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EFFECTS OF LIVESTOCK GRAZING EXCLOSURE ON AQUATIC MACROINVERTEBRATES IN A MONTANE STREAM, NEW MEXICO

John N. Rinne[1]

ABSTRACT.—Aquatic macroinvertebrate populations inhabiting reaches of a stream within areas excluded from livestock grazing for a decade were markedly different from those in grazed areas when density, biomass, biotic condition indices, and mean chi square indices of the two populations were compared. Increased densities and biomasses of more tolerant forms of macroinvertebrates were observed in grazed reaches. Because pretreatment data were not available, differences in macroinvertebrate populations and relative tolerances of taxa in grazed and ungrazed areas could be as easily attributed to linear changes in stream habitat as to removal of domestic livestock. Results of this study have implications for the design of future research on the effects of livestock grazing on stream environments and biota: (1) baseline/pretreatment information is prerequisite, and (2) the study should take a watershed (ecosystem) approach.

National forests, in their multiple-use role, are managed to serve and meet the needs of water, timber, wildlife, recreation, and grazing interests. The potential for conflict between respective uses is ever present. In recent years grazing has been implicated in having contributed significantly to widespread deterioration of habitat and decline of biota inhabiting riparian-stream areas (Platts 1982, Kauffman and Krueger 1984, Skovlin 1984). Thus, both managers and researchers recognize the need to better understand the impact of grazing on riparian ecosystems.

The Rio de las Vacas in northwestern New Mexico provides excellent trout sport fishing (Rinne 1988), and its valley provides camping opportunities and high-quality forage for livestock grazing. Because of concern over conflicts of use, this montane stream was partially fenced by the USDA Forest Service to exclude domestic livestock and thereby improve in-stream and near-stream habitat. Effects of grazing on in-stream and near-stream habitat and extant biota were examined a decade later. Results of several years of study indicated that apparent changes in habitat and biota had occurred (Rinne 1985, Rinne 1988, Szaro et al. 1985, Szaro and Rinne 1988).

Studies on how grazing impacts aquatic macroinvertebrates are nonexistent in the literature. This is largely attributable to the inherent difficulty of devising reliable sampling techniques required to adequately define temporal and spatial variability in stream macroinvertebrate populations (Resh 1979). Nevertheless, Buikema and Cairns (1980) and Munther (1985) have suggested that insects are very sensitive to environmental perturbations and therefore valuable in early detection of habitat changes resulting from particular management activities. A study was initiated in 1982 on the effects of grazing removal on stream habitat and biota (Rinne 1985, Rinne, in press). This paper reports the effects of exclusion of livestock grazing on aquatic macroinvertebrates in this particular montane stream.

STUDY AREA

The Rio de las Vacas (hereafter called the Vacas) is a third-order (Hynes 1975) montane stream draining the San Pedro Parks Wilderness Area, Santa Fe National Forest. The study area is at an elevation of 2,600 m, MSL, about 17 km southeast of the town of Cuba, Sandoval County, New Mexico. Two exclosures, each about 1 km in length and 50 m in width (ca 10 hectares), were established in 1972 and 1975 (Fig. 1). These exclosures were separated by private land holdings from a downstream grazed area (Fig. 1). The exclosures are within the Cuba Community Grazing Allotment, which comprises about 11,000 ha. Almost half of this allotment is classified as "no-allowable capacity" for grazing because of

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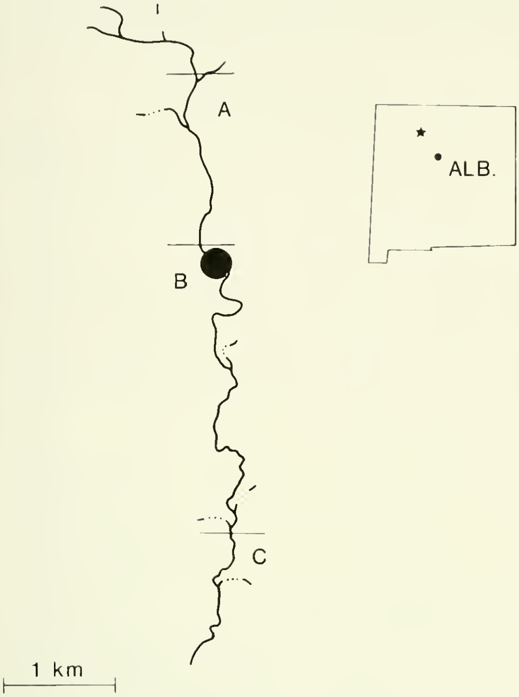

Fig. 1. Location and detailed map of study area indicating upstream ungrazed (A), grazed (B), and lower grazed (C) reaches. Sample site on private lands in 1984 is indicated by solid circle.

terrain and lack of forage. Near-equal percentages of lands (10% each of total allotment) are private holdings and riparian. Historically (1949–1980), the grazing strategy has been season-long (1 June to 31 October), at the average annual rate of 2,688 animal unit months (AUM). For further description and photos of the area, see Rinne (1985) and Szaro et al. (1985).

METHODS

Aquatic macroinvertebrates were sampled with a standard Surber sampler (1024-micron mesh size) during summer low flow conditions in reaches of stream in ungrazed (fenced) and grazed areas. Samples were preserved with 10% formalin in 1982 and with a glycerin-formalin-ethyl alcohol solution in 1983 and 1984. In 1982, samples were randomly taken in riffle habitats of the stream (Table 1) within the upstream exclosures and the downstream grazed area (Fig. 1). Because of variability in 1982 data (see RESULTS), which has been suggested to be the norm (Resh 1979), an attempt was made on ensuing (1983–84) samples to stratify within substrate composition categories to aid in reducing variability among samples (Table 1). In addition to sampling more for pebble-cobble substrate (17–256 mm), uniformity of water depth and velocity were likewise selected (Table 1). In 1984, because of suspected linear change in stream habitat, samples also were taken at stream sites within an upstream, contiguous area of private, grazed lands (Fig. 1).

In the laboratory invertebrate samples were sorted in gridded petri dishes with the aid of a compound dissecting microscope and identified to the lowest practicable taxon; taxonomic keys in Usinger (1956) and Merritt and Cummins (1984) were primary sources to aid in organism identification. Biomass of samples was estimated in 1982 and 1983 by using the length–dry weight equations of Smock (1980). Length was separated into seven size classes (0–6 mm); the largest individuals in the last class were measured directly because of the lack of an upper limit for this class. Because biomass values were not significant once stratified sampling was initiated in 1983, values were not estimated in 1984.

Tolerance quotients (TQ) were used to calculate biotic condition indices (BCI) following methods outlined in Winget and Mangum (1979) and discussed in Platts et al. (1983). The use of TQ reflects the tolerance of an invertebrate to alkalinity and sulfate content of the water in addition to its selectivity for or against fine substrate materials and stream gradient. Winget and Mangum (1979) have calculated TQ for a number of families and genera based on field data. Basically, if lower alkalinity and sulfate and greater gradients and larger substrate particle size are selected by an organism, its TQ will be lower. In turn, the BCI incorporates stream habitat, water quality, and TQ. The index is a function of a predicted community (TQ_p) divided by an actual community (TQ_a).

Chi-square indices (CSI) (Parrish and Wagner 1983) also were utilized to aid in comparing and quantifying community composition of stream macroinvertebrate populations in the grazed and ungrazed areas. Nonparametric Kolmogorov-Smirnov (KS) tests and standard t-tests (Statgraphics) were employed

TABLE 1. Characteristics of stream sample sites in grazed and ungrazed areas in the Rio de las Vacas, New Mexico, 1982–84. Ranges are in parentheses. Substrate classes are modified from Hynes (1975): (1) sand (< 2 mm); (2) gravel (3–16 mm); (3) pebble (17–64 mm); and (4) cobble (65–256 mm).

Year	Area	Velocity (cm/sec)	Depth (cm)	Substrate class 1	Substrate class 2	Substrate class 3	Substrate class 4	n
1982	Grazed	41.1 (13–60)	23.4 (10–35)	0	40	0	60	10
	Ungrazed	30.3 (10–75)	19.1 (8–40)	35	18	5	42	20
1983	Grazed	20.9 (13–25)	22.0 (21–25)	20	10	35	35	10
	Ungrazed	33.9 (10–40)	23.3 (19–30)	15	19	25	49	10
1984	Grazed	36.6 (27–40)	17.0 (15–22)	0	0	50	50	10
	Ungrazed	34.6 (23–40)	14.4 (10–17)	0	0	50	50	5

to determine distribution patterns and significance of density, biomass, TQ, and BCI in reaches of the stream in the two differently managed areas.

RESULTS

Eight orders, 39 families, and 71 genera of aquatic macroinvertebrates were collected and identified during the study (Table 2). As is common in streams, dipterans, trichopterans, ephemeropterans, odonatans, and coleopterans dominated the macroinvertebrate fauna in the Vacas in 1982 and 1983, comprising 88–97% of the total biomass (Table 3).

KS tests demonstrated uniform distribution of organism densities and biomasses in reaches of the stream within grazed and within ungrazed areas. Densities of macroinvertebrates were significantly greater in stream reaches within the grazed areas in 1982 and 1983, but not in 1984 (Table 4, Fig. 2). Samples from the upper and lower reaches of the stream taken within grazed areas in 1984 (B and C, Figs. 1 and 2) were separated and densities compared to densities in the ungrazed area (A). In 1984 differences were significant ($P = .02$) between samples from within the livestock grazing exclosures (A) and samples taken in the lower, grazed reaches (C) of the stream (Table 4, Fig. 2). In comparison, differences between densities from the ungrazed area (A) and those of the upper grazed area (B, Fig. 1), both taken in 1984, were not significantly different at the .05 confidence level (Table 4, Fig. 2).

Differences in macroinvertebrate biomass estimates in stream reaches within grazed areas were significantly higher than in ungrazed areas only in 1982 (Table 4, Fig. 3). There were no significant differences in biomass of macroinvertebrates between years (1982–83) in grazed or in ungrazed areas. Estimated biomass was consistently greater and more variable in the reaches of the stream within the grazed area in 1982 and 1983 (Fig. 3).

Average TQ was significantly higher in the ungrazed than in the grazed area in 1982, but the reverse was true in 1983 once stratified sampling was initiated. TQ was significantly higher in the lower grazed compared to the upper ungrazed area in 1984 (Table 4). BCI was significantly greater in the ungrazed reaches than in the grazed reaches of the stream in both 1983 and 1984.

Densities and BCI of four of the most common taxa in 1983 (i.e., those occurring in 10 or more samples) were compared to further illustrate that the overall tolerance of aquatic macroinvertebrates in the Vacas, as indicated by relative BCI, was greater in reaches of the stream in the grazed area. Two low-tolerance taxa, *Helicopsyche* sp. (TQ = 18) and *Ephemerella inermis* (TQ = 48), averaged 95 and 37 individuals/m^2, respectively, at stream sites in the grazed area. By comparison, these same taxa averaged 137 and 77 individuals/m^2 in the ungrazed area. Conversely, two

TABLE 2. Taxonomic listing and densities of aquatic macroinvertebrates per Surber sample collected in grazed (G) and ungrazed (U) reaches of the Rio de las Vacas, 1983–84. Chironomids were combined in 1984 because taxa were identified only to generic level in 1983.

	1983		1984	
Taxa	U	G	U	G
Ephemeroptera				
Ephemerella inermis	85	55	84	116
Paraleptophlebia sp.	32	29	11	34
Ameletus sp.	14	47	30	67
Baetis sp.	68	86	172	553
Cinygmula sp.	0	0	32	258
Tricorythodes minutus	40	396	4	45
Epeorus longimanus	19	11	256	85
Siphlonurus sp.	0	0	0	1
Ephemerella drunella grandis	0	0	0	2
Rhithrogena sp.	0	0	0	1
Diptera				
Culicoides sp.	11	11	0	0
Palpomyia sp.	22	38	0	0
Hydrophorus sp.	0	11	0	0
Hemerodromia sp.	11	0	0	8
Prosimulium sp.	22	0		
Simulium sp.	17	11	69	134
Tabanus sp.	14	11	2	1
Calopsectra sp.	19	87		
Cardiocladius sp.	14	32		
Corynoneura sp.	57	119		
Cryptochironomus sp.	0	22		
Hydrobaenus sp.	38	32		
Microtendipes sp.	36	18		
Paratendipes sp.	0	95		
Pentaneura sp.	11	11		
Polypedilum sp.	0	22		
Prodiamesa sp.	0	11		
Tanytarsus sp.	0	11		
Tendipes sp.	0	28		
Hexatoma sp.	11	33		
Ormosia sp.	16	0		
Tipula sp.	0	22		
Chironomidae			252	711
Hexatoma sp.			2	10
Maruina lanceolata			2	1
Picranota sp.			4	0
Ceratopogenidae			11	22
Deuterophebia sp.			2	0
Molophilus sp.			6	1
Ulomorpha sp.			0	1
Trichoptera				
Brachycentrus sp.	11	0	6	3
Agapetus sp.	11	0	30	15
Glossosoma sp.	15	0	54	61
Helicopsyche sp.	200	109	65	183
Hydropsyche sp.	47	78	265	318
Stactobiella sp.	11	0		
Lepidostoma sp.	46	0	0	6
Nectopsyche sp.	0	18		
Oecetis sp.	11	11	2	6
Asynarchus sp.	11	11		
Grammotaulis sp.	11	0		
Hesperophylax sp.	15	0	0	8
Polycentropus sp.	0	22	0	2
Amphicosmoecus sp.	2	9		

Table 2 continued.

	1983		1984	
Taxa	U	G	U	G
Plecoptera				
Alloperla (super genus)	0	11	0	0
Hastaperla sp.	0	27	0	0
Amphinemura sp.	19	11	26	16
Isoperla sp.	0	11	4	20
Unidentified genus	6	22		
Pteronarcella badia	0	11	6	1
Chloroperla sp.			17	10
Claasenia sp.			17	2
Nemoura (super genus)			0	1
Hemiptera				
Ambrysus sp.	22	18	2	24
Odonata				
Ophiogomphus sp.	22	19	2	2
Lepidoptera				
Parargyractis sp.	22	11	0	5
Coleoptera				
Helichus sp.	11	11		
Cleptelmis sp.	11	0		
Dubiraphia sp.	22	30		
Heterelmis sp.	11	0		
Optioservus sp.	25	11		
Stenelmis sp.	11	0		
Elmidae			136	168
Stenopelmus sp.			2	0
Dryopidae			0	2
Terrestrials	88	24	15	10

tolerant taxa (TQ = 108), *Calopsectra* sp. and *Tricorythodes minutus*, averaged 396 and 95/m^2 in the grazed compared to only 2.4 and 16 individuals/m^2, respectively, at sites in the ungrazed area.

Calculated mean chi square indices for the Vacas were 1.23 for 1983 and 2.3 for 1984. These high values indicate a marked difference in the structure of macroinvertebrate communities between the grazed and ungrazed areas (Parrish and Wagner 1983).

DISCUSSION

Use of the BCI and CSI on this group of macroinvertebrates in the Vacas suggests these organisms, indeed, may be a more suitable group than fishes to detect grazing effects on aquatic ecosystems. Although macroinvertebrate densities were variable in the Vacas, data (Table 4) serve to point out, as do selected water quality and fish data (Rinne 1988, Rinne, in press), that linear changes in physical stream habitat may have affected water quality and biota in this stream. To illustrate, during 1984 densities of aquatic macroinvertebrates estimated from samples collected in the ungrazed areas were not significantly ($P > .05$) different from pooled, downstream, disjunct, grazed (C, Fig. 1; $n = 5$) and contiguous, upstream, grazed (B, Fig. 1; $n = 5$) areas. However, separation of the 1984 samples into upper (contiguous) grazed and lower (disjunct) grazed sampling localities (Fig. 1) emphasized the greater densities of higher tolerance organisms in the downstream, disjunct, grazed area (Table 4, Fig. 2).

Significantly greater total alkalinity ($\bar{x}$ = 127 vs. 98 mg/l CaCO$_3$; $n = 15$) in summer combined with the slightly (ca 3 C) elevated water temperatures (Rinne, in press) in the downstream reach of the stream may have contributed to the greater macroinvertebrate and fish populations, as has been demonstrated elsewhere by Krueger and Waters (1983) and Egglishaw (1968). The lack of riparian vegetation in the lower grazed area of the Vacas (Rinne 1985, Rinne, in press) permitted

TABLE 3. Biomass (mg/m^2 wet weight) of the major groups of aquatic macroinvertebrates in ungrazed and grazed reaches of the Rio de las Vacas, 1982–83.

	1982		1983	
Order	Ungrazed	Grazed	Ungrazed	Grazed
Coleoptera	82.90	173.95	31.02	16.11
Diptera	127.90	174.23	39.47	191.83
Ephemeroptera	333.04	196.82	164.84	274.70
Hemiptera	2.09	111.37	1.03	13.17
Lepidoptera	0.00	9.57	8.80	5.79
Odonata	128.90	550.74	166.94	396.57
Plecoptera	10.53	3.48	0.20	6.68
Tricoptera	38.02	728.34	274.48	194.60
Totals	790.21	1960.34	764.70	1110.76

TABLE 4. Average number, biomass, tolerance quotients (TQ), and Biotic Condition Indices (BCI) for aquatic macroinvertebrates in grazed and ungrazed areas of the Rio de las Vacas, 1982–84. The number of samples is designated by n and significant (P ≦ .05) (single asterisk), and highly significant (P ≦ .01) (double asterisk) differences as determined by unpaired t-tests are indicated. Ranges are in parentheses.

Year	Treatment	Number/m^2	Biomass (g/m^2)	TQ	BCI	n
1982						
	Grazed	1897** (786–5216)	2.03* (.41–3.71)	64* (45–93)	88** (57–132)	10
	Ungrazed	787** (43–1691)	0.57* (.001–2.5)	78* (57–103)	71** (58–89)	20
1983						
	Grazed	1035* (701–2451)	1.01 (.31–3.04)	88* (68–101)	61** (52–78)	10
	Ungrazed	550* (108–1412)	0.69 (.02–2.61)	67* (44–89)	82** (60–121)	10
1984						
	Grazed					
	Upper	1509 (905–2058)	No Data	71 (60–85)	76 (62–87)	5
	Lower	4375* (2522–6283)		78* (69–82)	68* (64–76)	5
	Ungrazed	1541* (765–2974)		65* (51–77)	84* (69–103)	5

increased solar radiation to the stream and that, in turn, may have facilitated greater populations of macroinvertebrates, as reported in the Pacific Northwest by Murphy and Hall (1981), Murphy et al. (1981), and Hawkins et al. (1982). Also, significantly greater mean stream conductivity in summer in the grazed area than in the ungrazed area (216 umhos/cm vs. 164 umhos/cm, respectively) suggests that an increase in dissolved solids in stream water in the lower grazed area actually may be beneficial to aquatic macroinvertebrate communities. Winget and Mangum (1979) pointed out that high-elevation, "distilled water," first-order streams actually benefit from an increase in dissolved solids. Similarly, Egglishaw (1968) suggested that a more rapid turnover of organic matter resulted in more food availability for invertebrates in hard water (or high alkalinity) streams.

Mean TQs of aquatic macroinvertebrates were significantly less in 1982 in the grazed areas. Values, nevertheless, were significantly greater in both 1983 and 1984 when stratified sampling was initiated and when samples within the grazed area in 1984 were separated into upper and lower subsets. The combination of significant TQ and BCI suggests that although insect densities and biomasses were always greater in grazed

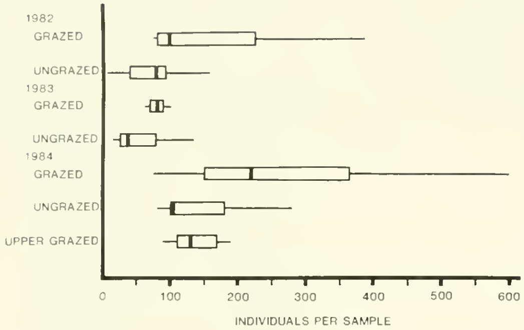

Fig. 2. Estimated number of aquatic macroinvertebrates per surber in grazed and ungrazed areas. Estimates for the upper (B, Fig. 1) and lower (C, Fig. 1) grazed samples taken in 1984 are also provided. Medians (large verticals), upper and lower quartiles (small verticals), ranges (single horizontals), and outlying values (dots) are indicated in plots.

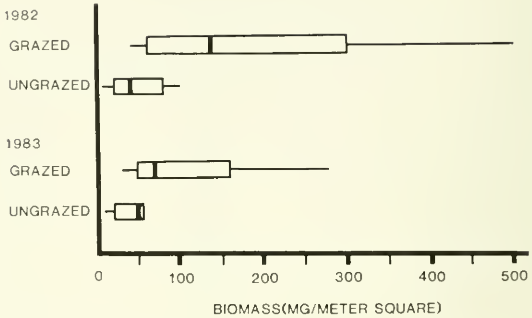

Fig. 3. Estimated biomass (mg/m^2) of aquatic macroinvertebrates in the Rio de las Vacas in grazed and ungrazed areas, 1982–83. Plots contain the same statistics as described for Fig. 2.

stream reaches (Table 4, Figs. 2, 3), these communities were comprised of the more-tolerant taxa. There was greater silt content in the substrate of the downstream reaches of the stream than in the upstream ungrazed area (Rinne 1988), which might, in part, explain the absence of some less-tolerant taxa. Further, the extremely high CSIs strongly suggest a difference in the structure of the aquatic macroinvertebrate community in the grazed compared to the ungrazed area of the Vacas. This difference, however, might as easily be attributed to naturally greater alkalinities, temperatures, and dissolved solids in the water column in the lower, disjunct grazed reaches of the stream, as to the effects of cattle grazing. Ward (1976) similarly reported that warmer water temperature and higher dissolved solid content of water paralleled increased macroinvertebrate populations in a Colorado mountain river.

In light of the stream continuum concept (Vannote et al. 1980) and watershed principles (Hynes 1975, Likens 1984) operating as discussed by Rinne (1988), it is perhaps surprising that there was any detectable difference between aquatic macroinvertebrate populations in stream reaches in the grazed and ungrazed areas of the Vacas. It is possible that the areas sampled were naturally different both in structure and in macroinvertebrate fauna prior to fencing. There was definitely a marked difference in streambank stability and vegetation between the two areas (Rinne 1985). Lack of any baseline, pretreatment data, however, precludes unequivocably stating that exclusion of grazing enhanced aquatic macroinvertebrate populations in the Vacas.

Based on limited literature and this study, we can state that aquatic macroinvertebrates are useful biological indicators of grazing impacts on stream ecosystems. However, case history studies of grazing effects on this group, on stream habitat, and on fishes (Rinne 1988) emphasize the importance of baseline, pretreatment definition of variability of factors within study areas prior to implementing treatments and ensuing research (Szaro and Rinne 1988).

CONCLUSIONS

Both qualitative and quantitative differences in habitat and biota were found in the Vacas, but lack of pretreatment (prefencing) data precludes making unequivocable statements regarding livestock grazing effects on the habitat and biota of this stream. Many factors create difficulties in a study of this problem (Rinne 1985). Future research on grazing effects on stream habitat and biota must be carefully designed (Rinne 1988) if viable, defendable information is to result.

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