Freshwater Biology

Freshwater Biology (2012) **57**, 204–217 doi:10.1111/j.1365-2427.2011.02706.x

APPLIED ISSUES

Effects of livestock exclusion on in-stream habitat and benthic invertebrate assemblages in montane streams

DAVID B. HERBST*, MICHAEL T. BOGAN*, SANDRA K. ROLL* AND HUGH D. SAFFORD†,‡

*Sierra Nevada Aquatic Research Laboratory, University of California, Mammoth Lakes, CA, U.S.A.

†United States Forest Service, Pacific Southwest Region, Vallejo, CA, U.S.A.

‡Department of Environmental Science and Policy, University of California, Davis, CA, U.S.A.

SUMMARY

1. Stream and riparian ecosystems in arid montane areas, like the interior western United States, are often just narrow mesic strands, but support diverse and productive habitats. Meadows along many such streams have long been used for rangeland grazing, and, while impacts to riparian areas are relatively well known, the effect of livestock grazing on aquatic life in streams has received less attention.
2. Attempts to link grazing impacts to disturbance have been hindered by the lack of spatial and temporal replication. In this study, we compared channel features and benthic macroinvertebrate communities (i) between 16 stream reaches on two grazed allotments and between 22 reaches on two allotments where livestock had been completely removed for 4 years, (ii) before and after the 4-year grazing respite at a subset of eight sites and (iii) inside and outside of small-scale fenced grazing exclosures (eight pairings; 10+ year exclosures) in the meadows of the Golden Trout Wilderness, California (U.S.A.).
3. We evaluated grazing disturbance at the reach scale in terms of the effects of livestock trampling on per cent bank erosion and found that macroinvertebrate richness metrics were negatively correlated with bank erosion, while the percentage of tolerant taxa increased.
4. All macroinvertebrate richness metrics were significantly lower in grazed areas. Bank angle, temperature, fine sediment cover and erosion were higher in grazed areas, while riparian cover was lower. Regression models identified riparian cover, in-stream substratum, bank conditions and bankfull width-to-depth ratios as the most important for explaining variability in macroinvertebrate richness metrics.
5. Small-scale grazing exclosures showed no improvements for in-stream communities and only moderate positive effects on riparian vegetation. In contrast, metrics of macroinvertebrate richness increased significantly after a 4-year period of no grazing.
6. The success of grazing removal reported here suggests that short-term removal of livestock at the larger, allotment meadow spatial scale is more effective than long-term, but small-scale, local riparian area fencing, and yields promising results in achieving stream channel, riparian and aquatic biological recovery.

Keywords: benthic macroinvertebrates, Golden Trout Wilderness, livestock grazing, Sierra Nevada, stream restoration

Correspondence: David B. Herbst, Sierra Nevada Aquatic Research Laboratory, University of California, 1016 Mt. Morrison Road, Mammoth Lakes, CA 93546, U.S.A. E-mail: herbst@lifesci.ucsb.edu

Introduction

Although streams of the arid intermountain region of western North America are usually small, they harbour diverse and productive riparian and aquatic ecosystems (Minshall, Jensen & Platts, 1989). Rangeland streams with confined riparian zones have been shown to be vulnerable to impacts from livestock grazing, exhibited as eroding banks, increased sedimentation, burial of spawning gravels, loss of vegetation cover and stature, increased water temperatures, reduced dissolved oxygen, nutrient enrichment and increased algal growth (Kauffman & Krueger, 1984; Kondolf, 1993; Armour, Duff & Elmore, 1994; Fleischner, 1994; Trimble & Mendel, 1995). About 80% of streams on western rangelands have been estimated to be damaged by livestock grazing (Belsky, Matzke & Uselman, 1999).

Although impacts to rangeland riparian areas are well known, the effect of livestock grazing on aquatic life in streams has received less attention. Habitat features and chemical water-quality monitoring alone are unlikely to detect cumulative non-point problems resulting from livestock grazing (MacDonald, Smart & Wissmar, 1991). More recently, monitoring efforts have incorporated invertebrate communities to assess grazing impacts because they are sensitive to many of the effects of grazing on stream habitat (Li *et al.*, 1994; Waters, 1995; Strand & Merritt, 1999; Scrimgeour & Kendall, 2003; Rios & Bailey, 2006; Braccia & Voshell, 2006, 2007; Ranganath, Hession & Wynn, 2009). Monitoring guidelines for forest land-use practices (MacDonald *et al.*, 1991) and livestock grazing (Bauer & Burton, 1993) suggest that bioassessment using macroinvertebrates is effective for gauging impacts associated with these types of landscape disturbances.

Studies of livestock grazing management on stream condition are often of limited value because they seldom include (i) integrated measures of habitat and aquatic biota, (ii) contrasts of differing spatial scales of livestock grazing exposures, (iii) control and treatment sites and replication, (iv) pre-treatment baseline conditions or (v) exposures over varied levels of grazing-related disturbance (Platts, 1991; Belsky *et al.*, 1999). Bank, channel and aquatic habitat components may respond over different temporal and spatial scales to disturbance or restoration, thereby often requiring long-term monitoring to detect integrated changes. Riparian vegetation can recover rapidly when grazing exposure is reduced or eliminated (e.g. Myers & Swanson, 1995). Channel structure and in-stream recovery of ecological health, however, may take longer and depend not only on local conditions but also on upstream catchment land use, sediment sources and the timing of sediment-flushing flows. Some rangeland streams may have degraded to altered ecological states from which they may not recover (Laycock, 1991).

The Golden Trout Wilderness (GTW) was established in 1978 as part of the ground work for rehabilitating the Kern Plateau and the genetic and habitat integrity of the state fish of California, the California Golden Trout (CGT), *Oncorhynchus mykiss aguabonita* (Moyle, 2002). One of the main perceived threats to stream habitat quality and stability in the GTW is grazing-induced stream degradation. Large herds of sheep caused significant changes in the riparian flora of the GTW by the end of the 19th century (Dull, 1999). Although grazing by sheep has been halted, cattle have continued to graze on the Kern Plateau under Forest Service lease since 1910. After creation of the GTW in 1978, and addition of the CGT to the USFWS candidate list for protection under the Endangered Species Act in 1991, public pressure mounted to reduce grazing impacts on CGT habitat. In 2001, the Inyo National Forest temporarily halted grazing on the Whitney and Templeton allotments because of livestock impacts on stream condition and hydrology. In upholding vacation of the allotment, the Deputy Regional Forester directed that a monitoring programme be developed to assess trends in stream and riparian meadow condition in both grazed and ungrazed allotments. The goal of this study was to provide an integrated evaluation of physical and biological stream conditions in grazed and ungrazed allotments within and adjacent to the GTW, and ultimately to provide information for the management of the GTW.

Methods

Study sites

The GTW is located on the Kern Plateau at the southern end of the Sierra Nevada, California (Fig. 1). The GTW includes 123 000 ha of montane, subalpine, and alpine ecosystems, and is located in the Inyo National Forest. Three of the four grazing allotments are situated entirely within the GTW, while the fourth (Monache) includes lands to the south of the GTW (Fig. 1). Bedrock in the study area is primarily Mesozoic granitic rock with scattered late Tertiary andesitic cones and basalt flows. Climate is montane Mediterranean, with dry summers and cold, wet winters. Mean annual precipitation ranges from 250 to 750 mm, deposited primarily as winter snow. Mean annual temperatures range from near freezing at higher altitudes to 12–13 °C at lower altitudes in the south

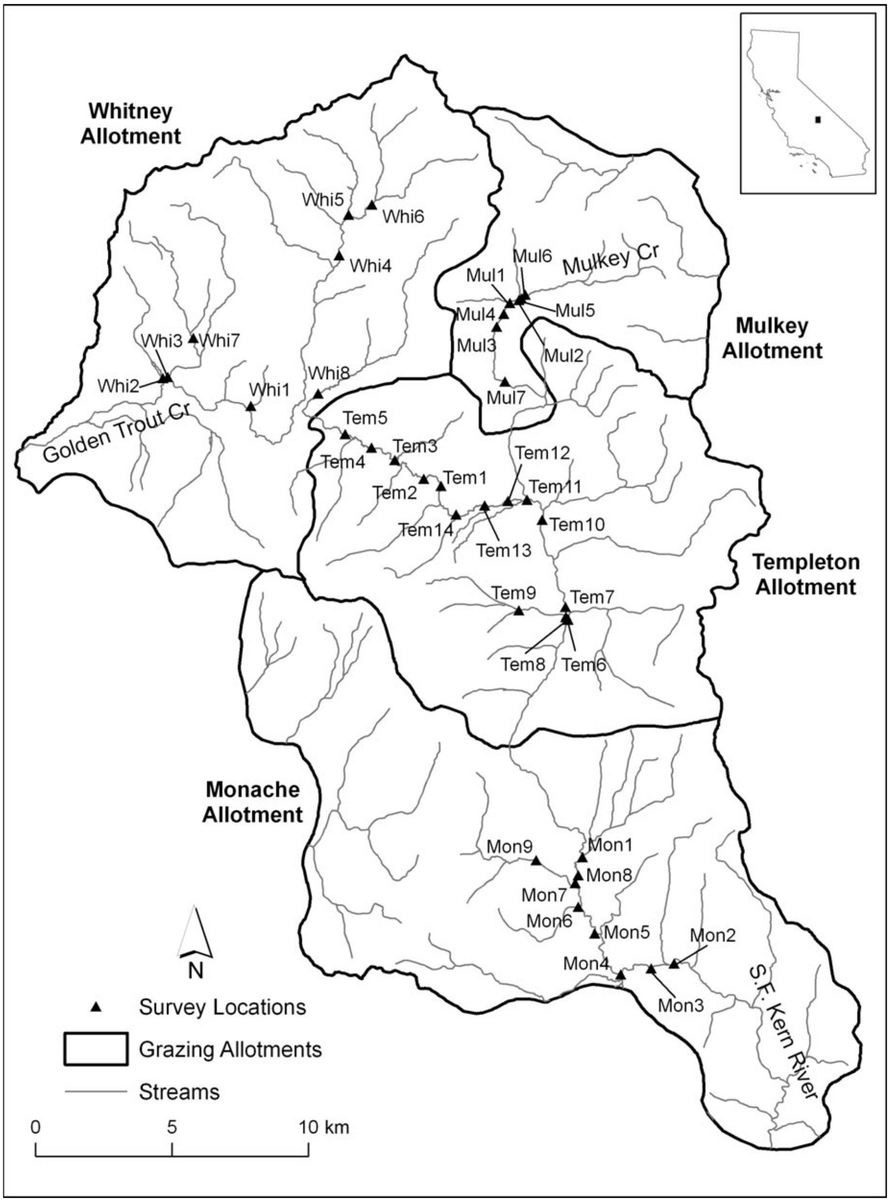

Fig. 1 Map of the allotments in the Golden Trout Wilderness region showing study reaches according to the site codes of Table 1. Golden Trout Creek flows to the west (all Whi except Whi8), and all other subcatchments and study reaches drain south into the South Fork Kern River. Allotment boundaries shown as bold black lines and streams as grey lines.

(Miles & Goudey, 1997). Altitudes at our sampling sites range from about 2400 m a.s.l. at the lower Monache Meadow sites to about 2950 m a.s.l. in Big Whitney Meadow. Vegetation in the study area is characterised by open conifer forests dominated by Jeffrey pine (*Pinus jeffreyi* Balfour) at lower altitudes, and lodgepole pine (*P. contorta* Douglas) and foxtail pine (*P. balfouriana* Balfour) at higher altitudes. All of the streams sampled were perennial, low gradient channels forming sinuous riffle-pool systems. Each study site was marked on a map and site altitude, latitude and longitude and stream order determined using TOPO! software (National Geographic, 2001). Slope for each reach was determined using a 30-m resolution digital altitude model in GIS (1-m vertical resolution). Physical data for the stream reaches are summarised in Table 1.

Field sampling

Stream surveys were conducted from mid-July until mid-September 2004 on two allotments that were removed from livestock grazing since 2000 (22 stream reaches), and in the two other allotments that continued to be grazed (16 reaches). At each site, 250 metres of stream length were

Table 1 Physical habitat features (means) of Golden Trout Wilderness stream reaches surveyed in the study of grazed and ungrazed allotments

Stream name	Site code	Latitude	Longitude	Stream order	Fenced exclosure	Reach slope (%)	% Bank erosion	Bank angle (degrees)	Mean riparian % Shade Cover	Bankfull W : D ratio	% Fines and sand	% Pebble and cobble	% Algae cover	Conductivity (μS cm^{-1})	Aquatic vegetation % cover	Wetted width (cm)
S.Fk. Kern	Mon1	36.2178	118.1723	4	Ai	0.4	32	150	20.8	19.1	35.4	58.5	53.8	200	4.6	753
S.Fk. Kern	Mon7	36.2091	118.1761	4	Ai	<0.4	42	153	11.9	14.6	56.9	0.0	44.6	200	16.9	565
S.Fk. Kern	Mon8	36.2124	118.1745	4	Ao	0.4	80	151	3.1	15.7	52.3	1.5	18.5	231	16.9	552
S.Fk. Kern	Mon4	36.1797	118.1570	4	Bi	<0.4	0	150	4.8	26.2	100.0	0.0	23.1	274	1.5	472
S.Fk. Kern	Mon5	36.1928	118.1678	4	Bi	<0.4	50	140	12.8	26.7	66.2	1.5	56.9	243	4.6	551
S.Fk. Kern	Mon6	36.2017	118.1744	4	Bo	<0.4	100	152	0.6	20.2	72.3	1.5	12.3	269	7.7	733
S.Fk. Kern	Mon2	36.1831	118.1345	5		<0.4	88	160	4.2	29.9	78.5	0.0	90.8	273	0.0	512
S.Fk. Kern	Mon3	36.1812	118.1437	5		<0.4	94	154	1.8	25.1	66.2	0.0	58.5	293	0.0	448
Soda	Mon9	36.2174	118.1919	2		<0.4	14	143	47.5	4.8	96.9	0.0	7.7	149	76.9	148
Bullfrog	Mul1	36.4010	118.2055	2	Ci	0.4	16	146	61.3	4.0	62.0	2.0	20.0	82	78.0	127
Bullfrog	Mul2	36.4018	118.2020	2	Co	<0.4	8	101	37.9	5.6	43.1	3.1	3.1	77	47.7	87
Mulkey	Mul3	36.3929	118.2101	3		<0.4	56	94	14.8	10.1	43.2	13.7	8.4	76	24.2	164
Mulkey	Mul4	36.3974	118.2074	3	Di	<0.4	22	111	31.3	7.4	44.6	16.9	38.5	72	24.6	162
Mulkey	Mul5	36.4026	118.2008	3	Di	<0.4	56	149	14.3	24.7	26.2	20.0	89.2	79	29.2	205
Mulkey	Mul6	36.4038	118.1993	3	Do	0.4	40	146	7.9	9.3	41.5	10.8	84.6	73	43.1	238
Mulkey	Mul7	36.3753	118.2072	3		0.8	28	118	17.9	7.7	55.4	1.5	9.2	75	12.3	227
S.Fk. Kern	Tem1	36.3404	118.2325	3		0.8	14	116	40.7	7.2	73.8	1.5	12.3	95	43.1	380
S.Fk. Kern	Tem10	36.3299	118.1900	4		<0.4	18	118	32.5	11.6	44.6	1.5	49.2	105	9.2	545
S.Fk. Kern	Tem11	36.3363	118.1975	3		<0.4	44	133	21.4	13.6	52.3	1.5	15.4	106	12.3	413
S.Fk. Kern	Tem12	36.3354	118.2054	3	Ei	1.6	30	137	22.9	13.4	15.4	13.8	20.0	115	43.1	506
S.Fk. Kern	Tem13	36.3341	118.2148	3	Eo	<0.4	32	139	17.3	16.7	52.3	3.1	32.3	101	29.2	435
S.Fk. Kern	Tem14	36.3309	118.2270	3		<0.4	48	111	33.6	7.6	26.2	10.8	46.2	90	24.6	398
S.Fk. Kern	Tem2	36.3429	118.2398	3		<0.4	20	133	36.0	6.1	46.2	4.6	32.3	93	56.9	439
S.Fk. Kern	Tem3	36.3504	118.2516	3		<0.4	16	119	33.9	8.2	83.1	4.6	32.3	88	33.8	378
S.Fk. Kern	Tem4	36.3529	118.2609	3		1.2	26	127	21.2	9.8	32.3	1.5	13.8	83	50.8	341
S.Fk. Kern	Tem5	36.3569	118.2722	3		0.4	26	122	27.9	11.6	41.5	3.1	41.5	82	33.8	337
S.Fk. Kern	Tem6	36.2968	118.1799	4		1.2	22	141	28.2	20.3	38.5	15.4	33.8	128	16.9	619
S.Fk. Kern	Tem7	36.3010	118.1811	4		<0.4	4	123	34.3	18.6	46.2	15.4	38.5	133	10.8	699
Strawberry	Tem8	36.2977	118.1811	2		1.2	22	116	39.1	10.6	10.8	37.4	21.5	55	21.5	110
Strawberry	Tem9	36.2994	118.2003	2		1.2	6	115	42.9	4.6	6.2	58.5	64.6	56	18.5	104
Groundhog	Whi1	36.3654	118.3105	1		2.4	12	95	61.3	3.6	81.5	0.0	24.6	102	33.8	83
L Whit. N. Fk.	Whi2	36.3753	118.3468	1		2.0	14	109	27.9	9.8	58.5	9.2	4.6	111	53.8	337
Johnson	Whi3	36.3756	118.3451	2		1.6	0	113	40.8	8.2	13.8	60.0	0.0	80	0.0	359
Golden Trout	Whi4	36.4164	118.2749	4		<0.4	24	98	37.3	6.0	16.9	60.0	0.0	72	1.5	291
Golden Trout	Whi5	36.4299	118.2710	3		<0.4	32	108	25.7	7.9	38.5	20.0	0.0	71	15.4	98
Stokes Stringer	Whi6	36.4331	118.2625	3		0.4	18	101	25.7	4.3	26.2	10.8	35.4	70	10.8	164
Salt Lick	Whi7	36.3889	118.3342	2		1.2	0	96	50.2	4.3	43.1	1.5	1.5	72	10.8	217
S.Fk. Kern	Whi8	36.3704	118.2834	3		<0.4	2	138	43.9	7.5	52.3	0.0	21.5	81	73.8	231

delineated, noting the length of each riffle and pool unit present (erosional and depositional environments). Water temperature, conductivity and pH were recorded using a portable Oakton pH/con10 m (Oakton Instruments, Vernon Hills, IL, U.S.A.). Recordings were taken at 13 channel cross-sections at 20-m intervals of wetted width, bank angles to the water surface (in degrees), bank erosion rating (unstable eroding, vulnerable or stable) and riparian cover (using a directional grid densiometer concave mirror to score the density of reflected vegetation at both banks and mid-stream up and down). Across each of the channel cross-sections, measures of depth, substratum size class (fines, sand, gravel, pebble, cobble, boulder; corresponding to <0.25, 0.25–2, 2–16, 16–64, 64–256, >256 mm, respectively), presence/absence of algae or macrophyte vegetation and current velocity at 0.6 depth (using a prop-driven current meter) were taken at five equally spaced points. Measures of bankfull width and depth were taken at four cross-section locations. Photographs were taken at fixed points in each study reach to contrast channel and riparian forms.

Benthic invertebrates were sampled from riffle habitats using a targeted-riffle composite method; samples from eight randomly selected locations within delineated riffles of each reach were taken and pooled. Samples were taken using a 30-cm wide D-frame net with a 500-μm mesh. The substratum within a 30×30 cm area immediately upstream of the net was disturbed, and invertebrates and organic matter washed into the net by the current. After collection, the pooled samples were placed in a bucket, larger rocks, sticks and leaves were removed and the sample was elutriated by repeated swirling and pouring off of lighter fractions (containing invertebrates, algae and organic matter). The lighter fractions were washed onto a 100-μm-mesh net and preserved in denatured ethanol. The remnant sand and gravel in the bucket was picked by hand in shallow, white trays to remove and preserve any remaining case-bearing caddisflies (Trichoptera) or shelled molluscs.

Sample processing

Samples were divided to obtain counts of *c.* 500–900 individuals using a Folsom plankton splitter. An additional search of the remnant sample for large and rare specimens was conducted using a magnifying visor. All individuals in the subsample were identified to genus or species (except oligochaetes and ostracods). To equalise counts of taxon richness, full-count data were re-sampled (using the R-program) to a fixed count of 500. For abundance data, counts were extrapolated from the original fraction and the per cent abundances calculated. Using this matrix, a number of metrics commonly used in bioassessment were calculated: total taxon richness; Ephemeroptera, Plecoptera and Trichoptera (EPT) richness and per cent abundance; number of sensitive taxa (having tolerance values of 0–2 according to listings for western streams; SAFIT, 2010); per cent of tolerant taxa (those with tolerance values 7–10); per cent dominance (per cent of total individuals counted belonging to the three most common taxa); overall density (abundance) and the per cent of total counts that are large (>5 mm), long-lived taxa that require stable habitat.

Grazing disturbance

Livestock had access to any part of the allotment deemed suitable during the summer grazing season. Herds are typically distributed into particular meadows or meadow groups in a regular seasonal pattern, with herd distribution managed by salt placement and herd riding. After cattle are driven to a given meadow group, they are allowed to drift back into the larger meadow(s) in the allotment (D. Hubbs, and H. Swartz, Inyo National Forest, pers. comm.). Unfortunately, livestock grazing records on public lands are often poorly maintained and incomplete, and standard animal unit month stocking levels are difficult to assign as actual exposure levels along streams. The records we examined from the GTW allotments were no exception and were further complicated by irregular summer season movement of cattle between meadows of different sizes and streamside accessibility. Qualitative evaluations of grazing used by Inyo National Forest range management staff did not match the few records of AUM stocking rates we felt were reliable. Because livestock distribution rules are based partly on soil moisture, certain parts of the GTW (especially in the Mulkey allotment) could be more heavily impacted in years of low or high precipitation.

Given these uncertainties, and the complete lack of records for some meadow allotments, we opted to assess habitat degradation associated with grazing pressure using a direct measure of observed channel alteration (i.e. bank erosion). While stocking rates would show exposure levels, they do not show actual grazing-related disturbance, which can vary from site to site depending on local stream vulnerability (soils, moisture, etc.) and on the vagaries of how and where grazing occurs near the stream. Direct grazing effects may be discerned from the physical habitat impacts that typify the syndrome of degradation that is caused by overgrazing. Trampling alongside streams erodes and flattens banks, reduces riparian cover and results in wide and shallow channel profiles (Trimble & Mendel, 1995; Belsky *et al.*, 1999). We quantified per cent bank erosion at the reach scale as the number of transect intercepts that showed erosion as degraded and sloughing bank structure where cattle trampling had occurred (20 transects per reach, at each bank for $n = 40$ observations per reach). We believe this is a conservative assessment of trampling damage because banks scored as vulnerable but not actively eroding were not included in counts.

Before and after livestock removal

To evaluate changes in the invertebrate community before and after the grazing moratorium, we compiled invertebrate community data prior to the period of grazing in the Templeton and Whitney allotments. The most useful data for comparative purposes came from US Forest Service stream surveys in the Whitney and Templeton allotments in 1999 and 2000, immediately before removal (M. Vinson, unpubl. data). These pre-removal data consisted of riffle samples collected using a Surber sampler from sites situated within reaches surveyed in our study. This yielded prior collections from eight matched sites, resampled in 2004. Sites within exclosures were not included.

Although specimen counts for comparing the before-and-after data sets were the same (in the range of 500–1000 organisms), many of the identifications in the pre-removal data set were determined only to the family level. Consequently, identifications from the current study were retracted back to family level so that all analyses were based on a common taxonomic resolution. Grouping the data to family decreased the sensitivity of the data set, because tolerance/sensitivity values vary within families; therefore, analyses based on these metrics were not performed. To assess the historical changes, mean abundance and richness were compared between samples taken before (1999, 2000) and after (2004) removal of grazing.

Exclosure studies

Small-scale fencing of riparian zones along streams in the GTW provided an opportunity to examine differences inside these exclosures to sites located immediately outside (upstream) of the exclosures. Fenced areas ranged from 10 to 13 years in age, and varied in length from several hundred to several thousand metres. In two of the longer exclosures, we sampled several inside reaches and compared each with a single outside reach. Four of the exclosures were in the grazed allotments, and one was in an ungrazed allotment. For the purpose of comparing grazed to ungrazed reaches (2000–2004) at the allotment scale, all sites in grazed allotments were treated as grazed, even if inside exclosures (7 of 16 sites), to account for impacts generated outside fenced areas.

Statistical analyses

Statistical analyses were carried out using XLStat (Addinsoft Corp., New York, NY, U.S.A.), PC-ORD v.5 (MJM software, Gleneden Beach, OR, U.S.A.; McCune & Grace, 2002), and Spatial Analysis in Macroecology SAM v.4 (Rangel, Diniz-Filho & Bini, 2010). We used simple linear regression to relate indices of taxon richness and stressor tolerance (total richness, EPT richness, sensitive taxa richness and per cent tolerant taxa), as well as channel and riparian features, to the extent of bank erosion due to livestock trampling across all sites.

Nonparametric Mann–Whitney *U*/Wilcoxon rank sum tests were used to evaluate differences in habitat features and the invertebrate community metrics of richness, tolerance, per cent with body size >5 mm (long-lived taxa) and dominance (the three most common taxa as per cent of total) contrasting sites on grazed allotments with the ungrazed allotments. As several study sites were located close to one another (<1 km) in the same catchment, spatial autocorrelation of community structure was examined at all sites using Mantel correlograms. Autocorrelation decreased from 0.349 (Pearson's r value) to 0.109 after 750 m, so seven sites within this proximity were excluded from statistical tests of grazing effects (excluded sites: Mon1, Mul1, Mul2, Mul6, Tem6, Tem8 and Whi2). Conditions inside versus outside grazing exclosures and before versus after livestock removal from the Whitney and Templeton allotments were compared using the paired Wilcoxon signed-rank test.

After examining Spearman's rank correlations among habitat variables, and eliminating related variables with correlations of $r > 0.70$, independent variables for multiple linear regressions were selected using the forward stepwise approach (using Number Cruncher Statistical Software 2007; Kaysville, UT, U.S.A.). Robust regression (using Huber's method) models were then built for total and EPT richness, and for sensitive taxa richness and per cent tolerant taxa, using only the best-fit variables. Normality of the dependent variables was verified with Q-Q plots and the Kolmogorov–Smirnov test.

Multivariate techniques were used to assess invertebrate community responses to grazing disturbance in the GTW study reaches. Non-metric multi-dimensional scaling (NMDS) ordination of all samples was used to examine patterns in community structure across the GTW and to determine which environmental factors were associated with the observed gradients in community structure. Environmental factors examined in the NMDS plots included all variables in Table 1. Additionally, the Pearson's correlation value of each taxon with the NMDS axes were examined to evaluate which taxa were driving the observed gradients in community structure. Multi-response permutation procedure (MRPP) tests were used to evaluate the distinctness of community groups based on grazing status.

Results

Stream reaches in GTW allotments that had been rested for 4 years (hereafter 'ungrazed sites') had significantly higher habitat quality and biological integrity than those with a continuous history of seasonal grazing (hereafter 'grazed sites'), even after accounting for spatial autocorrelation (Table 2). Benthic invertebrate richness metrics were significantly higher at ungrazed sites (for total, EPT and sensitive taxa). Large, long-lived invertebrates always comprised a small fraction of communities and did not differ with grazing status, nor did the dominance composed by the three most common taxa or total density. Collector-gatherer feeding groups made up the majority of organisms from all stream communities, and no trophic group differed between grazed and ungrazed sites (data not shown).

Ungrazed sites supported more stable (less eroded), steep-angled (more undercut) banks and had more riparian cover and vegetated stream banks than grazed sites. Substrata of grazed sites was significantly dominated by particle sizes <2 mm in diameter (fines and sands), and bankfull width-to-depth (W : D) ratios were higher at grazed sites. Conductivity was also significantly higher at grazed sites, especially downstream in lower Monache. On average, grazed sites were situated at slightly higher altitudes than ungrazed sites (mean altitude: grazed 2 535 m a.s.l., ungrazed 2676 m a.s.l.).

While livestock removal at the allotment scale was associated with greater biological diversity and improved habitat, these patterns were not found at the scale of local fenced exclosures (Table 3). Comparison of paired reaches inside and outside exclosure fences showed that only riparian shade cover and bank vegetation cover were significantly enhanced by the exclosure of cattle (although bank erosion was marginally higher outside exclosures, and total richness marginally lower).

The effect of removing livestock from entire allotments was also apparent when comparing invertebrate community samples from 1999 to 2000 (before or at the start of grazing removal) to those in 2004 (after 4 years without grazing) (Table 4). Even at the low level of taxonomic resolution used here, total richness and EPT richness showed significant increases after the 4-year period of no grazing.

Table 2 Contrasts of physical habitat factors and aquatic macroinvertebrate biotic metrics between grazed and ungrazed sites in and near the Golden Trout Wilderness (after the removal of autocorrelated sites)

	Grazed (n = 12)	Ungrazed (n = 19)	P-value
Physical habitat			
Slope	0.2 (±0.1)	0.6 (±0.3)	0.058
Altitude (m a.s.l.)	2535 (±134.6)	2677 (±58.2)	**0.045**
Percent bank erosion	52.5 (±20.9)	19.6 (±6.8)	**0.006**
Bank angle (degrees)	139.6 (±13.2)	118.0 (±6.9)	**0.004**
Mean riparian % shade cover	13.7 (±8.7)	34.2 (±5.3)	**0.000**
Bankfull width : depth ratio	17.8 (±5.6)	9.0 (±2.1)	**0.007**
Percent fines and sand	63.2 (±13.8)	41.7 (±10.6)	**0.016**
Percent algae cover	38.1 (±19.3)	25.3 (±9.9)	0.361
Percent riffles	35.7 (±11.0)	42.7 (±7.2)	0.174
Bank vegetation % cover	2.3 (±1.5)	5.8 (±0.9)	**0.000**
Percent pebble and cobble	4.7 (±4.7)	14.3 (±10.1)	0.052
Aquatic vegetation % cover	17.9 (±13.4)	27.0 (±9.7)	0.133
Conductivity (μS cm^{-1})	186 (±57.3)	89.2 (±8.9)	**0.023**
Temperature (°C)	16.7 (±3.6)	12.7 (±2.0)	0.057
pH	7.5 (±0.3)	7.9 (±0.2)	0.092
Biotic metrics			
Total richness	35.1 (±5.3)	41.7 (±3.5)	**0.033**
EPT richness	9.1 (±2.3)	13.5 (±2.0)	**0.004**
Sensitive richness	6.3 (±2.3)	10.4 (±2.0)	**0.003**
Percent tolerant taxa	28.3 (±4.3)	24.8 (±2.3)	0.105
Percent >5 mm	3.7 (±1.5)	3.3 (±0.9)	0.919
Percent dominance	53.7 (±5.6)	52.7 (±4.2)	0.612
Total density (ind m^{-2})	25 296 (±15 911)	27 518 (±11 672)	0.792

EPT, Ephemeroptera, Plecoptera and Trichoptera.
Mean values with 95% confidence intervals are shown along with Wilcoxon Rank–Sum test P-values. Significant bold values represents P > 0.05.

Bank erosion from livestock trampling was correlated with habitat features also known to be impacted by grazing. Shallow bank angles showing dished-out margins approaching 180° flattening increased with bank erosion (r = 0.26, P = 0.001), as did lower streamside vegetation cover (r = 0.73, P < 0.001) and higher bankfull W : D ratios, indicating broad shallow channel profiles (r = 0.58, P < 0.001). An example of this type of habitat degradation is shown in Fig. 2.

Responses in several of the invertebrate metrics were also related to the extent of bank erosion. Total, EPT and sensitive taxon richness decreased and tolerant taxa dominance increased with bank erosion (Figs 3 & 4). Multiple linear regression supported the conjecture that metric changes were driven by the deterioration of habitat quality because of grazing pressure (Table 5). Riparian cover and the amount of pebble and cobble substratum contributed to increased richness, while bank erosion and flattened bank angles were associated with decreased diversity; these models explained more than 50% of the variation in richness values. Per cent of tolerant taxa was best explained by increases in the prevalence of wide and shallow channel profiles, but overall less variation was explained compared with the diversity models.

Composition of invertebrate assemblages was correlated with many of the environmental factors used to describe disturbance. NMDS ordination of all invertebrate samples collected from the GTW and adjacent areas in 2004 resulted in a two-dimensional ordination (stress = 17.9%, $P = 0.004$; Fig. 5), which explained 83% of the variation in the original community space (axis 1, $R^2 = 0.58$, axis 2 $R^2 = 0.25$). Axis 1 was strongly correlated with a number of interrelated environmental variables associated with grazing disturbance, including increased bankfull stream width and W : D, bank angles, sandy substratum and decreased bank vegetation and riparian cover (vectors with $R^2 > 0.4$ shown in Fig. 5). Several midge taxa (*Cricotopus*, *Cladotanytarsus* and *Micropsectra*), siltation-tolerant *Tricorythodes* mayflies and burrowing *Ophiogomphus* dragonflies were among the taxa positively associated with axis 1 (the disturbance gradient), while two riffle beetles (*Cleptelmis* and *Optioservus*), *Ceratopsyche* caddisflies and *Baetis* mayflies were among the taxa negatively associated with axis 1. Axis 2 was not strongly associated with any abiotic variables, and explained only 25% of the variation in community structure among sites, but several taxa were correlated with axis 2: *Heleniella* (Chironomidae), *Dixa* (Dixidae), *Baetis* and *Rhyacophila* (Rhyacophilidae) were positively associated with axis 2, while *Micropsectra*, *Acentrella* (Baetidae), *Suwallia* (Chloroperlidae) and *Epeorus* (Heptageniidae) were negatively associated with axis 2. Examination of the NMDS ordination plot, however, revealed no distinct community types associated with either grazed or ungrazed sites, and MRPP analyses confirmed that discrimination between grazed and ungrazed communities was very weak ($A = 0.027$, $P = 0.009$). However, a group of heavily grazed sites in Monache meadows did form a tightly grouped cluster. At these sites, grazing pressure and stream conductivity were high, stream channels were wide and banks were often unstable with low bank angles (approaching 180°). At low values of axis 1 and middle

Table 3 Contrasts of physical habitat factors and aquatic macroinvertebrate biotic metrics between sites inside and outside of cattle grazing exclosures in and near the Golden Trout Wilderness. Table shows the group means (with 95% confidence intervals) for inside and outside, but tests were based on the paired difference Wilcoxon Signed-Rank test *P*-values (for $n = 8$ in-out contrasts)

	Inside exclosure	Outside exclosure	*P*-value
Physical habitat			
Percent bank erosion	31.0 (±15.4)	60.0 (±28.8)	0.080
Bank angle (degrees)	142.0 (±11.6)	142.4 (±14.6)	0.889
Mean riparian % shade cover	22.5 (±14.7)	9.8 (±10.6)	**0.014**
Bankfull W : D ratio	17.0 (±7.2)	14.1 (±4.5)	0.294
Percent fines and sand	50.8 (±22.2)	53.3 (±10.5)	0.834
Percent algae cover	43.3 (±19.8)	33.3 (±27.4)	0.363
Percent riffles	37.6 (±14.7)	30.2 (±9.5)	0.624
Bank vegetation % cover	3.8 (±2.5)	1.7 (±1.8)	**0.014**
Percent pebble and cobble	14.1 (±16.5)	4.2 (±3.4)	0.205
Aquatic vegetation % cover	25.3 (±21.4)	26.5 (±13.8)	0.735
Conductivity (μS cm^{-1})	158.1 (±67.4)	165.6 (±76.8)	0.624
Temperature (°C)	18.1 (±4.9)	16.3 (±4.8)	0.726
Biotic metrics			
Total diversity	36.6 (±5.3)	31.5 (±4.2)	0.059
EPT diversity	9.8 (±3.2)	7.8 (±2.5)	0.325
Sensitive diversity	6.6 (±2.1)	5.3 (±2.4)	0.325
Percent tolerant	26.2 (±6.3)	31.4 (±5.8)	0.080
Percent >5 mm	3.4 (±2.0)	2.6 (±1.4)	0.080
Percent dominance	57.5 (±9.3)	62.7 (±7.2)	0.234

EPI, Ephemeroptera, Plecoptera and Trichoptera.

Table 4 Contrasts of aquatic macroinvertebrate biotic metrics between samples taken before ($n = 8$) and after ($n = 8$) a grazing rest initiated in 2000 in the Golden Trout Wilderness area

	Before grazing rest (1999–2000)	After grazing rest (2004)	*P*-value
Total abundance (ind m^{-2})	26 222 (±23 355)	33 274 (±20 461)	0.183
Total taxa richness	21.2 (±4.2)	33.8 (±1.3)	**0.014**
EPT richness	11.0 (±3.7)	17.5 (±0.9)	**0.025**
EPT abundance (ind m^{-2})	4248 (±2548)	7750 (±3511)	0.183
Percent EPT abundance	24.6 (±10.3)	28.5 (±14.9)	0.529

Mean before and after grazing rest values with 95% confidence intervals are shown along with paired difference Wilcoxon Signed-Rank test *P*-values.

values of axis 2, a second group was composed of ungrazed sites from the Whitney and Templeton allotments, and five sites from the grazed Mulkey allotment (two of which were in ungrazed exclosures however, and two others which were subjected to low levels of grazing

Fig. 2 Grazed Mulkey Creek (above, Mul6) and ungrazed South Fork Kern River (below, Tem14), third-order stream examples of study reaches in the Golden Trout Wilderness.

pressure). These sites had narrow, deep to moderately deep stream channels with overhanging banks and moderate to high cover of both bank vegetation and in-stream macrophytes.

Non-metric multi-dimensional scaling ordination and MRPP showed no consistent differences in community composition between paired sites inside and outside of grazing exclosures in five areas: Bullfrog, Mulkey and Templeton Meadows, and two areas within Monache Meadows (MRPP: $A = -0.02$, $P = 0.642$). Communities were very similar inside and outside of exclosures at Mulkey (Mul4, 5 and 6), Bullfrog (Mul1 and 2) and Monache (Mon4, 5 and 6), and moderately different inside and outside the exclosure at Templeton (Tem12 and 13). One sample taken inside an exclosure (Mon7) was similar to the sample from outside the exclosure (Mon8), while a second sample taken inside an adjacent exclosure (Mon1) was dramatically different from the outside exclosure sample.

Discussion

Our results show (i) dependence of stream biological integrity on channel habitat conditions that are damaged by livestock, (ii) strong evidence of degraded stream habitat and invertebrate indicators on reaches in active livestock grazing allotments and (iii) trends of biological recovery from grazing effects after 4 years at the allotment scale, but not at the scale of local exclosures existing for 10 or more years. Using multiple approaches to measure grazing effects, we found that stream invertebrate richness decreased with increased level of grazing impact. Consistent with other studies of livestock effects on stream and riparian habitat (see Introduction), we found that grazed streams had more eroded banks, carried more sediment, had wider bankfull cross-sections, more algal growth and higher temperatures, and that all these stressors were correlated with decreases in invertebrate diversity and increases in stress-tolerant taxa. Our study also showed that streams of the GTW are resilient systems, with the ability to recover relatively quickly when livestock impacts are removed.

Grazing significantly altered bank and riparian habitat features, with greater bank erosion, low bank angles, less riparian cover and flattened channel profiles more prevalent at overgrazed sites. Although channel widening is a typical response to livestock trampling, floods can cause incision and head-cuts where stream beds and banks have been destabilised by grazing (Germanoski & Miller, 2004). Such channels are in a state of disequilibrium and as form evolves, a new W : D bankfull profile related to channel gradient becomes re-established within the down-cut terrace (Simon & Downs, 1995). Our study showed that more sediments, warmer temperatures, higher dissolved mineral content and algal growth were often associated with grazing. Although grazed and ungrazed site groups differed slightly in slope and altitude, these channels all conform to the same riffle-pool geomorphic form and are unlikely to be affected by these small differences.

Taxon richness and habitat quality revealed that conditions on Whitney and Templeton allotments were significantly improved 4 years after the elimination of grazing. The rapid improvement in conditions suggests that aquatic habitats may recover quickly if large portions of catchments are cleared of livestock influences. On the other hand, our results also showed that local fenced exclosures are largely ineffective at improving aquatic habitat conditions. In both respects, our results agree with earlier studies that compared effects of local, focused restoration efforts versus more comprehensive catchment-

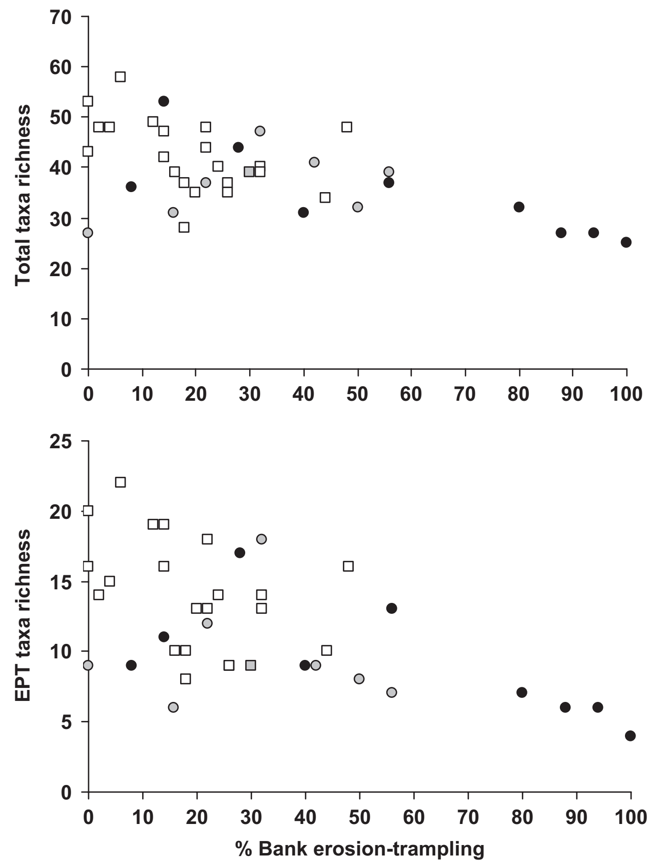

Fig. 3 Total richness (above) and Ephemeroptera, Plecoptera and Trichoptera richness (below) in relation to percent bank erosion. Streams on allotments that were not grazed 2000–2004 shown as squares and on grazed allotments as circles. Black circles are open to grazing and symbols with grey fill show a stream reach within a fenced exclosure (see Table 1).

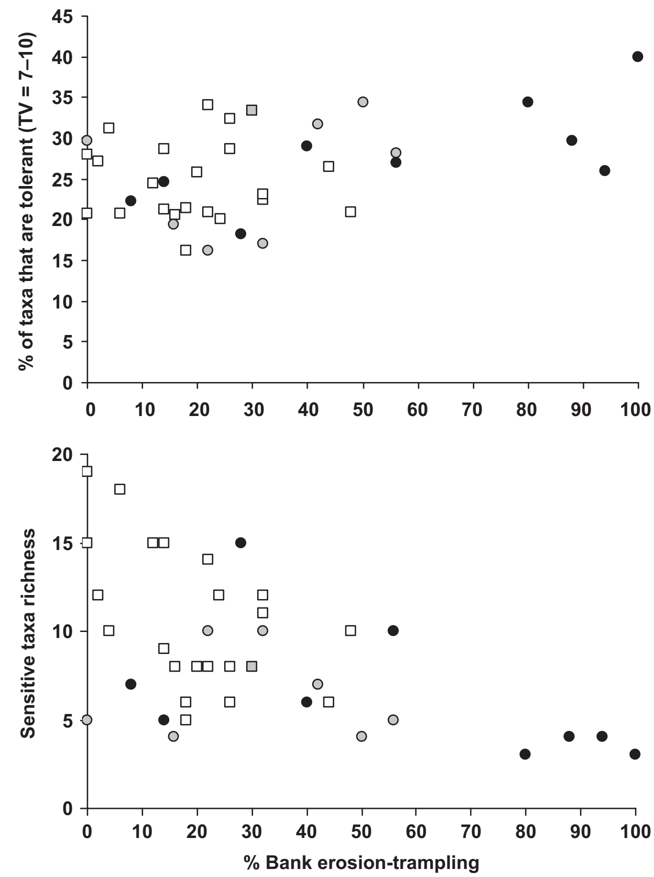

Fig. 4 Percent of taxa considered tolerant of pollution (above) and richness of taxa considered sensitive (below) in relation to percent bank erosion. Streams on allotments that were not grazed 2000–2004 shown as squares and on grazed allotments as circles. Black circles are open to grazing and symbols with grey fill show a stream reach within a fenced exclosure (see Table 1).

wide efforts on aquatic biological integrity (e.g. Roth, Allan & Erickson, 1996; Ranganath *et al*., 2009). Upstream of exclosures and along the adjacent uplands, livestock effects still persist and are propagated into the exclosures by water flow. This is compounded by the failure of fence lines to prevent cattle entry (observed in the field during this study).

Improvements in condition of stream habitat and aquatic life are likely to continue under management that minimises livestock damage to these montane meadow systems. While grazed streams in our study area experience channel and riparian degradation, aquatic ecosystems in the GTW appear to be relatively resilient, and, assuming livestock damage is not extreme, may be restored through passive processes when grazing pressure is reduced or discontinued. With no or low grazing activity, channels may rebuild and streams may experience ecological recovery. The rapid improvement in invertebrate diversity at most sites in the Whitney–Templeton allotments conforms to the ‘rubber-band’ concept of response to grazing exclusion (Sarr, 2002). Bank stability and riparian vegetation of sinuous low-gradient streams have also been found to improve rapidly in arid Great Basin streams of northern Nevada where livestock were completely removed or present only under periodic rest rotation (Myers & Swanson, 1996a,b). Under reduced grazing pressure, channel W : D has been shown to become reduced as riparian conditions recover and the stream profile narrows (Clary, 1999).

Sites we sampled along the South Fork Kern River in the Monache allotment exhibited the most pronounced effects of livestock use, both in terms of habitat conditions (Table 1) and the invertebrate assemblages (Fig. 5). Stream banks in incised, dry meadows of the Monache allotment were found to be about 10 times more susceptible to erosion than streams in meadows that retained their wet meadow characteristics (Micheli & Kirchner, 2002). Conversion of wet meadow habitats to dry

Table 5 Multiple regression results for richness and tolerance community metrics. Stepwise forward regression was used to select variables, followed by robust regression (Huber's method). All habitat variables in Table 1 used for selection and also included per cent riffle, per cent bank vegetation, water temperature and median particle size (D50)

Model		Variable	Coefficient	SE	β	P
Total richness						
Adj. R^2	0.515	Intercept	29.8	1.86		
F	20.6	Riparian cover	0.29	0.06	0.55	<0.0001
P	<0.0001	% Pebble-cobble	0.16	0.05	0.36	0.004
EPT richness						
Adj. R^2	0.512	Intercept	12.9	0.94		
F	20.4	% Bank erosion	−0.08	0.02	−0.47	0.0004
P	<0.0001	% Pebble-cobble	0.11	0.03	0.46	0.0005
Sensitive taxa richness						
Adj. R^2	0.519	Intercept	18.7	3.49		
F	14.3	Bank angle (°)	−0.08	0.03	−0.35	0.013
P	<0.0001	% Bank erosion	−0.05	0.02	−0.28	0.044
		% Pebble-cobble	0.09	0.03	0.37	0.004
% Tolerant taxa						
Adj. R^2	0.185	Intercept	21.3	1.63		
F	9.4	Bankfull W : D	0.35	0.12	0.45	0.004
P	0.004					

EPT, Ephemeroptera, Plecoptera and Trichoptera.
Based on all stream reaches surveyed (n = 38).

meadows through head-cutting, channel incision and water table retreat contributes to reduced bank stability, and may represent a state change in that this condition is coupled to wide-shallow stream channels and losses of riparian cover where it is difficult to re-establish stable fluvial function (Simon, 1989). We further observed a progressive decline in discharge from upper to lower portions of Monache meadows, suggesting this is a hydrologically losing reach where channel flows are drawn-off by a lowered water table. Stream conditions in the Monache allotment, where dry meadow erosion, downstream sediment accumulation and loss of flows are occurring, may be more difficult to reverse and restore to a stable/resilient state.

Although protective riparian exclosures have proven to be effective in the recovery of streamside vegetation (e.g. Odion, Dudley & D'Antonio, 1988), channel geomorphology and macroinvertebrates may remain in a degraded state or show no differences inside and outside of exclosures (Kondolf, 1993; Allen-Diaz, Jackson & Fehmi, 1998; Ranganath *et al.*, 2009). Other studies of livestock exclosures in riparian areas have shown that removal from grazing can improve stream physical conditions for the production of stream fisheries (Marcuson, 1977; Keller & Burnham, 1982; Claire & Storch, 1983; Stuber, 1985; Platts & Nelson, 1989). Density and biomass of CGT per unit-area in the GTW were found to be significantly higher in ungrazed exclosures versus neighbouring grazed reaches (Knapp & Matthews, 1996), even though the wider and shallower grazed reaches provided more spawning habitat relative to narrow, deep channels within the exclosures (Knapp, Vredenburg & Matthews, 1998). Although grazed reaches in the GTW may provide more spawning habitat than ungrazed reaches, the growth of individual fish in the dense juvenile populations in grazed areas is reduced and adult fish are small (Knapp *et al.*, 1998). Our studies showed that reaches with eroded banks and high W : D ratios have less variety of potential invertebrate food resource, which may further limit fish growth rates.

Livestock grazing intensity and associated fine sediment delivery have been linked to losses of aquatic diversity and sensitive invertebrate taxa (Braccia & Voshell, 2006, 2007). Our results support the use of invertebrates as indicators of ecological distress and recovery from livestock grazing. Sediments in streams have a pervasive effect on aquatic biological communities (Newcombe & MacDonald, 1991; Waters, 1995), and grazing often results in increased erosion and sedimentation in western intermountain streams (Chambers & Miller, 2004). Studies of sediment effects in natural and laboratory streams have shown that suspended and deposited sediments can impair and reduce populations of sensitive invertebrates (McClelland & Brusven, 1980; Lemly, 1982; Zweig & Rabeni, 2001). The scouring action of suspended sediments can also cause loss of invertebrates through drift (Culp, Wrona & Davies, 1986). Sediment deposition-driven changes in habitat have been documented to decrease invertebrate densities, and to induce shifts in taxonomic composition to sediment-tolerant taxa such as chironomids (Lenat, Penrose & Eagleson, 1979). Reduced cover of riparian trees in the presence of livestock grazing has also been shown to result in reduced total and EPT diversity (Rios & Bailey, 2006), but rapid recovery of an overgrazed Sierra stream has been linked to improved channel geomorphology and riparian cover (Herbst & Kane, 2009). Our findings agree with those from previous studies, suggesting that erosion- and sediment-related impacts of livestock use have strong negative influences on aquatic biological integrity.

Management decisions are difficult to make without guidance regarding which restoration practices work and where they work. Our results suggest that rapid recovery of stream habitats may accompany complete exclusion of livestock at the allotment scale (assuming that ecosystem degradation is not extreme), but riparian exclosures are ineffective, except perhaps in promoting denser riparian

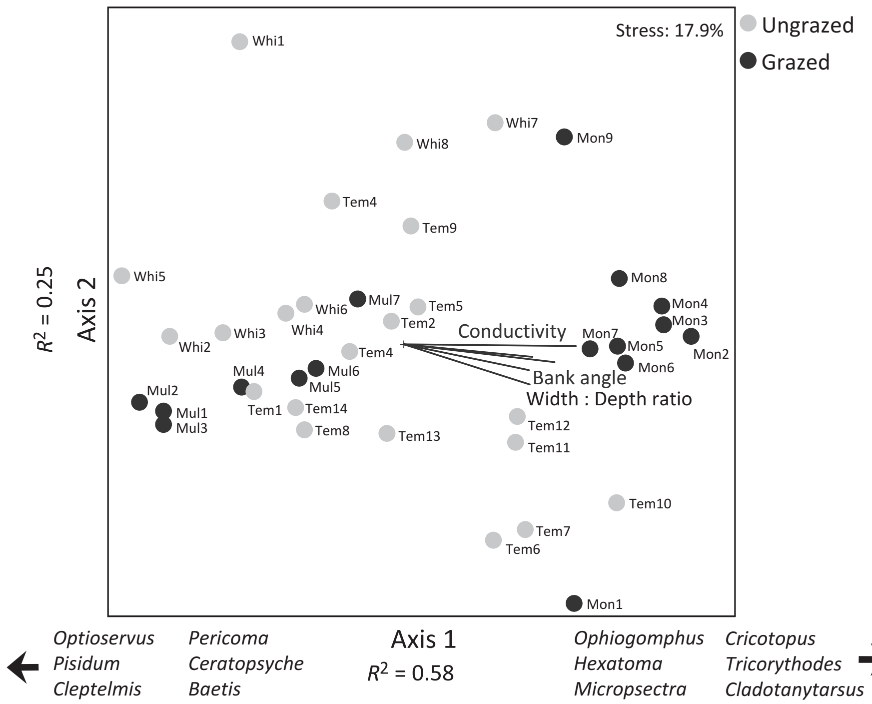

Fig. 5 Non-metric multi-dimensional scaling (NMDS) ordination of aquatic invertebrate community samples collected in 2004 from the Golden Trout Wilderness area. Grazed sites (black circles) were actively grazed from 2000 to 2004, while ungrazed sites (grey circles) were not grazed from 2000 to 2004. Vectors indicate environmental factors strongly correlated with NMDS axes ($R^2 > 0.4$). (*Note:* bankfull width is not labeled due to space limitations). Percentage of total community variation explained by each axis (R^2) are given below each axis labels. Twelve influential taxa that were positively (listed on right) or negatively (listed on left) associated with axis 1 (Pearson's $r > 0.4$) are listed below axis 1. These taxa comprise 49.9% of the total individual counts over all sites. No taxa were strongly influential on axis 2.

vegetation cover. Rest rotation of meadow grazing may provide the same potential system recovery depending on the interval of rest, but this may be contingent on the capacity for local recovery and whether the state of the channel has transitioned into an unstable geomorphology. Coupling the monitoring of aquatic biological communities to the status of channel morphology provides important insights into which management strategies are effective, and permits gauging of ecological responses and habitat suitability. The protocol we used in the GTW should continue to be useful in evaluating grazing management decisions in the GTW and elsewhere. Given the existence of baseline data, repeated monitoring of our sites would be especially useful in helping to determine the status of in-stream conditions if livestock are ultimately removed from the Mulkey and Monache allotments, or if they are returned to the Whitney and Templeton allotments.

Acknowledgments

Funding for this project was provided by the US Forest Service, the California Department of Fish and Game, and California Trout. Our appreciation for clarifying discussions, information and insights from Lisa Sims, Heather Schwartz, and Del Hubbs of the Inyo National Forest. The manuscript has been strengthened by anonymous peer reviews and the editing skills of Richard Johnson. Administrative assistance was provided by Dave Lentz of California Department of Fish and Game. Thanks to Trip Armstrong for field work and habitat data compilation, and to Scott Roberts for assisting with GIS and map preparation.

References

Allen-Diaz B., Jackson R.D. & Fehmi J.S. (1998) Detecting channel morphology change in California's hard-wood rangeland spring ecosystems. *Journal of Range Management*, **51**, 514–518.

Armour C.L., Duff D.A. & Elmore W. (1994) The effects of livestock grazing on western riparian and stream ecosystems. *Fisheries*, **19**, 9–12.

Bauer S.B. & Burton T.A. (1993) *Monitoring Protocols to Evaluate Water Quality Effects of Grazing Management on Western Rangeland Streams*. EPA 910R-93-017, Environmental Protection Agency Office of Water, Washington, DC.

Belsky A.J., Matzke A. & Uselman S. (1999) Survey of livestock influences on stream and riparian ecosystems in the western United States. *Journal of Soil and Water Conservation*, **54**, 419–431.

Braccia A. & Voshell J.R. (2006) Environmental factors accounting for benthic macroinvertebrate assemblage structure at the sample scale in streams subjected to a gradient of cattle grazing. *Hydrobiologia*, **573**, 55–73.

Braccia A. & Voshell J.R. (2007) Benthic macroinvertebrate responses to increasing levels of cattle grazing in Blue Ridge Mountain streams, Virginia, USA. *Environmental Monitoring and Assessment*, **131**, 185–200.

Chambers J.C. & Miller J.R. (2004) *Great Basin Riparian Ecosystems: Ecology, Management, and Restoration*. Island Press, Washington, DC.

Claire E.W. & Storch R.L. (1983) Streamside management and livestock grazing in the Blue Mountains of Oregon: a case study. In: *Proceedings of the Workshop on Livestock and Wildlife-Fisheries Relationships in the Great Basin* (Ed. J.W. Menke), pp. 111–128. University of California Agriculture Station, Science Special Publication 3301, Berkeley, CA.

Clary W.P. (1999) Stream and channel vegetation responses to late spring cattle grazing. *Journal of Range Management*, **52**, 218–227.

Culp J.M., Wrona F.J. & Davies R.W. (1986) Response of stream benthos and drift to fine sediment deposition versus transport. *Canadian Journal of Zoology*, **64**, 1345–1351.

Dull R.A. (1999) Palynological evidence for 19th century grazing-induced vegetation change in the Southern Sierra Nevada, California, U.S.A. *Journal of Biogeography*, **26**, 899–912.

Fleischner T.L. (1994) Ecological costs of livestock grazing in western North America. *Conservation Biology*, **8**, 629–644.

Germanoski D. & Miller J.R. (2004) Basin sensitivity to channel incision in response to natural and anthropogenic disturbance. In: *Great Basin Riparian Ecosystems – Ecology, Management and Restoration* (Eds J.C. Chambers & J.R. Miller), pp. 88–123. Island Press, Washington, DC.

Herbst D.B. & Kane J.M. (2009) Responses of aquatic macroinvertebrates to stream channel reconstruction in a degraded rangeland creek in the Sierra Nevada. *Ecological Restoration*, **27**, 76–88.

Kauffman J.B. & Krueger W.C. (1984) Livestock impacts on riparian ecosystems and streamside management considerations: a review. *Journal of Range Management*, **37**, 430–438.

Keller C.R. & Burnham K.P. (1982) Riparian fencing, grazing, and trout habitat preference on Summit Creek, Idaho. *North American Journal of Fisheries Management*, **2**, 53–59.

Knapp R.A. & Matthews K.R. (1996) Livestock grazing, golden trout, and streams in the Golden Trout Wilderness, California: impacts and management implications. *North American Journal of Fisheries Management*, **16**, 805–820.

Knapp R.A., Vredenburg V.T. & Matthews K.R. (1998) Effects of stream channel morphology on golden trout spawning habitat and recruitment. *Ecological Applications*, **8**, 1104–1117.

Kondolf G.M. (1993) Lag in stream channel adjustment to livestock exclosure, White Mountains, California. *Restoration Ecology*, **4**, 226–230.

Laycock W.A. (1991) Stable states and thresholds of range condition on North American rangelands: a viewpoint. *Journal of Range Management*, **44**, 427–433.

Lemly A.D. (1982) Modification of benthic insect communities in polluted streams – combined effects of sedimentation and nutrient enrichment. *Hydrobiologia*, **87**, 229–245.

Lenat D.R., Penrose D.L. & Eagleson K.W. (1979) *Biological Evaluation of Non-Point Source Pollutants in North Carolina Streams*. North Carolina Department of Natural Resources and Community Development, Biological Series 102, Raleigh, NC.

Li H., Lamberti G.A., Pearsons T.N., Tait C.K., Li J.L. & Buckhouse J.C. (1994) Cumulative effects of riparian disturbances along high desert trout streams of the John Day Basin, Oregon. *Transactions of the American Fisheries Society*, **123**, 627–640.

MacDonald L.H., Smart A.W. & Wissmar R.C. (1991) *Monitoring Guidelines to Evaluate Effects of Forestry Activities on Streams in the Pacific Northwest and Alaska*. Region 10, U.S. Environmental Protection Agency, EPA/910/9-91-001, Washington, DC.

Marcuson P.E. (1977) *Overgrazed Streambanks Depress Fishery Production in Rock Creek, Montana*. Montana Department of Fish and Game, Fisheries Division, Special Report Project No. F-20-R-21, II-a, Helena, MT.

McClelland W.T. & Brusven M.A. (1980) Effects of sedimentation on the behavior and distribution of riffle insects in a laboratory stream. *Aquatic Insects*, **2**, 161–169.

McCune B. & Grace J.B. (2002) *Analysis of Ecological Communities*. MjM Software Design, Gleneden Beach, OR.

Micheli E.R. & Kirchner J.W. (2002) Effects of wet meadow riparian vegetation on streambank erosion: 1. Remote sensing measurements of streambank migration and erodibility. *Earth Surface Processes and Landforms*, **27**, 627–639.

Miles S.R. & Goudey C.B. (1997) *Ecological Subregions of California*. USDA Forest Service, Pacific Southwest Region, San Francisco, CA.

Minshall G.W., Jensen S.E. & Platts W.S. (1989) *The Ecology of Stream and Riparian Habitats of the Great Basin Region: A Community Profile*. U.S. Fish and Wildlife Service Biological Rep. 85(7.24), Washington, DC.

Moyle P.B. (2002) *Inland Fishes of California*. University of California Press, Berkeley, CA.

Myers T.J. & Swanson S. (1995) Impact of deferred rotation grazing on stream characteristics in Central Nevada: a case study. *North American Journal of Fisheries Management*, **15**, 428–439.

Myers T.J. & Swanson S. (1996a) Long-term aquatic habitat restoration: Mahogany Creek, Nevada, as a case study. *Water Resources Bulletin*, **32**, 241–252.

Myers T.J. & Swanson S. (1996b) Temporal and geomorphic variations of stream stability and morphology: Mahogany Creek, Nevada. *Water Resources Bulletin*, **32**, 253–265.

National Geographic. (2001) *TOPO! Version 2.6.8*. National Geographic Maps, San Francisco, CA.

Newcombe C.P. & MacDonald D.D. (1991) Effects of suspended sediments on aquatic ecosystems. *North American Journal of Fisheries Management*, **11**, 72–82.

Odion D.C., Dudley T.L. & D'Antonio C.M. (1988) Cattle grazing in southeastern Sierran meadows: ecosystem change and prospects for recovery. In: *White Mountain Symposium Volume 3: Plant Biology of Eastern California* (Eds C.A.J. Hall & V. Doyle-Jones), pp. 277–292. University of California, Los Angeles, CA.

Platts W.S. (1991) Livestock grazing. In: *Influences of Forest and Rangeland Management on Salmonid Fishes and Their Habitats* (Ed. W.R. Meehan), pp. 389–423. American Fisheries Society Special Publication 19, Bethesda, MD.

Platts W.S. & Nelson R.L. (1989) Stream canopy and its relationship to salmonid biomass in the intermountain west. *North American Journal of Fisheries Management*, **9**, 446–457.

Ranganath S.C., Hession W.C. & Wynn T.M. (2009) Livestock exclusion influences on riparian vegetation, channel morphology, and benthic macroinvertebrate assemblages. *Journal of Soil and Water Conservation*, **64**, 33–42.

Rangel T.F., Diniz-Filho J.A.F. & Bini L.M. (2010) SAM: a comprehensive application for Spatial Analysis in Macroecology. *Ecography*, **33**, 1–5.

Rios S.L. & Bailey R.C. (2006) Relationship between riparian vegetation and stream benthic communities at three spatial scales. *Hydrobiologia*, **553**, 153–160.

Roth N.E., Allan J.D. & Erickson D.L. (1996) Landscape influences on stream biotic integrity assessed at multiple spatial scales. *Landscape Ecology*, **11**, 141–156.

SAFIT (Southwestern Association of Freshwater Invertebrate Taxonomists). (2010) List of tolerance values and functional feeding groups – Available at: http://safit.org/TVFFG.html.

Sarr D.A. (2002) Riparian livestock exclosure research in the western United States: a critique and some recommendations. *Environmental Management*, **30**, 516–526.

Scrimgeour G.J. & Kendall S. (2003) Effects of livestock grazing on benthic invertebrates from a native grassland ecosystem. *Freshwater Biology*, **48**, 347–362.

Simon A. (1989) A model of channel response in disturbed alluvial channels. *Earth Surface Processes and Landforms*, **14**, 11–26.

Simon A. & Downs P.W. (1995) An interdisciplinary approach to evaluation of potential instability in alluvial channels. *Geomorphology*, **12**, 215–232.

Strand M. & Merritt R.W. (1999) Impacts of livestock grazing activities on stream insect communities and the riverine environment. *American Entomologist*, **45**, 13–29.

Stuber R.J. (1985) Trout habitat, abundance, and fishing opportunities in fenced vs. unfenced riparian habitat along Sheep Creek, Colorado. In: *Riparian Ecosystems and Their Management: Reconciling Conflicting Uses* (Eds. R.R. Johnson, C.D. Ziebell & D.R. Patton), pp. 310–314. First North American Riparian Conference, USDA Forest Service General Tech. Rep. RM-120, Washington, DC.

Trimble S.W. & Mendel A.C. (1995) The cow as a geomorphic agent, a critical review. *Geomorphology*, **13**, 233–253.

Waters T.F. (1995) Sediment in streams: sources, biological effects and control. *American Fisheries Society Monograph*, **7**, 1–251.

Zweig L.D. & Rabeni C.F. (2001) Biomonitoring for deposited sediment using benthic invertebrates: a test on 4 Missouri streams. *Journal of the North American Benthological Society*, **20**, 643–657.

(Manuscript accepted 29 September 2011)