology of Freshwater Pollution. 2nd. edition. Langman Science and Technical, Wiley Publ. New York, N.Y.

McKim, J.M. et. al. 1975. "Effects of Pollution on Freshwater Fish." Journal Water Pollution Control Federation. 47(6): 1711-1768.

References continued;

Meyer, Carmille. 1991. "Non-point Source Control in Colorado: Inactive/Abandoned Mines Non-point Source Program." Proceedings of 1991 National Meeting of the American Society of Surface Mining and Reclamation. vol. 1.

Nelson, F.A. 1976. "The Effects of Metals on Trout Populations in the Upper Boulder River, Montana." Master's Thesis. Fish and Wildlife Management, Montana State University, Bozeman, MT.

Nriagu, Jerome O., J.B. Sprague. 1987. Cadmium in the Environment. vol. 19 in Wiley Series, Advances in Environmental Science and Technology. Wiley-Interscience Publ. New York, N.Y.

Phillips, Glen, K. Hill. 1985. "Evaluation of Sources and Toxicity of Copper and Zinc in the Boulder River Drainage, Jefferson County." MT. Fish, Wildlife and Parks Dept.

Platts, William, S.B. Martin, E,R,J. Edwards. 1987. "Water Quality in an Idaho Stream Degraded by Acid Mine Waters." USDA Forest Service Technical Report INT-67.

Roembild, George. 1991. Personal Communication. Conversation at Bozeman Office. Bozeman, MT.

Schaffer, William M. E. Spouts. 1989. "Treatment of Acid Mine Drainage Using Constructed Wetlands." Schaffer and Associates, presented to MT Fish, Wildlife and Parks Dept.

Shuttlewoth, Kay L., R.F. Unz. 1988. A Growth Inhibition Model For Thiobacillus Ferrooxidans. Mine Drainage and Surface Mine Reclamation Proceedings, 1988. vol 1.

Smock, Leonard. 1983. "The Influence of Feeding Habits on Whole-body Metal Concentrations in Aquatic Insects." Freshwater Biology. vol. 13.

Sonderegger, J. 1991. Personnel Communication. Conversation. Geology Dept. Montana Tech, Butte, MT.

Sprague, J.B. 1964. Avoidance of Copper-Zinc Solutions by Young Salmon in the Laboratory. Jour. of Water Pollution Control Federation. 26:990-1004.

Standard Methods for the Examination of Water and Wastewater. 1985. 16th edition. American Public Health Assn. Washington, D.C.

References cont;

Stoker, Stephen H., S. Seager. 1976. Environmental Chemistry: Air and Water Pollution. 2nd Edition. Glenview, IL.

Thomann, Robert V., J.A. Miller. 1987. Principles of Surface Water Quality Modeling and Control. Harper and Row Publ. New York, N.Y.

Vandenberg, Raymond J. 1974. "The Effects of Acid Mine Pollution on the Benthic Macroinvertebrates of the Dry Fork of Belt Creek Drainage." Masters Thesis, Biology Dept., M.S.U., Bozeman, MT.

Warren, Charles E. 1971. Biology and Water Pollution Control. W.B. Sanders Company. Philadelphia, PA.

Washington Public Power Supply System/Envirosphere Company. 1978. Copper in the Aquatic Environment: a literature review.

Wildeman, Tom, J. Gusek, G. Brodie. 1991. Handbook for Constructed Wetlands Receiving Acid Mine Drainage. American Society of Surface Mining and Reclamation.

Worester, Sam. 1992. Personal Communication. Conversation. Montana Tech, Butte, MT.

Zar, Jerrold H., 1974. Biostatistical Analysis. Prentice-Hall, Inc. publ. New York, N.Y.

APPENDIX

Area vs River Mile

South Fork Boulder

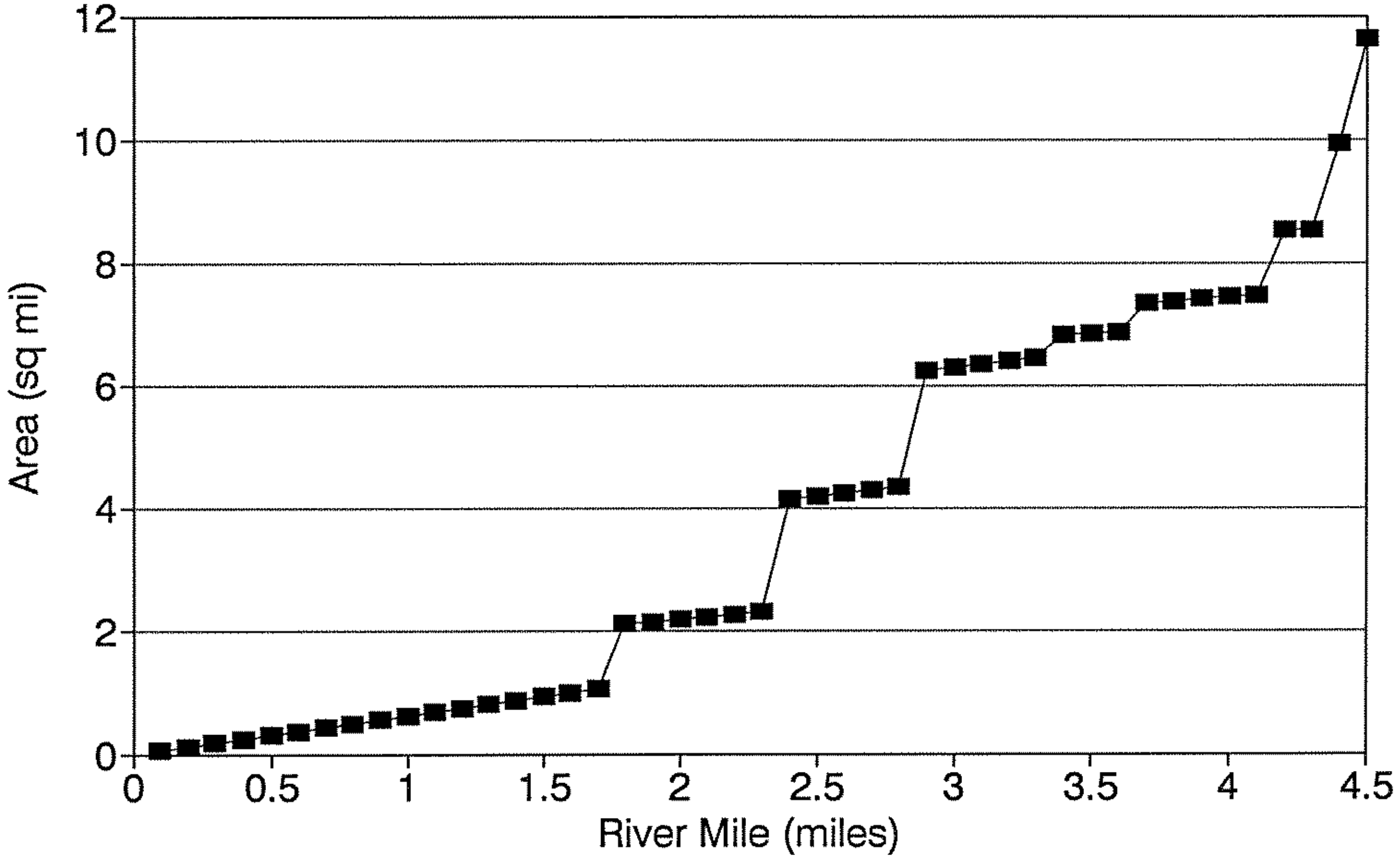


Table 1

SOILS ANALYSIS

Site	Lab #	Cd(ppb)	Cu(ppm)	Zn(ppm)	Mass Filtrate	Mass Soil
1	1	47	1.05	4.52	62.644	1.003
2	2	73	4.87	6.5	61.582	0.9861
5	3	665	46.8	51.3	62.466	1.005
6	4	372	18.2	25.5	62.828	1.034
7	5	207	8.6	17.4	63.286	1.0095
8	6	456	13.5	29.6	63.401	1.002
9	7	84	2.79	8.25	65.074	1.03
10	8	75	2.4	7.3	63.827	1.009
11	9	208	5.7	16.8	61.294	1.019
13	10	84	2.65	7.9	62.527	1.013
14	11	50	1.64	4.98	63.863	1.024
1	12	39	1.24	4.97	63.031	1.006
3	13	47	3.44	4.85	77.099	1.034
5	14	549	38.5	42.3	74.799	1.039
6	15	281	14.8	20	75.313	1.043
7	16	186	7.5	14.4	67.707	1.019
8	17	420	12.6	27.8	67.507	1.011
10	18	38	1.14	3.39	70.754	0.61
11	19	188	5.2	12.1	71.699	1.031
13	20	58	1.72	5.2	74.452	1.003
F6	21	260	14.4	29.8	75.033	0.267
1	22	32	0.89	3.25	83.46	1.008
14	23	30	1.11	3.14	82.749	1.007
5	24	570	37.3	40.3	66.077	1.011
F8	25	606	22.5	67	62.791	1.024
F13	26	65	2.02	9.35	63.497	0.524
F14	27	58	4.73	10.4	63.294	0.509
F9	28	136	4.11	16.8	63.546	0.365
3	29	58	3.76	5.15	65.694	1.004
11	30	200	5.3	13.2	65.249	1.009
BLANK	31	9	0.0969	0.0162	63.37	0
8	32	460	13.7	30.4	64.802	1.041

Table 2

FISH AND INVERTEBRATE LAB DATA

Site	Lab #	Cd(ppb)	Cu(ppm)	Zn(ppm)	SampleWt(gm)
14	1	13.5	0.6	0.53	0.1646
14	2	105	4.37	1.6	0.32
14	3	73	1.64	1.23	0.2492
13	4	264	4.83	1	0.253
13	5	96	1.23	0.91	0.268
13	6	388	4.1	1.48	0.343
13	7	95	1.55	0.76	0.218
13	8	85	0.47	1.06	0.393
13	9	95	1.58	0.95	0.321
13	10	130	1.65	1.28	0.416
6	11	148	1.07	2.14	0.375
6	12	171	1.47	5.3	1.026
6	13	436	4.19	2.9	0.545
7	14	117	0.82	2.37	0.755
7	15	102	0.67	4.81	1.634
7	16	197	1.56	7.1	0.717
8	17	956	3.58	3.42	0.435
8	18	459	2.32	2.43	0.349
8	19	835	2.41	3.36	0.502
9	20	25	0.73	0.98	0.373
9	21	284	0.135	0.88	1.715
9	22	21	1.3	0.95	0.305
9	23	55	0.45	0.55	0.129
10	24	258	0.46	1.31	0.623
10	25	503	2.24	2.72	0.613
10	26	116	0.73	0.81	0.223
10	27	351	0.193	1.27	1.188
11	28	297	1.48	0.81	0.446
11	29	182	3.5	1.67	0.3778
11	30	531	2.25	3.02	0.577
Blank	31	10	0.0218	0.0184	0
F8	32	165	0.9	0.38	0.226
F8	33	286	2.45	0.77	0.239
F8	34	115	0.61	0.37	0.085
F13	35	158	1.53	0.5	0.121
I13	37	63	0.42	3.26	0.561
F9	38	87	0.82	1.09	0.383
F14	39	27	0.27	0.29	0.088
F13	40	814	9	3.17	0.559
F9	41	75	0.21	0.97	0.31
F9	42	244	1.52	0.87	0.295
F14	43	48	2.22	0.88	0.322
F14	44	205	3.04	1.46	0.441
I14	45	123	3.55	17.5	2.785

Table 3 Mean/Standard Deviation
Fish Table Raw Data

Site #	Cu(ug/g)	Zn(ug/g)	Cd(ug/g)
6	83.8560	169.7280	11.0400
	42.7080	154.4330	4.7080
	229.4420	158.6200	23.4450
Mean	118.6687	160.9270	13.0643
Std Dev	80.1097	6.4537	7.7821
7	31.7160	93.4410	4.2520
	11.9010	87.9730	1.6890
	64.3600	296.3010	7.8240
Mean	35.9923	159.2383	4.5883
Std Dev	21.6287	96.9436	2.5159
8	245.3930	234.5930	65.2410
	197.5530	207.3010	38.5960
	142.7211	199.6970	49.3030
Mean	195.2224	213.8637	51.0467
Std Dev	41.9480	14.9830	10.9474
Fall 8	116.5750	48.0000	20.5750
	304.7950	94.3430	34.6440
	207.6000	124.0940	37.0590
Mean	209.6567	88.8123	30.7593
Std Dev	76.8543	31.3104	7.2686
9	56.9600	77.3400	1.2060
	1.9800	15.0720	4.7930
	125.7250	91.6330	1.0820
	99.5810	123.6280	10.4660
Mean	71.0615	76.9183	4.3868
Std Dev	46.8310	39.4445	3.8132
Fall 9	62.5220	83.9370	6.0310
	18.2130	92.0900	6.2900
	152.3590	86.6030	23.7970
Mean	77.6980	87.5433	12.0393
Std Dev	55.8063	3.3942	8.3146
10	21.1010	62.1960	11.9420
	108.5580	132.2150	24.1270
	95.2740	106.4930	14.2600
	4.3230	31.6060	8.6110
Mean	57.3140	83.1275	14.7350
Std Dev	45.2392	38.8848	5.7823

Fish Table Raw Data

Site #	Cu(ug/g)	Zn(ug/g)	Cd(ug/g)
11	98.0850	53.2470	19.3050
	276.1940	131.1490	13.6580
	115.8510	156.0620	27.0880
Mean	163.3767	113.4860	20.0170
Std Dev	80.1029	43.7928	5.5058
13	570.1420	116.3950	30.1190
	135.2460	99.8060	9.6270
	356.6940	127.8370	33.0610
	210.3130	102.0050	11.6970
	34.2140	79.5110	5.7250
	145.6260	87.0650	7.9440
	117.4180	90.9810	8.6540
Mean	224.2361	100.5143	15.2610
Std Dev	168.6108	15.6495	10.4892
Fall 13	373.9340	119.4050	36.6940
	481.8350	169.1380	43.1480
Mean	427.8845	144.2715	39.9210
Std Dev	53.9505	24.8665	3.2270
14	105.3830	93.2440	0.6380
	407.7440	148.2750	8.9060
	194.8070	145.8590	7.5840
Mean	235.9780	129.1260	5.7093
Std Dev	126.8249	25.3916	3.6264
Fall 14	84.6140	92.5910	5.7950
	204.1080	80.2730	3.5400
	205.3200	98.0680	13.2650
Mean	164.6807	90.3107	7.5333
Std Dev	56.6178	7.4416	4.1561
	Cu(ug/g)	Zn(ug/g)	Cd(ug/g)
Fall 9	73.589	201.663	3.579
Fall 13	21.294	173.347	2.834
Fall 14	38.006	188.312	1.217

Table 4. Mean/Standard Deviation
Sediment Raw Data

Sediment Table

Site #	Cu(ppm)	Zn(ppm)	Cd(ppm)
1	59.5270	281.2920	2.3730
	71.8210	310.3800	1.8800
	65.6670	267.7510	1.9040
Mean	65.6717	286.4743	2.0523
Std Dev	5.0190	17.7848	0.2270
2	298.0800	404.9140	3.9970
3	249.2740	360.4270	2.8330
	239.6850	335.9160	3.2060
Mean	244.4795	348.1715	3.0195
Std Dev	4.7945	12.2555	0.1865
5	2902.8410	3187.5560	40.7740
	2764.6910	3044.0670	38.8750
	2431.5220	2632.8710	36.6660
Mean	2699.6847	2954.8313	38.7717
Std Dev	197.8295	235.0760	1.6787
6	1099.9820	1548.4490	22.0560
	1061.6820	1442.9930	19.6410
Mean	1080.8320	1495.7210	20.8485
Std Dev	19.1500	52.7280	1.2075
Fall 6	4019.4930	8369.9170	70.5370
7	533.0630	1089.7980	12.4130
	491.8960	955.7250	11.7610
Mean	512.4795	1022.7615	12.0870
Std Dev	20.5835	67.0365	0.3260
8	848.0740	1871.8990	28.2840
	834.8630	1855.1940	27.4430
	846.7900	1891.3840	28.0750
Mean	843.2423	1872.8257	27.9340
Std Dev	5.9482	14.7890	0.3575
Fall 8	1373.7430	4107.4020	36.6080
9	170.1460	520.2000	4.7380
Fall 9	698.6750	2922.0370	22.1100
10	145.6890	460.7560	4.1750
	120.9890	391.3280	3.3640
Mean	133.3390	426.0420	3.7695
Std Dev	12.3500	34.7140	0.4055

Sediment Table cont

Site	Cu(ppm)	Zn(ppm)	Cd(ppm)
11	337.0330	1009.5650	11.9700
	354.8860	840.3460	12.4480
	336.4690	852.5570	12.3510
Mean	342.7960	900.8227	12.2563
Std Dev	8.5520	77.0539	0.2063
13	157.5890	486.6240	4.6290
	120.4820	384.7800	3.6370
Mean	139.0355	435.7020	4.1330
Std Dev	18.5535	50.9220	0.4960
Fall 13	233.0360	1131.0460	6.7860
14	96.2370	309.5730	2.5570
	83.2500	256.6940	1.7260
Mean	89.7435	283.1335	2.1415
Std Dev	6.4935	26.4395	0.4155
Fall 14	576.1250	1291.2220	6.0930