Effects of Mining on Benthic Macroinvertebrate Communities and Monitoring Strategy

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Chapter E20 of

Integrated Investigations of Environmental Effects of Historical Mining in the Animas River Watershed, San Juan County, Colorado

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Chapter E20
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Abstract

The benthic macroinvertebrate community of the Animas River watershed study area has been impaired by more than a century of hard-rock mining. Currently, remediation of mine sites is being done to restore ecosystem health in this watershed. Ecological recovery of the community depends on the degree that remediation improves water quality and reduces erosion of mill tailings into the river, resulting in improved habitat quality. Benthic macroinvertebrate data were collected in fall 1996, spring 1997, and fall 1997 to measure existing ecosystem health and to evaluate the response of the benthic community to remediation activities. Macroinvertebrate samples were obtained from 25 impaired and 11 unimpaired sites. Sampling methods were proposed to determine time of year, frequency, and location for collection of pre-remediation data and to qualitatively and statistically compare those data to the post-remediation data. Two methods were proposed: (1) a single site assessment where only one sample site exists downstream from each remediation site, and (2) a multiple site assessment where a minimum three sample sites are used for each remediation site. At least 3 years of pre-remediation data and 3 years of post-remediation data from the same site must be collected to distinguish between the effectiveness of remediation and temporal variation inherent in natural populations. If post-remediation data are not significantly different from pre-remediation data, then subsequent data collection should be delayed until further remediation is completed, the benthos has had more time to recover, or macroinvertebrates have had more time to recolonize the impaired reaches. If a significant difference is evident between pre- and post-remediation data, a minimum of 2 years of additional post-remediation data should be collected.

Introduction

The Animas River watershed study area has been extensively affected by more than a century of hard-rock mining, resulting in impaired benthic macroinvertebrate and fish communities throughout the watershed. The purpose of remediating abandoned mine sites is to restore ecosystem health (von Guerard and others, this volume, Chapter B). Two indicators of aquatic ecosystem health are species composition and species diversity, which researchers can evaluate by determining number of taxa (taxa richness) and relative densities of the taxa or the taxa evenness (May, 1988). In general, healthy aquatic ecosystems are reflected by high numbers of taxa and even taxa distribution across their habitat.

Benthic biological communities include periphyton, macroinvertebrates, and some species of fish. Periphyton (or biofilm) is composed mostly of the primary producer, algae, and grows on top of rocks and other benthic substrates. Benthic macroinvertebrates feed on periphyton, dead organic material and a wide range of small organisms, and are themselves an important food resource for fish, amphibians, reptiles, birds, and mammals. Of the fish, some species, such as suckers and fathead minnows, graze primarily upon periphyton. Trout and speckled dace, however, rely primarily upon benthic macroinvertebrates for their food.

Acid mine drainage may reduce species diversity in stream ecosystems through both acute and chronic mechanisms, including toxicity of metals in stream water, sediment, and interstitial water (Besser and others, this volume, Chapter D; Besser and Brumbaugh, this volume, Chapter E18; Besser and Leib, this volume, Chapter E19) as well as physical effects of metal-bearing precipitates (colloids) on aquatic organisms and their habitats (Milhous, this volume, Chapter E21).

One of the major goals of the Animas River watershed study was to evaluate the effectiveness of remediation (Buxton and others, 1997). Although monitoring water chemistry in streams provides information on exposure concentrations and chemical loading, it does not directly address species composition and diversity or the quality of benthic community habitat (Karr and others, 1986; Yoder and Rankin, 1999). Because of the abundance and mobility of benthic macroinvertebrates, coupled with their ability to recolonize impaired ecosystems, this group of taxa is commonly selected for use in monitoring studies of aquatic ecosystem health throughout the United States (Davis and others, 1996). The methods for collection, analysis, and interpretation of benthic macroinvertebrate data are well established through works including the Rapid Bioassessment Protocol (RBP) developed for the Environmental Protection Agency (Shackleford, 1988; Plafkin and others, 1989; Barbour and others, 1992, 1995, 1996; Hayslip, 1993), the benthic Index of Biotic Integrity (IBI; Kerans and Karr, 1994; Fore and others, 1996), and the Invertebrate Community Index (DeShon, 1995). Their use as indicators of water and habitat quality has been justified by Hutchinson (1993), Karr and Chu (1999), Resh and Jackson (1993), and Rosenburg and Resh (1993). In addition, long holding times for preserved samples and the establishment of voucher collections that may be evaluated by other investigators for quality control are advantageous. Moreover, using macroinvertebrates for monitoring and assessing the health of an aquatic ecosystem directly addresses the concept of species diversity by recognizing that benthic macroinvertebrates integrate a variety of variables of concern over both space (such as benthic habitat and water quality) and time (because they live in the stream year-round and any ephemeral change in water quality will appear in the composition of the macroinvertebrate community). Limitations to the use of macroinvertebrates for monitoring include the relative difficulty of sampling. In contrast to invertebrates, periphyton is relatively difficult to identify.

Community metrics can be calculated from counts of individuals or counts of individual taxa in an ecosystem. Most metrics incorporate knowledge of the type of taxa, taxa richness, and relative numbers (evenness). There have been several attempts to apply a single number to taxa richness and evenness. The most widely accepted attempt is the Shannon-Weiner diversity index. The problem is that such an integrative index loses important information and may not wholly define and encompass all the intricacies of a macroinvertebrate community (Hurlbert, 1971; Purvis and Hector, 2000).

How the community will respond to remediation efforts (Stone and Wallace, 1998; O'Neill, 1999) is also unknown. The response will depend on existing macroinvertebrate communities within the watershed as well as the effect that remediation has on water and substrate quality. Potential responses include additions of or increasing densities of intolerant taxa and (or) loss of or decreasing densities of tolerant taxa. Therefore, descriptions of the macroinvertebrate community data should include species identification and the number of taxa as well as relative densities With this information, a variety of community metrics may be used to evaluate the effectiveness of remediation and to compare pre-remediation data to post-remediation data (Resh and others, 1988). Because the exact response of the macroinvertebrate community to remediation is unknown, identifying specific metrics for the purpose of evaluating the effectiveness of remediation cannot be completed until post-remediation data are obtained and thoroughly compared to the pre-remediation data. Distinguishing between temporal variability and actual recovery of the ecosystem is also necessary to accurately evaluate the effectiveness of remediation (Chapman, 1999).

Other processes may also help in evaluation of the effectiveness of remediation and may help identify underlying or mechanistic reasons for recovery or lack thereof. They include measurements at the suborganismal (for example, bioassays, Besser and Leib, this volume) to the ecosystem level (for example, nutrient cycles, Stone and Wallace, 1998; Adams and others, 2002; Resh and others, 1988). However, such procedures may not be feasible because of their complexity and cost. In conjunction with other research, both included in this volume and by other agencies, this study is an attempt to meet some of the objectives outlined by Michener (1997), Kondolf (1995), and Kondolf and Micheli (1995) in assessment of the effectiveness of remediation practices.

Purpose and Scope

The primary purpose of this study was to

- Gather benthic macroinvertebrate pre-remediation data to evaluate whether mine-site remediation in the Animas River watershed study area will increase the diversity of benthic macroinvertebrates

- Propose guidelines for conducting post-remediation sampling of the benthic community in the Animas River

- Propose methods to evaluate the effectiveness of remediation using biological data.

Acknowledgments

I acknowledge the help of Dave Gerhardt of the USDA Forest Service, San Juan National Forest; William Simon, Animas River Stakeholders; and Barbara Horn, Colorado Division of Wildlife. Each contributed significantly to the design and implementation of this research. The San Juan National Forest provided funds, personnel, and equipment for the collection and processing of the macroinvertebrate samples. Bob Brantlinger and Jamey O'Leary helped considerably in the data collection.

Methods

We used biomonitoring and assessment methods to obtain pre-remediation data in 1996 and 1997 at 25 impaired and 11 reference sites, from the northernmost headwaters of the Animas River (the watershed study area as such) to the confluence with the San Juan River (figs. 1 and 2; table 1). The pre-remediation data were collected prior to the bulk of

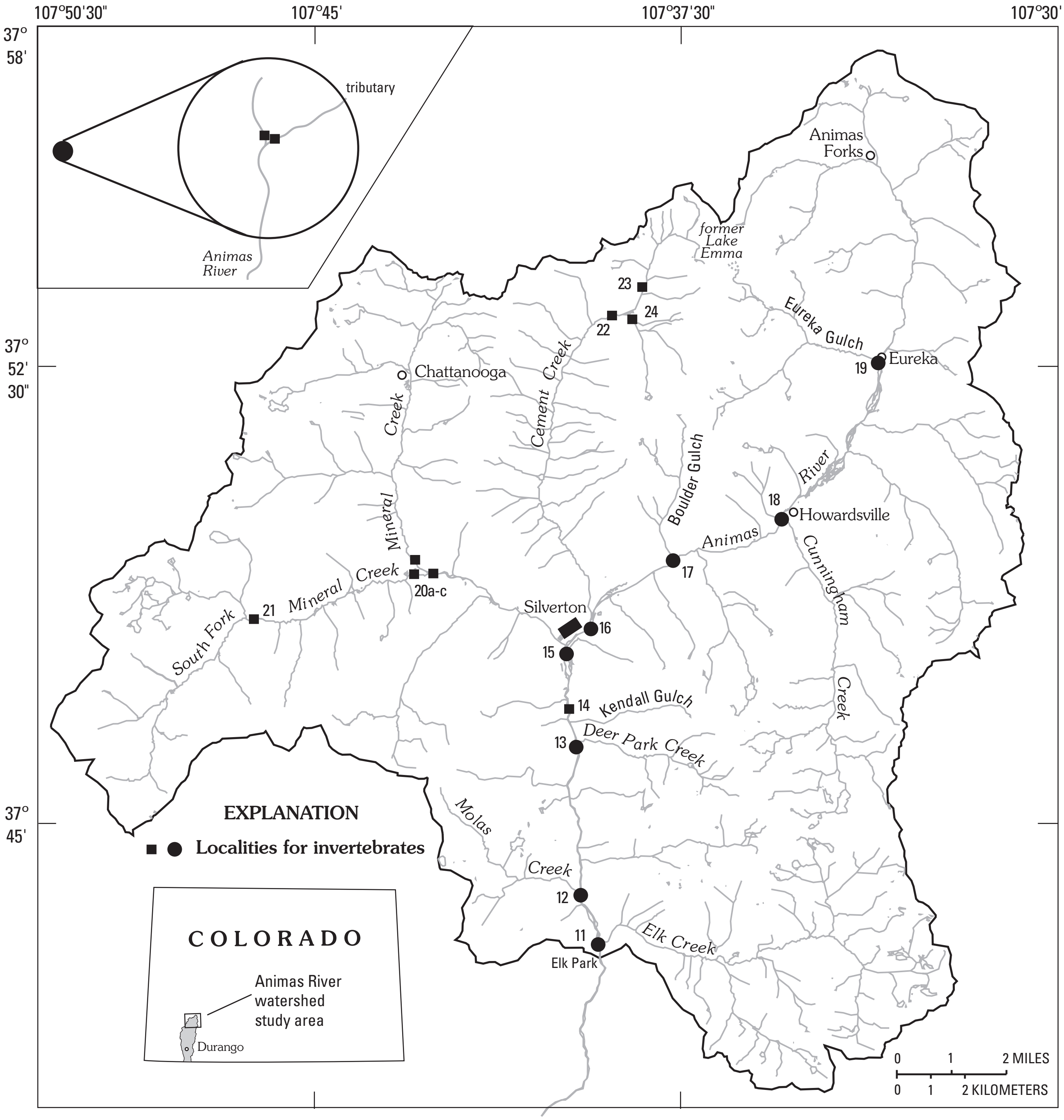
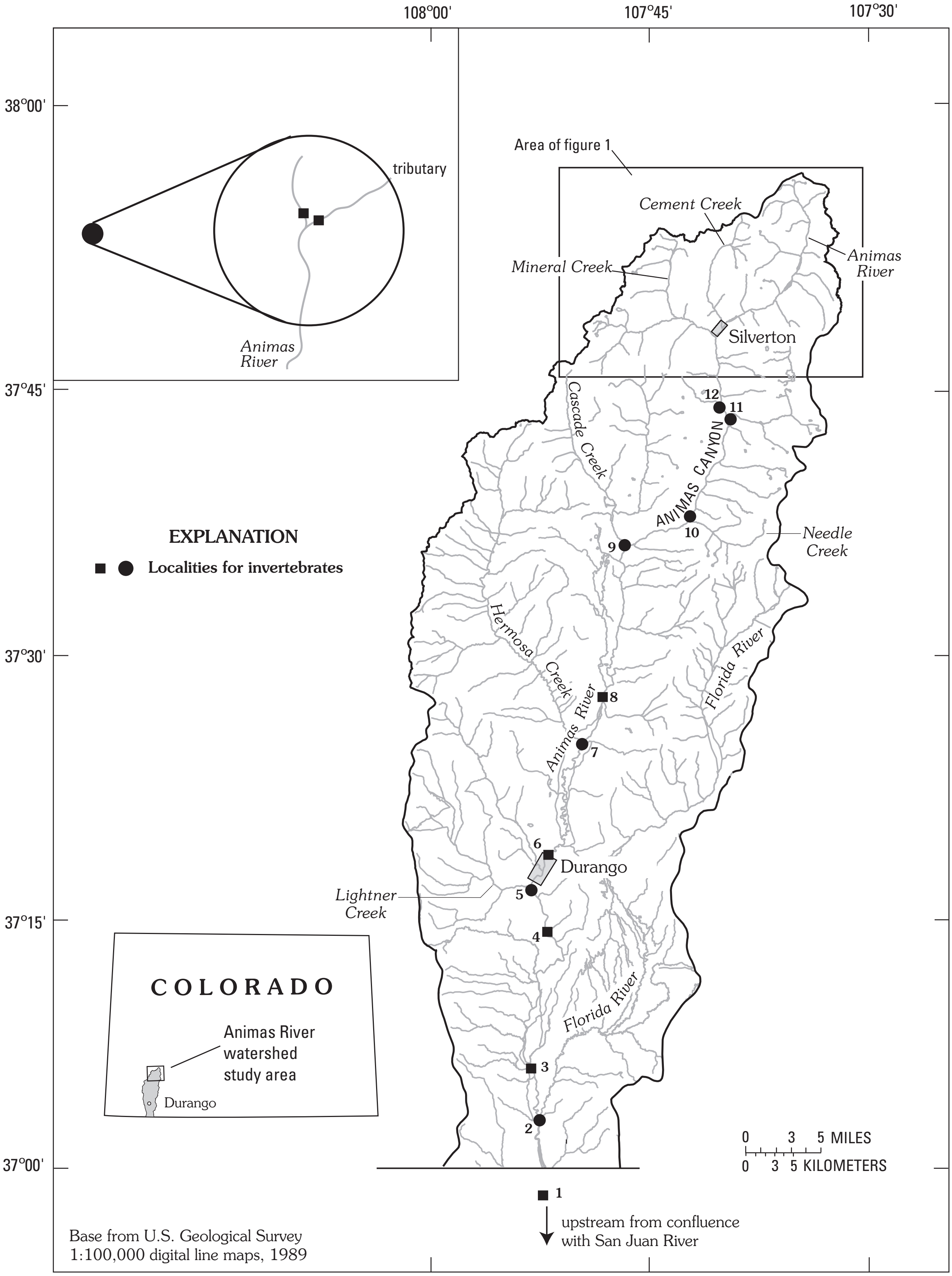


Figure 1. Sample sites, Animas River watershed study area, from confluence of Elk Creek north to the headwaters of the study area streams. Dot indicates two samples taken at site, one in mainstem upstream of confluence and one in the tributary (see inset in figure).


108°00'
107°45'
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tributary
Animas
River
Area of figure 1
Cement Creek
Mineral Creek
Animas
River
Silverton
12
11
Cascade Creek
ANIMAS CANYON
10
9
Needle
Creek
EXPLANATION
Localities for invertebrates
Hermosa
Creek
Florida River
Animas River
8
7
6
Durango
Lightner
Creek
5
4
COLORADO
Animas River
watershed
study area
Durango
Florida River
3
2
0 3 5 MILES
0 3 5 KILOMETERS
1
upstream from confluence
with San Juan River
Base from U.S. Geological Survey
1:100,000 digital line maps, 1989

remediation work (Finger and others, this volume, Chapter F). Sites were sampled in fall 1996, and spring and fall 1997. The majority of the sample sites were at major tributaries of the Animas River where, for the response variable, samples were taken from the Animas River immediately upstream from the tributaries. These sample sites were chosen for two reasons: (1) to use the tributary as a reference and (2) to avoid mixing zones. Where tributaries were far apart, especially downstream from the Animas River canyon, samples were taken at established sample sites that were not necessarily associated with a tributary but where we knew that data had been collected previously (Peter Butler, Robert Owen, and William Simon, Unpublished report to Colorado Water Quality Control Commission, Animas River Stakeholders Group, 2001).

Reference samples were obtained for the purpose of measuring potential community metrics given various levels of remediation. Each tributary station had similar physical habitat features as the associated Animas River stations, such as substrate type (granite, sandstone, limestone, for example) and composition (boulder, cobble, gravel, for example) but were dissimilar in stream order (size) and gradient. At each sample site, we also obtained data for benthic habitat.

Field sampling procedures were modeled after the EPA's Rapid Bioassessment Protocol (RBP) for macroinvertebrates. Because macroinvertebrate habitat in the Animas River is primarily boulder-cobble substrate, the single-habitat approach was selected as opposed to the multi-habitat approach. RBP protocol emphasizes compositing samples over an area of greater than 2 m^2, but the primary goal of RBP is assessment of stream ecosystem health. The goal of this research was to establish pre-remediation data from which to gauge the effectiveness of mine-site remediation. To accomplish this, we needed to determine sampling errors and densities of particular taxa. Therefore, instead of compositing the samples as in the RBP protocol, we took from 3 to 6 samples from each site. The area sampled for each one was limited to an area of 0.5 m directly upstream of the net, resulting in an area of 0.25 m^2. Because of the high flows and volumes of water found in the incised Animas River canyon (sites 8–14, figs. 1 and 2), a modified rectangular dip-net was constructed for this research. The net had more surface area to allow greater quantities of water to flow into and through the net. The opening measured 23 cm high by 50 cm wide. The net measured 50 cm long and had approximately 1,000 cm^2 of 500-µm mesh netting. Laboratory sample processing of the macroinvertebrate samples followed protocols outlined by the National Aquatic Monitoring Center (Vinson and Hawkins, 1996). Taxa were identified to the lowest possible level (see Appendix, included on CD-ROM, Sole and others, this volume, Chapter G), and voucher specimens are stored at the USDA San Juan National Forest aquatics laboratory.

Figure 2 (facing page). Sample sites, Animas River watershed south of study area to Colorado State line. Sample site 1 on the Animas River upstream from the confluence with the San Juan River in New Mexico is not shown. Dot indicates two samples taken at site, one in mainstem at confluence and another in the tributary (see inset in figure).

The total number of taxa per sample site and the means of the densities of each taxon per sample site were calculated, tabulated, and graphed to determine the pre-remediation baseline. Unknown specimens that clearly did not fit in any of the other taxonomic categories were not included in calculations of taxa richness. Data were assessed on the basis of (1) the composition of a diverse macroinvertebrate community within the Animas River watershed, (2) the total number of taxa (taxa richness) for each sample site and how this metric varies by river mile within the Animas River and in the tributaries, and (3) the relative densities of taxa (number of individuals/taxon/m^2) within each of the most important lotic (flowing streams and rivers) aquatic groups: true flies (Diptera), mayflies (Ephemeroptera), stoneflies (Plecoptera), caddis flies (Trichoptera), beetles, and non-insects for both the Animas River and tributaries.

Minor modifications were made to the original 1996 sampling plan. In 1997, the sampling plan included stations upstream of and in Hermosa Creek (fig. 2, site 7) and Kendall Gulch (site 14, fig. 1 and table 1). Note: Because of the large number of springs along the Animas River upstream from Molas Creek (site 12T), data from this site were not included in the analysis.

Macroinvertebrate communities at impaired sites were compared to reference sites using Morisita's index of similarity and percent similarity index for the following: upstream from Cunningham Creek site (site 18) to Cunningham Creek at mouth (site 18T), upstream from Boulder Gulch (site 17) to Boulder Gulch at mouth (site 17T), upstream from Deer Park Creek (site 13) to Deer Park Creek at mouth (site 13T), upstream from Elk Creek (site 11) to Elk Creek at mouth (site 11T), upstream from Needle Creek (site 10) to Needle Creek at mouth (site 10T), upstream from Cascade Creek (site 9) to Cascade Creek at mouth (site 9T), upstream from Hermosa Creek (site 7) to Hermosa Creek at mouth (site 7T), and Mineral Creek upstream from South Fork Mineral Creek (site 20a) to South Fork Mineral Creek at mouth (site 20b, fig. 1).

Annual variation of pre-remediation data was estimated utilizing 1992, 1996, and 1997 data both within the Animas River canyon (sites 8–14, figs. 1, 2) and at Mineral Creek upstream from South Fork Mineral Creek (site 20a, fig. 1). Within the canyon a regression line was fit, utilizing least-squares analysis, to total taxa data collected at each station for each year. At the Mineral Creek site the mean and standard error of the 1992, 1996, and 1997 total taxa data was calculated.

Table 1. Sample sites.

[Site number (T denotes associated tributary); distance in river miles upstream from confluence of Animas River with San Juan River, N. Mex., see figs. 1 and 2, degree of impairment and reference sites]

Site No.[1]	Site name	River distance	Impairment
	Animas River sites		
19	Upstream from Eureka Gulch	82.1	Impaired
18	Upstream from Cunningham Creek	78.2	Impaired
17	Upstream from Boulder Gulch	75.8	Impaired
16	Upstream from Cement Creek	74.2	Impaired
15	Upstream from Mineral Creek	73	Impaired
14	Upstream from Kendall Gulch	72	Impaired
13	Upstream from Deer Park Creek	71.7	Impaired
11	Upstream from Elk Creek	67.4	Impaired
10	Upstream from Needle Creek	61	Impaired
9	Upstream from Cascade Creek	54.7	Impaired
8	Downstream from Bakers Bridge	43.3	Impaired
7	Upstream from Hermosa Creek	38.5	Impaired
6	32nd St. Bridge, Durango	25.8	Impaired
5	Upstream from Lightner Creek	23.5	Impaired
4	Purple Cliffs	21	Impaired
3	Weaselskin Bridge	19	Impaired
2	Upstream from Florida River	17	Impaired
1	Mouth	0.3	Impaired
	Animas River tributary sites		
19T	Eureka Gulch mouth	82	Impaired
18T	Cunningham Creek mouth	78.1	Slightly Impaired/Reference
17T	Boulder Gulch mouth	74.2	Unimpaired/Reference
13T	Deer Park Creek mouth	71.6	Unimpaired/Reference
11T	Elk Creek mouth	67.3	Unimpaired/Reference
10T	Needle Creek mouth	60.9	Unimpaired/Reference
9T	Cascade Creek mouth	54.7	Unimpaired/Reference
7T	Hermosa Creek mouth	38.5	Unimpaired/Reference
5T	Lightner Creek mouth	23.1	Unimpaired/Reference
2T	Florida River mouth	17	Unimpaired/Reference
	Cement Creek basin		
16T	Mouth	74.1	Impaired
22	Downstream from South Fork Cement Creek	80.5	Impaired
23	Upstream from North Fork Cement Creek	81.4	Impaired
24	South Fork Cement Creek Mouth	80.6	Impaired
	Mineral Creek basin		
15T	Mouth	73	Impaired
20a	Upstream from South Fork Mineral Creek	76.5	Impaired
	South Fork Mineral Creek subbasin		
20b	Mouth	76.5	Slightly Impaired/Reference
21	South Fork Mineral Creek Forest Service Campground	80.1	Slightly Impaired/Reference

[1]See figures 1 and 2.

Results

Macroinvertebrate Communities in Reference Tributaries

Data from the unimpaired tributaries, when compared with data from the Animas River and Cement and Mineral Creeks, showed relatively higher numbers of taxa and densities and an even distribution of the number of taxa within the Orders (figs. 3, 4; table 1). Except for South Fork Mineral Creek mouth (site 20b, fig. 1) and Lightner Creek mouth (site 5T, fig. 2), the communities in the tributaries were dominated by mayflies (fig. 4). As in the Animas River upstream from Boulder Gulch, Cunningham Creek at mouth, and Boulder Gulch at mouth (sites 17T and 18T, fig. 1; table 1), macroinvertebrate communities were dominated by the heptageniid mayflies, *Rhithrogena* and *Epeorus*, and the ephemerelid mayfly, *Drunella* (fig. 5). South Fork Mineral Creek at mouth and South Fork Mineral Creek at the United States Department of Agriculture (USDA) Forest Service campground (site 21, fig. 1) had fairly diverse and even macroinvertebrate communities reflected in both taxa richness (23 and 17 respectively) and evenness (fig. 4). Because this tributary, as well as Cunningham Creek and lower Boulder Gulch, was affected by mineralization, the composition of the macroinvertebrate community of these tributaries may be important in determining what community compositions to expect as the Animas River macroinvertebrate community recovers. (See objective 4, Appendix 6b, Macroinvertebrate Report, Unpub. Report to Colorado Water Quality Commission, ARSG, 2001.)

Boulder Gulch is a high-gradient drainage wholly within the Silverton caldera. Since this subbasin was set aside as the community water supply, essentially no mining activity has occurred within the subbasin. The subbasin is underlain by propylitically altered bedrock (Bove and others, this volume, Chapter E3), and the water quality is good (Wright and others, this volume, Chapter E10). The stream contains a diverse assemblage of macroinvertebrates (figs. 3–8). This tributary represents the best reference reach for the macroinvertebrate community in the caldera. It is easily accessible for sampling, and pre-remediation geochemical and biological baseline data are available.

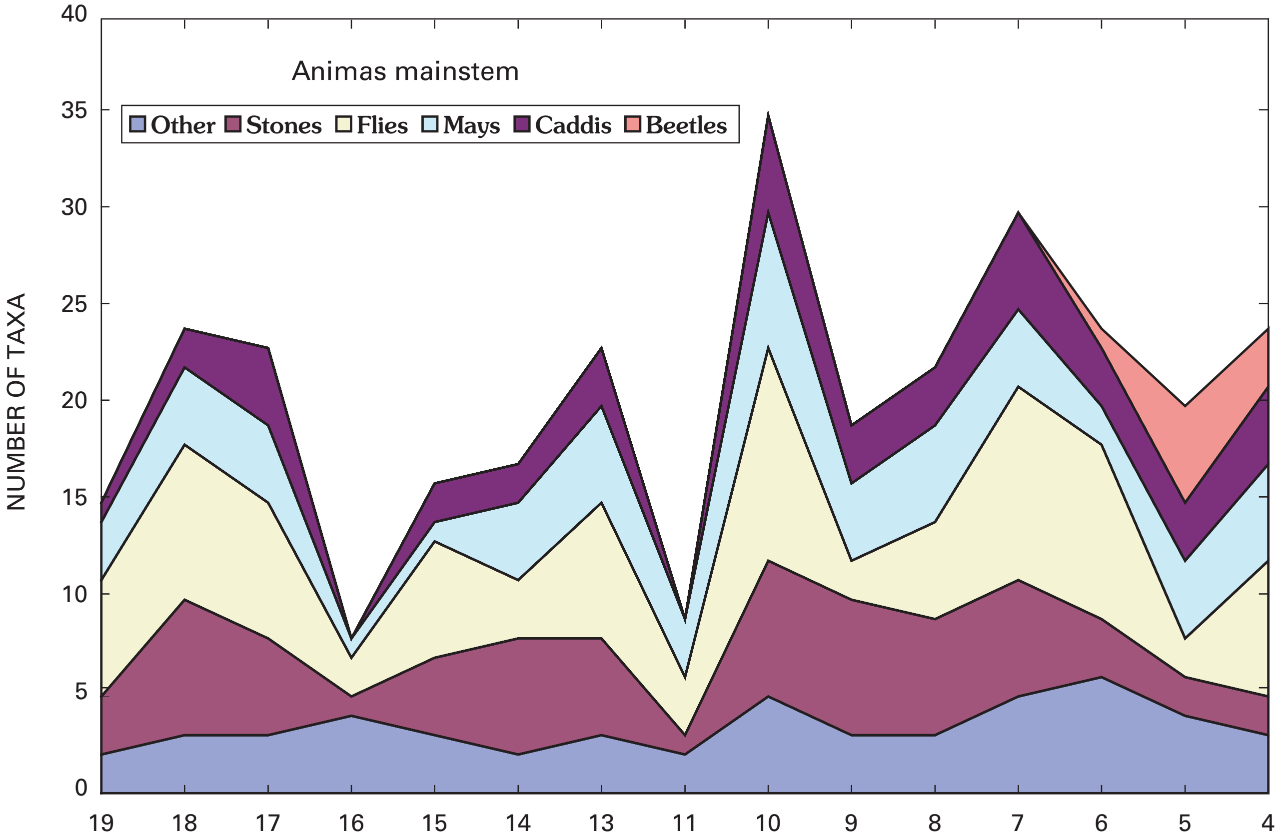
A diverse macroinvertebrate community in a tributary outside the Silverton caldera was best represented by Deer Park Creek (site 13T, fig. 1), 1.3 miles downstream from the caldera margin (Yager and Bove, this volume, Chapter E1). Deer Park Creek had a relatively large number of taxa (35) existing at relatively high densities (1,274 individuals per square meter; fig. 3). The numbers of taxa within each Order were distributed relatively evenly with a slight dominance by mayflies (fig. 4). Within the mayfly Order, there were seven taxa dominated by two genera, *Rhithrogena* and *Drunella* (303 and 130/m^2, fig. 5 and Appendix (Sole and others, this volume)). Although the density of mayflies in Deer Park Creek (533/m^2) was less than in Cunningham Creek (773/m^2), there were more mayfly taxa in Deer Park Creek (7) than in Cunningham Creek (4) as well as more taxa of the other major Orders such as caddis flies (fig. 6 and Appendix). In Deer Park Creek, there were six taxa of caddis flies dominated by *Rhyacophila* (Appendix) and seven taxa of stoneflies dominated by both *Zapada* (142/m^2) and a taeniopterygid (120/m^2; fig. 7 and Appendix). There were relatively few dipterans (fig. 8) in Deer Park Creek compared to South Fork Mineral Creek at mouth site (20b, fig. 1).

Similarity of macroinvertebrate communities in the Animas River mainstem to reference tributaries reflected patterns in taxa richness and evenness in the mainstem (fig. 9) except at Hermosa Creek (sites 7 and 7T). Similarity declined due to changes in taxa composition in Hermosa Creek at mouth compared to other tributaries and the mainstem.

Examples of less diverse and even macroinvertebrate communities exist in Cement and Mineral Creeks (sites 20a, 16T, and 22–24, fig. 1). Both tributaries had relatively few taxa (9 and 16 respectively, collected at confluence with Animas River) with either low densities (64 and 143/m^2) or communities that were dominated by only one or two taxa (figs. 3 and 4; Appendix).

Taxa Richness and Densities of Specific Taxa in the Animas River

Compared to conditions in Boulder Gulch and other reference tributaries, both taxa richness and density of macroinvertebrates in the Animas River upstream from Eureka Gulch (site 19, fig. 1) were low, but both metrics increased from Eureka Gulch to Boulder Gulch (sites 19 through 17T, fig. 1 and fig. 3). The peak in taxa richness and densities in the Animas River from the reach upstream of Cunningham Creek and Boulder Gulch represented a fairly diverse macroinvertebrate community, although not as diverse as the communities within either the mouth of Cunningham Creek or Boulder Gulch themselves. Taxa richness and densities in the Animas River decreased dramatically downstream from Boulder Gulch to Mineral Creek. Downstream, the taxa richness in the Animas River increased to previous levels found around Cunningham Creek near the confluence with Deer Park Creek, but invertebrate densities did not increase substantially until 28 miles downstream at the Downstream from Bakers Bridge site (site 8, fig. 2 and fig. 3; table 2). Presumably this recovery of the invertebrate community is attributable to improved water and habitat quality and changes in toxicity of dissolved metals. Taxa richness more or less levels off from Bakers Bridge to the confluence with the San Juan River (sites 8 through 1, fig. 2), but densities continue to increase downstream to Weaselskin Bridge (site 3, figs. 2, 4).


Animas mainstem
Other
Stones
Flies
Mays
Caddis
Beetles
NUMBER OF TAXA
0
5
10
15
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25
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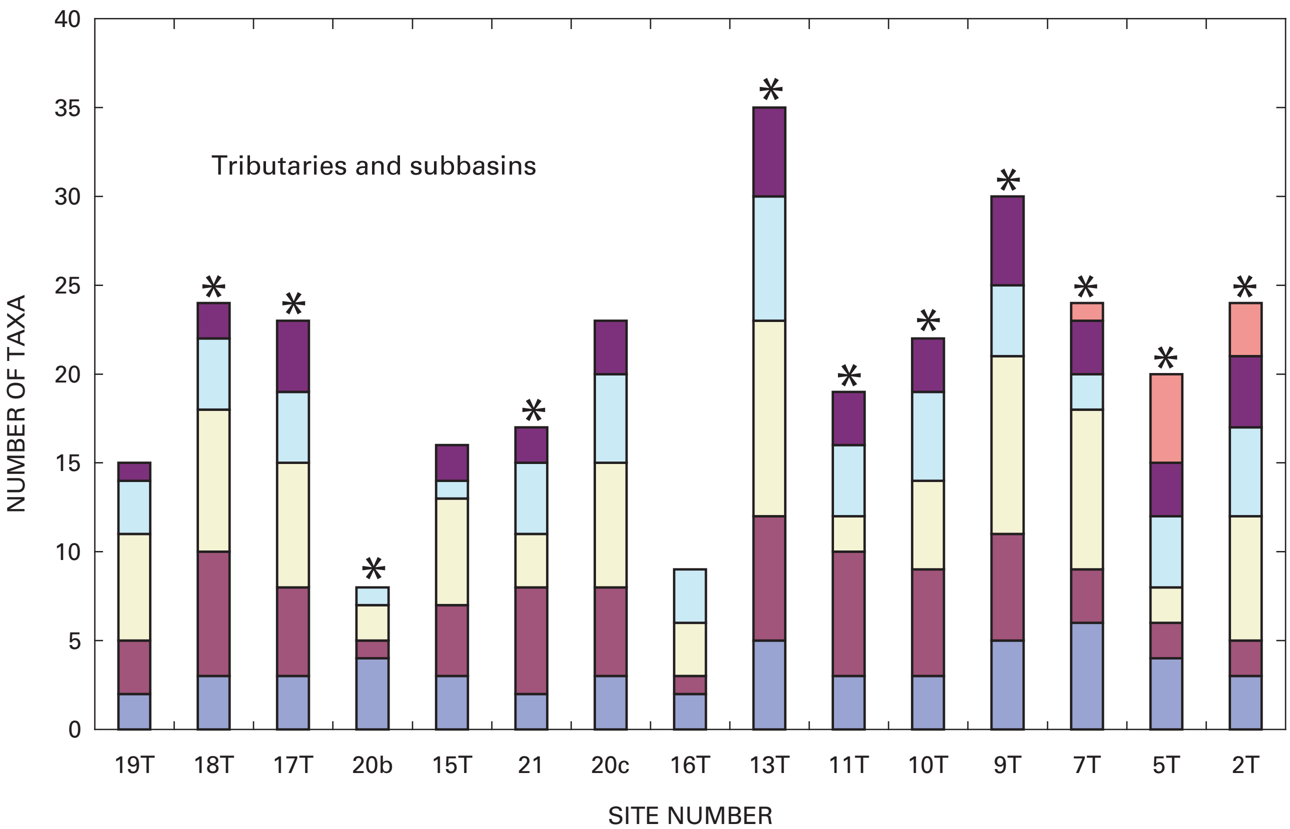

Tributaries and subbasins
NUMBER OF TAXA
0
5
10
15
20
25
30
35
40
19T
18T
17T
20b
15T
21
20c
16T
13T
11T
10T
9T
7T
5T
2T
SITE NUMBER

Evenness of Taxa in the Animas River

The number of taxa found within each Order of macroinvertebrates is relatively even throughout the Animas River. The evenness, however, as illustrated by the number of individuals per taxon/m^2 within the Animas River upstream from Cunningham Creek, shows domination by dipterans (240/m^2 out of total of 399/m^2; fig. 3 and Appendix). Within the mayfly community in the Animas River upstream from Cunningham Creek, *Rhithrogena* (36/m^2) and *Baetis* (27/m^2, total mayfly density=73/m^2) are dominant (fig. 5 and Appendix). In the reach from Eureka Gulch to Boulder Gulch, the composition of the mayfly community improves: an increase in taxa richness (from two to four) and densities (from 2.67 to 500/m^2) with dominance by *Rhithrogena* (385/m^2), *Baetis* (59/m^2), *Epeorus* (38/m^2), and *Drunella* (18/m^2). The number of taxa of mayflies remains relatively constant throughout the reach (fig. 3), but the densities of the mayfly community do not recover in the Animas River until south of Durango, 55 mi downstream (site 5, fig. 2; fig. 5). *Rhithrogena* essentially disappears downstream at Cement Creek (site 16, to 0.67/m^2) and reappears upstream from Elk Creek (site 11, fig. 1; 9.33/m^2) reaching a peak density of 42/m^2 at the site upstream from Needle Creek (site 10, fig. 2).

Upstream from Boulder Gulch, there is less disparity of relative densities of mayflies (500 mayflies per square meter compared to 630 total taxa per square meter) between the Animas River and its associated tributaries—Cunningham Creek (site 18T; 773 / 1,121) and Boulder Gulch (site 17T; 564 / 1,138)—than there is in the Animas River downstream from the confluence with Mineral Creek, where a greater disparity exists between relative densities of mayflies in the Animas River and associated tributaries—Deer Park, Elk, Needle, Cascade, and Hermosa Creeks (figs. 1, 2, 5; Appendix). In the Animas River upstream from the confluence with Mineral Creek (site 15, fig. 1), dipteran communities were dominated by orthoclads and an unknown chironomid, whereas associated tributaries, Boulder Gulch and Cunningham Creek, had relatively high diversity of true flies (fig. 8).

Downstream from the confluence of Cascade Creek (site 9, fig. 2), the number of taxa and their densities increased sharply in the Animas River. Upstream from Bakers Bridge (site 8, fig. 2), densities of caddis flies (fig. 6) were low (84/m^2), dominated by *Arctopsyce* (75/m^2). Downstream from Bakers Bridge, additional taxa were found along with a dramatic increase in densities (fig. 6). The Animas River caddis fly community, when compared to that of tributaries such as Boulder Gulch and Deer Park Creek, appeared to be reduced more than the other Orders of macroinvertebrates (fig. 6). The densities of *Brachycentrus* increased substantially upstream from Hermosa Creek (site 7, fig. 2), but the total number of taxa of caddis flies did not increase in the Animas River until downstream from Durango (Appendix). Below Bakers Bridge, the Animas River's setting changes dramatically from a steep, incised canyon to a relatively wide valley and a meandering stream. Agriculture, gravel mining, housing developments as well as urban landscapes dominate land use.

Stonefly densities throughout the Animas River were low (fig. 7). The communities were dominated by *Zapada* upstream from Deer Park Creek with the greatest diversity upstream from the confluence with Cement Creek. Downstream from the Mineral Creek confluence, an unknown taeniopterigid dominated the stoneflies in the Animas River south to Bakers Bridge through the Animas River canyon. Relatively high numbers of taxa of stoneflies occurred in the Animas River upstream from Boulder Gulch (five), but this community was absent in the reach downstream from the confluence with Cement Creek and did not reappear until Needle Creek (fig. 7), reflecting the effects of more acidic and metal-rich water flowing from Cement and Mineral Creeks into the Animas River (Kimball and others, this volume, Chapter E9).

Figure 3 (facing page). Number of taxa in the major stream Orders (stoneflies, mayflies, true flies, caddis flies, beetles, non-insects) for sample sites in Animas River upstream of tributaries (upper graph) and in tributaries (lower graph). Sample stations sorted from upstream (left) to downstream (right) on *X*-axis. Sample site with asterisk, reference site.

Similarity Among Animas River and Reference Tributary Sites

The composition of the macroinvertebrate communities in the Animas River when compared to reference tributaries utilizing Morisita's and Percent Similarity Indices revealed similar trends to the number of taxa and evenness metrics. Similarity peaked upstream of Silverton with a decline at and downstream from Silverton, and a recovery of the community in the Animas River canyon. At the Hermosa Creek sites (site 7 versus 7T) a decline in similarity occurred (fig. 9).

Discussion

The rate of recovery of the macroinvertebrate community after remediation depends on (1) the rate that water quality improves, (2) the rate that the benthos (stream substrate and associated organisms and detritus) recovers, and (3) the rate of recolonization by macroinvertebrates. Therefore, when and how often to conduct post-remediation sampling should depend on the rate that all the components (physical, chemical, and biological) of impaired segments recover after remediation. The processes by which these components recover operate along different spatial and temporal scales and may be specific to a particular location in the watershed.

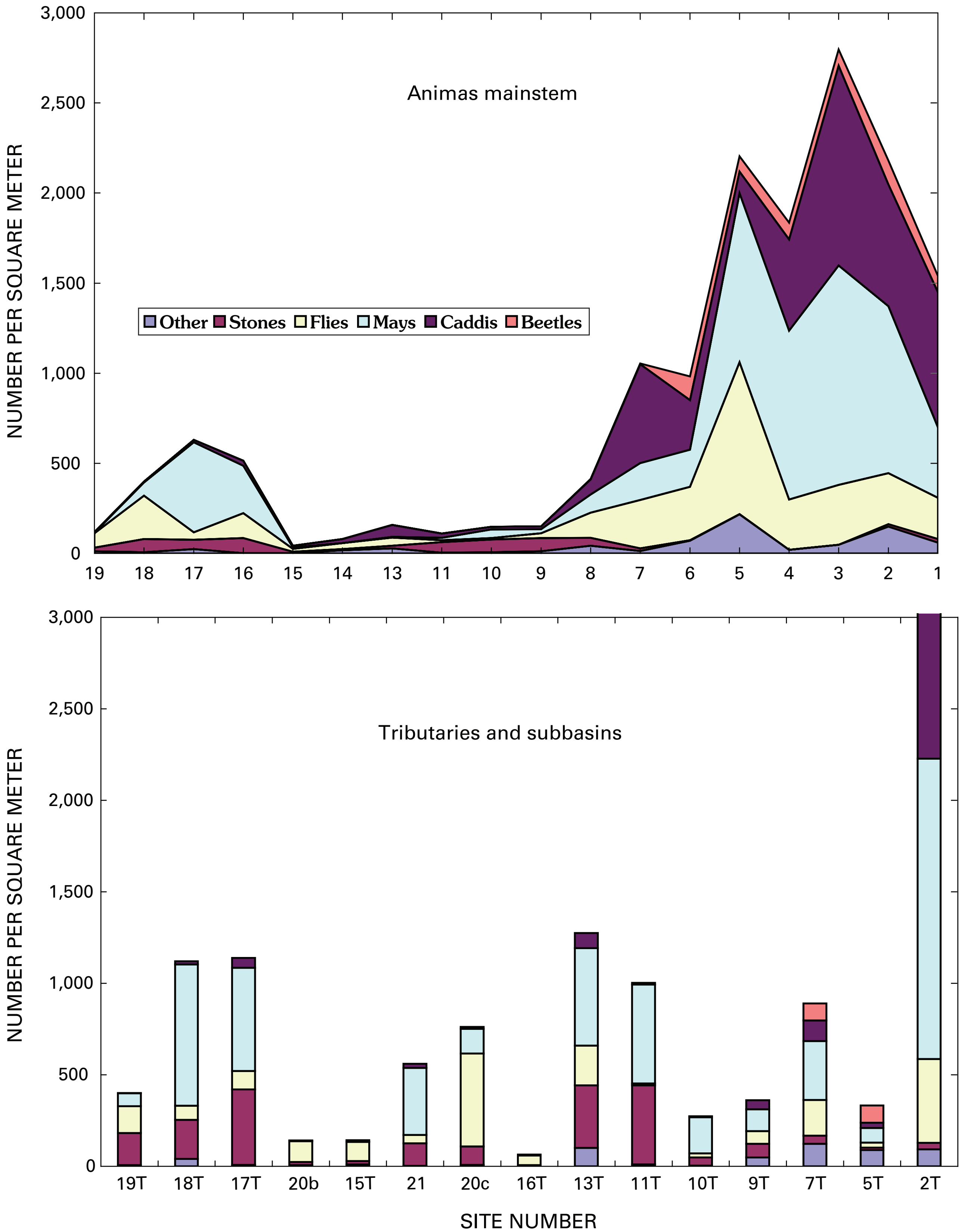


Figure 4. Densities (number of individuals per square meter) of taxa within major stream Orders (stoneflies, mayflies, true flies, caddis flies, beetles, non-insects) for sample sites within Animas River upstream of tributaries (upper graph) and within tributaries (lower graph).

The assumption is that water quality from remediation sites will improve almost immediately after remediation is completed, but how long it will take for the sediment quality (Church, Fey, and Unruh, this volume, Chapter E12) and the biological community to recover (Besser and others, this volume; Milhous, 1998) is unknown. The physical and chemical processes that may affect the recovery of the benthos are stream gradient, stream size, frequency and extent of floods, and changes in pH and background concentrations of metals in the water (Mast and others, this volume, Chapter E7).

Aluminum and iron have many common sources in the Cement and Mineral Creek basins (Bove and others, this volume), and the substrate of the Animas River has been significantly affected by aluminum and iron precipitates (Church, Fey, and Unruh, this volume; Kimball and others, this volume). These precipitates have coated rock surfaces and filled interstitial spaces, reducing the habitat available to macroinvertebrates (Parkhurst, 1999; Church, Fey, and Unruh, this volume; Milhous, this volume).

Parkhurst and others (1999) found significant relationships between invertebrate metrics and total aluminum, iron, and manganese concentrations, but found no such relationship with cadmium, copper, and zinc. Iron and aluminum were present primarily in the particulate phase, and manganese, cadmium, copper, and zinc were present primarily in dissolved forms. Their conclusion was that the physical effects of the precipitate, not the toxic effects of dissolved metals, were the primary reason for the suppression of the benthic macroinvertebrate community. Current research has found correlations between dissolved metals that are at or near toxic levels for fish and invertebrates, levels of these metals in tissues of fish and invertebrates and impacts on populations of trout and invertebrates (Besser and others, this volume; Besser and Brumbaugh, this volume; Besser and Leib, this volume).

A scenario may be hypothesized wherein the majority of sources of mining impacts are eliminated from the Animas River and the remaining metals affecting water quality and benthic habitat exist only in the benthos. The rate that these metals move through the river system and out of the benthos will be a function of the depth of contamination and the rate that metals are brought to the surface of the benthos. Floods will have a greater impact where the stream gradient is high and will tend to move metals to depositional areas (margins of the streams or pools) and farther down the Animas River where the gradient is lower. As metals move downstream, they become less available to erosion and their concentrations become more dilute.

Chemical processes that govern metal bioavailability of the benthos include changes in pH (from ground-water infiltration, surface water, and geology), amount of dissolved metal in surface water, and bioaccumulation of primary producers (Besser and Brumbaugh, this volume). An increase in the pH will reduce the potential for toxic metals to be dissolved from the benthos into the water column and will also increase the rate of precipitation of most metals, contributing to the toxicity of precipitates to benthic organisms (Church, Fey, and Unruh, this volume). Suspended precipitates, as colloids or flocculates, will work their way downstream and out of the watershed and are flushed out each year during high flow.

Biological processes may also significantly contribute to cleaning up the benthos. Studies have shown that macroinvertebrates can move large quantities of particulates downstream through their foraging behaviors (Wallace and Whiles, 1993). Therefore, as the macroinvertebrate community recovers through time, the rate that metals could move out of the watershed will increase by this mechanism alone. Biological processes will also produce more organic matter that tends to bind metals in less toxic, more innocuous or unavailable forms.

Because the adult stages of macroinvertebrates are winged and terrestrial, and larvae can easily drift downstream into impaired reaches from unimpaired or from less impaired reaches of the Animas River and its tributaries where there are more diverse communities of macroinvertebrates, colonization is most likely not a limiting factor to the rate of recovery of the macroinvertebrate community.

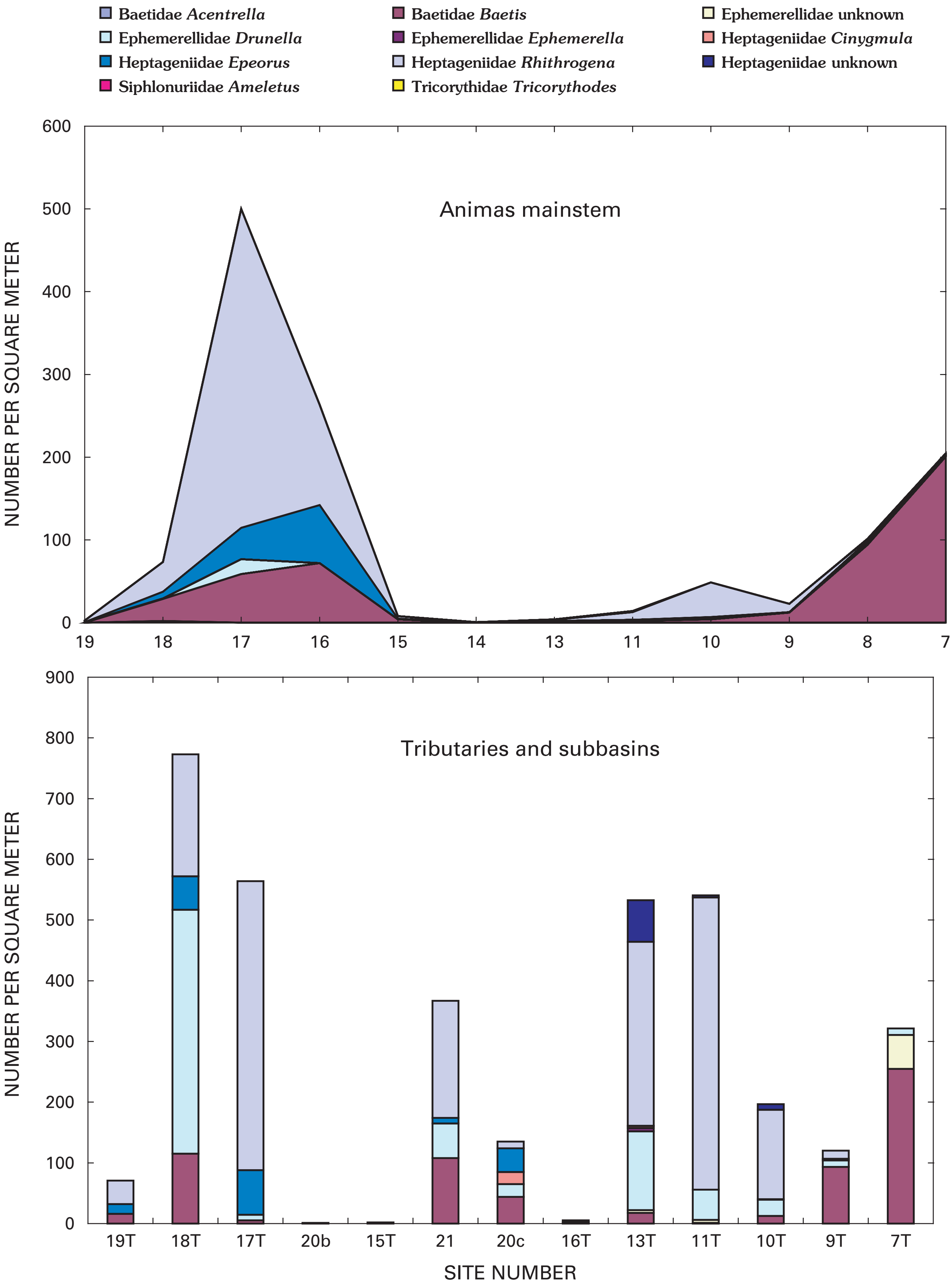
Proposed Sampling Plan and Analysis

Pre-Remediation Baseline Sampling

The overall concept of sampling upstream of major tributaries and utilizing the mouth of unimpaired tributaries for macroinvertebrate reference sites appeared to be adequate and sound, and has historical precedence (Clements and others, 2000; Hughes and others, 1986). In any design of an evaluation of remediation practices, pre-remediation baseline sample sites should be located downstream from proposed remediation sites and upstream from significant changes in water or habitat quality. For economic reasons, the number of sample sites should be kept at a minimum as long as the accuracy and efficacy of the program are not compromised. Sample sites must be located in segments where, other than remediation, no anthropogenic changes occurred to the segments upstream from or between the sample stations. For this reason, stations downstream from Bakers Bridge should be eliminated from the analysis (sites 7–1, fig. 2) because they were subject to a variety of anthropogenic disturbances, including non-point pollution from erosion, agriculture, livestock, and septic tanks, as well as municipal discharge and gravel mining.

Pre-remediation baseline data should include data on water chemistry and benthic habitat. Such data will not only allow for assessments of particular remediation strategies and insight into the mechanisms and underlying processes of recovery (Adams and others, 2002), but will also help in decisions on when and where to conduct post-remediation sampling for biological indices.

Both physical accessibility and cost must be considered in a decision on which season to sample. High flow, low temperatures, and snow severely inhibit access and limit the efficiency and accuracy of sampling in the Animas River during


Baetidae *Acentrella*
Ephemerellidae *Drunella*
Heptageniidae *Epeorus*
Siphlonuriidae *Ameletus*
Baetidae *Baetis*
Ephemerellidae *Ephemerella*
Heptageniidae *Rhithrogena*
Tricorythidae *Tricorythodes*
Ephemerellidae unknown
Heptageniidae *Cinygmula*
Heptageniidae unknown
Animas mainstem
NUMBER PER SQUARE METER
0
100
200
300
400
500
600
19
18
17
16
15
14
13
11
10
9
8
7
Tributaries and subbasins
NUMBER PER SQUARE METER
0
100
200
300
400
500
600
700
800
900
19T
18T
17T
20b
15T
21
20c
16T
13T
11T
10T
9T
7T
SITE NUMBER

winter, spring, and late fall. At high flow, difficulty of access is compounded by the high gradient in the Animas River canyon (Blair and others, 2002). In the spring, a small window opens between snowmelt and high flow. If this window is missed due to weather or some other circumstance such as accessibility, then biomonitoring for that season will not be accomplished. In the fall, the sampling window between the time that high flow has receded and the time of inhibiting snowfall is much larger. Therefore it is suggested that macroinvertebrate sampling be done in the fall. Sampling in the fall, during low flow, was and will be necessary for efficiency and consistency of results between the pre- and post-remediation sampling.

Post-Remediation Sampling

Post-remediation sampling is important to conduct during the same season that the pre-remediation data were gathered, and procedures for field sampling, sample processing, and taxonomic identification must be exactly the same. Given the scarcity of data on how benthic macroinvertebrate communities will react to mine-site remediation, sample processing should include at least a 300-count subsample, and taxa should be identified to the lowest possible level. Furthermore, if taxonomic unknowns are included in the data analysis, they must be systematically used in the same way throughout the study period and must be known to not fit into any of the other taxonomic classifications.

When and how soon post-remediation sampling occurs after remediation is completed must be decided based on when and where mine-site remediation occurs (which is governed by factors not related to testing the effectiveness of remediation) and by the effect that remediation has on water quality. Remediation in the Animas River watershed is currently underway and will continue in various amounts in various subbasins (William Simon, written commun., 2003). Robert Owens (Unpub. water quality and metal loading report, Colo. Dept. of Public Health and Environment, 1997) predicted that remediation must occur at several sites throughout each basin before appreciable reductions in metal loading will be detected. Walton-Day and others (this volume, Chapter E24) have shown that remediation in Cement Creek will have little difference in instream metal loads, but that remediation in the Animas River downstream from Eureka could result in lowering of toxic metal loads. Therefore, after an appreciable amount of remediation has taken place in the Animas River basin, the effects of this remediation should be detected in changes in the water chemistry, and post-remediation macroinvertebrate sampling should be conducted to demonstrate possible improvement in the macroinvertebrate community. After there has been remediation throughout the study area, post-remediation sampling should take place downstream from Silverton. This scenario implies that an adequate pre-remediation data set for water chemistry exists to detect changes in water chemistry (Unpub. report to Colorado Water Quality Control Commission, ARSG, 2001).

Figure 5 (facing page). Densities of taxa within mayfly Order (Ephemeroptera) for stations within Animas River upstream of tributaries (upper graph) and within tributaries (lower graph).

Statistical Analysis to Assess Effectiveness of Remediation

At least 3 years of pre- and post-remediation data is necessary to adequately assess the degree of annual variation of the invertebrate community and to determine if the difference between the post-remediation and the pre-remediation data is significant or is simply within the limits of annual variation (Chapman, 1999; O'Neill, 1999). If possible, a better strategy would be to sample annually while remediation procedures were implemented and sufficiently beyond the remediation period until the metrics reach some level of stability. Stations used to assess annual variation should include ones that vary in degree of impairment, from highly impaired to unimpaired. Unimpaired tributaries essentially have saturated macroinvertebrate communities and thus will not vary from year to year in the same manner that an impaired stream with only a few species will vary. Although not as accurate, year-to-year variation may be defined using historical data as long as the differences in sampling methods, sample processing, and taxonomic identification are defined.

Two types of analysis may be performed to evaluate the effectiveness of remediation. The analyses may be performed on a number of metrics, including richness, percent Ephemeroptera–Plecoptera–Trichoptera genera, percent Chironomidae individuals, Hilsenhoff Biotic Index (Plafkin and others, 1989), and others. In impaired tributaries where the number of stations was limited, it would be necessary to assess the effectiveness of remediation using only one sample site, a "Single Site Assessment." Downstream from such a site where the tributary enters the Animas River, any changes to the tributary due to remediation would be masked by the influence of the water chemistry in the Animas River. At such sites, pre-remediation baseline means may be calculated from three uncomposited samples taken from an equal area of benthic habitat and statistically compared to post-remediation means. If a significant difference between the pre- and the post-remediation means is found, then the difference should be compared to and be greater than the annual variation in order to imply that the difference is attributable to remediation efforts. Annual variation may be determined by obtaining at least 3 years of pre-remediation data and 3 years of post-remediation data. Instead of obtaining three uncomposited samples from an equal area of benthic habitat, a composited sample may be obtained and, from processing the whole sample and through rarefaction analysis (Hurlbert, 1971), means and variances may be calculated for each sample site and compared to post-remediation data (for example, *ECOSIM*, Gotelli and Entsminger, 2001).

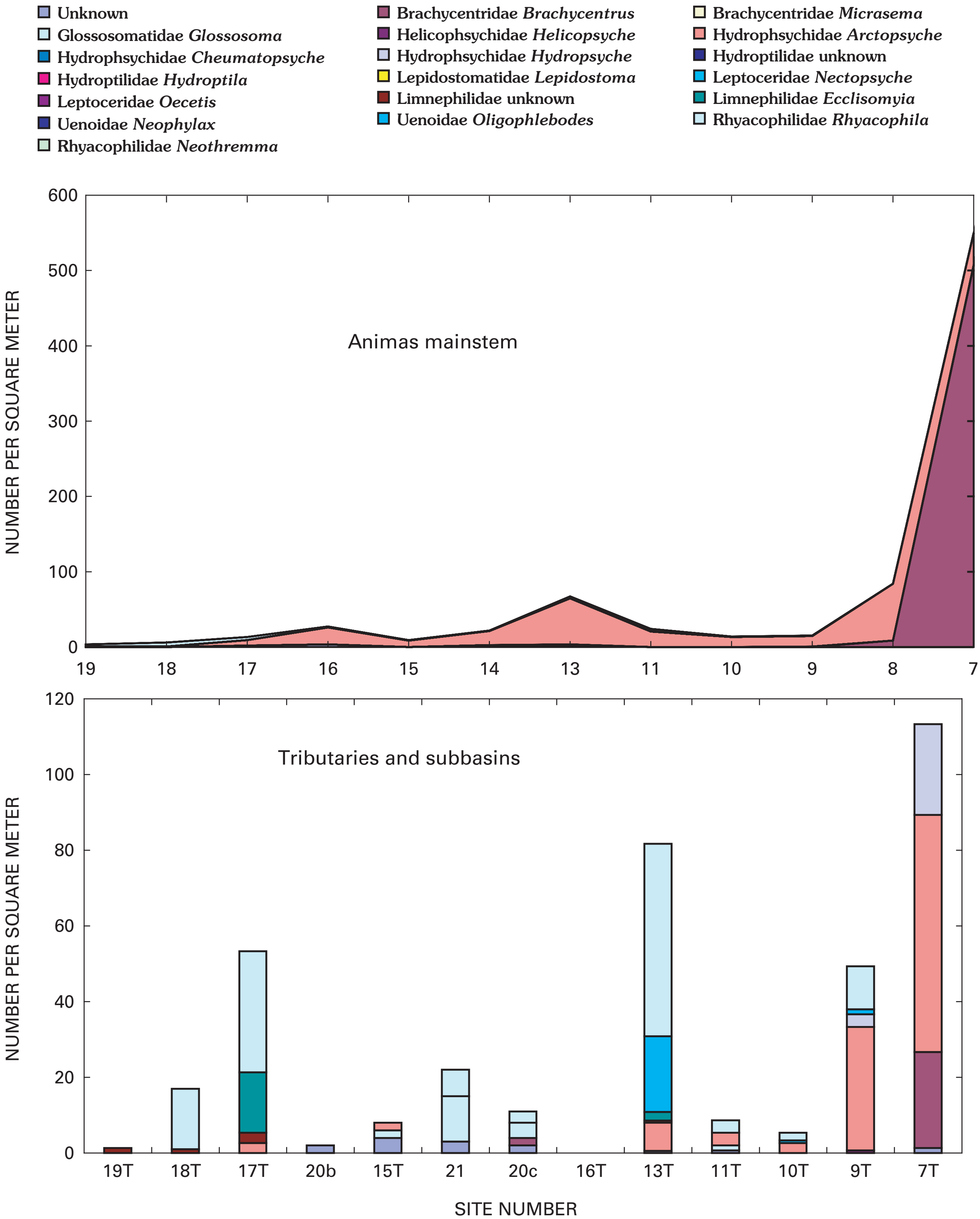


Figure 6. Densities of taxa within caddis fly Order (Trichoptera) for stations within Animas River upstream of tributaries (upper graph) and within tributaries (lower graph).

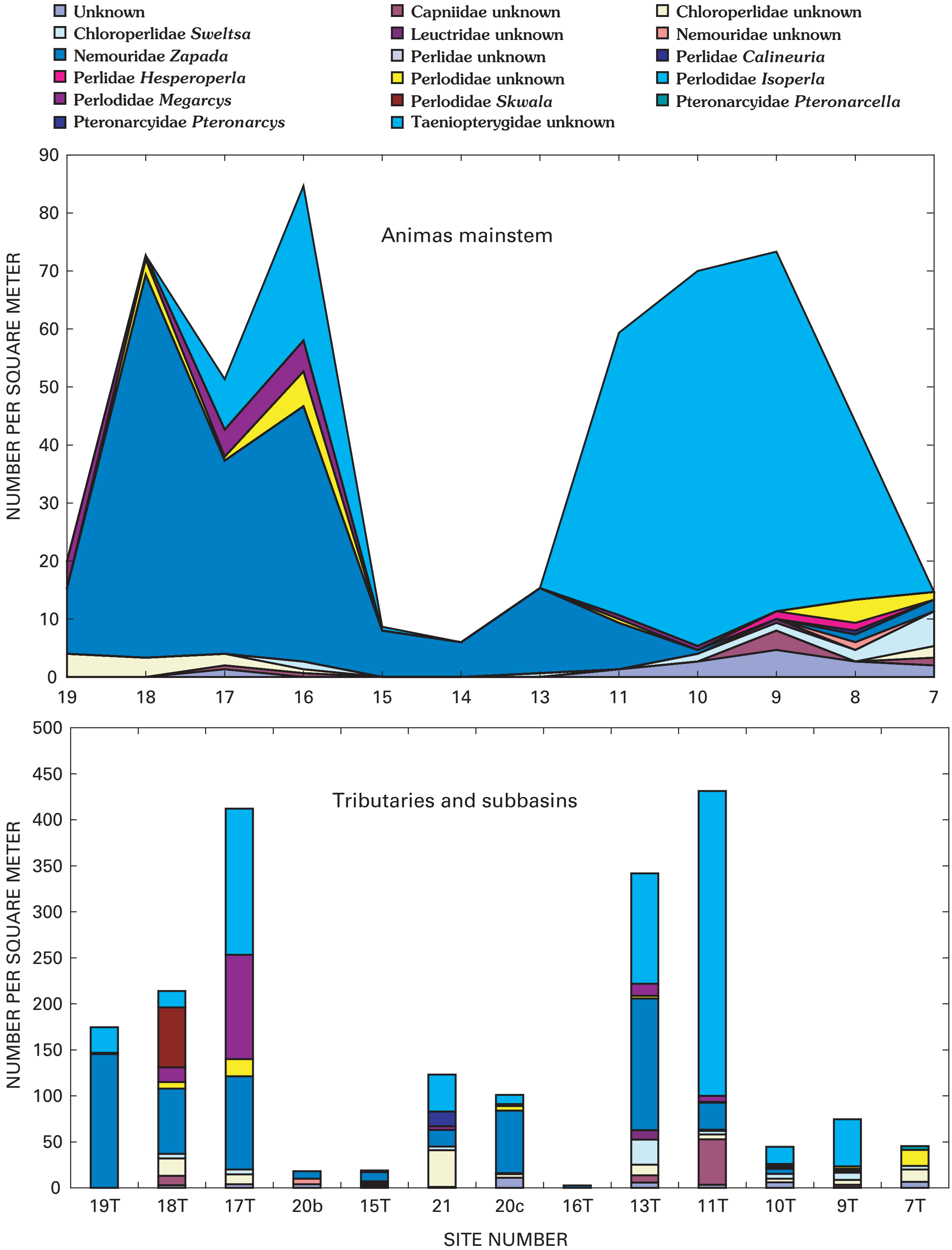


Figure 7. Densities of taxa within stonefly Order (Plecoptera) for stations within Animas River upstream of tributaries (upper graph) and within tributaries (lower graph).

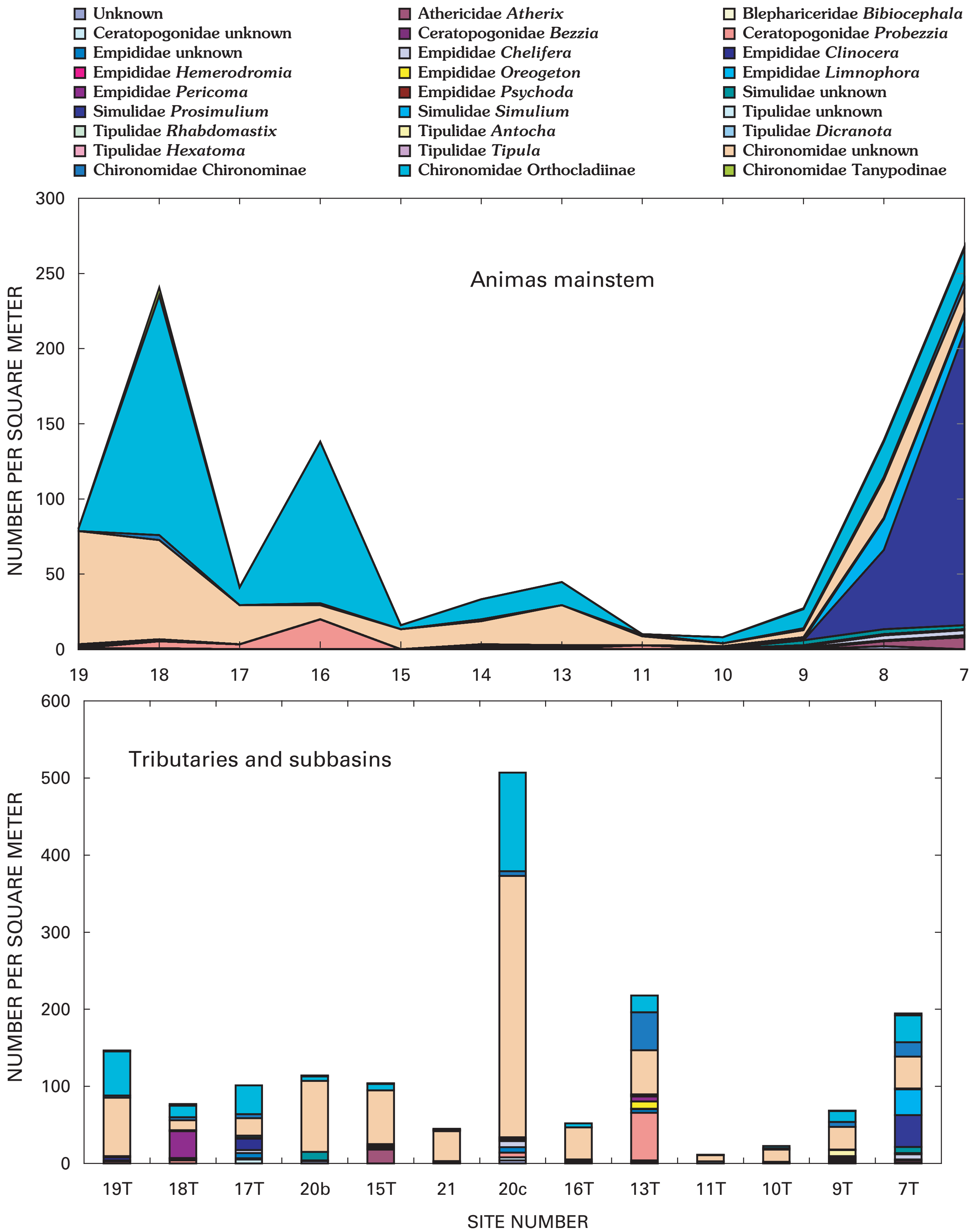


Figure 8. Densities of taxa within true fly Order (Diptera) for stations within Animas River upstream of tributaries (upper graph) and within tributaries (lower graph).

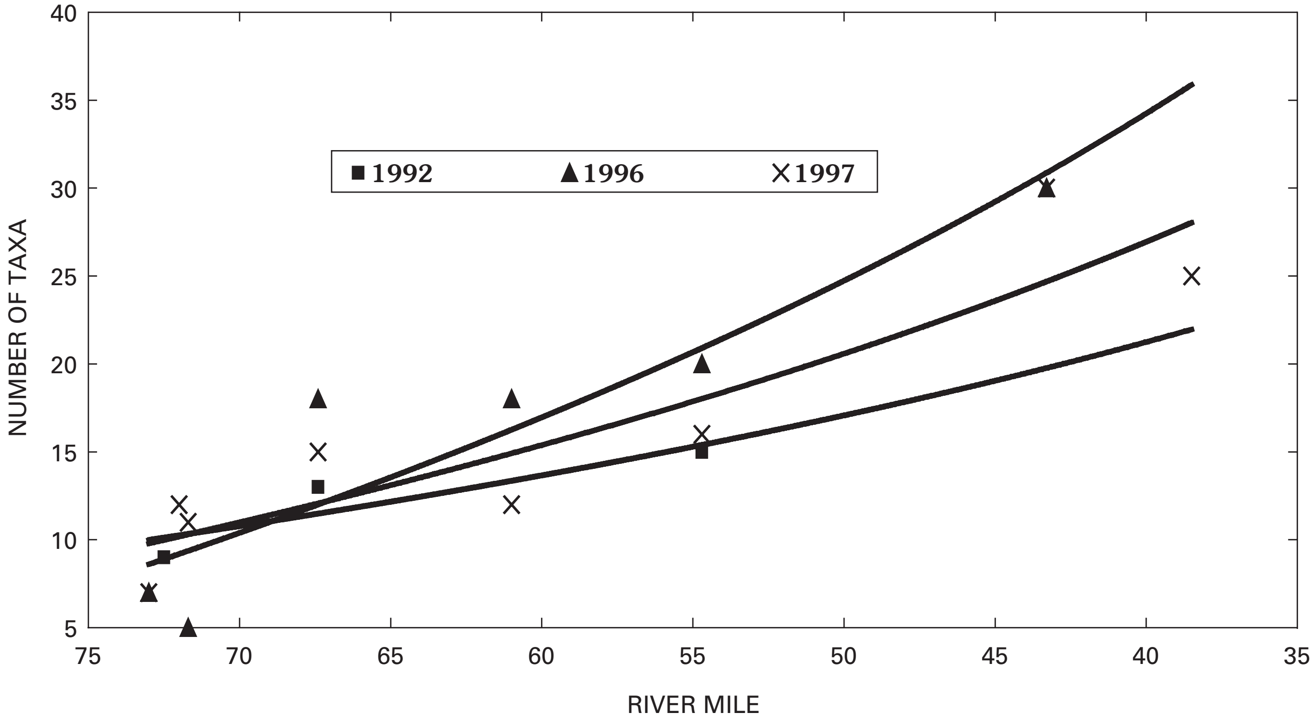


Figure 9. Morisita's Index of similarity (Cλ) and Percent Similarity between upstream sites in Animas River and associated reference tributaries (north is at left, that is, upstream). Mileages coordinated with sites in tables 1 and 2.

An example where a Single Site Assessment is appropriate is on Mineral Creek upstream from South Fork Mineral Creek (site 20a, fig. 1). South Fork Mineral Creek is similar in size to Mineral Creek, and 3.5 miles downstream, Mineral Creek drains into the Animas River. Any assessment of remediation that occurred in the Mineral Creek drainage at a site downstream from the confluence with South Fork Mineral Creek would be masked by the influence of the quantity and quality of water from South Fork Mineral Creek. The annual variation may be estimated from fall 1992 data collected by the Colorado Department of Public Health and Environment and the fall 1996 and 1997 data collected for this study. The range is from a low of 2 taxa in 1992 to a high of 12 taxa in 1997. To assume that mine-site remediation has improved the diversity and evenness of the macroinvertebrate communities, 3 years of consecutive, fall, post-remediation data collected at the same sample sites must be statistically different from the pre-remediation data and the annual variation. For a Single Site Assessment, the statistical test to determine the effectiveness of mine-site remediation is a *t*-test, thus the necessity of at least three samples.

In the Animas River, where at least three sample sites were available to establish pre-remediation data and where (1) the tributaries or the Animas River between the stations will not have any type of anthropogenic interference other than

Table 2. Pre-remediation baseline data for the Animas River canyon.

[Total number of taxa for macroinvertebrates per square meter by year from upstream of Mineral Creek downstream to Bakers Bridge; distance given in river miles upstream from confluence of Animas River with San Juan River, N. Mex.; --, no data]

Sample site	Distance (mi)	1992	1996	1997
Downstream from Bakers Bridge	43.3	--	30	30
Upstream from Cascade Creek	54.7	15	20	16
Upstream from Needle Creek	61.0	--	18	12
Upstream from Elk Creek	67.4	13	18	15
Upstream from Deer Park Creek	71.7	--	5	11
Upstream from Kendall Gulch	72.0	9	--	12
Upstream from Mineral Creek	73.0	--	7	7
Regression equation		y = –0.3027 ln(x) + 31.968	y = –41.10 ln(x) + 185.64	y = –24.803 ln(x) + 118.24
R^2 value		R^2 = 0.8248	R^2 = 0.8371	R^2 = 0.6003

remediation occurring between the time the pre-remediation data were collected and the time the post-remediation sampling was conducted or (2) a regression line can be fitted to the data such that the R^2 value is not unreasonably high, the effectiveness of remediation may be evaluated using multiple sample sites on the Animas River, a "Multiple Site Assessment." At such sample sites, samples may be composited. Examples within the Animas River where a Multiple Site Assessment may be utilized are in the reach from Eureka Gulch to Boulder Gulch (sites 19–17, fig. 1), from Boulder Gulch to Cement Creek (sites 17–16, fig. 1), and from Mineral Creek to Bakers Bridge (sites 15–8, figs. 1 and 2). Regression lines fitted to the pre-remediation data may be compared to post-remediation regressions using an analysis such as ANOVA. If a significant difference between the pre- and the post-remediation data is found, then the difference must be compared to and be greater than the annual variation to imply that the difference is attributable to remediation efforts. For example, in the reach from the confluence with Mineral Creek to Bakers Bridge (sites 15–8, figs. 1 and 2), regression analysis of the data may be fitted to 3 years of pre-remediation data: 1992 data collected by the Colorado Department of Public Health and Environment and the fall 1996 and 1997 data collected for this study (table 2 and fig. 10). To assume that mine-site remediation has improved the diversity and evenness of the macroinvertebrate communities, 3 years of consecutive, fall, post-remediation data collected at the same sample sites must be statistically greater than the pre-remediation data. If no significant difference is observed between the pre-remediation data and the post-remediation data, then either further remediation is required or more time for recovery of the benthos or colonization by macroinvertebrates is necessary before more post-remediation data are collected. In statistical assessment of the effectiveness of remediation, critical statistical values, α and β, do not necessarily have to be at the generally accepted 0.05 and 0.20 but may rather be set by the regulatory process and may be more or less stringent (Fleiss, 1981).

For either of these scenarios, composited samples should be taken from unimpaired streams to determine the degree that impaired communities are becoming more similar to unimpaired communities (fig. 10). Unimpaired streams are assumed to have healthy macroinvertebrate communities and thus accurately represent the goal of stream remediation.

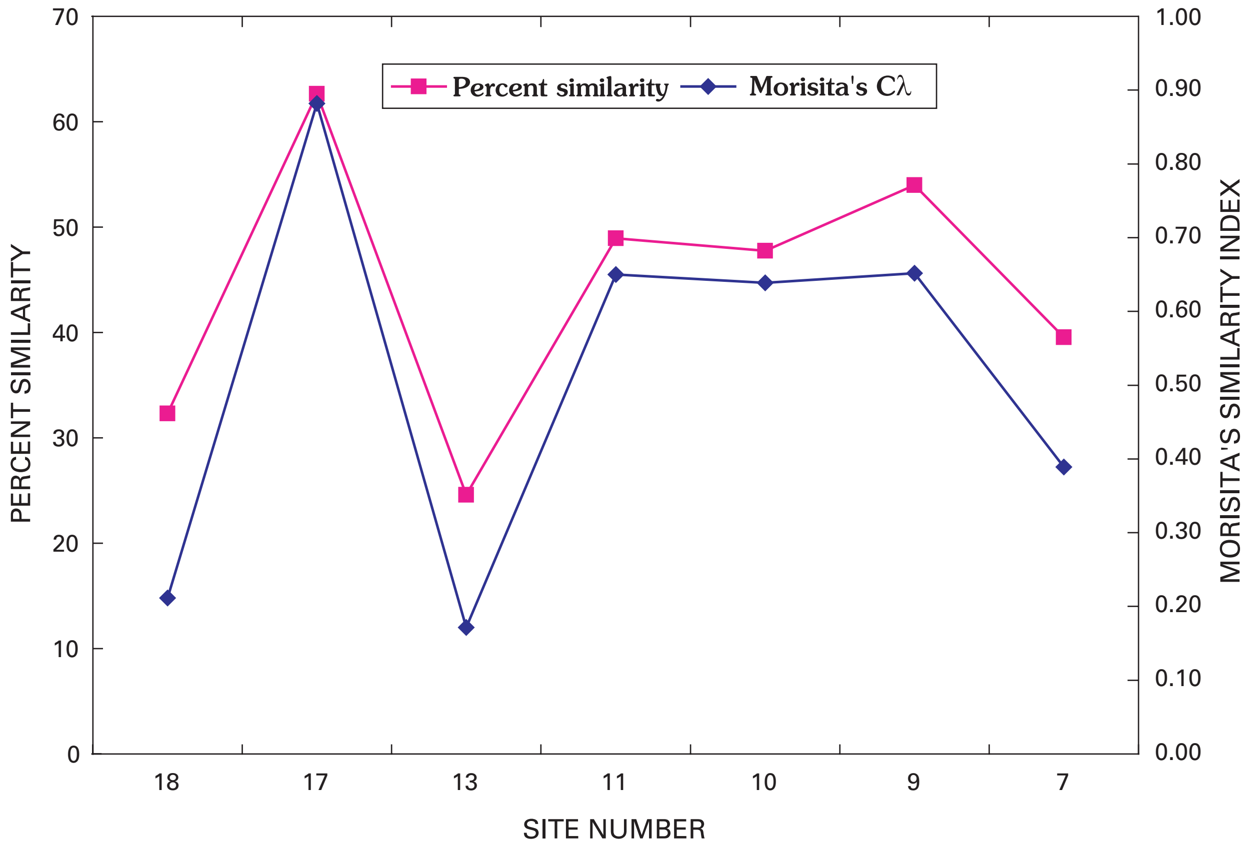


Figure 10. Pre-remediation baseline data for a Multiple Site Assessment. Variation of taxa richness within the Animas River canyon downstream from Silverton to the upstream from Hermosa Creek site for fall 1992 data collected by Colorado Department of Public Health; fall 1996 and fall 1997 data collected by USDA Forest Service, San Juan National Forest, and Animas River Stakeholders Group. Three years of consecutive, fall, post-remediation data collected at the same sample sites must be statistically different from the pre-remediation data to imply that mine-site remediation has improved diversity of macroinvertebrate communities. Variable for *Y*-axis may include any appropriate population metric.

Summary

The primary objectives of establishing pre-remediation baseline data were achieved by our obtaining multiple uncomposited samples for three sampling dates: fall 1996, spring 1997, and fall 1997, from 25 impaired sites in the Animas River watershed study area and beyond (figs. 1 and 2). These samples will allow for a statistical analysis to post-remediation data. Multiple, uncomposited samples from 11 reference streams in the watershed were also collected (table 1). Specimens have been archived at the USDA Forest Service, San Juan National Forest laboratory, and identifications may be validated for comparison to post-remediation data. Methods were proposed to evaluate the effectiveness of mine-site remediation and to decide when and where to conduct post-remediation sampling. The greatest limitation to the use of this data set in assessing the success of remediation will be the lack of information on temporal variation. A long-term, annual sampling program at a variety of sites with differing degrees of impairment will be important to an assessment of the effectiveness of future mine-site remediation (Chapman, 1999).

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