Responses of Aquatic Macroinvertebrates to Stream Channel Reconstruction in a Degraded Rangeland Creek in the Sierra Nevada

David B. Herbst and Jeffrey M. Kane

ABSTRACT

Streams on western rangelands are sometimes degraded to the point that reestablishing lost ecological values requires rebuilding their physical structure, shape, and form. We evaluated the biological response to reconstruction of a small rangeland stream channel through comparisons of aquatic invertebrate communities before and after engineering activities and in relation to local and regional reference stream conditions. We measured geomorphic and riparian habitat features along with benthic macroinvertebrates for two years prior to restoration actions and for two years afterward. Stream restoration activities included the construction of a new channel to replace an incised meadow stream reach (including addition of coarse rock substrate, erosion control fabric, and willow planting) and the rehabilitation of gullies and roads in the meadow and its watershed. In postproject monitoring, we found statistically significant changes in the macroinvertebrate community and trophic structure at the restored site. These were exhibited as increases in EPT taxa (mayflies, stoneflies, caddisflies) and the proportion and diversity of sensitive taxa, decreased tolerant taxa, and an increase in consumers of riparian organic matter (shredders) and decrease in fine organic particle filter-feeders. A site monitored downstream of the restoration activities exhibited increased deposition of fines and sand one year after project construction, but was similar to preproject conditions in the second year, and the macroinvertebrate community was unchanged. Improved biological integrity at the restored site showed that rapid recovery can occur in rebuilt streams within rehabilitated watersheds, but neither degraded nor enhanced conditions were transferred to downstream habitat, at least over the initial postproject period.

Keywords: livestock grazing impacts, macroinvertebrate bioassessment, rangeland improvement, Sierra Nevada, stream restoration

The restoration of disturbed rangelands must balance demands for the recovery of native ecosystems with the need for sustainable land use practices. The problem has long been recognized in the arid and mountainous regions of western North America where degraded soil and vegetation conditions are often found in the presence of livestock grazing (Jones 2000). Grazing is known to alter watershed hydrology and stream channel morphology and lead to losses of soil, riparian vegetation, wildlife, and water quality at both local and landscape scales (Belsky et al. 1999). The reduction of vegetative cover and disturbance of soil at the watershed scale promote the concentration of surface runoff over exposed soils, which may create gullies across the landscape, increase erosion and stream sediment loads, and alter the geomorphology of receiving channels (Trimble and Mendel 1995). In the riparian corridors, trampling by livestock accelerates the erosion of stream banks and produces channel widening, loss of shade, and increased water temperatures. For streams typified by fine unconsolidated alluvium, deep incision of the channel may result, disconnecting the stream from its floodplain and lowering the surrounding water table. In such cases, including the subject of this study, Bagley Valley Creek, loss of connectivity often results in conversion of adjacent wetland meadow and riparian habitats to drier and less productive upland community types (Kauffman and Krueger 1984).

Where the restoration of impacted rangeland watersheds and streams is desired, management practices that reduce or eliminate grazing exposure may allow the natural capacity of streams to recover (i.e., passive restoration; Kauffman et al. 1997, Sarr 2002, Agouridis et al. 2005). But often impacts are so severe that passive restoration of natural channel morphology and habitat characteristics may be slow or not occur at all (Kondolf 1993, Sidle and Sharma 1996, Sarr 2002). Where land use disturbance has altered natural dynamic channel form and processes, active restoration efforts in both streams and their watersheds

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may be necessary to restore ecological processes and desired rangeland productivity (Kauffman et al. 1997).

Evaluating Stream Restoration

Public lands management on rangelands has devoted considerable effort to restoring ecosystem function to watersheds but has seldom verified that ecological goals have been attained. There is widespread recognition of the importance of evaluating the efficacy of stream restoration projects, yet few available records indicate that project assessment or monitoring has been performed. Many opportunities to learn from successes and failures and to improve future practices are being lost (Bernhardt et al. 2005, Moerke and Lamberti 2004, Alexander and Allan 2006). Even where monitoring and evaluation are performed, agreement on what constitutes an ecologically successful restoration project may be lacking (Palmer et al. 2005), and use of ineffective indicators (Bash and Ryan 2002) or an inadequate study design (Jansson et al. 2005) often hamper evaluation of project objectives. In California, many well-intentioned projects have been unnecessary, unsuccessful, or even detrimental to aquatic habitat (Kondolf 1998). Even though construction-phase project implementation is often defined as success, monitoring seldom documents effectiveness in improving conditions for aquatic life (Lake 2001, Bond and Lake 2003). Evaluating project success or failure requires the use of both baseline and reference data in a monitoring program that includes measurement of both structural (state) and functional (process) attributes; consideration of local and landscape spatial scales; information on current, historical, and anticipated future conditions; and it is based on ecologically grounded objectives and hypotheses (NRC 1992, Kondolf and Larson 1995, Kondolf and Micheli 1995).

In this study, we evaluated stream restoration success for two years following project implementation, through physical habitat measures such as substrate heterogeneity and biological monitoring of the aquatic macroinvertebrate community (bioassessment). Benthic invertebrate community metrics offer a sensitive ecological indicator of recovery of in-stream ecological structure and function in response to improved physical habitat conditions and have been used to monitor biological responses in a variety of stream restoration efforts (Larson et al. 2001, Muotka et al. 2002, Moerke et al. 2004, Harrison et al. 2004, Suren and McMurtrie 2005). Restoration monitoring is a critical project component for enabling adaptive management and documenting whether ecological objectives are attained. Defining ecological goals is a first step in restoration planning; then using appropriate measures and standardized study design provides an unambiguous process for recording the progress and success of each project (Noss and Cooperrider 1994). Integration of this type of monitoring into most restoration projects is seldom achieved because desired ecological outcomes are not made explicit, data gathering is inadequate and has no baseline, or monitoring expenses are not allocated (Bernhardt et al. 2005).

Restoration Project Description

Livestock grazing of the Bagley Valley Creek watershed began in the late nineteenth century. By the 1990s, the main Bagley Valley Creek channel had incised up to 6 m below the plane of its associated meadow and up to 15 m in width, and a network of gullies had propagated along perennial and ephemeral tributaries upstream. The Humboldt-Toiyabe National Forest purchased 800 ha within Bagley Valley in 1994 and later suspended cattle grazing and began planning active restoration of the incised channel and the watershed.

Bagley Valley Creek (Alpine County, California) is a second-order perennial stream in the Carson River Basin, on the east side of the Sierra Nevada (Figure 1, Table 1). The elevation of the watershed ranges from 1,920 to 2,720 m, with 3.5 km of perennial stream channel and a basin relief ratio of 0.23 (ratio of catchment elevation change to maximum basin length along main drainage line, indicating a relatively steep basin, prone to erosion). Mean annual precipitation is on the order of 80 cm (NWCC 1999), falling mostly as snow between December and April. Aside from irrigation ditches and roads, few or no structures are present in the watershed, and land use is largely recreational.

The restoration project focused on the entire watershed and had two objectives: 1) to restore the hydrological function by connecting Bagley Valley Creek with its historic floodplain; and 2) to stop the progressive erosion of the network of gullies that had formed in Bagley Valley (Table 2). Project design was based on explicit recognition of the need for physical habitat reconstruction, but ecological endpoints (aquatic life) were implicit in monitoring success. The design was based on previous U.S. Forest Service meadow restoration work using Rosgen stream classification, but was also informed by field measurements of bankfull discharge and the dimensions of historical channel fragments to determine final channel alignment and shape. Connecting the stream channel to the meadow plane and preventing additional gullying required the construction of structures using natural materials, but the designers intended to allow the stream channel to migrate and determine its course within the lower meadow.

The most intensive construction activities were within the upper 150 m of the meadow, where a new channel was excavated and constructed using erosion control fabric, imported substrate, and willow (*Salix* spp.) plantings.

Table 1. Summary of physical measurements for the four study sites, including the Restoration site before versus after project construction. For mean values, reported variability is one standard deviation. Latitude and longitude are datum NAD27.

Stream:	Bagley Valley Creek		Bagley Valley Creek	Slinkard Creek	Tributary Silver King Creek
Site:	**Prerestoration**	**Postrestoration**	**Downstream**	**Restored Reference**	**SKC Reference**
Latitude (°N)	38.5986		38.5923	38.6003	38.5519
Longitude (°W)	119.6480		119.6547	119.5680	119.6080
Elevation (m)	1,945		1,935	1,877	2,024
Slope (%)	2.40	2.42	1.79	1.25	8.30
Sinuosity	1.2	1.1	1.2	1.3	1.2
Mean Width (cm)	133 ± 54	154 ± 70	130 ± 72	81 ± 23	84 ± 29
Mean Depth (cm)	13 ± 8	7 ± 4	19 ± 13	22 ± 9	7 ± 5
Mean Velocity (cm/s)	51 ± 66	13 ± 15	14 ± 21	27 ± 23	5 ± 12
Discharge (l/s)	67 ± 26	12 ± 6	17 ± 9	36 ± 17	3 ± 4
Proportion Riffle, Pool (%)	56, 19	78, 11	33, 57	47, 29	59, 15

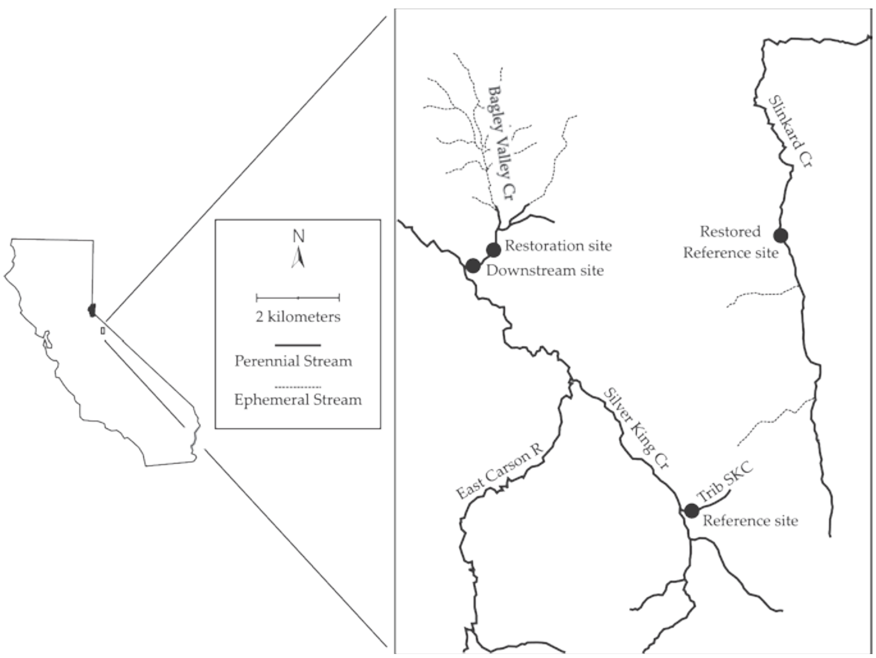

Figure 1. Location map in California for study sites showing the high density of ephemeral washes in the Bagley Valley Creek watershed.

At the upper end of the constructed channel, a stream bank reinforced with boulders approximately 2 m in height was constructed to direct the flow away from its existing alignment at the meadow's lateral edge to the meadow proper. The bank was needed to dissipate energy that otherwise would have led to the creek recapturing the abandoned channel. Downstream from the constructed channel, only a few rock weirs were placed to direct flow and dissipate energy while allowing a new channel to form in the lower meadow (further downstream, as the valley became more confined, flow then reentered the existing channel). The abandoned, deeply incised channel was filled by material borrowed from elsewhere in the meadow, which created an adjacent pond that had not existed previously. Lower portions of the meadow were also covered with erosion fabric and pulp fiber and revegetated where heavy equipment had disturbed soils. Reconstruction of the creek channel and other restoration activities were performed in 2001 (Figure 2).

The total project cost was approximately $2.6 million (2001 dollars), of which $2 million was spent directly on project construction and implementation (Table 3). An additional $1 million worth of measures to fill additional gullies and re-create tributary stream channels in other areas of the watershed were identified, but no funding was available to implement this work. The relative remoteness of the watershed, accessible only by a four-wheel-drive road and foot trails, and presence of historical and cultural resources (Native American artifacts) contributed to project cost. Funding for project monitoring described in this study amounted to approximately 1.5% of total project cost.

Methods

Study Sites and Design

To detect changes at the restoration site, we compared conditions in a reach of the former channel before restoration with the reconstructed channel. To evaluate downstream transfer of project effects, we also monitored an unrestored control reach of the creek below the project site at the same time. To provide a comparison to expected conditions, we monitored two reference streams in adjacent watersheds—one of which had been the subject of a similar restoration project 10 years

Fig. 2. Restoration of stream habitat in Bagley Valley Creek. The degraded, incised channel in 1999 (above right); the reconstructed channel upstream of the previous photo in 2002, the first year after restoration (above left); and the same location in 2003 showing extensive geomorphic and vegetative recovery (right). Photos by David Herbst

earlier, and another with low exposure to grazing and other land use activities. These four sites are 1) Bagley Valley Creek ("Restoration"), located in an incised channel disconnected from the meadow prior to project construction and at the top of the meadow project site within the new channel after construction; 2) Bagley Valley Creek ("Downstream"), below the restored meadow in an existing channel approximately 500 m downstream of the Restoration site, as close as possible to indirect influences that might be transported below the restoration area; 3) Slinkard Creek ("Restored Reference"), located 7 km from the Restoration site within a meadow on a first-order stream where restoration work had been completed in 1990; and 4) Tributary Silver King Creek ("SKC Reference"), located 6 km from the Restoration site on a first-order stream selected a priori to represent a similar landscape but with less land use disturbance (Figure 1).

We conducted preproject surveys in both 1999 and 2000 at each of the Bagley Valley Creek sites and once in

Table 2. Restoration activities in the Bagley Valley Creek channel and its watershed (based on the USDA Forest Service project pamphlet (2000) and M. Joplin, USDA Forest Service, Carson Ranger District, pers. comm.).

Channel Construction and Meadow Restoration	
Length	150 m of channel constructed to realign creek to meadow proper and floodplain (500 m downstream meadow channel reclaimed)
Structures	25 rock weirs/channel constrictions, rock drops, and bank armor installed in the channel and elsewhere in the meadow
Materials	600 m³ of earthen fill and boulders concentrated on the outside bank of a critical bend directing flow to the meadow proper
	Biodegradable erosion-control fabric on channel banks
	Gravel, cobble, and boulder substrate added to channel
	Willow cuttings/starts planted on banks
Abandoned channel	12,000 m³ of fill excavated adjacent to meadow (creating a pond) to regrade incised channel
Watershed Restoration	
Roads	2.6 km realigned away from meadow areas, with the former road obliterated, recontoured, and revegetated
	1.6 km regraded to provide for drainage and repair erosion damage
Irrigation ditches and berms	5 km recontoured and revegetated to restore natural drainage surfaces

2000 at the two reference sites. We conducted postproject surveys at all sites in both 2002 and 2003. Sample dates were concurrent at sites within each year, occurring between July 8 and August 23 in different years (within the period used to compare bioassessment data for this region). This selection of sites and years allowed us to evaluate 1) whether a short-term response occurred at the Restoration site; 2) whether conditions at the Restoration site approached those of the Reference sites; and 3) whether project construction altered conditions at the Downstream site. Physical habitat survey information enabled us to evaluate the comparability of physical features between sites over time, assess the relationship of physical habitat features to the macroinvertebrate community, and consider the mechanism(s) involved in macroinvertebrate community responses, if any.

Table 3. Estimated project costs for planning, construction, and monitoring (M. Joplin, pers. comm.).

Project Item	Cost
Aerial survey/orthophotography	$34,000
Hydrologic analysis	$33,000
Heritage resources survey and testing	$258,000
NEPA/CEQA planning documentation	$29,000
Engineering design, specs, contract preparation	$222,000
Construction/implementation	$2,000,000
Benthic macroinvertebrate and physical habitat monitoring	$40,000
Total	**$2,616,000**

Physical Habitat

We defined each site as a 150-m-length reach; measured slope with a survey level and stadia rod, and sinuosity as the ratio of the 150 m thalweg to straight-line distance; and delineated longitudinal distribution and length of riffle and pool habitats over the length of the reach. At each of 15 transects spaced at 10-m intervals, we recorded water depth, substrate size-class (i.e., fine [< 0.25 mm], sand [0.25–2 mm], gravel [> 2–64 mm], cobble [> 64 mm]), and current velocity at five equidistant points (n = 75 for each measure over the study reach), along with wetted stream width (n = 15), the type of structural cover on both banks (n = 30), and overhanging riparian canopy cover. Dominant bank structure cover between water level and bankfull channel level was classified as open ground, vegetated (presence/absence of herbaceous/woody [brush or trees] plants intersected at each bank on the transect), or armored (rock or log). We also measured riparian canopy cover density on each transect at each stream edge and at midstream facing upstream and downstream (n = 60) using a concave densiometer mirror to count overhanging vegetation reflected at 17 grid points (Platts et al. 1987). We calculated discharge from each transect as the averaged sum of one-fifth the width times depth and current velocity at each of the five transect points.

Benthic Macroinvertebrates

We collected benthic macroinvertebrates from riffle habitats within each study reach for each of five separate replicate samples Each replicate sample consisted of a composite of three fixed-area (900 cm²) collections taken with a 250-μm mesh D-frame net (30 cm width) at locations within riffle segments (chosen as points using a random number table that corresponded to linear reach distances where riffle habitat occurred). To sample, we placed the net against the substrate at each sample point and collected from a 30 × 30 cm area immediately above the net by turning and brushing rock and other substrates by hand while current carried dislodged invertebrates into the net. We processed samples in the field to clean and remove large rock and wood debris and removed invertebrates both by repeated filtering of the suspended fraction swirled from a bucket through a 100-μm mesh aquarium net and by hand-picking of remnant heavier organisms (snails and caddisflies) from shallow inspection pans. We then preserved collections in ethanol.

We subsampled macroinvertebrate field samples in the laboratory using a rotating drum splitter to obtain a minimum count of 250 organisms for identification (in practice generally yielding 300–500 organisms). We sorted the subsample split under a 10× microscope and counted and identified to the lowest practical taxonomic level (including Chironomidae), usually genus or species, based mainly on Merritt and Cummins (1996), Wiggins (1996), Stewart and Stark (2002), and Thorp and Covich (2001). Oligochaete worms and ostracods were not further classified. All stages of sample processing, identification, and data entry were rechecked to verify data quality.

Data Analysis

The data are presented here as contrasts of habitat conditions and invertebrate community metrics before and after the restoration project, as paired comparisons of individual metrics using parametric techniques, in comparison to a regional multimetric index of biological integrity (IBI) score, and as an ordination analysis of changing community similarity. Sample replication before and after restoration was insufficient for analysis of variance using a BACI (before-after, control-impact) design (Underwood 1994).

Within-site replication permitted assessment of sampling variability at each site during each date, such that differences betweens sites and dates could be evaluated and the results placed in a statistical context (Oksanen 2001, Hawkins 1986). We expected that the reconstructed stream would exhibit predictably improved biological indicators (e.g., greater diversity and more sensitive taxa) and conditions approaching those of the reference sites (selected a priori), and therefore that any statistically significant response could most likely be attributed to the restoration actions. Natural background variability in relation to restoration expectations was evaluated via before-after changes at the treatment site relative to year-to-year variations at downstream and reference sites, as well as from the consistency of response among multiple biological and physical indicators.

We compiled taxon lists and counts for each replicate sample to calculate relative abundance, total areal density of each taxon, and composite metrics. Each taxon was assigned a functional feeding group category (shredder, predator, filterer, collector, or grazer) and a tolerance value (TV) to disturbance and/or pollution based on a published source (Barbour et al. 1999). Metrics of community biotic integrity included mean and total taxa richness, composite community tolerance (modified Hilsenhoff biotic index), proportion of sensitive and tolerant taxa, proportion of functional feeding groups, dominance, and others. For metrics expressed as a proportion (% tolerant taxa), we used arcsine-square-root transformation for statistical computations to improve normality (Sokal and Rohlf 1995, Zar 1999). We inspected the distribution of each metric for violations of the assumption of normality. Finally, we screened all tests for unequal variances using critical values of the *F* distribution for variance ratio tests.

For each metric, we calculated 95% confidence intervals for the mean at each site for each sampling event. Two-sample *t*-tests using variance pooled by sampling event (i.e., year) and site were performed to evaluate 1) whether improvements in metric values were observed after project construction at the Restoration site; 2) whether metric values at the Restoration site were different than at reference sites before or after project construction (did the Restoration site differ prior to construction and did it approach the reference condition after); and 3) whether metric values at the Downstream site were different before compared to after project construction (did project construction impact the benthic community downstream).

In addition to comparison of the Bagley Valley Creek sites with the local reference streams, it was also possible to evaluate biological integrity in the context of a regional reference data set from a wider geographic coverage. Using an IBI developed for streams on the east slope of the central Sierra Nevada (Herbst and Silldorff 2006, and unpub. data), the study reaches were compared to conditions across streams from a similar climatic and geological setting. The IBI was developed by screening metrics for correlation with stressor gradients, minimal overlap with test site distributions, and high signal-to-noise ratio (reference-test difference to metric variation). This approach resulted in the selection of 10 metrics, rescaled and combined as an IBI that was calculated based on a 500-count random resampling of specimens identified from each site and sample date.

We used an ordination of the community composition using nonmetric multidimensional scaling (NMS) to compare community composition between sites and sample dates and to investigate shifts in community composition (e.g., was the postproject restoration site more similar to reference sites). Community dissimilarity for use with NMS was calculated using the Sørensen distance metric for the mean untransformed relative abundance of each taxon at each site for each sampling event. Sørensen is a distance metric that has better empirical performance than Euclidean alternatives for community datasets characterized by a sparse matrix (McCune and Grace 2002). All analyses were completed using PC-ORD (Version 5.05, MjM Software, Gleneden Beach OR). Analyses were run with taxa that occurred in two or more of the 14 site-date survey samples. Taxa that occurred in only a single site-date survey were removed to eliminate the effect of these rare taxa, since they provide little information on the overlap of community composition. The effect size and significance of potentially relevant groups apparent in the ordination were evaluated using the multiresponse permutation procedure (MRPP), a nonparametric technique for comparing group differences. There is circularity of logic in using MRPP to estimate the significance of groups identified post hoc from an ordination; however, its use in this way serves to describe and contextualize the effect size of any relevant groups identified. The MRPP was completed using community dissimilarity calculated as above using PC-ORD.

Results

Physical Habitat

Elevation, sinuosity, slope, and mean width, depth, and discharge were generally of similar magnitudes at each site, indicating comparability of physical characteristics between sites (Table 1). The SKC Reference site was the least similar of the four. The average slope at this site was 8% compared to a range of 1% to 2% for the other sites, its measured discharge and mean current velocity was about an order of magnitude lower, and it had the most extensive riparian cover. This site nonetheless exhibited pool-riffle morphology, and, consistent with all sites, invertebrate sampling was conducted only in lower gradient riffles. The Restored Reference site had only herbaceous cover along the channel, but bank grasses provided extensive shading and stable, undercut banks. The Bagley Valley reaches prior to project construction were generally wider, with more shallow bank angles and open banks and a greater extent of fine particle deposition than reference reaches. These differences are consistent with an eroded, exposed channel with limited riparian cover. The Downstream site was the only site with a higher proportion of pool than riffle habitat.

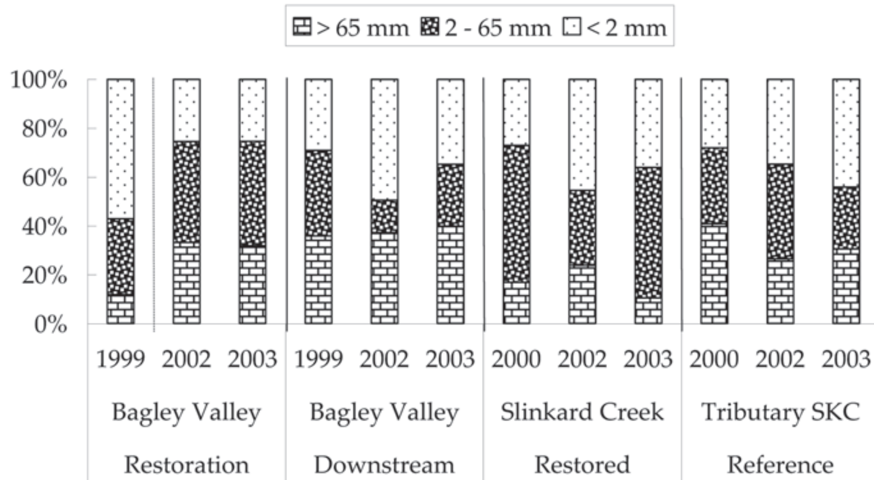

Figure 3. Percent of substrate particle size classes (from 75 point-counts taken as 5 measurements on each of 15 transects) at each site before and after project completion in 2001.

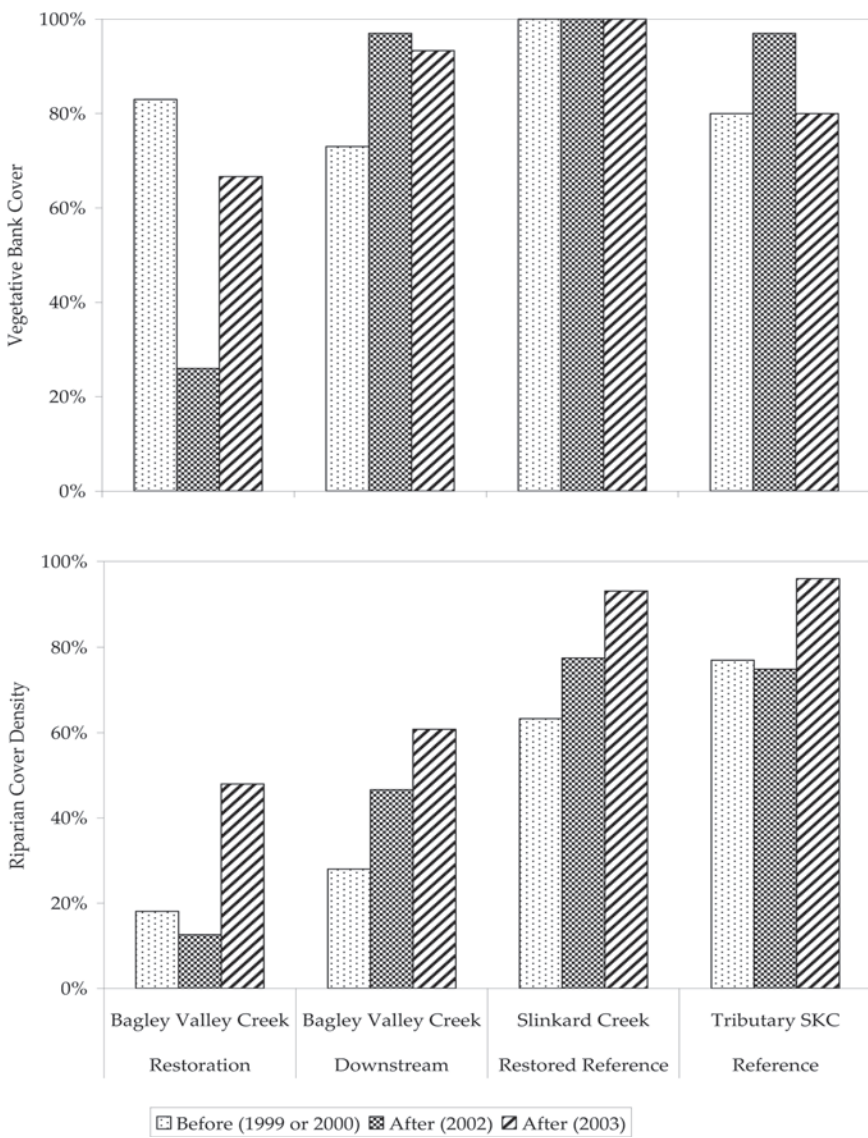

Figure 4. Vegetation cover (herbaceous or woody) on bank transects shown in upper panel, and density of riparian canopy cover (measured with a densiometer mirror) shown in lower panel for each site and sample date.

Postproject physical habitat measurements at the Restoration site indicated that the constructed channel was straighter, wider, shallower, and had a higher proportion of riffle habitat than the preproject channel (Table 1). Substrate particle size at the Restoration site also shifted from 57% fine + sand particles to a more even distribution among size classes, with only 25% fine + sand in both postproject years, resembling the Reference site (Figure 3). The proportion of fine + sand particles at the Downstream site increased from 29% to 49% between 1999 and 2002, possibly indicating an impact from project construction. But by 2003 the particle size distribution resembled that of 1999 (35% fine + sand), indicating that if project construction did cause an accumulation of fine and sand particles at the Downstream site, the effect appeared to be transient. Particle deposits found during a given sampling event may depend on the amount of sediment transport occurring during the antecedent peak runoff flows. Preproject flows were average to above average for 1999–2000, low-flow drought for the following construction year (2001), and moderate drought in 2002. Flows increased in 2003 to near average for spring runoff conditions. Flows may therefore have permitted sediment flushing in 2003 after accumulation under drought conditions in 2001–2002.

Vegetative bank cover decreased at the Restoration site following project construction (2002), but had reestablished by 2003 (Figure 4). By then, the density of riparian canopy also showed an increase, nearly threefold greater than in 1999. Bank cover included woody vegetation from willow plantings rather than just grasses, which dominated in the incised channel. Field observations indicated that the groundwater table elevation increased rapidly following the filling of the incised stream channel and rerouting the creek, and salvaged wetland vegetation took hold aggressively in the meadow (M. Joplin, USFS, pers. comm). Vegetative bank cover remained about the same at the other study reaches, but the density of riparian cover doubled over the years of the study at the Downstream site and increased at both reference sites (Figure 4).

Benthic Macroinvertebrates

At the Restoration site, two-sample *t*-tests of mean metric values before versus after project construction using variance pooled by sampling event indicated significant differences ($p \leq 0.05$) for metrics that characterized community sensitivity. The collective richness of Ephemeroptera (mayflies), Plecoptera (stoneflies), and Trichoptera (caddisflies) taxa (EPT taxa) increased by seven taxa ($p < 0.001$), and the ratio of EPT richness to total richness increased from 27% to 38% ($p = 0.001$, Figure 5). The proportion of tolerant organisms decreased from 21% to 5% ($p < 0.001$), and the proportion of sensitive taxa increased from 2% to 11% ($p = 0.002$, Figure 6). For most of these same metrics, the Restoration site was significantly different in the year prior to project construction compared to both of the reference sites, but not in either of the years following project construction. Compared to the Restoration site, the SKC Reference site had nine more EPT taxa in 2000 ($p = 0.02$), and five ($p = 0.26$) and four ($p = 0.21$) more EPT taxa in 2002 and 2003, respectively. Similarly, the Restored Reference site had six more EPT taxa ($p = 0.01$) in 2000, the same number in 2002, and two fewer ($p = 0.36$) in 2003. The EPT taxa that had not previously occurred or were rare before reconstruction but were found at the Restoration site afterward included the mayflies *Diphetor hageni, Paraleptophlebia*, *Ironodes*, and *Epeorus*; the stoneflies *Malenka* and *Skwala*; and the caddisflies *Lepidostoma*, *Rhyacophila* (mixed species groups), and *Ceratopsyche*. Differences in biotic index values mirrored those of EPT richness, declining after project construction (more of the community comprised sensitive taxa with low tolerance values). At the Downstream site, none of the community tolerance metrics were significantly different before versus after the project.

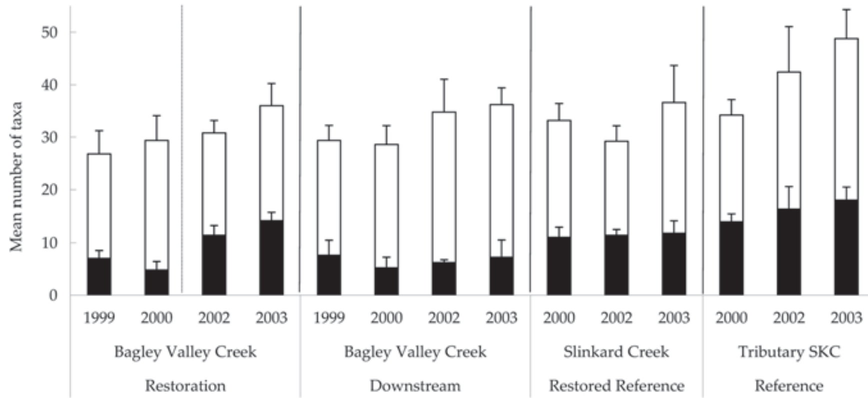

Figure 5. Mean taxa richness and inclusive EPT richness (black inset) with 95% confidence intervals (*n* = 5).

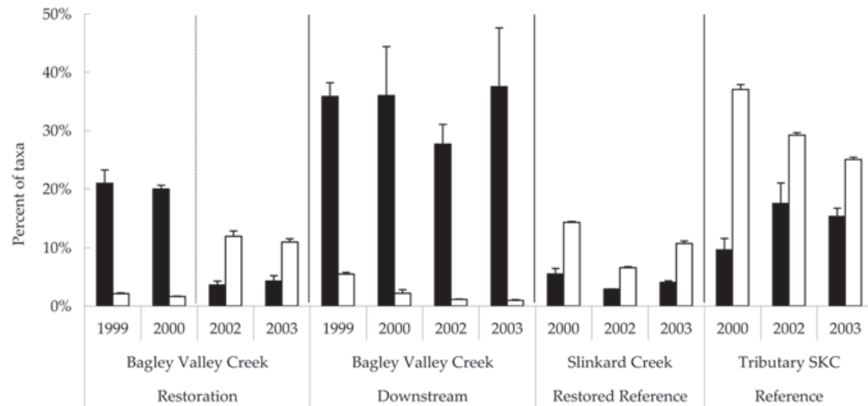

Figure 6. Mean proportion of tolerant (black bars; TV = 7–10) and sensitive (white bars; TV = 0–2) taxa with 95% confidence limits (*n* = 5), calculated using arcsine-square-root transformed values.

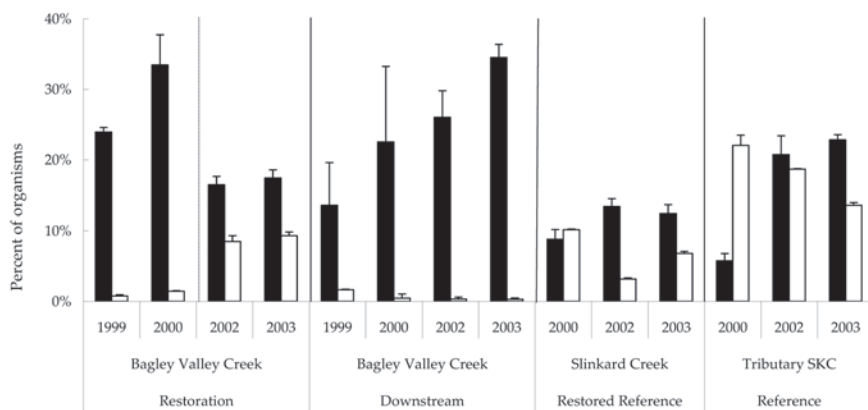

Figure 7. Mean proportion of filterers (black bars) and shredders (white bars) with 95% confidence limits (*n* = 5), calculated using arcsine-root transformed values.

Examination of functional feeding group composition at the Restoration site indicated that the mean proportion of shredders increased from 1% before to 9% after project construction ($p < 0.01$, Figure 7). The mean proportion of filterers decreased from 29% to 17%, but this was a marginally significant difference ($p = 0.07$). Trends were not apparent with other feeding

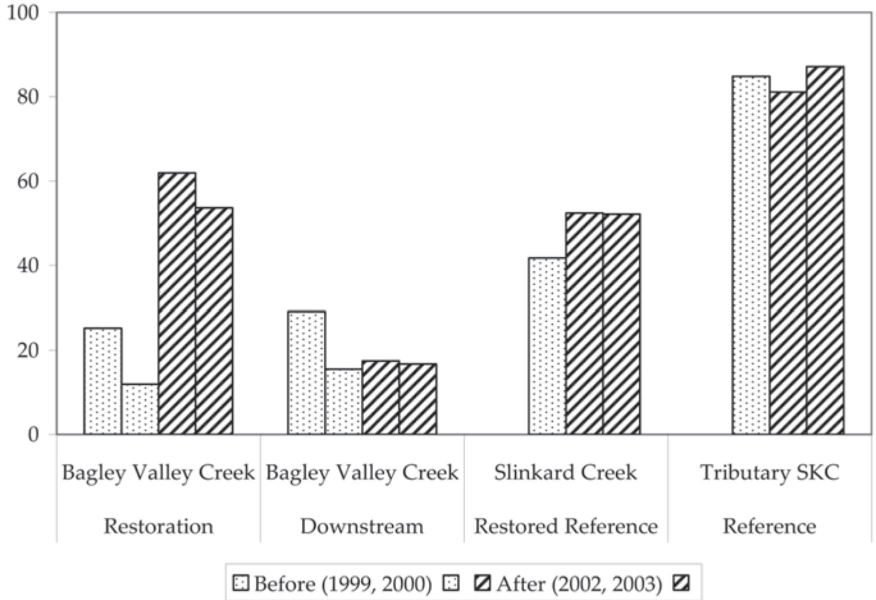

Figure 8. Index of Biological Integrity values calculated for each site and sample date.

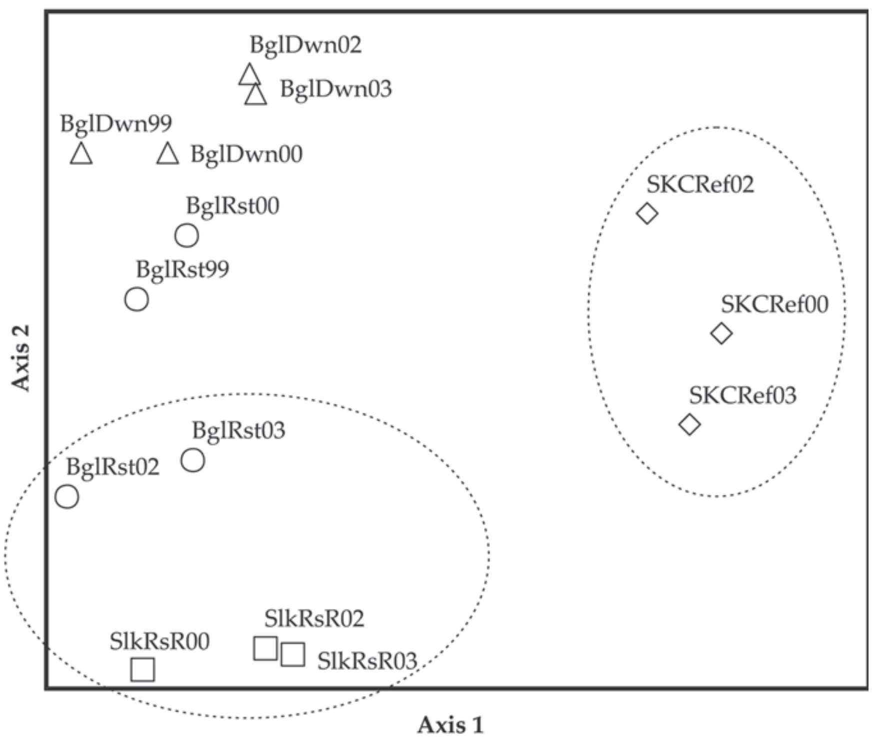

Figure 9. NMS ordination of community similarity among study sites and sample dates using mean macroinvertebrate taxa relative abundance for taxa that occurred in two or more samples. Sites are coded by symbol and samples are labeled with year as follows: ○ BglRst = Restoration site; Δ BglDwn = Downstream site; □ SlkRsR = Restored Reference site; and ◊ SKCRef = Silver King Creek Reference site.

guilds (predator, collector, and grazer). Collectors were generally the most dominant feeding guild, especially at both Bagley Valley Creek sites, where collectors ranged from 35% to 72% of the collected organisms.

Mean taxa richness increased by five taxa ($p = 0.04$) at the Restoration site after project construction, but a similar increase was also found at the SKC Reference site, tempering a conclusion that the restoration project resulted in an increase in diversity. While overall richness increased at both these sites, the EPT richness component more than doubled at the Restoration site compared to a smaller proportional increase at the Reference site. Total taxa richness (cumulative taxa richness in the five replicates collected from each site at each sampling event) showed similar trends. Between-year macroinvertebrate density was variable across all sites and showed no consistent trend before versus after restoration.

Mean dominance (percent of the community comprising the most abundant taxon) at the Restoration site was 32% ± 1% (95% CI) in the 2 years prior to project construction. Following construction, it increased to 48% ± 5% in 2002 and then decreased to 29% ± 1% in 2003. Mean dominance values were relatively consistent over the years at the Reference (15%–19%), Restored Reference (23%–36%), and Downstream (26%–39%) sites. Some taxa that were common before reconstruction disappeared from the new channel or became rare, including the amphipods *Hyalella*, being replaced by *Gammarus*, and Chironomidae (midges), which declined overall from 19%–20% relative abundance in 1999–2000 to 7%–10% in 2002–2003. Although changes in these and the EPT taxa noted above resulted in differing community composition following channel reconstruction, the three most common taxa remained the same—the mayfly *Baetis*, the blackfly *Simulium*, and the riffle beetle *Optioservus*.

The Eastern Sierra Region IBI scores were 14 to 23 before construction at the Restoration site and improved to 57 to 61 after (Figure 8). Scores for other sites changed less than 16 points over time. The relative consistency of scores at each site between events indicates that this multimetric index was more stable to changes in community composition than component metrics over the course of the study and provides strong evidence of a treatment effect. In the dataset used to develop the IBI, reference condition streams scored above 62, indicating that the Restoration site had nearly met this standard for biological integrity. The Restored Reference also scored just below the reference threshold, but the Downstream site consistently scored in the impaired range.

A combined total of 152 taxa were identified in the samples, of which 35 (23%) were found in a single site survey. The NMS analysis for all but these rare taxa yielded a two-dimensional solution that converged on a global minimum final stress of 11, with axes cumulatively explaining 86% of the variation in the original data (Figure 9). A three-dimensional solution reduced the stress further, but did not substantially improve explanatory power nor illuminate additional pattern. The ordination indicated that each site was most similar with itself in all sample events and well separated in ordination space, with the exception of the Restoration site, which was similar to the Downstream site before the project, but shifted in similarity to the Restored Reference site in 2002 and 2003 (Figure 9). Accordingly, MRPP analysis was performed on three groups: 1) the Downstream site and preproject Restoration site; 2) the Restored Reference site and the postproject Restoration site; and 3) the SKC Reference site. The analysis indicated that within-group homogeneity for these three groups was fairly high relative to that expected by chance ($A = 0.30$, $p < 0.001$).

Discussion

The physical habitat and benthic macroinvertebrate community measurements made in Bagley Valley indicate that riparian and in-stream ecological conditions improved over preproject conditions at the Restoration site within the first 2 years following project construction. The community shifted from being dominated by pollution- and disturbance-tolerant taxa to one comprising more sensitive taxa and more closely resembling the composition found at the two reference sites. The proportion of shredders increased and that of filterers declined, indicating that changes in resource availability yielded a more mixed trophic composition. Shredders rely on allocthonous coarse particulate organic matter typical of a fully functioning riparian stream ecosystem, while filterers flourish where suspended fine particulate organic matter is abundant (Cummins and Klug 1979). In a study of grazed and ungrazed sites on a Michigan stream (Strand and Merritt 1999), sampling from mixed substrates and habitat (pool and riffle) showed that both shredders and filterers increased in the less erosion-prone ungrazed habitat (though least on rock substrata). Our samples, taken only in riffle habitat from relatively clean rock surfaces, showed that filtering organisms were more abundant in the unrestored Bagley Creek sites and were reduced in the restored channel where fewer fine particles were found (Figures 3 and 7). These results suggest that under some conditions (rock substrate, riffles), fine particle deposition can provide a subsidy for growth of filter feeders, but in excess, or in limited supply, these organisms may decline (Hynes 1973, Rosenberg and Wiens 1978). The collector-gatherer functional feeding group may also benefit from increased availability of deposited fine organics, but only if not accompanied by fouling of water and habitat quality. The addition of boulder clusters to straightened, homogenous stream channels has been shown to increase retention and storage of detritus, which may yield greater macroinvertebrate abundance, especially shredders (Negishi and Richardson 2003, Lepori et al. 2005). Similar increases in habitat heterogeneity and storage of coarse organic matter likely contributed to the increase in proportion of shredders and the overall enhanced diversity of the macroinvertebrate community at the Restoration site.

Monitoring data for each of the first two years following project construction revealed some of the dynamics of recolonization and appeared consistent with our expectations. The early phases of stream habitat restoration involve substrate and macrohabitat changes (i.e., increased substrate particle size and proportion of riffle habitat), while secondary phases involved riparian establishment (i.e., increased shade, bank stability, and vegetative litter inputs) and stabilization of food resources from algal colonization and organic matter retention. This sequence should lead to reduced dominance, colonization by sensitive taxa, and stabilization of productivity and macroinvertebrate biomass. Many of the metrics, the IBI score, and overall community composition showed progressive improvement in biotic integrity at the restoration site in both postproject years. Dominance increased substantially in the first postproject year, possibly a transient disturbance effect, but returned to preproject levels in the second year. This is consistent with dynamics observed in disturbed communities, where one or a few taxa, usually generalists and opportunistic colonizers with short, multivoltine lifecycles (i.e., weedy species), dominate by taking advantage of uninhabited space (e.g., Wallace and Gurtz 1986). We observed temporary impacts from project construction at the Restoration site (i.e., channel disturbance and reduced riparian vegetation in the first postproject year), but the biological community recovered relatively quickly, as has been observed in other studies (e.g., Friberg et al. 1998, Biggs et al. 1998). Overall, this study provides evidence that active restoration projects of this nature have the potential to quickly enhance the biological integrity of degraded stream habitats.

Road construction activities that disturb the stream channel can cause significant increases in suspended sediment concentrations, causing short-term declines in benthic macroinvertebrate abundance and diversity downstream (Extence 1978, Barton 1977, Lenat et al. 1981, Fossati et al. 2001). Although stream restoration projects may also impact downstream communities, available studies indicate that impacts to macroinvertebrate communities may only last a few months (Tikkanen et al. 1994, Biggs et al. 1998). This study indicated that benthic macroinvertebrate communities at the Downstream site were not significantly impacted by project construction. However, this site had relatively poor biological integrity at the inception of the study, so the potential for causing and detecting further impairment may have been limited. The Downstream site did not suffer from the local erosion and incision characteristic of the upstream portions of the watershed, having a lower gradient than upstream (1.8% vs. 2.4%) characterized by depositional habitat (57% pool habitat). Thus rates of upstream sediment production would be expected to have the greatest influence on the biological community at this site. Streambed substrate measurements indicated that the proportion of fine + sand particles increased substantially in the first postproject year, but that they more closely resembled preproject conditions in the second year. If continued, this trajectory may indicate that upstream restoration efforts led to improved biological conditions at the Downstream site as well. This also illustrates the advantage of continued monitoring on a downstream reach that may be integrating effects of upstream restoration activities.

Comparison with the two reference sites enabled us to contextualize an ecologically meaningful response at the Restoration site over the two years of postproject monitoring, and to anticipate additional improvement that may be possible in the long term. In many respects, including taxonomic composition (Figure 9), total and EPT diversity (Figure 5), proportion of tolerant taxa (Figure 6), macroinvertebrate density, and dominance, the Restoration site was in a condition very similar to the Restored Reference site within two years following project construction. Conditions at the Restored Reference site were expected to be an intermediate goal, as this stream had been reconstructed only ten years earlier. The SKC Reference site conditions were considered indicative of the community that might be achieved at the Restoration site in the long term, although the communities at these two sites may never be identical given the local differences that exist between any two sites and the dynamic nature of benthic stream communities.

The monitoring methods employed in this study characterized the response from the combined restoration actions while allowing for some insight into the mechanisms responsible for improvement in indicators of biological integrity at the Restoration site. Our data are consistent with the "Field of Dreams" hypothesis (Palmer et al. 1997) in that construction of a new channel facilitated rapid recruitment of a more diverse assemblage of benthic invertebrates, characterized by more EPT taxa, an increase in sensitive taxa, and a functional shift in resource use toward consumers of decomposing coarse-particle organic matter. These changes were also accompanied by declines in tolerant taxa and fine particle feeders, consistent with expectations. Habitat linkages corresponding to these changes were larger substrate particles and less deposition of fine and sand particles, fabric revetment–stabilized banks, increased willow riparian cover, retention structures for coarse particulate organic matter, and the creation of a greater proportion of riffle habitat. The unrestored channel conditions favored sediment-inhabiting organisms that can tolerate turbidity during transport events and a stream bed dominated by fine particle deposition (Waters 1995). Over the long term, maintaining improved conditions may be contingent on the degree of success of the watershed rehabilitation efforts in reducing upstream erosion and sediment production.

This study demonstrates significant, short-term response in the biological community of a reconstructed headwater stream channel. The use of benthic macroinvertebrates as indicators and a study design that included collection of preproject samples and the sampling of reference sites provided a context with which to compare the restoration effort. At least for small streams where there is greater interface and connectivity with the riparian zone, this study also demonstrates that restoration of both a stream and its watershed can lead to rapid improvement in the ecological community of a headwater habitat severely impacted by livestock grazing. This short-term monitoring program was completed for approximately 1.5% of the total project budget and established a dataset that will facilitate long-term monitoring and assessment of this project, inform additional restoration activities in the watershed, and possibly other restoration efforts on small streams.

The results of this study might have been more rigorous had we been able to implement a study design that included greater replication in time, with less emphasis on within-event, within-site replication. Such an approach would have had the potential to better characterize environmental variability over time and permit a standard BACI analysis using ANOVA techniques (e.g., Underwood 1994). We nevertheless conclude that the study design and results demonstrate a restoration response, and the statistical tests show the degree of that response relative to preproject and potential future conditions. In many cases, especially with restoration efforts in remote locations such as this, logistical constraints may preclude repeated sampling over longer or more frequent time intervals. In addition, as was the case in this study, initial monitoring of project site itself may need to be accomplished before project funding or a larger study design is in place. Our results show that in such situations valuable insight and evidence may still be gained.

We emphasize that this study assessed only the short-term response to the project and can only speculate whether the completed landscape-level restoration activities, observed improvement in hydrologic conditions, and constructed stream channel improvements will persist over time and varied environmental conditions. Additional monitoring over an extended period (e.g., 10 years, Kondolf 1995) will be necessary to demonstrate the long-term efficacy of the restoration effort, although "how much is enough" may always be a moving target (Jansson et al. 2005), and endpoints must incorporate a recognition of the dynamic nature of nonequilibrium communities (Palmer et al. 1997).

Summary for Managers

- Intensive, active restoration of degraded rangeland headwater stream habitat can result in rapid recovery of biological integrity in benthic aquatic invertebrate life.
- Feasible engineering techniques are available to successfully restore severe incision and gullying in relatively remote headwater stream channels.
- Benthic macroinvertebrates can be effective indicators of within-stream ecological response to stream restoration at a modest cost.
- Concomitant sampling of reference streams and a downstream control site was essential in establishing ecological context and environmental variability as a background for evaluating restoration treatment effects.
- Even where repeated sampling over long time frames before and after restoration is not possible, use of multiple lines of physical and biological evidence may permit inference about project effectiveness.

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