

Influences of water and substrate quality for periphyton in a montane stream affected by acid mine drainage

Dev K. Niyogi

Center for Limnology, Cooperative Institute for Research in Environmental Sciences, University of Colorado, Boulder, Colorado 80309-0216

Diane M. McKnight

Institute for Arctic and Alpine Research, University of Colorado, Boulder

William M. Lewis, Jr.

Center for Limnology, Cooperative Institute for Research in Environmental Sciences, University of Colorado

Abstract

St. Kevin Gulch, a headwater stream of the Rocky Mountains of Colorado, receives acid mine drainage that maintains low pH, high concentrations of heavy metals, and high rates of metal hydroxide deposition. An acid-tolerant alga, *Ulothrix* sp., is present below the source of mine drainage in St. Kevin Gulch, but its biomass is limited by the deposition rates of iron hydroxides, which are especially high near the source. An experimental diversion of the mine drainage increased the quality of water and improved the substrate condition through a reduction of deposition rates. During the first year of the experiment, *Ulothrix* became abundant in this reach. During the second year, pH increased to the point at which aluminum hydroxides precipitated from the stream water onto the streambed; this change inhibited the growth of all periphyton, including *Ulothrix*. The deposition rate of aluminum hydroxides, however, was less than that of iron hydroxides in stream reaches with high *Ulothrix* biomass, suggesting that metal hydroxides vary by type in their effect on periphyton.

Acid mine drainage affects many streams throughout the world and causes a variety of stresses on aquatic ecosystems. Acidity in mine drainage is generated by the oxidation of sulfide minerals, such as pyrite, which produces sulfuric acid. High concentrations of heavy metals, including zinc, cadmium, and copper, may be found in waters that receive acid mine drainage. Additionally, deposition of metal oxide precipitates onto the streambed can affect biota in receiving waters. Our objective in this study was to examine how these types of stresses (acidity, dissolved metals, hydroxide deposition) and their interactions affect periphyton in a stream prior to and following diversion of acid mine drainage, as might occur through remediation of a stream segment.

The effects of mine drainage have been studied in a variety of aquatic ecosystems (Kelly 1988). As pH decreases, structural changes may occur in periphyton communities as acid-tolerant taxa become dominant. These community changes are similar to those occurring in aquatic systems affected by acidic precipitation (Elwood and Mulholland 1989; Planas 1996). Metal toxicity also alters the structure of periphyton communities (Clements 1991; Genter 1996). While species composition of periphyton may change as pH declines or metal concentrations increase, community biomass may be stable or even increase with the release of

grazing pressure from invertebrates, which are usually more sensitive than some algae to the changes in water chemistry (Elwood and Mulholland 1989).

Ecological effects of the deposition of metal hydroxides have been less studied than metal or acid toxicity. McKnight and Feder (1984) found that the deposition of iron and aluminum hydroxides below the confluence of an acidic and a neutral stream had larger effects on periphyton and invertebrates than pH or dissolved metals. Sheldon and Skelly (1990) found that deposition of metal oxides associated with bacteria in a stream below a source of iron and manganese limited growth and colonization by algae. Precipitation of metal hydroxides is a characteristic feature of some streams receiving mine drainage. For example, the term “yellow boy” was used by miners to describe the characteristic color of metal precipitates on streambeds.

In streams receiving acid mine drainage, the chemical characteristics and deposition rate of metal oxides vary spatially and temporally. The deposition rate is typically greatest near the inflow of mine seeps or at the confluence of streams carrying mine drainage with pristine streams because at these locations an increase in pH causes the solubility of the metal oxides to be exceeded. Kimball et al. (1994) showed that the concentration of dissolved iron decreased rapidly below a mine seep inflow because of precipitation of iron oxide and that the decrease could be modeled as a first-order process. Similarly, an experiment in which the pH of an acid mine drainage stream was raised to 6.2 showed that aluminum hydroxide precipitation occurred primarily in the first 50 m below the point at which pH increased (Broshears et al. 1996).

Acknowledgments

This research was supported by the Toxic Substances Hydrology Program of the U.S. Geological Survey and the University of Colorado.

We acknowledge assistance in algal identification by D. Dufford and S. Spaulding, and comments on the manuscript by B. Kimball, B. Kreiser, A. West, and two anonymous reviewers.

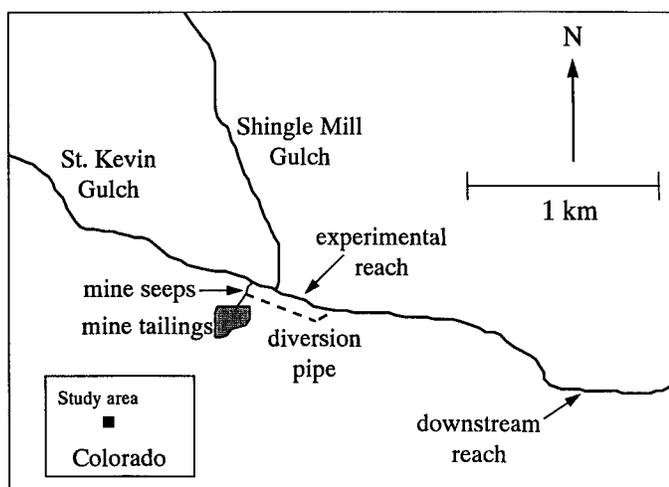


Fig. 1. Map of St. Kevin Gulch, west-central Colorado.

The net deposition rate of metal hydroxides reflects the rate at which the precipitate forms and coagulates to a size large enough to be deposited on the streambed, as well as the stability of the precipitate after its deposition. These processes are influenced by the stream chemistry, especially pH and DOC concentration, and by hydrologic and environmental conditions. For iron oxides, photoreduction of the oxide surface releases ferrous iron back to the water and slows the rate of deposition and crystallization of the iron oxides on the streambed (McKnight et al. 1988). Aluminum oxides are not subject to photoreduction and may be more stable once deposited.

We examined controls on periphyton recovery following an experimental diversion of acid mine drainage in St. Kevin Gulch, a headwater stream in the Rocky Mountains of Colorado. We report here results from several years of monitoring the periphyton communities, water chemistry, and rates of metal hydroxide deposition in this experimental reach.

Material and methods

Study site—St. Kevin Gulch, which is near Leadville, Colorado, drains a basin of about 10 km² in the Upper Arkansas River watershed. Several abandoned mines along St. Kevin Gulch contribute acidity and metals to the stream. The main source of mine drainage is a mine dump at an elevation of about 3,200 m (Fig. 1). Upstream of this source, other mines and natural weathering reactions account for some acidity and metals in the stream. A pristine tributary of neutral pH, Shingle Mill Gulch, joins St. Kevin Gulch just downstream of the main mine drainage source (Fig. 1). The two streams have approximately equal discharge, and the flow below the confluence is about 10 liters s⁻¹ under baseflow conditions.

Downstream of the mine on St. Kevin Gulch, periphyton is present where the pH of the water is <4 and concentrations of dissolved zinc are >10 mg liter⁻¹. A filamentous green alga, *Ulothrix* sp., comprises more than 99% of the biomass (McKnight 1988). After peak discharge from snow-melt (usually in June), *Ulothrix* accumulates throughout

summer until fall when ice cover begins. Water temperatures in St. Kevin Gulch during summer vary between 2°C at night and 12°C during warm days. Concentrations of dissolved nutrients are low during baseflow (PO₄-P, 2.0 μg liter⁻¹; inorganic N, <20 μg liter⁻¹; Tate et al. 1995).

Experimental procedures—In July 1994, we set up an experimental diversion of the main source of mine drainage in the watershed. Mine drainage from several seeps was collected in a plastic pipe and released back into St. Kevin Gulch about 100 m downstream, which was 80 m below the confluence with Shingle Mill Gulch (Fig. 1). The diversion collected about 75% of the mine drainage from this area, as estimated by changes in metal loading to the stream (unpubl. data). The remaining 25% continued to flow into the stream, primarily as groundwater. Our goal was to measure changes in stream chemistry and stream communities in this experimental reach following reduction in the amount of mine drainage. We also monitored a downstream reach, which was located about 1.3 km downstream from the confluence with Shingle Mill Gulch. This downstream reach was also dominated by *Ulothrix* sp., and its water quality was not altered by the experimental diversion. We collected data on water and substrate quality and periphyton under baseflow conditions during late summer (late August to early September) in 1993 (prior to the diversion), 1994 (just after the diversion), 1995, and 1996.

Sampling and analysis of water and sediment follow methods of Kimball et al. (1994). The pH was measured in the field with an ion-specific electrode. Water samples for analysis of dissolved metals were filtered through nitrocellulose membrane filters (0.1-μm pore size), acidified with ultrapure nitric acid, and analyzed by ICP-AES.

The rate of metal hydroxide deposition was estimated as the rate of accumulation of ash mass that was collected from cobbles placed in the stream for known periods of time. Deposition rates were assumed to be constant over time and were calculated as the ash mass per unit rock area per day. Preliminary data indicated that deposition rates do change with flow, but are relatively constant during baseflow conditions, as measured in this study. Cobbles from the stream were used and are the main substrate type in St. Kevin Gulch. During deployment, cobbles were brushed clean and placed in the stream in areas of surface velocity >10 cm s⁻¹ and not in the few pools found along the stream where deposition rates were much higher. After 3–6 weeks, the same cobbles were carefully removed from the stream and placed in a bag. In the laboratory, deposited material was rinsed and brushed from the cobbles with water into a tared weighing boat for determination of dry and ash mass. Samples were dried at 60°C to a constant weight, ashed at 450°C for 24 h, and reweighed. Subsamples of the deposited material were also used to determine the amount of metals (Fe, Al) as a percentage of dry or ash mass. Subsamples were analyzed by ICP-AES or atomic absorption spectrometry after acid digestion (Kimball et al. 1994).

We also estimated the loss of aluminum from the dissolved phase as an indication of the amount of precipitation in the experimental reach. Aluminum enters the experimental reach from St. Kevin Gulch and Shingle Mill Gulch and

Table 1. Characteristics of sites in St. Kevin Gulch. Discharge values are averages of daily mean flows during August and September. pH and concentrations of dissolved zinc are representative of baseflow during late summer (August–September). Deposition rates are means and standard errors (SE) of 4–8 replicates.

Reach, year	Discharge (liters s ⁻¹)	pH	Dissolved Zn (mg liter ⁻¹)	Deposition rate (g m ⁻² d ⁻¹)	
				Mean	SE
Exptl, 1993	12.5	3.7	10	1.54	0.08
Exptl, 1994	9.1	4.5	5.0	0.31	0.05
Exptl, 1995	19.8	5.4	2.0	0.34	0.04
Exptl, 1996	15.0	4.9	3.2	0.20	0.02
Downstream, 1993–1996	—	3.7	10	0.41	0.06

from groundwater. Because flow rates in Shingle Mill Gulch and St. Kevin Gulch above the confluence are typically equal, we can estimate the concentration of aluminum transported into the experimental reach from upstream sources (McKnight et al. 1992). We also can estimate the contribution of aluminum from groundwater sources entering the experimental reach. Because zinc transport is conservative (Kimball et al. 1994) and concentrations of zinc and aluminum in groundwater are known, we can calculate the groundwater aluminum contribution from the concentrations of dissolved zinc through the experimental reach. We then can calculate potential concentrations of dissolved aluminum, from conservative mixing (without precipitation), and compare these to actual measured values.

Periphyton biomass was quantified as amount of chlorophyll *a* per unit area. Algae were scraped and pooled from three rocks collected at random locations along a transect. There were 10 replicated transects at each site. Algae were filtered onto glass-fiber filters and analyzed spectrophotometrically for chlorophyll *a* following hot ethanol extraction (Lewis et al. 1984). Rock area was estimated from the mass

of aluminum foil required to cover the surface of the rock and correcting for foil mass from a measured area (Steinman and Lamberti 1996).

Results and discussion

In 1993, below the confluence with Shingle Mill Gulch, St. Kevin Gulch had a pH <4 and the concentration of dissolved zinc was ~10 mg liter⁻¹ for several kilometers downstream (Table 1). The rate of metal hydroxide deposition, however, decreased with distance from the mine (Fig. 2). Periphyton biomass accumulated only when the deposition rate decreased below ~1.0 g m⁻² d⁻¹. Periphyton biomass reached more than 80 mg Chl *a* m⁻² where the deposition rate was low, despite the low pH and high concentrations of dissolved zinc. Species composition was dominated by *Ulothrix* sp., which comprised >99% of algal cells and biovolume. During all 4 yr of monitoring, algal biomass was high in the downstream reach (Fig. 3). The values for pH, dissolved zinc, and deposition rate of metal oxides in the down-

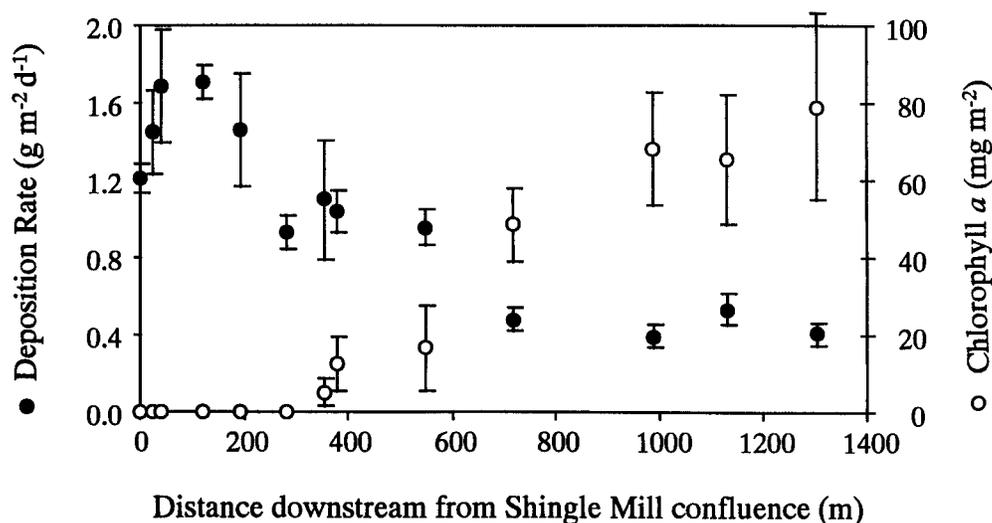


Fig. 2. Changes in deposition rate of metal oxides (●) and algal biomass (as chlorophyll *a*—○) in St. Kevin Gulch downstream of its confluence with Shingle Mill Gulch. Data are means ± 1 SE for 3–10 replicates collected late August to early September 1993. The downstream reach is 1,300 m below the confluence with Shingle Mill Gulch.

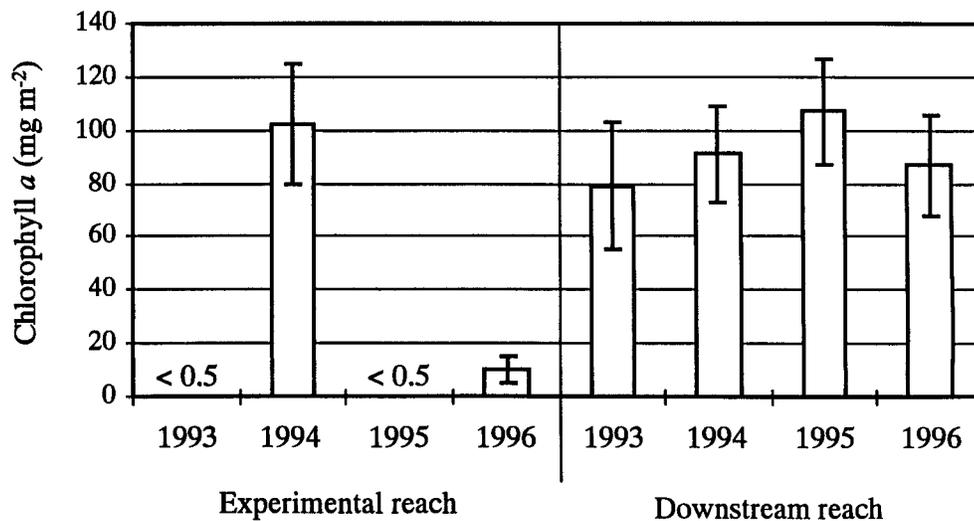


Fig. 3. Annual changes in algal biomass (as chlorophyll *a*) between 1993 and 1996 in the experimental and downstream reaches in St. Kevin Gulch. Data are means \pm 1 SE for 10 replicates collected late August to early September of each year.

stream reach were similar in all years monitored to the values for 1993 (Table 1).

The composition and distribution of periphyton in St. Kevin Gulch demonstrate the importance of multiple stresses associated with acid mine drainage. *Ulothrix* sp. is tolerant of the low pH and high concentrations of dissolved metals characteristic of St. Kevin Gulch. In fact, *Ulothrix* sp. is abundant even in acidic seeps undiluted by stream water. Other tolerant taxa, especially the diatom *Eunotia exigua*, can be found in these seeps as well (McKnight 1988). Algal biomass in the seeps can be extremely high (>500 mg Chl *a* m⁻²), despite extremely low pH (2.7) and high concentrations of zinc (100 mg liter⁻¹). Algal biomass and primary production rates in downstream reaches of St. Kevin Gulch and the acidic seeps exceed those of pristine reference streams nearby (unpubl. data).

The deposition of metal precipitates onto the streambed is the main factor controlling the biomass of *Ulothrix* downstream of the source of mine drainage in St. Kevin Gulch (Fig. 2). Other factors, including pH, concentrations of zinc and nutrients, and light levels, were relatively constant for 1.5 km downstream of the mine. As the deposition rate decreased to <1.0 g m⁻² d⁻¹, *Ulothrix* was able to accumulate and grow on rocks, wood, or metal precipitates on the streambed.

During summer 1993, before the experimental diversion, no periphyton accumulated in the 200 m downstream from the source of mine drainage for St. Kevin Gulch (Fig. 2), including the experimental reach (Fig. 3). Following the diversion in 1994, pH increased in the experimental reach to 4.5, and dissolved zinc decreased to 5 mg liter⁻¹. While residual iron hydroxides were still present on the streambed, the deposition rate had decreased to about 0.3 g m⁻² d⁻¹. Periphyton quickly colonized the experimental reach and was comprised almost exclusively of *Ulothrix* sp. ($>99\%$). Within 6 weeks of the diversion, *Ulothrix* biomass exceeded 100 mg Chl *a* m⁻² in this reach (Fig. 3).

In summer 1995, a treatment system at another mine in the upper watershed increased the pH of St. Kevin Gulch upstream of the diversion. In addition, greater winter snowpack in St. Kevin Gulch in 1995 led to a higher flow (Table 1), which increased pH and decreased concentrations of metals. As a result, pH in the experimental reach was higher (5.4) and concentrations of dissolved zinc were lower (Table 1). The deposition rate of metal hydroxides was not significantly different from 1994. Even so, no periphyton accumulated in the reach during summer (Fig. 3).

In summer 1996, pH was lower in the experimental reach than in 1995 and the deposition rate decreased to 0.20 g m⁻² d⁻¹. Periphyton (*Ulothrix* sp.) did accumulate in the reach during summer, but biomass levels remained lower than those observed in the downstream reach or in the experimental reach in 1994 (Fig. 3).

To summarize, the diversion of mine drainage created an experimental reach with improved quality of water and substrate. During the first year of the diversion, 1994, the pH increased from 3.7 to about 4.5, and the deposition rate of metal precipitates decreased from 1.5 to 0.31 g m⁻² d⁻¹. *Ulothrix* quickly became established in this reach and attained high biomass within several weeks of the diversion. The following 2 yr showed little or no periphyton growth in the experimental reach, despite a further increase in pH and a similar deposition rate of metal precipitates. The downstream reach in St. Kevin Gulch had higher biomass of *Ulothrix* for all years studied than the experimental reach did in 1995 or 1996 (Fig. 3), even though the downstream reach had lower pH, higher dissolved zinc concentrations, and higher deposition rates.

The seemingly contradictory periphyton growth responses may be explained by a change in deposition from iron to aluminum hydroxides in the experimental reach. Aluminum hydroxides generally precipitate at a higher pH (about 4.9: Nordstrom and Ball 1986) than iron hydroxides. Table 2 presents potential concentrations of dissolved aluminum based

Table 2. Dissolved aluminum (mg liter^{-1}) and pH in the experimental reach from 1994 to 1996: potential concentrations from conservative mixing (if there were no precipitation) and measured concentrations. Calculations are explained in methods.

Year	Upstream St. Kevin Gulch		Shingle Mill Gulch		Ground-water Al contrib.	Experimental reach		
	Al	pH	Al	pH		Pot. Al	Measured	
							Al	pH
1994	3.59	3.8	0.08	6.8	1.40	3.23	3.17	4.5
1995	0.87	4.7	0.03	6.9	0.55	1.00	0.12	5.4
1996	3.09	4.1	0.06	6.8	1.08	2.65	1.94	4.9

on conservative mixing and measured concentrations in the experimental reach. In 1994, most aluminum remained dissolved upon mixing in the experimental reach. In 1995, however, aluminum did not behave conservatively, as only a fraction remained dissolved at the higher pH (5.4) of the water below the confluence. The remainder of the aluminum was deposited on the streambed. In 1996, a slightly smaller amount of aluminum was lost from the dissolved phase in the experimental reach.

Elemental composition of the deposited metal compounds onto the streambed confirms the difference between precipitates in the experimental and downstream reaches (Table 3). In 1996, iron comprised 31.5% of the ash mass of material deposited onto cobbles at the downstream site, and aluminum made up only 2.8%. In the experimental reach in 1996, aluminum comprised 34.6% of the ash mass, and iron made up 7.5%. A change in the color of the precipitates from orange (iron hydroxides) to white (aluminum hydroxides) also confirmed the increase in aluminum deposition in 1995 and 1996. These results agree with previous studies on sediment chemistry in St. Kevin Gulch (Smith et al. 1991), which characterize precipitates in the downstream reach as about 25% iron and 5% aluminum of dry mass (ash mass is typically about 75% of dry mass at this site). Broshears et al. (1996) documented a precipitation of aluminum compounds in St. Kevin Gulch following an experimental increase in pH.

Previous studies have demonstrated the ecological significance of metal oxide deposition in controlling the composition and abundance of stream periphyton (Sode 1983; McKnight and Feder 1984; Sheldon and Skelly 1990; Wellnitz and Sheldon 1995). Deposition of metal oxides has obvious physical effects on the streambed, but little is known about the exact mechanisms by which it limits colonization by epilithic biota. It is interesting that only one algal species, *Ulothrix* sp., can flourish in St. Kevin Gulch under conditions of iron hydroxide deposition. Other algal species, in-

cluding a few diatom and cyanobacterial species, are found rarely in the downstream reach of St. Kevin Gulch (McKnight 1988). While iron hydroxides apparently prevent colonization of the streambed by most algal species, *Ulothrix* sp. is able to grow in association with them; however, even *Ulothrix* filaments can become covered by iron hydroxides if they do not grow faster than the rate of deposition.

It is uncertain why the type of metal hydroxide, iron vs. aluminum, would have such a large effect on periphyton abundance in our experimental reach. Although dissolved aluminum is generally more toxic than iron (Genter 1996), high concentrations of dissolved aluminum (1 mg liter^{-1} in the downstream reach and $>10 \text{ mg liter}^{-1}$ in the acidic seeps) do not limit *Ulothrix* biomass at our study sites. One difference between iron and aluminum hydroxides is that iron is photoreduced in St. Kevin Gulch (McKnight et al. 1988), but aluminum is not. When iron hydroxides settle onto *Ulothrix* filaments, they may redissolve if the iron hydroxides are photoreduced to more soluble ferrous iron. Deposition rates on algae may be lower than on rocks, in part due to their greater exposure to sunlight. Additionally, *Ulothrix* may be a source of oxidizable carbon, through exudation, that enhances photoreduction of iron from filaments. This would cause our iron deposition rate measurements to overestimate the actual rate of deposition on *Ulothrix*. In contrast, aluminum hydroxides are not subject to photochemical dissolution and would thus present a greater stress on growing periphyton.

Another explanation is that *Ulothrix* can grow in association with iron hydroxides, but not aluminum hydroxides, because of a physical difference in the precipitates. Some of the *Ulothrix* present in the experimental reach in 1996 grew attached to branches and other detritus, as opposed to the precipitate-covered rocks on the streambed.

Iron and aluminum hydroxides also have different effects on *Ulothrix* biomass in other Colorado streams affected by mine drainage (unpub. results). These streams show characteristic zones of orange precipitates (iron hydroxides) with *Ulothrix* and sometimes other algae, and zones of white precipitates (aluminum hydroxides) with little or no algae. We are currently investigating possible mechanisms that may explain the differential effects of aluminum vs. iron hydroxides in St. Kevin Gulch and other streams affected by mine drainage.

Conclusions

The experimental diversion of mine drainage in St. Kevin Gulch demonstrates the interactions between water quality

Table 3. Aluminum and iron composition (percent of ash mass) of precipitates collected on cobbles from experimental and downstream reaches in 1996.

Reach, year	Percent Al		Percent Fe	
	Mean	SE	Mean	SE
Exptl, 1996	34.6	3.8	7.5	0.7
Downstream, 1996	2.8	1.3	31.5	2.5

and substrate quality in streams affected by acid mine drainage. In acidified environments, the pH of the water is often seen as the primary variable that affects aquatic biota, but pH also acts as a master variable that affects many other chemical and physical factors in aquatic ecosystems (e.g., Frost et al. 1999). In St. Kevin Gulch, the slight increase in pH from 1994 to 1995 led to a change in metal hydroxide deposition that limited periphyton growth. In general, hydrologic and chemical factors determined water quality, which in turn affected substrate conditions. The combined effect of both water quality and substrate conditions subsequently controlled periphyton in this stream.

These interactions demonstrate that acid mine drainage is a mechanistically complex agent of stress; its low pH affects the biota not only directly, but also indirectly through substrate quality because of the pH dependence of metal hydroxide solubility. Because of the interaction between quality of water and substrate, recovery of streams affected by acid mine drainage may be different than recovery from single stresses. Different stresses may become important at different stages during recovery, as highly acidic sites go through a stage of precipitation and deposition of iron oxides and then aluminum oxides, en route to achieving neutral pH and low concentrations of dissolved metals. The responses of stream biota to the varying physical and chemical stresses imposed by acid mine drainage will thus be complex and variable in nature.

References

- BROSHEARS, R. E., R. L. RUNKEL, B. A. KIMBALL, D. M. MCKNIGHT, AND K. E. BENCALA. 1996. Reactive solute transport in an acidic stream: Experimental pH increase and simulation of controls on pH, aluminum, and iron. *Environ. Sci. Technol.* **30**: 3016–3024.
- CLEMENTS, W. H. 1991. Community responses of stream ecosystems to heavy metals: A review of observational and experimental approaches, p. 363–391. *In* M. C. Newman and A. W. McIntosh [eds.], *Ecotoxicology of metals: Current concepts and applications*. CRC.
- ELWOOD, J. W., AND P. J. MULHOLLAND. 1989. Effects of acidic precipitation on stream ecosystems, p. 85–135. *In* D. C. Adriano and A. H. Johnson [eds.], *Acid precipitation*. V. 2. Springer.
- FROST, T. M., AND OTHERS. 1999. Multiple stresses from a single agent: Diverse responses to the experimental acidification of Little Rock Lake, Wisconsin. *Limnol. Oceanogr.* **44**: 784–794.
- GENTER, R. B. 1996. Ecotoxicology of inorganic chemical stress to algae, p. 403–468. *In* R. J. Stevenson et al. [eds.], *Algal ecology: Freshwater benthic ecosystems*. Academic.
- KELLY, M. 1988. *Mining and the freshwater environment*. Elsevier.
- KIMBALL, B. A., R. E. BROSHEARS, K. E. BENCALA, AND D. M. MCKNIGHT. 1994. Coupling of hydrologic transport and chemical reactions in a stream affected by acid mine drainage. *Environ. Sci. Technol.* **28**: 2065–2073.
- LEWIS, W. M., JR., J. F. SAUNDERS III, D. W. CRUMPACKER, SR., AND C. M. BRENDENCKE. 1984. *Eutrophication and land use*. Springer.
- MCKNIGHT, D. M. 1988. Metal-tolerant algae in St. Kevin Gulch, Colorado, p. 113–117. *In* U.S. Geol. Surv. Open-File Rep. 87-764.
- , AND OTHERS. 1992. Sorption of dissolved organic carbon by hydrous aluminum and iron oxides occurring at the confluence of Deer Creek with the Snake River, Summit County, Colorado. *Environ. Sci. Technol.* **26**: 1388–1396.
- , AND G. L. FEDER. 1984. The ecological effect of acid conditions and precipitation of hydrous metal oxides in a Rocky Mountain stream. *Hydrobiologia* **119**: 129–138.
- , B. A. KIMBALL, AND K. E. BENCALA. 1988. Iron photo-reduction and oxidation in an acidic mountain stream. *Science* **240**: 637–640.
- NORDSTROM, D. K., AND J. W. BALL. 1986. The geochemical behavior of aluminum in acidified surface waters. *Science* **232**: 54–56.
- PLANAS, D. 1996. Acidification effects, p. 497–530. *In* R. J. Stevenson et al. [eds.], *Algal ecology: Freshwater benthic ecosystems*. Academic.
- SHELDON, S. P., AND D. K. SKELLY. 1990. Differential colonization and growth of algae and ferromanganese-depositing bacteria in a mountain stream. *J. Freshwater Ecol.* **5**: 475–485.
- SMITH, K. S., J. F. RANVILLE, AND D. L. MACALADY. 1991. Predictive modeling of copper, cadmium, and zinc partitioning between streamwater and bed sediment from a stream receiving acid mine drainage, St. Kevin Gulch, Colorado, p. 380–386. *In* U.S. Geol. Surv. Water Resour. Invest. Rep. 91–4034.
- SODE, A. 1983. Effect of ferric hydroxide on algae and oxygen consumption by sediment in a Danish stream. *Arch. Hydrobiol. Suppl.* **65**, p. 134–162.
- STEINMAN, A. D., AND G. A. LAMBERTI. 1996. Biomass and pigments of benthic algae, p. 295–313. *In* F. R. Hauer and G. A. Lamberti [eds.], *Methods in stream ecology*. Academic.
- TATE, C. M., R. E. BROSHEARS, AND D. M. MCKNIGHT. 1995. Phosphate dynamics in an acidic mountain stream: Interactions involving algal uptake, sorption by iron oxide, and photoreduction. *Limnol. Oceanogr.* **40**: 938–945.
- WELLNITZ, T. A., AND S. P. SHELDON. 1995. The effects of iron and manganese on diatom colonization in a Vermont stream. *Freshwater Biol.* **34**: 465–470.