Cattle Grazing Effects on Macroinvertebrates in an Oregon Mountain Stream

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Abstract

Cattle grazing effects on aquatic macroinvertebrates were assessed in a 4-year experiment of a mountain stream in northeastern Oregon. From 1996 through 1999, 10 cow–calf pairs were introduced into 6 experimental units along the stream for 42 days between July and September, and effects on aquatic macroinvertebrates were compared with 3 units in which no grazing occurred. Streambank and geomorphological variables were also measured to provide context for interpretation of effects on aquatic macroinvertebrates. Macroinvertebrate response to grazing was subtle, indicated by significantly lower abundance in grazed units. We measured more profound effects on streambanks: grazing caused an average decrease of 18% in bank length of the highest stability/cover class and caused an average increase of 8% in the lowest condition class over the course of each summer. By June of each following year, banks had recovered to their previous June condition, but grazing each summer caused a progressively larger decline in bank condition by September. Streambank effects were accompanied by an increase in cobble embeddedness over time in grazed units and were correlated with grazing-associated stream widening. Treatment effects were overwhelmed, however, by a profound decline in the abundance of most macroinvertebrates over the study period, with a drop in September 1999 to 14% of the initial September abundance of 1997. While the drop was more precipitous in grazed units, declines were common to all study units, suggesting that something more widespread affected the system during this time. Logging on lands just upstream of the study area in 1998 and 1999, in which trucks drove through the study stream without the benefit of a culvert, sent sediment plumes into the study area each of those 2 years and could have caused the precipitous decline in aquatic macroinvertebrates.

Resumen

En un experimento de cuatro años, realizado en un río en una montaña del nordeste de Oregon, se evaluaron los efectos del apacentamiento de los bovinos sobre los macroinvertebrados acuáticos. De 1996 a 1999, 10 pares de vaca-becerro se introdujeron en seis unidades experimentales a lo largo del río, por un periodo de 42 días entre julio y septiembre, y los efectos sobre los macroinvertebrados acuáticos se compararon contra tres unidades experimentales en las que no hubo apacentamiento. También se midieron variables geomorfológicas y del talud del río, para proveer un contexto para la interpretación de los efectos sobre los macroinvertebrados acuáticos. La respuesta de los macroinvertebrados al apacentamiento fue muy sutil, indicada por la abundancia significativamente baja de las unidades apacentadas. Medimos efectos más profundos en el talud: el apacentamiento causó una disminución promedio de 18% en la longitud del talud de la clase de mayor estabilidad/cobertura, y ocasionó un aumento promedio del 8% en la clase de menor condición en el curso de cada verano. En junio del siguiente año, los taludes habían recuperado la condición que tenían en junio del año anterior, pero el apacentamiento de cada verano causó una disminución progresivamente mayor en la condición del talud registrada en septiembre. Los efectos del talud estuvieron acompañados por un aumento a través del tiempo del empedrado con guijarros en las unidades apacentadas y estuvieron correlacionadas con ensanchamiento de la corriente por efecto asociado del apacentamiento. Sin embargo, los efectos de los tratamientos fueron confundidos por una profunda disminución de la abundancia de macroinvertebrados a través del estudio, con una caída en septiembre de 1999 a 14% de la abundancia inicial registrada en septiembre de 1997. Mientras que la caída fue más precipitada en las unidades apacentadas, las disminuciones fueron comunes en todas las unidades del estudio, sugiriendo que algo más general afectó el sistema durante este tiempo. La tala de las tierras, justo aguas arriba del área de estudio, en 1998 y 1999, cuando los camiones circularon sobre la corriente de agua en estudio, sin el beneficio de una alcantarilla, envió los sedimentos al área de estudio durante esos dos años, y pudo haber causado la disminución precipitada de los macroinvertebrados acuáticos.

INTRODUCTION

In the arid inland west, livestock management issues often pivot around the use of riparian areas (Platts 1991), where water and shade attract livestock in the hottest summer months (Stuth 1991). Working in a mountain riparian zone, Roath and Krueger (1982) estimated that 81% of the vegetation used by livestock came from a streamside meadow representing only 2% of the grazing allotment area. Several other studies have shown that both wild ungulates and livestock use riparian areas disproportionately more than adjacent upland areas (Kauffman and Krueger 1984; Marlow and Pogacnik 1986). Moreover, it has been shown that heavy livestock grazing in riparian areas can reduce aquatic community integrity and water quality by removing vegetation (Leege et al. 1981), increasing bank instability through trampling (Moring et al. 1985; Platts 1986; Marlow et al. 1987), increasing rates of sedimentation (Duff 1979), and causing streams to widen and become more shallow (Platts 1986). These geomorphological changes can in turn decrease water quality and can negatively influence components of aquatic biodiversity, particularly macroinvertebrates and the salmonids that feed on them.

Although comparative studies tend to show that lighter grazing intensities typically result in lower riparian impacts (Hayes 1978; Stephenson and Street 1978; Skovlin 1984; Maloney et al. 1999), most studies have assessed grazing effects under relatively high stocking levels. For example, high grazing intensities have been shown to cause high levels of streambank breakdown (Dobson 1973), reduced infiltration through compaction (Bryant et al. 1972), reduced riparian revegetation (Glinski 1977), and declines in habitat quality for salmonid fish (Skovlin 1984). The impacts of heavy livestock grazing are exemplified by the consistent demonstration of dramatic ecosystem recovery following cessation of heavy grazing (Busse 1989; Beschta et al. 1991). By contrast, studies on grazing effects under more moderate intensities are less common (Hanson et al. 1970; Van Poolen and Lacey 1979; Blackburn 1984; Clary et al. 1996; Clary 1999; McInnis and McIver 2001), so we understand less about how cattle grazing affects riparian areas for the more commonly used modern grazing regimes. Moreover, many cattle grazing studies have focused on effects observed over fairly short time periods, usually a season or two, and so there is relatively little information on how grazing impacts observed during one season are mitigated over the course of the fall, winter, and spring (Buckhouse et al. 1981; Kauffman et al. 1983). Finally, because cattle have the potential to cause a variety of linked effects that include streambank breakdown (McInnis and McIver 2001) and changes in both channel morphology and aquatic biodiversity (Skovlin 1984), we need integrated, multivariate studies to capture the entire picture of impact.

In an effort to provide multivariate, longer-term information on moderate cattle grazing effects, we conducted a 4-year experiment to assess how moderate intensity grazing (0.77 ha · animal unit month^{-1} [AUM]) affected the aquatic macroinvertebrate community of a mountain stream in northeastern Oregon. Our study was conducted within the context of a broader study on economics and cattle behavior of alternative livestock management schemes in riparian areas (Dickard 1998; Stillings 1998; Parsons et al. 2003; Stillings et al. 2003). We measured streambank effects, as well as in-stream channel morphology variables in an attempt to identify mechanisms associated with observed changes in the macroinvertebrate community. It has been well documented that streambank cover and stability are 2 critical factors that influence erosion potential, which over the long term may lead to changes in water quality and channel morphology, with accompanying changes in water storage and wildlife habitat in riparian areas (Kauffman and Krueger 1984; Platts 1986; Bohn 1986; Elmore and Kauffman 1994; McInnis and McIver 2001). Because of their widely accepted utility as indicator organisms for water quality and as critical food for salmonids (Barbour et al. 1999), the aquatic macroinvertebrate assemblage was assessed relative to the cattle grazing treatment. Our objectives were to measure the influence of cattle grazing against ungrazed controls and to provide insight on potential thresholds in impact on streambanks in terms of recovery to preexisting levels and with respect to channel morphology and the abundance and diversity of macroinvertebrates.

STUDY SITE

The study was conducted between 1996 and 1999 on the Hall Ranch unit (lat 45°7′41″N, long. 117°42′45″W) of the Eastern Oregon Agricultural Research Center, approximately 19 km southeast of Union, Oregon. Over the past 20 years, mean annual precipitation at Hall Ranch was 660 mm, with approximately 60% occurring as snow. Elevation ranges from 1050 to 1250 m. Hall Ranch includes 2 distinct riparian zones, the larger on Catherine Creek, a tributary of the Grande Ronde River, and the smaller on Milk Creek, a tributary of Catherine Creek. The 101-ha study area included the entire Hall Ranch riparian zone of Milk Creek, a 2.4-km section beginning at a private boundary on the north and ending at Highway 203 a few hundred meters from its confluence with Catherine Creek (Fig. 1). The study area had been grazed lightly from mid-July to mid-August in the 5 years prior to the beginning of this study (1992–1996) at an average rate of 1.75 ha · AUM^{-1}.

MATERIALS AND METHODS

In May 1996, 9 experimental units of similar area were established along the 2.4-km reach of Milk Creek (Fig. 1). Treatments were distributed among blocks because of obvious differences in riparian habitat from the southern to northern section of Milk Creek: block 1 was forested with hawthorn (*Crataegus columbiana* Howell) and ponderosa pine (*Pinus ponderosa* Douglas), block 2 had components of both forest and meadow, and block 3 was primarily meadow, dominated by a variety of introduced grass species favored by livestock. These included rushes (*Juncus* spp.), sedges (*Carex* spp.), and a variety of grasses: Kentucky bluegrass (*Poa pratensis* L.), several species of brome (*Bromus* spp.), meadow foxtail (*Alopecurus pratensis* L.), and timothy (*Phleum pretense* L.). At the beginning of the experiment, the 3 experimental units within each block were randomly assigned to either the grazing treatment (2 units) or to the ungrazed control (1 unit), and
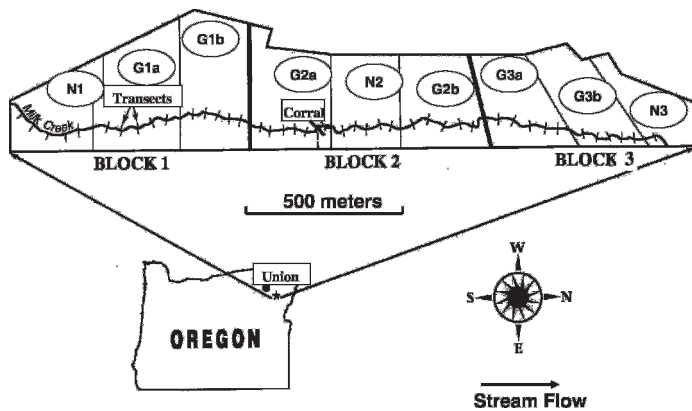
Figure 1. Map of Milk Creek study area, Hall Ranch, Eastern Oregon Agricultural Research Station (Oregon State University), northeastern Oregon. Ungrazed units are designated N1, N2, and N3; grazed units are designated G1a, G2a, G3a and G1b, G2b, and G3b.

these assignments were maintained by fencing throughout the 4-year study period.

Within each of the 6 grazed experimental units, 10 cow–calf pairs were introduced for 42 days between early July and early September for the years 1996 through 1999. This grazing intensity (0.77 ha · AUM^{-1}) was a little more than twice the grazing intensity of the previous 5 years. The length of grazing time and the stocking rate were chosen with the objective of achieving a moderate grazing intensity of approximately 50% utilization of grass within each experimental unit. Actual utilization over the 4-year study averaged about 40% for grazed pastures (Dickard 1998; Parsons et al. 2003).

Environmental assessment was undertaken during the second through fourth grazing seasons (1997–1999). Because the same treatment design was used for all 4 seasons, measuring environmental variables in the second through the fourth year allowed us to assess the cumulative effect of 4 consecutive years of grazing. Three classes of variables were measured: 1) streambank condition—2 variables, 2) channel morphology—3 variables, and 3) aquatic macroinvertebrates—61 variables. Variables were chosen to assess direct effects on the streambank and indirect effects that bank breakdown would be expected to cause on channel morphology, sediment delivery, and in the macroinvertebrate assemblage.

Streambank condition was measured before (mid-June) and after (late September) cattle grazing by pacing each side of Milk Creek and recording the appropriate streambank cover and stability class within plots defined lengthwise as a step (0.5 m in length) taken parallel to the stream. Plot width was defined as the vegetation located below the bank full level but above the scour line (Bauer and Burton 1993). Plots were first examined for the presence of hoof prints and then assessed for bank cover and stability. Streambank plots were considered covered if they contained any of the following features: perennial vegetation ground cover greater than 50%, roots of deeply rooted vegetation such as shrubs or sedges covering more than 50% of the bank, at least 50% of bank surface protected by rocks of cobble size or larger, or at least 50% of the bank surface protected by logs of 10-cm diameter or larger (Bauer and Burton 1993). Otherwise, banks were rated "uncovered." Banks were rated "unstable" if they exhibited

any of the following features: blocks of banks broken away and lying adjacent to the bank breakage ("breakdown"), bank sloughed into stream channel ("slump"), bank cracked and about to move into stream ("fracture"), or bank was uncovered as defined above and exhibited an angle visually estimated steeper than 80 degrees from the horizontal (Bauer and Burton 1993). Otherwise, banks were rated "stable." Each step of the observer was thus rated according to streambank cover and stability and grouped into 4 classes: covered/stable, covered/unstable, uncovered/stable, and uncovered/unstable. To test hypotheses about grazing impacts on streambank cover and stability, data were summarized by experimental unit for 3 replicates of the ungrazed control and 6 replicates of the 2 grazing treatments for each experiment. We tested grazing effects by analyzing changes in the "covered/stable" and "uncovered/unstable" condition classes.

Channel morphology and macroinvertebrates variables were measured along 42 permanent transects, each placed perpendicular to the stream along the entire length of the study area (Fig. 1). Seven of the 9 experimental units had 5 transects placed at uniform intervals within each unit. The ungrazed control of block 3 had sufficient stream length for only 1 transect, while one of the grazing units of block 2 (G3) had sufficient stream length for 6 equally spaced transects. Because streambank condition was also measured relative to the transect system, we had the opportunity to determine spatially specific linkages in effects among the 3 classes of variables.

Three channel morphology variables were measured: stream width/depth ratio, substrate type (silt, sand, gravel, cobble), and cobble embeddedness. Stream width/depth and substrate type were measured at all 42 transects (Fig. 1) in June (before grazing), August (during grazing), and September (after grazing). At each transect, stream width was measured as bank full, and depth at 5 equidistant points was averaged to obtain width-to-depth ratio. Substrate type (silt, sand, gravel, cobble) was assessed visually and given an index value (silt = 1, sand = 2, gravel = 3, cobble = 4) at 5 equidistant points across the width of the stream, and the index values of these 5 subsamples were averaged for a single estimate. Embeddedness was estimated for 30 cobbles within 2 m of transects 1, 3, and 5 for each experimental unit. Each cobble was examined for adhering vegetation, slime layer, and variation in color due to exposure or embeddedness, and the average percent of cobble embeddedness was estimated visually by assuming a spherical rock shape. The mean of the value for 30 rocks was calculated as a single subsample of embeddedness for each transect.

Aquatic macroinvertebrates were sampled in June (before grazing), August (during grazing), and September (after grazing), in 1997, 1998, and 1999. Kick-net samples were taken in the riffle nearest each of 26 transects (transects 1, 3, and 5 of each experimental unit) (Fig. 1). Each sample consisted of 2 "kicks" taken from the thalwag and from near the western edge of the stream that were combined for a total estimated sampling area of 0.187 m^2. A "kick" consisted of the observer placing a 20 cm-wide D-net in the selected spot and kicking the substrate upstream of the net for 15–20 seconds and then rubbing the substrate by hand for another 10–15 seconds to dislodge and capture any macroinvertebrates present. The 2-kick subsamples were then combined in the

field, preserved in 80% alcohol, and taken back to the laboratory. Samples were cleaned in the laboratory and sent to the laboratory of Aquatic Biology Associates (Robert Wisseman) in Corvallis, Oregon, where specimens were extracted, identified, and counted.

A total of 61 macroinvertebrate variables were chosen for analysis, including individual taxa, abundance indices, and diversity indices. First, 42 species or generic-level taxa that were found in at least 50% of the samples during the 4-year study period were analyzed with split-plot analysis of variance (ANOVA). Second, 17 "abundance" indices were also analyzed, including: total abundance; percent dominance measured as the abundance of the 3 most common taxa; abundance of Ephemeroptera (mayflies) + Plecoptera (stoneflies) + Trichoptera (caddisflies) (EPT abundance); percent abundance of noninsect taxa; percent abundance of oligochaetes (earthworm relatives), mayflies, stoneflies, caddisflies, beetles, flies, and midges; and percent abundance of the functional feeding guilds—collector-filterers, collector-gatherers, parasites, predators, shredders, and scrapers (functional feeding groups as defined by Merritt and Cummins 1996). Third, 12 "diversity" indices were analyzed, including: total taxa richness; richness of mayflies + stoneflies + caddisflies (EPT taxa); richness of noninsect taxa; richness of tolerant and intolerant taxa (the latter 2 after Robert Wisseman, Aquatic Biology Associates, personal communication, 2001); the species richness of mayflies, stoneflies, caddisflies, flies, midges, and beetles; and the Hilsenhoff tolerance index. Tolerance values for calculation of the "Hilsenhoff tolerance index" were taken from Hilsenhoff (1988). These abundance and diversity indices listed above were chosen because they are known to be diagnostic for aquatic impairment (Barbour et al. 1999) and because they represent a wide range of potential aquatic response.

Four statistical analyses were utilized: 1) ANOVAs on the 63 dependent variables separately, 2) spatially explicit correlation among dependent variables to identify potential linkages at the transect level, 3) multivariate analysis of macroinvertebrate abundance and community composition to determine how the whole assemblage responded to treatment, and 4) path analysis (a form of structural equation modeling; Kaplan 2000) to test our hypothesized concept on how the system as a whole would respond to treatment (McCune and Grace 2002). In general, key variables were normally distributed, and relationships among variables were linear; thus, no transformations were performed on the data.

ANOVA was conducted for the 4-year grazing experiment, using data collected in September (end of year) in the years 1997, 1998, and 1999 (1996 data were not collected) (SPSS 2005). This was a randomized complete block design, with 3 blocks, 2 grazing treatments, and repeated measures (1997, 1998, 1999). The grazing treatment was replicated within each block, so there were 9 experimental units, 3 in each of 3 blocks. Two experimental units within each block were grazed, and 1 was not.

After analysis of individual dependent variables in the ANOVA, we constructed a single data set in which the change in value from June to September of each year was calculated for each variable at each transect point. By aligning the observations based on transect location, we could then correlate observed effects for pairs of dependent variables. We were particularly interested in how streambank effects correlated with channel morphology and macroinvertebrate responses. For correlations involving streambanks, we used the bank segment just upstream of the transect that was used to sample other variables to account for suspected drift effects. Correlations between variables allowed us to identify potential links between environmental effects observed in different components of the aquatic ecosystem.

To determine how macroinvertebrates *as a group* responded to treatment, we ordinated species by sample, using the nonparametric multivariate method "nonmetric multidimensional scaling" (NMS; McCune and Grace 2002). NMS uses relative abundance of taxa to classify samples and then orders these samples in multidimensional space based on their relative similarity to one another. It is the preferred ordination method for data that are nonlinear and nonnormal. The value of this tool is that the primary sources of variation present during the study period (year, block, treatment) could be described in relation to one another, as they influenced the numerically dominant ensemble of species. These patterns of variation could then be used to better interpret the ANOVAs for the suite of individual macroinvertebrate variables. We constructed 2 matrices for the ordination. A "main matrix" consisted of the 54 taxa found in more than 50% of the samples (42 species/genera + 12 higher taxa). These 54 taxa represented over 97% of the total abundance sampled. A "secondary matrix" (or "environmental matrix") was also constructed, consisting of the categorical variables (block, treatment, year) plus the 5 quantitative variables (percent covered/stable banks, percent uncovered/unstable banks, width-to-depth ratio, percent embeddedness, and mean substrate particle size). Categorical and quantitative variables within this environmental matrix could then be used to help interpret the ordination derived from the main matrix. In this article, we present the ordination for the September samples because it was assumed that these would best represent the grazing effect for each of the last 3 years of the experiment.

Because the entire study area had been only lightly grazed over the preceding several years, we expected that the doubling of grazing intensity would cause significant grazing impacts on streambanks relative to ungrazed controls (Fig. 2). Regarding the 3 channel morphology variables, we expected that in grazed units, stream width-to-depth ratio and cobble embeddedness would increase and that mean substrate particle size class would decline over time because of greater sediment deposition caused by streambank breakdown. The strength of this relationship however, was hypothesized to be less than the strength of the relationship between grazing and bank decline (indicated by narrower arrow width in Fig. 2). For the aquatic macroinvertebrate community, we expected that grazing would cause an overall decline in abundance and richness, particularly in those taxa that were regarded as "sensitive" (mayflies, stoneflies, caddisflies) and for those metrics that reflected the sensitivity of species that were either tolerant (e.g., many noninsect taxa) or intolerant of sediment (e.g., EPT richness). This relationship is depicted by a narrow dashed line in Figure 2, indicating that we expected habitat decline (increase in embeddedness or decrease in mean substrate particle size) to be negatively related to invertebrate abundance. Finally, we expected time to contribute unknown variation to all de-

Hypothetical Effects
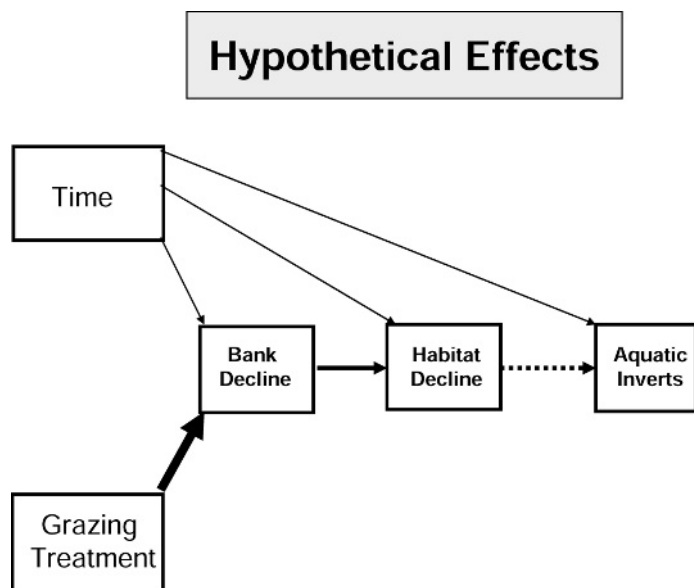
Figure 2. Conceptual path model of expected relationships among independent variables (grazing treatments and year) and dependent variables (streambanks, habitat [channel morphology], and bugs [aquatic macroinvertebrates]). Width of arrows visually indicates strength of relationship; solid line indicates positive and dashed line indicates negative relationship.

pendent variables because of factors outside the control of our experimental system. These factors would include variation in sediment pulses generated upstream and interannual variation in the relative abundance of particular invertebrate species. This conceptual path model was parameterized by data from the experiment. Model estimation was based on maximum likelihood, and the adequacy of model fit was evaluated using the model chi-square and its associated P values. We used the software package AMOS 5.0.1 (James Arbuckle, SmallWaters Corporation), now available as AMOS 7.0 through SPSS (http://www.spss.com/amos/index.htm).

RESULTS

As expected, grazing caused a significant change in streambank condition over the course of each grazing season, with a decrease in the percentage of covered and stable banks ($P < 0.05$; Fig. 3A) and an increase in the percentage of uncovered and unstable banks ($P < 0.05$; Fig. 3B). No changes in streambank condition were observed throughout each summer in ungrazed units ($P > 0.05$). Although bank recovery occurred by June 1998 and 1999, the percentage of streambanks in the highest condition class (covered/stable) declined more by September each successive year (Fig. 3A). Similarly, recovery by June was indicated in the percentage of streambanks classified in the lowest condition class (uncovered/unstable), but increases in this class were greater by September each successive year (Fig. 3B).

Clear patterns of grazing impact on streambanks were accompanied by much more subtle patterns in other channel morphology variables. In particular, when data from all 3 years are analyzed together, cobbles in grazed units were significantly more embedded in September ($P = 0.05$; Fig. 4). Recovery of
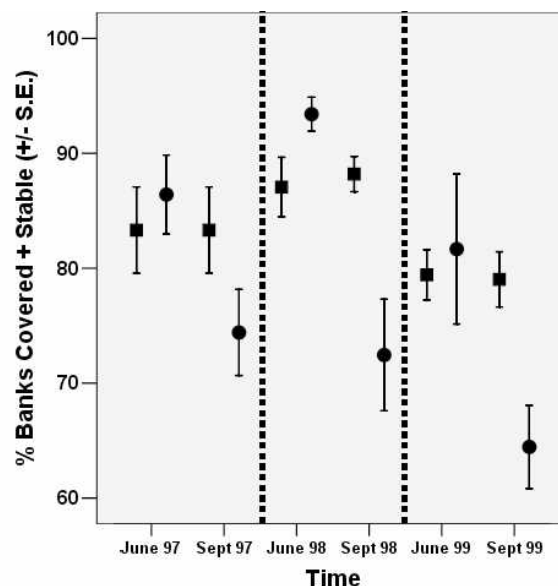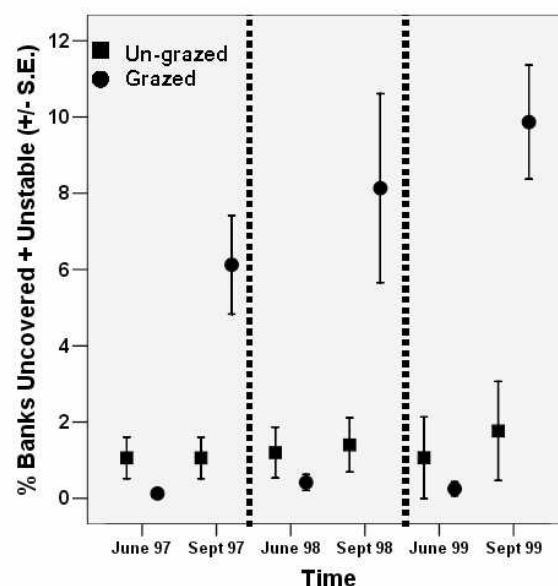

Figure 3. Percent of streambanks in covered/stable condition class (**A**) and in uncovered/unstable condition class (**B**) in grazed and ungrazed units over the course of the 3-year measurement period (Oregon State University Hall Ranch).

embeddedness occurred by the time measurements were taken the following June, however, paralleling the pattern observed in streambank condition. In addition, measurements of stream width taken at the same locations as streambank assessments were significantly correlated ($r^2 = 0.34$; $P < 0.003$) in 1998 and 1999, suggesting that grazing might have caused an increase in stream width. All evidence taken together indicates that macroinvertebrate habitat quality was reduced slightly in grazed versus nongrazed units during the course of each grazing season, with some recovery occurring by June of each successive year.

An estimated total of just over 300 000 invertebrates were collected over the 3-year sampling period, 93% of them insects (Table 1). Among noninsect groups, mites were the most commonly collected (13 569 individuals), followed by oligo-
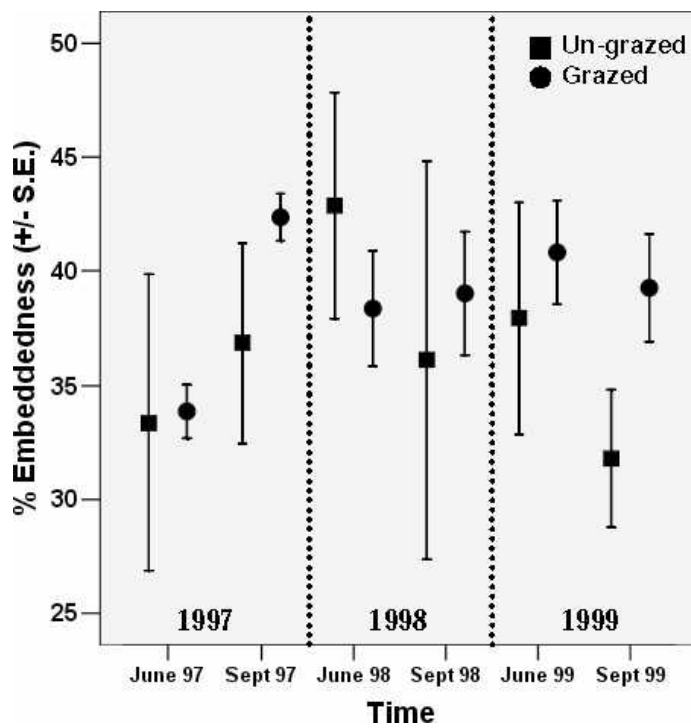
Figure 4. Percent embeddedness (± SE) in grazed and ungrazed units over the course of the 3-year measurement period (Oregon State University Hall Ranch).

chaetes (4 227) and snails (1 812). Among insects, we collected representatives from 6 orders. The most commonly collected order was the true flies (92 715 individuals), followed by the mayflies (72 693) and the beetles (72 277). Nonbiting midges were the most commonly collected of all families, comprising 20% of the total. In terms of both abundance and species richness, the 3 typically most intolerant orders (Ephemeroptera, Plecoptera, Trichoptera) were well represented in our collection, making up 38% of the total number of aquatic invertebrate individuals and 39% of the total taxa richness.

The most conspicuous pattern observed in the macroinvertebrate community was a pronounced decline in the abundance of most species over the course of the experiment in both grazed and ungrazed experimental units (Fig. 5A). Total invertebrate abundance · m^{-2} averaged about 10 200 individuals in June 1997 but significantly declined to 3 750 in June 1999 ($P < 0.001$). This pattern of decline was even more pronounced for the September samples, with a high of over 17 700 · m^{-2} in 1997 significantly declining to just 2 480 individuals · m^{-2} in 1999 (1 400 for grazed and 3 560 for ungrazed units) ($P < 0.001$). Declines occurred in most taxonomic groups, including more sensitive taxa such as Ephemeroptera (mayflies) + Plecoptera (stoneflies) + Trichoptera (caddisflies) (EPT Taxa) ($P < 0.001$) and generally less sensitive taxa such as oligochaetes (earthworm relatives; $P < 0.001$). For only 2 higher taxonomic groups did we observe a significant increase in abundance from 1997 to 1999:

Table 1. Summary of higher taxa of invertebrates represented in the collection: number of distinct taxa recognized and the estimated number of individuals collected over the 3-year sample period. The number of taxa is related in part to the effort made to identify specimens. Thus, for some groups (e.g., mites, nematodes, oligochaetes), taxa richness is underestimated because taxonomic expertise was not easily available for identification of species.

Taxonomic rank	Taxonomic group	No. taxa	No. collected			Total
			1997	1998	1999	
Phylum	Nematoda (roundworms)	1	189	252	188	629
Phylum	Annelida (segmented worms)					
Class	Oligochaeta (earthworms)	1	2 488	514	1 225	4 227
Phylum	Arthropoda (joint-foot animals)					
Class	Crustacea (crustaceans)					
Order	Copepoda (copepods)	1	4	109	143	256
Order	Ostracoda (ostracods)	1	15	24	91	130
Order	Decapoda (crabs, crayfish)	1	249	307	17	573
Class	Arachnida (spiders, mites)					
Order	Acari (mites)	1	9 215	3 477	877	13 569
	Total noninsects	6	12 160	4 683	2 541	19 384
Class	Insecta (insects)	163	155 811	92 103	31 846	279 761
Order	Ephemeroptera (mayflies)	19	36 632	24 562	11 499	72 693
Order	Plecoptera (stoneflies)	21	10 477	8 899	1 587	20 963
Order	Trichoptera (caddisflies)	28	12 246	7 134	1 619	20 998
Order	Megaloptera (dobsonflies)	1	63	27	25	115
Order	Coleoptera (beetles)	7	37 871	26 619	7 787	72 277
Order	Diptera (true flies)	87	58 522	24 863	9 330	92 715
Family	Nonmidge families	31	23 537	7 148	2 596	33 281
Family	Chironomidae (midges)	56	34 985	17 715	6 734	59 433
	Total	169	167 971	96 786	34 387	299 145
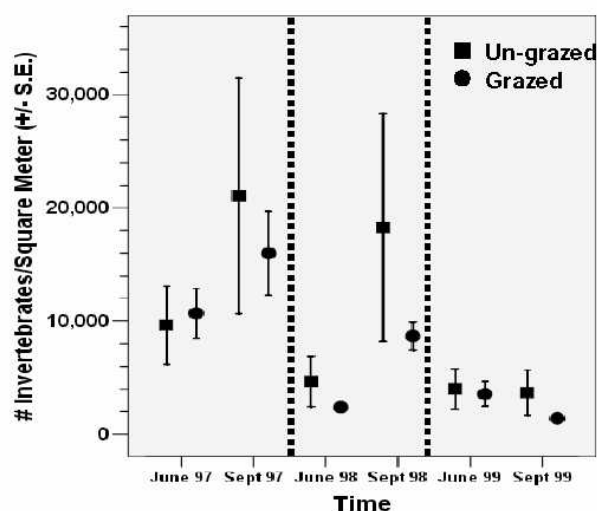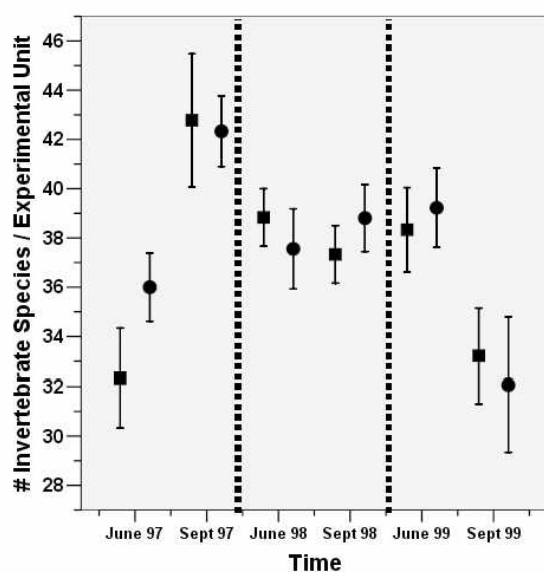
Figure 5. Mean (± SE) invertebrate abundance (**A**) and mean (± SE) number of invertebrate species per experimental unit (**B**) in June, August, and September in grazed and ungrazed units over the course of the 3-year measurement period (Oregon State University Hall Ranch).

percent abundance of noninsect species ($P < 0.01$) and percent abundance of midge species ($P < 0.02$); all other higher taxonomic groups declined over the 3-year measurement period. For the 6 functional feeding groups analyzed, we observed decreases over the years in the percent abundance of parasites ($P < 0.001$) and scrapers ($P < 0.001$), while the percent abundance of collector-gatherers increased ($P < 0.01$). Finally, of 42 individual taxa analyzed, 28 showed significant year effects, in each case a decline over the 3-year period of time. Taken together, these data indicate a broad-spectrum decline in macroinvertebrate abundance, with a slight shift in abundance toward more sediment-tolerant, noninsect species.

Changes in total species richness over the 3-year period were somewhat less profound than patterns for abundance but generally showed a shift toward increased richness of more sediment-tolerant species. Total invertebrate species richness observed in the September samples declined over the 3-year

period (Fig. 5B; $P < 0.001$), and this was accompanied by significant declines in the richness of most higher taxa, including the orders Ephemeroptera + Plecoptera + Trichoptera (EPT taxa: $P < 0.001$), Coleoptera ($P < 0.001$), and Diptera ($P < 0.001$) and the family Chironomidae ($P < 0.001$). The only higher taxa for which an increase in richness was observed were the number of noninsect species ($P < 0.01$) and the number of "tolerant" species ($P < 0.001$). The Hilsenhoff tolerance index was consistent with these richness patterns, indicating an increase in index value over the 3-year period ($P < 0.001$). Taken together, patterns in richness suggest that overall habitat quality in Milk Creek declined significantly over the 3-year study period.

There were significantly higher numbers of total invertebrates in the nongrazed units overall ($P = 0.04$) despite considerable variability among experimental units (Fig. 5A). We found similar patterns (lower numbers in grazed units) in the percentage of Coleoptera individuals ($P < 0.001$), in the abundance of the most dominant taxa ($P < 0.001$), and in the abundance of species classified as "scrapers" ($P < 0.001$). There was a strong trend toward lower numbers of individuals in the orders Ephemeroptera, Plecoptera, and Trichoptera (EPT taxa; $P < 0.07$) in the grazed units as well. The percentage of individuals classified as parasites was higher in the grazed units ($P < 0.05$). When interpreted together, these results indicate that habitat quality was generally higher in ungrazed units relative to grazed units.

At the species level, there were significant grazing effects for 6 of the 42 taxa analyzed, all showing higher abundances in the no-grazing treatment. No taxa showed higher abundances in the grazed units. The 6 taxa for which significantly lower abundances were observed in grazed units included the mayfly *Ephemerella inermis/infrequens*-complex ($P < 0.01$), the stoneflies *Zapada cinctipes* (Banks) and *Z. oregonensis* (Claassen) ($P = 0.01$ and $P = 0.03$), the crane fly *Antocha* sp. ($P = 0.02$), and the midges *Micropsectra* sp. ($P < 0.001$) and *Eukiefferiella* sp. ($P < 0.01$). In each of these taxa except *Antocha* sp., there was a significant year effect as well, with general decline in abundance from 1997 to 1999. These combined effects indicated that decline over the years was more steep for these taxa than for others in the grazing units. Finally, the significant grazing effects for 4 species were not uniform across the blocks: for *Antocha* sp., *Micropsectra* sp., and *Z. cinctipes*, the grazing effect was highest in block 3, while for *Zapada oregonensis* (Claassen), the grazing effect was highest in block 2.

When the 54 most prevalent invertebrate taxa sampled in September were ordinated in sample space, the dominant year effect is clearly seen (Fig. 6) in the display of axis 1 versus axis 3 (graphs of axis 1 vs. 2 and 2 vs. 3 show very similar patterns). With the exception of 1 sample (open triangle 99-3: ungrazed, 1999, block 3), the 9 samples from each year can be neatly arranged into 3 groups, 1 for each year. Axis 3 clearly separates the 1997 samples from both the 1998 and the 1999 samples, while axis 1 clearly separates the 1999 samples from both 1997 and 1998; samples delineated by ellipses are significantly different from one another (multiresponse permutation procedure [MRPP]; $P < 0.01$). The only environmental variables significantly correlated with the 2 ordination axes are the percentage of streambanks classified as covered and stable (CS;
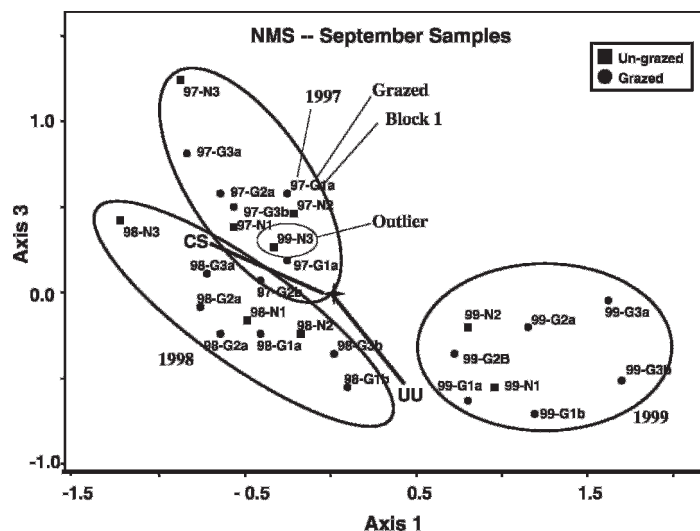
Figure 6. Ordination (NMS, or nonmetric multidimensional scaling) of samples based on species in grazed and ungrazed units over the course of the 3-year measurement period (Oregon State University Hall Ranch). Circles denote grazed units; squares denote no grazing units. Number code defined as follows: Circle 97-3 = grazed, 1997, block 3. Ellipses drawn by hand to emphasize major groupings. The 2 significant correlations ($r^2 > 0.04$) of environmental variables (UU = unstable, uncovered banks; CS = stable, covered banks) are drawn with their pattern of correlation to the main matrix indicated: thus, higher UU values were found in samples toward the lower right of the graph (mainly 1999 samples), while high CS values were found in samples toward the upper left (mainly 1997 samples).

$r^2 = 0.35$) and as uncovered and unstable (UU; $r^2 = 0.35$). The CS variable is more correlated with axis 1 than axis 3, thus explaining more of the variation between 1999 and the other 2 years, while the UU variable is more correlated with axis 3, thus explaining more of the variation between 1997 and the other 2 years. Thus, changes in the macroinvertebrate community parallel changes in streambank condition in terms of both the highest condition class (CS) and the lowest condition class (UU). There were no significant correlations, however, between the ordination axes and the channel morphological variables that would be expected to indicate changes in invertebrate habitat quality, such as mean substrate particle size, embeddedness, or the ratio between width and depth.

The only sample that did not fit neatly within the pattern described above was the 1999 ungrazed sample in block 3 (labeled outlier in Fig. 6). This sample represents the most downstream location in the study area (Fig. 1). When ordinated, the sample fell squarely within the 1997 ellipse but was still separated clearly from both the 1997 and the 1998 samples taken at the same location (open triangles 97-3, 98-3; Fig. 6). It is apparent from the ordination that this location was fairly distinct each year, with both the 1997 and the 1998 samples lying at the edge of their groups, respectively. Furthermore, the species that most indicate the position of ungrazed 99-3 within the ordination are the same set of species that best explain both axes; in other words, species that were typically rare in 1999, because of decreases in abundance from 1997 to 1998 and from 1998 to 1999, were still common in this ungrazed unit of block 3. This most downstream sample

location therefore served as a possible refuge in 1999 for a distinct group of indicator species.

The 10 taxa that explained most of the variation along axis 3 were segmented worms (Oligochaeta), the mayfly *Ephemerella inermis/infrequens*-complex, the stonefly *Skwala* sp., the beetle *Cleptelmis addenda* (Fall), the caddisfly *Glossosoma* sp., and the midges *Rheotanytarsus* sp., *Thienemanniella* sp., *Tvetenia bavarica*-group, *Cricotopus* sp., and *Eukiefferiella* sp. Correlations with axis 3 for these 10 taxa were each greater than −0.68, showing a general decrease in abundance from 1997 to 1998. The ten taxa that explained most of the variation along axis 1 were the mayflies *Epeorus* sp. and *Paraleptophlebia* sp.; the beetles *Optioservus* sp., *Zaitzevia* sp., and *C. addenda*; the caddisfly *Rhyacophila brunnea* Banks; the midges *Orthocladius*-complex and *Tvetenia bavarica*-group; total midge pupae; and the owl fly *Pericoma* sp. Correlations for these 10 taxa were each greater than −0.55, showing a general decline in abundance from 1998 to 1999. These indicator taxa span a wide range of expected tolerance to sediment. For example, the caddisfly (*Glossosoma* sp.) and the mayfly (*Epeorus* sp.) are typically rated as "sensitive" species, while oligochaetes and the beetle *Optioservus* sp. are typically rated as "tolerant" taxa, yet all these taxa indicate a decline in habitat quality over the course of the 3-year sampling period. Within the year-defined groups, there are no clear patterns of separation due to either block or treatment (Fig. 6). Thus, if sediment from streambank breakdown influenced macroinvertebrates, any influence was so systemic that it masked the more subtle differences among treatments observed in the ANOVAs. The ordination indicates that the entire system was affected, from the upstream units of block 1 to most of the downstream units of block 3 (with the possible exception of the nongrazed units in block 3, the most downstream in the study area).

Structural equation model (SEM) analysis showed that the data were consistent with the structure of the initial model (Fig. 2), though not necessarily with the quantitative expectations. An overall model χ^2 value of 7.6 was obtained for goodness of fit, with $P = 0.10$. Note that, in SEM, a P value greater than 0.05 is indicative of an absence of major deviations between model and data and thus indicates model adequacy. The findings confirm the overall relationships that emerged from ANOVA of individual variables, from correlations among variables, and from the ordination of species (Fig. 7). Grazing had a strong positive effect on bank decline. Declines in habitat quality, as expressed by embeddedness, caused a subtle negative effect on macroinvertebrate abundance, as expected. Time also had an influence, but the effect was much greater than expected and also negative, supporting the conclusion from previous analyses that macroinvertebrates declined over time. Overall, the path model reflects the fact that the macroinvertebrate community collapsed by the third measurement year regardless of treatment effect. In fact, of the macroinvertebrate variation explained by this model (43%), the great majority (97%) is explained by direct decline in abundance over the 3-year period, with only 3% explained by the grazing effect itself.

DISCUSSION

The most convincing sign of cattle grazing effect in the current study is the seasonal degradation of the banks along Milk
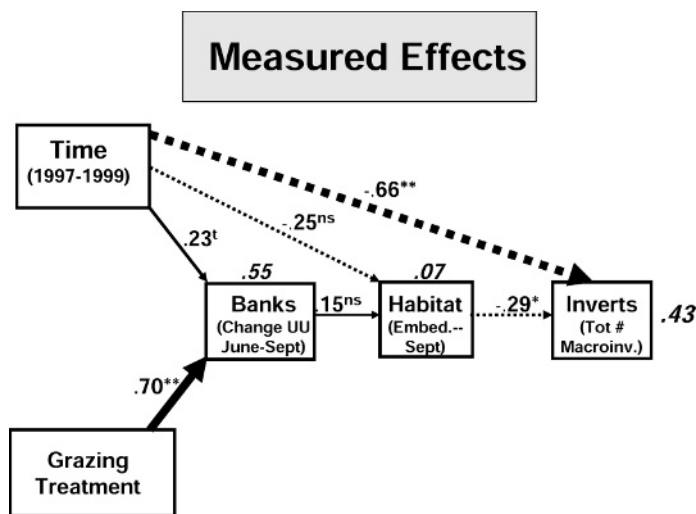
Figure 7. Parameterized path model (using data from the experiment) depicting the nature of relationships among independent variables (year and grazing effects) and dependent variables (bank decline throughout each year, habitat quality as measured by September embeddedness, and bugs as measured by total invertebrate numbers). Numbers in italics give the proportion of variation for indicated dependent variable explained by the model. Numbers on arrows give the path coefficients relating pairs of variables; ns = nonsignificant ($P > 0.15$), t = trend ($0.15 < P < 0.05$), * = significant ($P < 0.05$), and ** = highly significant ($P < 0.01$). Width of arrows visually indicate strength of relationships; solid lines indicate positive and dashed lines indicate negative relationships.

Creek. Each year they were measured, between 12% and 20% of streambank length was degraded from the highest to a lower condition class, accompanied by an increase of between 6% and 10% in the representation of the lowest condition class. Streambanks showed signs of having recovered over the fall, winter, and spring of 1997–1998 and 1998–1999, although there was a trend toward greater decrease in quality from June to September each successive year. This pattern was observed with respect to both the decline in the highest category of condition (covered, stable) and the increase in the lowest category of condition (uncovered, unstable). The accumulation of effects in streambanks corresponded to a slight increase in stream width and percent embeddedness in the grazed units over the 3-year measurement period. Although these effects were relatively subtle, they were consistent with the doubling of grazing intensity that occurred in the experiment (0.77 ha $\cdot$ AUM^{-1}) relative to the intensity observed in the previous 5 years (1.75 ha $\cdot$ AUM^{-1}).

The response of the macroinvertebrate community was more subtle and more complex. The weight of evidence suggests that the macroinvertebrate community shifted slightly toward more sediment-tolerant taxa in grazed units. This conclusion is bolstered by the observation that total invertebrate numbers, percent Coleoptera individuals, percent EPT individuals, and the abundances of 6 taxa were all lower in grazed units, while no taxa were significantly more abundant in grazed units. The observation that Coleoptera and EPT taxa showed a negative response to grazing is consistent with results from a recent study on an Index of Biotic Integrity from southern California,

in which the percentages of Coleoptera and EPT species were diagnostic of relatively healthy streams (Ode et al. 2005). Yet it should also be noted that some of the variables that were significantly different between ungrazed and grazed units showed patterns that could not be explained by the simple argument that grazing decreased habitat quality.

With respect to the change in the macroinvertebrate community over the 3-year measurement period, the observed declines between 1997 and 1999 were striking. Importantly, the declines occurred in *both* grazed and ungrazed units, with the single exception of the most downstream ungrazed pasture of block 3. Thus, the cause of the precipitous drop in the abundance of some macroinvertebrates cannot be ascribed to grazing impacts, unless it is assumed that the major mechanism for the decline was sediment produced by grazing within the study area that drifted throughout the entire system. A more likely explanation for the decline in macroinvertebrate abundance and species richness is that something else happened to Milk Creek that affected the entire system simultaneously during the summer of 1999. In fact, 2 significant events occurred upstream during that time, each of which could have contributed to the observed effects (Rick Wagner, Oregon Department of Forestry, La Grande, OR, personal communication). First, a logging operation on private land upstream from and adjacent to Hall Ranch was carried out in the summer of 1999. Although the logging occurred in the uplands and did not directly influence the riparian area, logging trucks were observed to have driven across Milk Creek repeatedly without the benefit of a bridge or culvert, less than 300 m upstream from the study area. Also, 2 small culverts located 2 km upstream from Hall Ranch were replaced with larger culverts during the summer of 1999. Each of these events created sediment plumes that not only reached the Hall Ranch study area but also penetrated several hundred meters downstream. On August 13, 1999, 1 sediment plume from the logging operation was still visible by eye as far downstream as the beginning of block 3 (J.D. McIver, unpublished date). It is noteworthy that the only pasture within the study area that did not experience a precipitous decline in macroinvertebrate abundance and richness was the pasture farthest downstream (ungrazed, block 3). It is therefore possible that the ungrazed pasture of block 3 functioned as a refuge for more sensitive species because it was just far enough away from the upstream boundary of the study area to be largely shielded from the effects of sediment produced in the private lands upstream.

It is clear that heavy livestock grazing can reduce aquatic community integrity and water quality by removing vegetation (Leege et al. 1981) and by increasing bank instability through trampling (Moring et al. 1985; Platts 1986; Marlow et al. 1987). Once again, however, most studies have involved the assessment of environmental effects of grazing treatments with very high stocking levels. While studies on grazing effects of more moderate intensities are less common (Blackburn 1984), Hanson et al. (1970) showed that decreasing grazing intensity from "heavy" to "light" resulted in half the annual runoff, suggesting that managed grazing systems are an improvement over unrestricted systems. Indication that light to moderate cattle grazing may be compatible with relatively healthy riparian systems was noted by Clary (1999), who found that

previously degraded riparian systems recovered equally well in ungrazed, lightly grazed, and moderately grazed treatments in terms of vegetation and streambank stability. While these studies and others suggest that cattle grazing strategies can reduce impacts on sensitive riparian areas, what is really needed are experiments that link cattle grazing intensity and time with bank erosion, sediment release, and instream habitat effects. Such studies are essential if we are to understand the thresholds beyond which cattle-induced bank breakdown becomes a problem for sediment yield, macroinvertebrate habitat quality, and fish production.

MANAGEMENT IMPLICATIONS

The significant streambank effects due to grazing observed in this study beg the question: would the magnitude of these effects be expected to result in eventual channel morphology changes or to contribute to declines in native fish populations? The answer to this question depends on whether streambanks recover over the course of the year and whether the bank breakdown along Milk Creek created enough sediment to cause permanent changes in aquatic habitat quality for macroinvertebrates and fish. Several studies have reported significant channel morphology effects as a consequence of chronic, intensive livestock grazing (Marlow et al. 1987; Rinne 1988), but few have attempted to follow recovery rates year to year, especially after more moderate grazing. Kauffman et al. (1983), working on a stream adjacent to Milk Creek (Catherine Creek), found that a stocking rate of 1.3–1.7 ha · AUM^{-1} (compared to 0.77 ha · AUM^{-1} measured in the current study) caused significantly greater bank erosion compared to ungrazed controls during 2 seasons of grazing. They also found that while overwinter erosion was not different among treatments, livestock grazing was enough to cause an overall increase in streambank losses over the study period. On the other hand, there is evidence that moderate bank damage can be mitigated by natural processes in the following year. Buckhouse et al. (1981) reported that while moderate cattle grazing caused measurable bank effects in a single season, any differences between grazed and ungrazed treatments were erased the following year by ice effects and peak flows. While their experiment was not able to isolate cattle grazing effects per se, results underscore the difficulty in understanding the role of grazing for sediment production in the context of the annual cycle of sediment release. Similarly, Marlow et al. (1987) reported that stream flow and cattle use were both correlated with degree of change in stream channel profile. In particular, streambank alteration resulted from a combination of high soil moisture, high stream flow, and cattle use. Thus, cattle impacts could be judged only within the context of the annual cycle of natural events typical of their study site. In general, because at least 30 variables are involved in the sediment transport process (Heede 1980), few studies have been able to isolate the effect of ungulate grazing from the natural background of erosion that occurs over the course of the year (Skovlin 1984). Given these considerations, it would be interesting to measure the extent of bank recovery over many years of moderate grazing at the Milk Creek study site.

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